CHAPTER 4

Faecal pollution and water quality

Faecal pollution of recreational water can lead to health problems because of the presence of infectious microorganisms. These may be derived from human sewage or animal sources.

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).

4.1 Approach

Water safety or quality is best described by a combination of sanitary inspection and microbial water quality assessment. This approach provides data on possible sources of pollution in a recreational water catchment, as well as numerical information on the actual level of faecal pollution. Combining these elements provides a basis for a robust, graded, classification as shown in Figure 4.1.

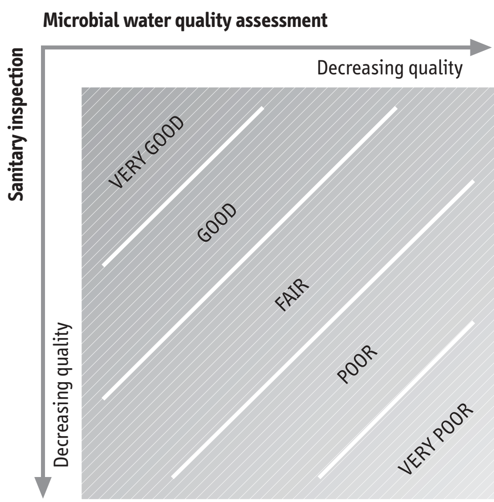


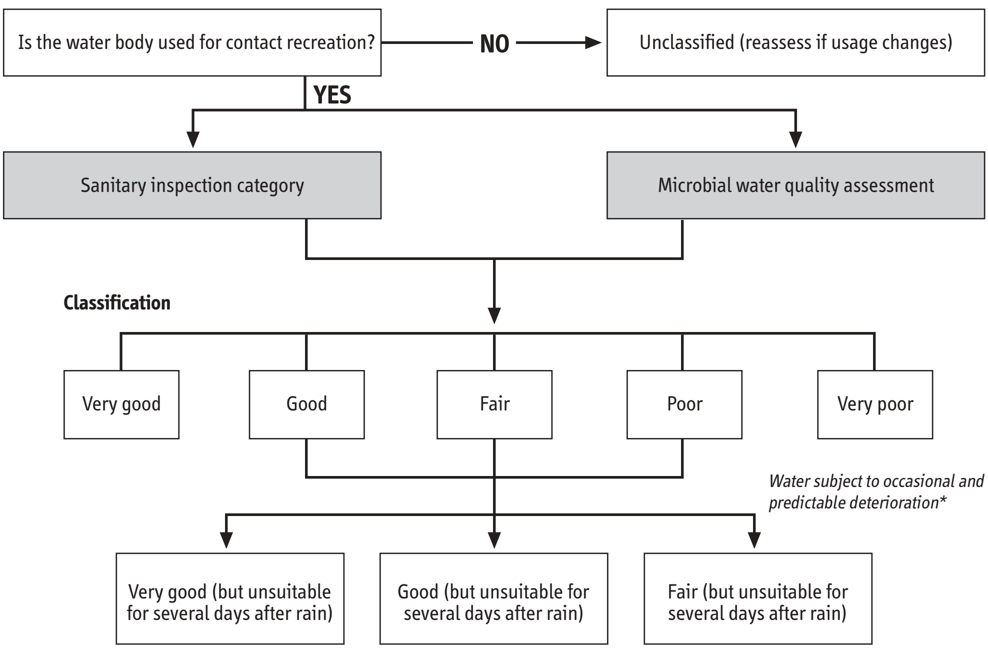
FIGURE 4.1. SIMPLIFIED CLASSIFICATION MATRIX

The results of the classification can be used to:

- grade beaches in order to support informed personal choice;
- provide on-site guidance to users on relative safety;
- assist in the identification and promotion of effective management interventions; and
- provide an assessment of regulatory compliance.

In some instances, microbial water quality may be strongly influenced by factors such as rainfall leading to relatively short periods of elevated faecal pollution. Experience in some areas has shown the possibility of advising against use at such times of increased risk and, furthermore, in some circumstances that individuals respond to such messages. Where it is possible to prevent human exposure to pollution hazards in this way this can be taken into account in both grading and advice. Combining classification (based on sanitary inspection and microbial quality assessment) with prevention of exposure at times of increased risk leads to a framework for assessing recreational water quality as outlined in Figure 4.2.

The resulting classification both supports activities in pollution prevention (e.g., reducing stormwater overflows) and provides a means to recognise and account for local cost-effective actions to protect public health (e.g., advisory signage about rain impacts).



* where users can be shown to be effectively discouraged from entering the water following occasional and predictable water quality deteriorations (linked to, for example, rainfall), the area may be upgraded to reflect the water quality that users are exposed to, but only with the accompanying explanatory material.

FIGURE 4.2. SIMPLIFIED FRAMEWORK FOR ASSESSING RECREATIONAL WATER ENVIRONMENTS

4.2 Health effects associated with faecal pollution

Recreational waters generally contain a mixture of pathogenic and non-pathogenic microorganisms. These microorganisms 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, domestic animals (such as dogs) and wildlife. In addition, recreational waters may also contain free-living pathogenic microorganisms (chapter 5). These sources can include pathogenic organisms that cause gastrointestinal infections following ingestion or infections of the upper respiratory tract, ears, eyes, nasal cavity and skin.

Infections and illness due to recreational water contact are generally mild and so difficult to detect through routine surveillance systems. Even where illness is more severe, it may still be difficult to attribute to water exposure. Targeted epidemiological studies, however, have shown a number of adverse health outcomes (including gastrointestinal and respiratory infections) to be associated with faecally polluted recreational water. This can result in a significant burden of disease and economic loss.

The number of microorganisms (dose) that may cause infection or disease 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., 1994; Teunis, 1996; Haas et al., 1999; Okhuysen et al., 1999; Teunis 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 pathogens in sewage will differ depending on the incidence of disease and carrier states in the contributing human and animal populations and the seasonality of infections. Hence, numbers will vary greatly across different parts of the world and times of year. A general indication of pathogen numbers in raw sewage is given in Table 4.1.

In both marine and freshwater studies of the impact of faecal pollution on the health of recreational water users, several faecal index bacteria, including faecal streptococci/intestinal enterococci (see Box 4.1), have been used for describing water quality. These bacteria are not postulated as the causative agents of illnesses in swimmers, but appear to behave similarly to the actual faecally derived pathogens (Prüss, 1998).

Available evidence suggests that the most frequent adverse health outcome associated with exposure to faecally contaminated recreational water is enteric illness, such as self-limiting gastroenteritis, which may often be of short duration and may not be formally recorded in disease surveillance systems. Transmission of pathogens that can cause gastroenteritis 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).

TABLE 4.1. EXAMPLES OF PATHOGENS AND INDEX ORGANISM CONCENTRATIONS IN RAW SEWAGE[a]

Pathogen/index organism	Disease/role	Numbers per 100 ml
Bacteria		
Campylobacter spp.	Gastroenteritis	10^4–10^5
Clostridium perfringens spores	Index organism	$6 \times 10^4 - 8 \times 10^4$
Escherichia coli	Index organism (except specific strains)	10^6–10^7
Faecal streptococci/intestinal enterococci	Index organism	$4.7 \times 10^3 - 4 \times 10^5$
Salmonella spp.	Gastroenteritis	0.2–8000
Shigella spp.	Bacillary dysentery	0.1–1000
Viruses		
Polioviruses	Index organism (vaccine strains), poliomyelitis	180–500 000
Rotaviruses	Diarrhoea, vomiting	400–85 000
Adenoviruses	Respiratory disease, gastroenteritis	not enumerated[b]
Norwalk viruses	Diarrhoea, vomiting	not enumerated[b]
Hepatitis A	Hepatitis	not enumerated[b]
Parasitic protozoa[c]		
Cryptosporidium parvum oocysts	Diarrhoea	0.1–39
Entamoeba histolytica	Amoebic dysentery	0.4
Giardia lamblia cysts	Diarrhoea	12.5–20 000
Helminths[c] **(ova)**		
Ascaris spp.	Ascariasis	0.5–11
Ancylostoma spp. and *Necator* sp.	Anaemia	0.6–19
Trichuris spp.	Diarrhoea	1–4

[a] Höller (1988); Long & Ashbolt (1994); Yates & Gerba (1998); Bonadonna et al. 2002.

[b] Many important pathogens in sewage have yet to be adequately enumerated, such as adenoviruses, Norwalk-like viruses, hepatitis A virus.

[c] Parasite numbers vary greatly due to differing levels of endemic disease in different regions.

A cause–effect relationship between faecal or bather-derived pollution and acute febrile respiratory illness (AFRI) and general respiratory illness 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 gastroenteritis, probabilities of contacting AFRI are generally lower and the threshold at which illness is observed is higher.

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 index organisms (van Asperen et al., 1995). Associations between ear infections and microbiological indices of faecal pollution and bather load have been reported (Fleisher et al., 1996a). When compared with gastroenteritis, the statistical probabilities are generally lower and are associated with higher faecal index concentrations than those for gastrointestinal symptoms and for AFRI.

BOX 4.1 FAECAL STREPTOCOCCI/INTESTINAL ENTEROCOCCI

Faecal streptococci is a bacterial group that has been used as an index of faecal pollution in recreational water; however, the group includes species of different sanitary significance and survival characteristics (Gauci, 1991; Sinton & Donnison, 1994). In addition, streptococci species prevalence differs between animal and human faeces (Rutkowski & Sjogren, 1987; Poucher et al., 1991). Furthermore, the taxonomy of this group has been subject to extensive revision (Ruoff, 1990; Devriese et al., 1993; Janda, 1994; Leclerc et al., 1996). The group contains species of two genera—*Enterococcus* and *Streptococcus* (Holt et al., 1993). Although several species of both genera are included under the term enterococci (Leclerc et al., 1996), the species most predominant in the polluted aquatic environments are *Enterococcus faecalis*, *E. faecium* and *E. durans* (Volterra et al., 1986; Sinton & Donnison, 1994; Audicana et al., 1995; Borrego et al., 2002).

Enterococci, a term commonly used in the USA, includes all the species described as members of the genus *Enterococcus* that fulfil the following criteria: growth at 10 °C and 45 °C, resistance to 60 °C for 30 min, growth at pH 9.6 and at 6.5% NaCl, and the ability to reduce 0.1% methylene blue. Since the most common environmental species fulfil these criteria, in practice the terms faecal streptococci, enterococci, intestinal enterococci and *Enterococcus* group may refer to the same bacteria.

In order to allow standardization, the International Organization for Standardization (ISO, 1998a) has defined the intestinal enterococci as the appropriate subgroup of the faecal streptococci to monitor (i.e., bacteria capable of aerobic growth at 44 °C and of hydrolysing 4-methylumbelliferyl-β-D-glucoside in the presence of thallium acetate, nalidixic acid and 2,3,5-triphenyltetrazolium chloride, in specified liquid medium). In this chapter, the term intestinal enterococci has been used, except where a study reported the enumeration of faecal streptococci, in which case the original term has been retained.

It may be important to identify human versus animal enterococci, as greater human health risks (primarily enteric viruses) are likely to be associated with human faecal material—hence the emphasis on human sources of pollution in the sanitary inspection categorisation of beach classification (see Table 4.12). Grant et al. (2001) presented a good example of this approach. They demonstrated that enterococci from stormwater, impacted by bird faeces and wetland sediments and from marine vegetation, confounded the assessment of possible bather impact in the surf zone at southern Californian beaches. There will, however, be cases where animal faeces is an important source of pollution in terms of human health risk.

Increased rates of eye symptoms have been reported among swimmers, and evidence suggests that swimming, 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).

Some studies have reported increased rates of skin symptoms among swimmers, and associations between skin symptoms and microbial water quality have also been reported (Ferley et al., 1989; Cheung et al., 1990; Marino et al., 1995; see also chapter 8). Controlled studies, however, have not found such association and the relationship between faecal pollution and skin symptoms remains unclear. Swimmers with exposed wounds or cuts may be at risk of infection (see also chapter 5) but there is no evidence to relate this to faecal contamination.

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. Considering the strong evidence for transmission of self-limiting gastroenteritis, much of which may be of viral etiology, transmission of infectious hepatitis (hepatitis A and E viruses) and poliomyelitis is biologically plausible, should exposure of susceptible persons occur. However, poliomyelitis was not found to be associated with bathing in a 5-year retrospective study relying on total coliforms as the principal water quality index (Public Health Laboratory Service, 1959). Furthermore, sero-prevalence studies for hepatitis A among windsurfers, waterskiers and canoeists who were exposed to contaminated waters have not identified any increased health risks (Philipp et al., 1989; Taylor et al., 1995). However, 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). Also, significantly higher rates of typhoid have been observed in Egypt among bathers from beaches polluted with untreated sewage compared to bathers swimming off relatively unpolluted beaches (El Sharkawi & Hassan, 1982).

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

Outbreak reports have noted cases of diverse health outcomes (e.g., gastrointestinal symptoms, typhoid fever, meningoencephalitis) 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 swimming 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 at a Hawaiian beach (Reynolds et al., 1998). *Leptospira* sp. are usually associated with animals that urinate into surface waters, and swimming-associated outbreaks attributed to *Leptospira* sp. are rare (see chapter 5). Conversely, outbreaks of acute gastrointestinal infections with an unknown etiology are common, with the symptomatology of the illness frequently being suggestive of viral infections. The serological data shown in Table 4.3 suggest that Norwalk virus has more potential than rotavirus to cause swimming-associated gastroenteritis (WHO, 1999), although these results were based on a limited number of subjects. Application of reverse transcriptase-polymerase chain reaction technology has 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 CHILDREN 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 WHO (1999).

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

4.3 Approaches to risk assessment and risk management

Regulatory schemes for the microbial quality of recreational water have been largely based on percentage compliance with faecal index organism 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 indices of faecal pollution are also derived from other sources. The response to non-compliance, however, typically concentrates on sewage treatment or outfall management as outlined below.

- There is poor interlaboratory comparability of microbiological analytical data.
- Beaches are classified as either safe or unsafe, although there is, in fact, a gradient of increasing variety and frequency of health effects with increasing faecal pollution of human and animal origin.

Traditionally, regulation tends to focus response upon sewage treatment and outfall management as the principal, or only, interventions. Due to the high costs of these measures coupled with the fact that local authorities are generally not the sewerage undertaker, local authorities may be relatively powerless, 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 may often not justify the proposed investments which may also be ineffective in securing regulatory compliance, particularly if non-human, diffuse faecal sources and/or stormwaters are major contributor(s) (Kay et al., 1999). Furthermore, the costs may be prohibitive or may divert 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 largely be overcome by a monitoring scheme that combines microbial 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.

4.3.1 Harmonized approach and the "Annapolis Protocol"

A WHO expert consultation in 1999 formulated a harmonized approach to assessment of risk 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 4.3 (Bartram et al., 2001).

The "Annapolis Protocol" (WHO, 1999; Bartram & Rees, 2000—chapter 9) represents an adaptation of the "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 developments recommended in the Annapolis Protocol were:

- the move away from the reliance on numerical values of faecal index bacteria as the sole compliance criterion to the use of a two component qualitative ranking of faecal loading in recreational water environments, supported by direct measurement of appropriate faecal indices; and

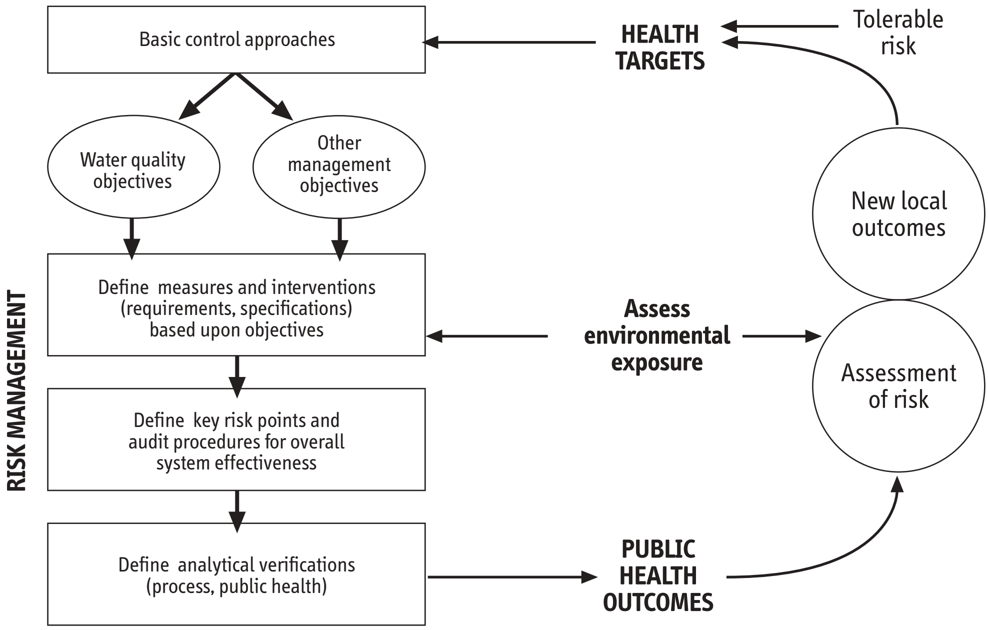


FIGURE 4.3. HARMONIZED APPROACH TO ASSESSMENT OF RISK AND RISK MANAGEMENT FOR WATER-RELATED EXPOSURE TO PATHOGENS (ADAPTED FROM BARTRAM ET AL., 2001)

- provision to account for the impact of actions to discourage water use during periods, or in areas, of higher risk.

The protocol has been tested in various countries, and recommendations resulting from these trials 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 safe Recreational Water Environments*, which is described in sections 4.5 and 4.6.

4.3.2 Risk assessment

Assessing the risk associated with human exposure to faecally polluted recreational waters can be carried out directly via epidemiological studies or indirectly through quantitative microbial risk assessment (QMRA). Both methods have advantages and limitations.

Epidemiological studies have been used to demonstrate a relationship between faecal pollution (using bacterial index organisms) and adverse health outcomes (see section 4.2 and Prüss, 1998). Some types of epidemiological studies are also suitable to quantify excess risk of illness 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 reviewed by Prüss (1998).

From a review of the literature, one (or more) key epidemiological study may be identified that provides the most convincing data with which to assess quantitatively the relation between water quality (index organism) data and adverse health outcomes. The series of randomized epidemiological investigations, conducted in the United Kingdom, provide such data for gastroenteritis (Kay et al., 1994), AFRI and ear ailments associated with marine bathing (Fleisher et al., 1996a). These studies are described in more detail in section 4.4.1.

QMRA can be used to indirectly estimate the risk to human health by predicting infection or illness rates given densities of particular pathogens, assumed rates of ingestion and appropriate dose-response models for the exposed population. Application of QMRA to recreational water use is constrained by the current lack of specific water quality data for many pathogens and the fact that pathogen numbers, as opposed to faecal index organisms, 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. However, caution is required in interpretation because the risk of infection or illness from exposure to pathogenic microorganisms is fundamentally different from the risk associated with other contaminants, such as toxic chemicals. Several of the key differences between exposure to pathogens and toxic chemicals are:

- exposure to chemical agents occurs via an environment-to-person pathway. Exposure to pathogens can occur via an environment-to-person pathway, but can also occur due to person-to-person contact (secondary spread);
- whether a person becomes infected or ill after exposure to a pathogen may depend on the person's pre-existing immunity. This condition implies that exposure events are not independent;
- infectious individuals may be symptomatic or asymptomatic;
- 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; and
- pathogens are generally not evenly suspended in water.

Although the differences between exposure to chemical agents and pathogenic microorganisms are widely acknowledged, the conceptual framework for chemical risk assessment (Table 4.4) has been commonly employed for assessing the risk associated with exposure to pathogenic microorganisms. Frameworks have been developed specifically to assess the risks of human infection associated with exposure to pathogenic microorganisms and to account for some of the perceived shortcomings of the chemical risk framework with respect to properties unique to infectious microorganisms. However, to date, these frameworks have not been widely adopted.

In employing the chemical risk framework to carry out a SLRA, a representative pathogen is used to conservatively characterize its microbial group. For example, the occurrence of adenovirus, with its associated dose–response curve, may be used as a predictor for enteric viruses. 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 an order of magnitude estimate of risk, whether or not further data are required and if risks are likely to be dominated by a single class of pathogen or source (potentially defining options for risk management). It should be emphasized that this SLRA approach presumes that little net error is made by not accounting for either person-to-person transmission of disease or immunity.

TABLE 4.4. RISK ASSESSMENT PARADIGM FOR ANY HUMAN HEALTH EFFECT[a]

Step	Aim
1. 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 hazard identification steps in order to estimate the magnitude of the public health problem and to evaluate variability and uncertainty.

[a] Adapted from NRC, 1983.

Given the somewhat limited array of microorganisms for which a dose–response relationship has been estimated, SLRAs are currently limited to a few microorganisms, such as rotavirus, adenovirus, *Cryptosporidium parvum*, *Giardia lamblia* and *Salmonella* spp. (Haas et al., 1999). A screening-level QMRA approach is outlined for a recreational water example in Box 4.2 (adapted from Ashbolt et al., 1997).

A more comprehensive alternative to the SLRA approach is to employ a population based disease transmission model to assess the risks of human disease associated with exposure to pathogenic microorganisms. In this population-based approach, the potential for person-to-person transmission and immunity are accounted for (Eisenberg et al., 1996; Soller, 2002), however, the models require substantially more epidemiological and clinical data than SLRA models. Application of the disease transmission modelling approach may, therefore, be more limited than the SLRA approach.

The primary advantages of QMRA studies are that the potential advantages and limitations of risk management options may be explored via numerical simulation to examine their potential efficacy, and that risk below epidemiologically detectable levels may be estimated under certain circumstances. The limitations of QMRA studies, as noted earlier, are that limited data are available to carry out these assessments and, in many cases, the data that are available are highly uncertain and variable. Nevertheless, it may be inferred from several of the available QMRA studies

(Sydney and Honolulu) (Mamala Bay Study Commission, 1996; Ashbolt et al., 1997) that they provide supporting evidence for the results of various epidemiological studies.

BOX 4.2 SCREENING-LEVEL QMRA APPROACH FOR BATHER RISK (ADAPTED FROM ASHBOLT ET AL., 1997)

For a predominantly sewage-impacted recreational water, the concentration of pathogens in waters may be estimated from the mean pathogen densities in sewage and their dilution in recreational waters (based on the numbers of index organisms; see Table 4.5 below). As an initial conservative approximation of pathogen numbers in recreational 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 recreational waters may be attempted (Reynolds et al., 1998). Next, a volume of recreational water ingestion is required to determine the pathogen dose, in this instance 20–50 ml of water per hour of swimming has been assured.

TABLE 4.5. GEOMETRIC MEAN INDEX ORGANISMS AND VARIOUS PATHOGENS IN PRIMARY SEWAGE EFFLUENT IN SYDNEY, AUSTRALIA[a]

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

[a] Index bacteria and parasite data are from Long & Ashbolt (1994).

[b] Total enteric virus estimate of 5650 for raw sewage is 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 have been determined, 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 health protective 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). Employing the framework described above for chemical agents, 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, QMRA can be a useful tool for screening the risk to public health at recreational water sites and for determining the potential efficacy of management alternatives through the integration of a wide array of disparate data. Finally, QMRA provides credible scientific analysis that can be used in conjunction with or, at times, in lieu of epidemiological investigations to assess risk to human health at recreational water sites.

4.3.3 *Risk management*

To meet health targets ultimately based on a tolerable risk of illness (see section 4.4), achievable objectives need to be established for water quality and associated management. Hazard analysis and critical control point (HACCP) provides an example of a possible approach. It is a risk management tool that promotes good operational/management practice and is an effective quality assurance (QA) system that is used in the food and beverage industry (Deere et al., 2001). It 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 and WHO Codex Alimentarius Commission. Water Safety Plans (WSP) for drinking-water have been developed from the HACCP approach (WHO, 2003).

For recreational waters, the HACCP approach has been interpreted as described in Table 4.6. 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.

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 subareas of a single recreational water environment. It may be 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 water contact during these periods may be inexpensive (especially where large point sources are concerned), greater cost effectiveness and improved possibilities for effective local management intervention are possible.

4.4 Guideline values

In many fields of environmental health, guideline values 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, such as DDT (*p,p′*-dichlorodiphenyl trichloroethane) and copper.

For other chemicals in drinking-water, such as genotoxic carcinogens, there is no "safe" level of exposure. In these cases, guidelines (including WHO guideline values; WHO, 1996) 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

TABLE 4.6. IMPLEMENTATION OF HACCP APPROACH FOR RECREATIONAL WATER MANAGEMENT

Initial steps	Implementation
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 recreational water 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 recreational water 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 of different sources of faecal contaminants to be pinpointed and any detected contamination to be traced.
Core principles	
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.
Critical control points	• Identify those points or locations at which management actions can be applied to reduce the presence of, or exposure to, hazards to acceptable levels. Examples include municipal sewage discharge points, treatment works operation, combined sewer overflows, illegal connections to combined sewers, etc.
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).
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 also include, for example, visual inspection of potential sources of contamination in catchment or flow/overflow gauges.
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.
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.
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.

100 000 excess incidence of cancer over a lifetime of exposure. Such levels may be termed tolerable risk levels.

Guideline values and standards for microbial water quality were originally developed to prevent the occurrence of outbreaks of disease. However, there was limited information available concerning the degree of health protection they provided. 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 any given range of water quality. Further information on this is available in section 4.4.1, which illustrates the association of gastrointestinal illness and respiratory illness with microbial 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, or tolerable, in other circumstances (such as carcinogens in drinking-water). However, 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 and would detract from the beneficial effects of recreational water use.

The approach adopted here therefore recommends that a range of water quality categories be defined and individual locations be classified according to these (see sections 4.4.3 and 4.6). The use of multiple categories provides incentive for progressive improvement throughout the range of qualities in which health effects are believed to occur.

4.4.1 *Selection of key studies*

Numerous studies have shown a causal relationship between gastrointestinal symptoms and recreational water quality as measured by index 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, although various biases can occur with all epidemiological studies (Prüss, 1998).

[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.

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 index 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.

The randomized controlled trials conducted in marine waters in the United Kingdom (Kay et al., 1994; Fleisher et al., 1996a; Kay et al., 2001) provide the most convincing data. 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.

These trials therefore form the key studies for derivation of guideline values for recreational waters (Box 4.3). However, it should be emphasized that they are primarily indicative for healthy adult populations in sewage impacted 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 UK studies may therefore systematically underestimate risks to children.

BOX 4.3 KEY STUDIES FOR GUIDELINE VALUE DERIVATION

The randomized trials reported by Kay et al. (1994) and Fleisher et al. (1996a) were designed to overcome significant "misclassification" (e.g., attributing a daily mean water quality to all bathers) and "self-selection" (e.g., the exposed bathers may have been more healthy at the outset) biases present in earlier studies. Both effects would have led to an underestimation of the illness rate.

This was done by recruiting healthy adult volunteers in urban centres during the four weeks before each of the four studies (i.e., the volunteers may not represent the actual population at a beach as well as did participants in the earlier prospective studies), conducted from 1998 to 1992 at United Kingdom beaches that were sewage impacted but passed existing European Union "mandatory" 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 ten min and immersed their heads three times during that period. The water in the recreational area was intensively sampled during the swimming 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 indices were measured synchronously at three depths during this process. Enumeration of indices was completed using triplicate filtration to minimize bias caused by the imprecision of index organism measurement in marine waters. All volunteers were interviewed on the day of exposure and at one week post-exposure, and they completed a postal questionnaire at three 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 index organism concentration encountered at the time and place of exposure.

Gastroenteritis rates in the bather group were predicted by faecal streptococci (i.e., intestinal enterococci) measured at chest depth (with gastroenteritis being based on accepted definitions in Europe and North America such as loose bowel motions, fever and vomiting). This relationship was observed at three of the four study sites; at the fourth, very low concentrations of this index organism were observed.

Only faecal streptococci, measured at chest depth, showed a dose–response relationship for both gastrointestinal illness (Kay et al., 1994) and AFRI (Fleisher et al., 1996a) in marine waters. Bathers had a statistically significant increase in the occurrence of AFRI at levels at or above 60 faecal streptococci/100 ml. While a significant dose–response relation with gastroenteritis was identified when faecal streptococci concentrations exceeded approximately 32/100 ml. No dose–response relationships with other illnesses were identified.

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

The maximum level of faecal streptococci measured in these trials was 158 faecal streptococci/100 ml (Kay et al., 1994). The dose–response curve for gastroenteritis derived from these studies, and used in deriving the guidelines below, is limited to values in the range commencing where a significant effect was first recorded, 30–40 faecal streptococci/100 ml, to the maximum level detected. The probability of gastroenteritis or AFRI at levels higher than these is unknown. In estimating the risk levels for exposures above 158 faecal streptococci/100 ml, it is assumed that the probability of illness remains constant at the same level as exposure to 158 faecal streptococci/100 ml (i.e., an excess probability of 0.388), rather than continuing to increase. This assumption is likely to underestimate risk and may need review as studies become available that clarify the risks attributable to exposures above these levels.

Discussion has arisen concerning the steep dose–response curve reported in these studies, compared with previous studies. The best explanation of the steeper curve appears to be that with less misclassification and other biases, a more accurate measure of the association between index organism numbers and illness rates was made. In addition, the key studies examined beaches with direct sewage pollution, and it is possible that other pollution risks may result in a different (lower) risk. A 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).

4.4.2 *The 95th percentile approach*

Many agencies have chosen to base criteria for recreational water compliance upon either percentage compliance levels, typically 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 statistical 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. When calculating percentiles it is important to note that there is no one correct way to do the calculation. It is therefore desirable to know what method is being used, as each will give a different result (see Box 4.4).

4.4.3 *Guideline values for coastal waters*

The guideline values for microbial water quality given in Table 4.7 are derived from the key studies described above. The values are expressed in terms of the 95th percentile of numbers of intestinal enterococci 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 4.6.

4.4.4 *Guideline values for fresh water*

Dufour (1984) discussed the significant differences in swimming-associated gastrointestinal illness rates in seawater and freshwater swimmers at a given level of faecal index organisms. The illness rate in seawater swimmers was about two 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, although it should be noted that the research groups used very different methodologies. At the same intestinal enterococci densities, the 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 index 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 index organism densities are identical, which would logically lead to a higher swimming-associated gastrointestinal illness rate in seawater swimmers.

BOX 4.4 PERCENTILE CALCULATION

Individual regulatory authorities should decide on the most appropriate percentile calculation approach, based on data availability, statistical considerations and local resources. Two main approaches can be used. In the parametric approach it is assumed that the samples have been drawn from a particular distribution. This is typically the $\log_{10}$ normal distribution for microbiological data and so one uses the 95 percentile of that distribution, calculated from the mean and standard deviation of the logarithms of the data. The nonparametric approach does not assume any particular distribution and uses data ranking.

The parametric approach is outlined in Bartram & Rees (2000). This approach requires sufficient data to define the mean and standard deviations of the $\log_{10}$ bacterial enumerations. It also assumes that the dilution policy applied by the microbiology laboratories has been applied so as to not produce data items reported as, for example, <100 per 100 ml. For data sets with sufficient entries and appropriate dilution policy, the 95 percentile point of the probability density function (PDF) is defined as follows:

$$\text{Log}_{10}\ 95\%\text{ile} = \text{Arithmetic mean } \log_{10} \text{ bacterial concentration} + (1.6449 \times \text{standard deviation of } \log_{10} \text{ bacterial concentration})$$

In calculating this statistic for a column of bacterial data acquired from one beach, all enumerations should be converted to $\log_{10}$ values and the calculations of mean and standard deviation should be completed on the $\log_{10}$ transformed data.

Sample percentiles can also be calculated by a two-step non-parametric procedure. Firstly the data are ranked into ascending order and then the "rank" of the required percentile calculated using an appropriate formula—each formula giving a different result. The calculated rank is seldom an integer and so in the second step an interpolation is required between adjacent data using the following formula:

$$X_{0.95} = (10 - r_{\text{frac}})X_{r_{\text{int}}} + r_{\text{frac}} X_{r_{\text{int}}+1}$$

where $X_{0.95}$ is the required 95 percentile, $X_1, X_2, \ldots, X_n$ are the n data arranged in ascending order and the subscripts r_{frac} and r_{int} are the fractional and integer parts of r.

RANKING FORMULAE

Three formulae are in use in the water industry (Ellis 1989), covering the range of estimates that may be made: Weibull, Hazen and Excel™. Their formulae are: $r_{\text{Weibull}} = 0.95(n + 1)$, $r_{\text{Hazen}} = \frac{1}{2} + 0.95n$, and $r_{\text{Excel}} = 1 + 0.95(n - 1)$. An example calculation using the Weibull formula is presented in Bartram & Rees (2000, Table 8.3). It needs at least 19 samples to work, and always gives the highest result. The Hazen formula needs only 10 samples to work, while the Excel™ formula needs only one sample and always gives the lowest result.

EXAMPLE CALCULATION

Say that we have 100 data of which the six highest are: 200, 320, 357, 389, 410, 440. Then we have $r_{\text{Hazen}} = 95.5$ and so the 95 percentile estimated by the Hazen formula is $X_{0.95} = (0.5 \times 200) + (0.5 \times 320) = 260$.

Note that using the Weibull formula we have $r_{\text{Weibull}} = 95.95$ and so the 95 percentile estimated by the Weibull formula is $X_{0.95} = (0.05 \times 200) + (0.95 \times 320) = 314$, while for the method used in Excel™ we have $r_{\text{Excel}} = 95.05$ and so the 95 percentile estimated by the Excel formula is $X_{0.95} = (0.95 \times 200) + (0.05 \times 320) = 206$—much lower than the Weibull result.

TABLE 4.7. GUIDELINE VALUES FOR MICROBIAL QUALITY OF RECREATIONAL WATERS

95th percentile value of intestinal enterococci/100 ml (rounded values)	Basis of derivation	Estimated risk per exposure
≤40 **A**	This range is below the NOAEL in most epidemiological studies.	**<1% GI illness risk** **<0.3% AFRI risk** The upper 95th percentile value of 40/100 ml relates to an average probability of less than one case of gastroenteritis in every 100 exposures. The AFRI burden would be negligible.
41–200 **B**	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** **0.3–1.9% AFRI risk** The upper 95th percentile value of 200/100 ml relates to an average probability of one case of gastroenteritis in 20 exposures. The AFRI illness rate at this upper value would be less than 19 per 1000 exposures, or less than approximately 1 in 50 exposures.
201–500 **C**	This range 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 risk** This range of 95th percentiles represents a probability of 1 in 10 to 1 in 20 of gastroenteritis 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 **D**	Above this level, there may be a significant risk of high levels of minor illness transmission.	**>10% GI illness risk** **>3.9% AFRI risk** There is a greater than 10% chance of gastroenteritis per single exposure. The AFRI illness rate at the 95th percentile point of >500/100 ml would be greater than 39 per 1000 exposures, or greater than approximately 1 in 25 exposures.

Notes:

1. Abbreviations used: A–D are the corresponding microbial water quality assessment categories (see section 4.6) used as part of the classification procedure (Table 4.12); 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 of swimming involving three head 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 8).
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 further illness outside the range of the data (i.e., at concentrations above 158 intestinal enterococci/100 ml; see Box 4.3). Thus, the estimates of illness rate reported above this value are likely to be underestimates of the actual disease incidence attributable to recreational water exposure.
5. The estimated risks were derived from sewage-impacted marine waters. Different sources of pollution and more or less aggressive environments may modify the risks.
6. This table is derived from risk to healthy adult bathers exposed to marine waters in temperate north European waters.

TABLE 4.7. *Continued*

7. This table may not relate to children, the elderly or the immunocompromised, who could have lower immunity and might require a greater degree of protection. There are presently no adequate data with which to quantify this, and no correction factors are therefore applied.
8. Epidemiological data on fresh waters or exposures other than swimming (e.g., high-exposure activities such as surfing, dinghy boat sailing or whitewater canoeing) are currently inadequate to present a parallel analysis for defined risks. Thus, a single series of microbial values is proposed, for all recreational uses of water, because insufficient evidence exists at present to do otherwise. However, it is recommended that the length 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).
9. Where disinfection is used to reduce the density of index organisms in effluents and discharges, the presumed relationship between intestinal enterococci (as an index of faecal contamination) and pathogen presence may be altered. This alteration is, at present, poorly understood. In water receiving such effluents and discharges, intestinal enterococci counts may not provide an accurate estimate of the risk of suffering from gastrointestinal symptoms or AFRI.
10. 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 for faecal streptococci was assumed. If the true standard deviation for a beach is less than 0.8103, then reliance on this approach would tend to overestimate the health risk for people exposed above the threshold level, and vice versa.
11. Note that the values presented in this table do not take account of health outcomes other than gastroenteritis and AFRI. Where other outcomes are of public health concern, then the risks should also be assessed and appropriate action taken.
12. 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.

BOX 4.5 DIFFERENTIAL DIE-OFF OF INDEX BACTERIA AND PATHOGENS IN SEAWATER AND FRESH WATER

Salinity appears to accelerate the inactivation of sunlight-damaged coliforms in marine environments, such that coliforms are appreciably less persistent than intestinal enterococci in seawater. Cioglia & Loddo (1962) showed that poliovirus, echovirus and coxsackie virus were inactivated at approximately the same rate in marine and fresh waters (Table 4.8), 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 4.8. SURVIVAL OF ENTEROVIRUSES IN SEAWATER AND RIVER WATER[a]

	Die-off rates (in days)[b]	
Virus strain	**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).

It appears likely that bacterial index organisms have different die-off characteristics in marine and fresh waters, while human viruses are inactivated at similar rates in these environments.

Thus, application of the guideline values derived above for seawaters (Table 4.7) to fresh waters would be likely to result in a lower illness rate in freshwater users, providing a conservative (i.e., more protective) guideline in the absence of suitable epidemiological data for fresh waters.

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

Studies using a randomized trial design have been conducted in Germany at freshwater sites. These have yet to be reported in the peer-reviewed literature. Initial reports (Wiedenmann et al., 2002) suggest that these studies have identified similar thresholds of effect to those reported in Kay et al. (1994). Until the full results of these investigations become available, there is inadequate evidence with which to directly derive a water quality guideline value for fresh water.

The guideline value derived for coastal waters can be applied to fresh water until review of more specific data has been undertaken.

4.4.5 Adaptation of guideline values to national/local circumstances

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 (gastroenteritis, 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 sociocultural, economic, environmental and technical aspects, as well as competing health risk from other diseases including those 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 than those presented in Table 4.7. What signifies an acceptable or tolerable risk is not only a regional or local issue, however, as even within a region or locality children, the elderly 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 4.7 were derived from studies involving healthy adult bathers swimming in sewage impacted marine waters in a temperate climate. Thus, the Guidelines do not relate specifically to children, the elderly or immunocompromised, who may 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 of, or different, pathogens), and stricter standards may be judged appropriate by local authorities.

If a 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 studies in which the "exposure" was a minimum of ten minutes of swimming involving three head immersions. They may therefore underestimate risk for activities involving higher risks of water ingestion or longer periods of water contact. Recreational water uses involving lesser degrees of water contact (such as windsurfing and sea canoeing) will usually result in less water ingestion and thus may require less stringent guideline values to achieve equivalent health protection.

When information on "typical" swimmers (e.g., age, number of swimming events per swimming season per swimmer, average amount of water swallowed per swimming 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" swimmer over a fixed period of time.

Use of a range of categories, rather than a simple pass/fail approach, supports the principle of informed personal choice. It also allows achievable improvement targets to be set for high-risk areas, rather than an "across the board" target which may result in less overall health gain.

Pathogens and faecal index organisms are inactivated at different rates, dependent on physicochemical conditions. Therefore, any one index organism is, at best, only an approximate index of pathogen removal efficacy in water (Davies-Colley et al., 2000; Sinton et al., 2002; Box 4.5). This suggests that factors influencing faecal index organism die-off should be taken into consideration when applying the guideline values in Table 4.7, depending on local circumstances. This is particularly true where sewage is disinfected prior to release, as this will markedly affect the pathogen/index organism relationship.

Objective input for the adaptation of guidelines to standards may be informed by quantitative microbial risk assessment (QMRA), as outlined in section 4.3.2. Thus, a screening-level QMRA is recommended where differential persistence of faecal index organisms 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 (such as disinfection) prior to discharge.

Adaptation of guideline values to national or local circumstances may be informed by reference levels of risk using, for example, disability adjusted life years per person per year, comparing risks considered tolerable for drinking-water, for example, with risks from recreational water use. Alternatively, exposure to recreational 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, 0.9–9.7% from a range of marine and freshwater studies (Cabelli et al., 1982; Kay et al., 1994; van Asperen et al., 1998). Based on the key studies of coastal bathers in the United Kingdom, Wyer et al. (1999) provided an example of tolerable risk in terms of faecal index 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 index organism exposure can then be used in conjunction with the distribution of faecal index bacteria in recreational water to yield prospective microbial water quality criteria or actual expected disease burden at a particular recreational water location.

The guideline values, defined in Table 4.7, were derived using an average value for the standard deviation of the PDF for faecal streptococci of 0.8103 (as a $\log_{10}$ faecal streptococci/100 ml value), calculated from a survey of 11 000 European recreational 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). Thus, the effect of using a fixed standard deviation for all recreational water environments is variable.

The adaptation of guidelines to form national standards, for example, and the subsequent regulation of recreational waters is also examined in section 4.7.3 and chapter 13.

4.4.6 Regulatory parameters of importance

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

- have a 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 low cost to test;
- make low demands on staff training; and
- require basic equipment that is readily available.

Microorganisms commonly used in regulation include the following:

- **Intestinal enterococci** meet all of the above.
- ***E. coli*** is intrinsically suitable for fresh waters but not marine water; however, as discussed in section 4.4.4, there are currently insufficient data with which to develop guideline values using this parameter in fresh water.
- **Total coliforms** are inadequate for the above criteria, in particular as they are not specific to faecal material.

- **Thermotolerant coliforms**, although a better index than total coliforms, include non-faecally derived organisms (e.g., *Klebsiella* can derive 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.
- **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 low infectivity and typically relatively low numbers in sewage, which, when combined with their rapid inactivation in waters, particularly seawaters, suggest 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 is 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 guideline values. Their direct health significance varies from negligible (e.g., vaccine strains) to very high.

4.5 Assessing faecal contamination of recreational water environments

The two principal components required for assessing faecal contamination of recreational water areas are:

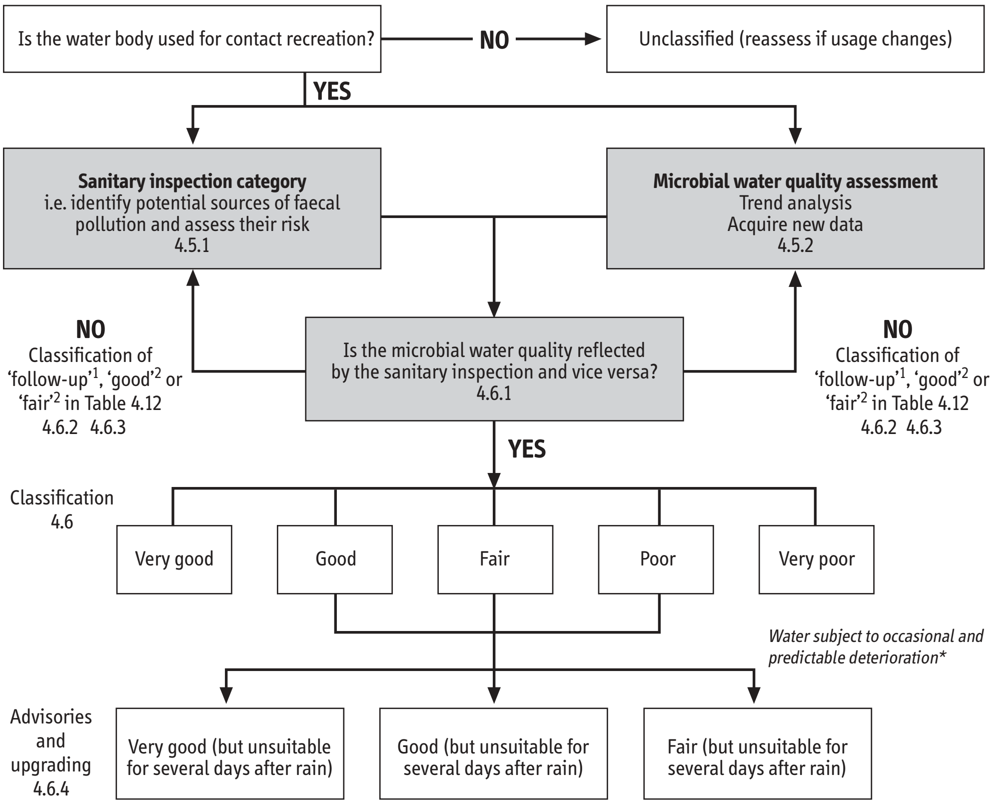
- assessment of evidence for the degree of influence of faecal material (i.e., derivation of a sanitary inspection category); and
- counts of suitable faecal index bacteria (a microbial water quality assessment).

These would be done for the purposes of classification only where a recreational water is used for whole-body contact recreation (i.e., where there is a meaningful risk of swallowing water). The two components are combined (as outlined in section 4.6 and Figure 4.4) in order to produce an overall classification.

4.5.1 Sanitary inspection category

Sources of faecal pollution have been outlined in section 4.2. The sanitary inspection should aim to identify all sources of faecal pollution, although human faecal pollution will tend to drive the overall sanitary inspection category for an area.

The three most important sources of human faecal contamination of recreational water environments for public health purposes are typically 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 contamination from bathers (including excreta). Other sources of human faecal contamination include septic tanks near the shore (leaching directly into groundwater seeping into the recreational water environment) and shipping and local boating (including moorings and special events such as regattas).



* where users can be shown to be effectively discouraged from entering the water following occasional and predictable water quality deteriorations (linked to, for example, rainfall), the area may be upgraded to reflect the water quality that users are exposed to, but only with the accompanying explanatory material.

FIGURE 4.4. FRAMEWORK FOR ASSESSING RECREATIONAL WATER ENVIRONMENTS (NUMBERS REFER TO SECTIONS IN CHAPTER)

Information to be collected during sanitary inspections should at least cover the following:

- Sewage outfalls, combined sewer overflows, stormwater discharges
  - 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 the bathing season

- Bather shedding
  - —Bather density in the swimming season
  - —Dilution (mixing of water in recreational water area)

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 or water release (e.g., dam-controlled rivers); and
- coastal physiography.

Index organism densities in recreational waters can be increased to high levels following rainfall because treatment plants may be overwhelmed (causing sewage to bypass treatment) or because of animal wastes washed from forestland, pastureland and urban settings. Resuspension 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. In adapting guidelines, 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 to 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 to present 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 most readily categorized, in terms of its sanitary inspection, according to the single most significant source of pollution.

The following subsections assist in placing recreational water environments into an appropriate sanitary inspection category indicative of susceptibility to human faecal pollution, but cannot account fully for local and regional factors.

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

Sewage-related risk arises from 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" 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, outfall length is generally less important than proper location and effective diffusion, which should 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 cooperate with appropriate authorities to eliminate this practice or to minimize recreational use of affected areas. 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 reaching the recreational area.

In public health terms, it is generally assumed that the processes of dispersion, dilution, sedimentation and inactivation (through sunlight, predation, natural die-off, etc.) following discharge into the aquatic environment from a piped outfall will lead to a certain degree of safety. A number of confounding factors reduce the efficiency of this in practice. Most important 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 slicks 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, such as advisory notices (sections 4.6.4 and 4.7.1), zoning or banning of swimming and other water contact activities, taken as appropriate. Coastal currents and tides may give rise to similar problems and may 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 swimming season, water users are effectively protected 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 swimming 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, the geographical configuration of the area and environmental factors as discussed above.

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

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

Of these, lagooning, primary plus secondary treatment, tertiary treatment and disinfection will effect a significant reduction in index organism and pathogen contamination. Some treatments, notably disinfection (in particular, chlorination), may affect the validity of the microbial water quality assessment due to differential attenuation of index and pathogenic organisms. This will tend to lead to underestimates of risk, particularly with disinfection-resistant enteric viruses and chlorine-resistant *Cryptosporidium*. Where the principal human faecal pollution source is disinfected sewage, it is suggested that supplementary investigations be undertaken because of the likely underestimate of health risk based on Table 4.7.

Urban stormwater runoff and outputs from combined sewer overflows are included 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, hydrometeorological and oceanographic conditions. The potential risk to human health through exposure to sewage through outfalls can be categorized as shown in Table 4.9.

2. *Riverine discharges*

Rivers discharging into recreational water 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 and rural stormwater overflows (including natural water courses - torrents - that only drain storm water) 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

TABLE 4.9. RELATIVE RISK POTENTIAL TO HUMAN HEALTH THROUGH EXPOSURE TO SEWAGE THROUGH OUTFALLS (INCLUDING STORMWATER RUNOFF AND COMBINED SEWER OVERFLOWS)

	Discharge type		
Treatment	**Directly on beach**	**Short outfall**[a]	**Effective outfall**[b]
None[c]	Very high	High	NA[d]
Preliminary	Very high	High	Low
Primary (including septic tanks)	Very high	High	Low
Secondary	High	High	Low
Secondary plus disinfection[e]	—	—	—
Tertiary	Moderate	Moderate	Very low
Tertiary plus disinfection[e]	—	—	—
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 if active during the bathing season (a history of total non-discharge during the bathing season can be treated as "Low").

[d] NA = not applicable

[e] Additional investigations recommended to account for the likely lack of prediction with faecal index organisms as outlined in Table 4.7.

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 (see sections 4.6.4 and 4.7.1).

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 resuspension of sediment. Rivers may be receiving environments for sewage effluents which may be treated to varying degrees. Much lower levels of effluent dilution may occur in riverine environments than in their coastal equivalents, and differential pathogen–index organism relationships may exist between saline and non-saline waters (see section 4.4.4, Box 4.5).

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. Where human faecal waste is not present but animal waste from, for example, animal husbandry is present this should be taken into account.

3. *Bather shedding*

Bathers themselves can influence water quality directly (Eisenberg et al., 1996). For example, Papadakis et al. (1997) collected water and sand samples from two beaches, counted the swimmers present on the beaches and conducted microbiological tests for counts of coliforms, thermotolerant coliforms, enterococci, *Staphylococcus aureus*, yeasts and moulds. There was a significant correlation between the number of swimmers present on the beach and *S. aureus* counts in water samples, the correlation being more pronounced on the more popular of the two beaches. Yeasts of human origin in water samples also were correlated with the number of swimmers on the more popular beach.

TABLE 4.10. RELATIVE RISK POTENTIAL TO HUMAN HEALTH THROUGH EXPOSURE TO SEWAGE THROUGH RIVERINE FLOW AND DISCHARGE

	Treatment level				
Population and flow characteristics[a,b]	**None**	**Primary**	**Secondary**	**Secondary plus disinfection**[c]	**Lagoon**
High population with low river flow	Very high	Very high	High	—	Moderate
Low population with low river flow	Very high	High	Moderate	—	Moderate
Medium population with medium river flow	High	Moderate	Low	—	Low
High population with high river flow	High	Moderate	Low	—	Low
Low population with high river flow	High	Moderate	Very low	—	Very low

[a] The population factor includes, in principle, 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 of primary concern is the lowest typical flow during the bathing season (excluding combined sewer overflow and stormwater; see Table 4.9).

[c] Additional investigations recommended to account for the likely lack of prediction with faecal index organisms as outlined in Table 4.7.

The effect of bathers on water quality is most commonly seen as microbial buildup 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 epidemiological studies (Calderon et al., 1991), several outbreaks of disease (see section 4.2) 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.

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.

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.

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 resuspended by water users and/or rainfall events. The health risks associated with resuspended sediments remain poorly understood, but should be noted as a potential risk during sanitary surveys.

4. *Animal inputs*

Although the sanitary inspection category is principally driven by human faecal inputs, it is important to determine major sources of animal faecal pollution. These will often be less important in terms of human health risk than human pollution, although in some instances they can have a significant impact on microbial water quality and health risk (see 4.6.2).

4.5.2 Microbial water quality assessment

The various stages involved in an assessment of the microbial quality of a recreational water environment are described elsewhere (Bartram & Rees, 2000 chapter 9) and are summarized as follows:

- **Stage 1:** Initial sampling to determine whether significant spatial variation exists. Sampling at spatially separated sampling sites should be carried out during the initial assessment 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 of 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 inspection category and microbial water quality assessment (section 4.6). It is suggested that microbial water quality for all recreational waters is classified into four categories (A–D) using the 95th percentile of the intestinal enterococci distribution as shown in Table 4.7.
- **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 a repeat of the sanitary inspection. If the subsequent classification (section 4.6, Table 4.12) is 'very good' or 'very poor', less frequent monitoring can be justified (Table 4.13).

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, should 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, but should be used further to characterize the area and impacts of adverse events.

It is important that sufficient samples are collected to enable an appropriate estimation of the index organism densities to which recreational water users are exposed. Previous recommendations based on 20 or fewer samples are considered to be inappropriate given the usual variation in faecal index organisms as the precision of the estimate of the 95th percentile is low. Increasing sample numbers, for instance towards 100 samples, would increase precision.

The number of results available can be increased significantly—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 from 100 samples from a 5-year period and a rolling 5-year data set be used for microbial water quality assessment purposes. In many situations, a much shorter period will be required, where, for example, more extensive sampling is undertaken. In some circumstances, fewer samples may be required—for instance, where the water quality is very poor, however, it is suggested that 60 samples from a 3-year period should be the minimum considered.

Data sets that contain numerous values below the limit of detection can be difficult to manage. Where the use of such data is unavoidable, the Hazen method (Box 4.4) is a robust method for calculating the 95th percentile. It should be the preferred method as it gives very close estimates of the actual 95th percentile whether or not there are results that fall below the limit of detection (Hunter, 2002). In subsequent analyses, however, appropriate dilutions should be employed to ensure that non-detects are rare or completely avoided.

Various index bacteria, including *E. coli*, thermotolerant coliforms and intestinal enterococci, are used for the monitoring of recreational waters (see section 4.4.6). Several methods are available for estimating bacterial numbers at recreational water areas (outlined in Bartram & Rees, 2000). Where a change is made between index organisms (e.g., from thermotolerant coliforms to intestinal enterococci, or a change in the microbiological method employed), a limited number of data may be available in the initial years of implementation. In order to overcome this, correction factors appropriate to local conditions may be applied to historical records to enable their use. Such conversion factors would normally be driven by comparative studies of the results of local analyses. Another strategy that has been employed is to collect both old and new index organism data during a transition period. Although costs are increased this does provide a 'break-in' period.

4.6 Classification of recreational water environments

Classification of recreational water is achieved by combining the sanitary inspection category and the microbial water quality assessment using a matrix such as that shown in Table 4.12. The overall approach is summarized in Figure 4.4 (see section 4.5).

The classification emphasizes 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. 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 which may therefore represent a significantly lower risk to human health. As a result, the use of faecal bacteria alone as an index of risk to human health may significantly overestimate risks where the index organisms derive from sources other than human excreta. Nevertheless, there are human health risks associated with pollution of recreational waters from animal excreta, and some pathogens, such as *Cryptosporidium parvum*, *Campylobacter* spp. and *E. coli* O157:H7 can be transmitted through this route. Thus, local knowledge of possible sources and environmental pathways of animal pathogens to humans should form part of the sanitary inspection.

The assessment framework (Figure 4.4) enables local management to respond to sporadic or limited areas of pollution and thereby upgrade a recreational water's classification provided appropriate and effective management action is taken to control exposure (section 4.6.4). This form of classification (as opposed to a pass/fail

TABLE 4.12. EXAMPLE OF A CLASSIFICATION MATRIX FOR FAECAL POLLUTION OF RECREATIONAL WATER ENVIRONMENTS[3,4]

		Microbial Water Quality Assessment Category (95^{th} percentile intestinal enterococci/100 ml)				
		A ≤40	**B** 41–200	**C** 201–500	**D** >500	**Exceptional circumstances**
Sanitary Inspection Category (susceptibility to faecal influence)	**Very low**	Very good	Very good	Follow up[1]	Follow up[1]	
	Low	Very good	Good	Fair	Follow up[1]	
	Moderate	Good[2]	Good	Fair	Poor	Action
	High	Good[2]	Fair[2]	Poor	Very poor	
	Very high	Follow up[2]	Fair[2]	Poor	Very poor	
	Exceptional circumstances	Action				

Notes:

1 implies non-sewage sources of faecal indicators (e.g., livestock), and this should be verified (section 4.6.2).

2 indicates possible discontinuous/sporadic contamination (often driven by events such as rainfall). This is most commonly associated with Combined Sewer Overflow (CSO) presence. These results should be investigated further and initial follow-up should include verification of sanitary inspection category and ensuring samples recorded include "event" periods. Confirm analytical results. Review possible analytical errors (see section 4.6.2).

3 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 (section 4.6.5).

4 Exceptional circumstances (see section 4.6.5) relate to known periods of higher risk, such as during an outbreak with a pathogen that may be waterborne, sewer rupture in the recreational water catchment, etc. Under such circumstances, the classification matrix may not fairly represent risk/safety.

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 and is thereby supportive of informed personal choice. It assists in identifying the principal management and monitoring actions likely to be appropriate.

4.6.1 *Initial classification*

The outcome of the sanitary inspection and the microbial water quality assessment, based on Table 4.12 and Figure 4.4, is a five-level classification for recreational water environments—very good, good, fair, poor and very poor. In addition, there is a follow-up category or requirement where there is potential discrepancy between the results of the microbial water quality assessment and the sanitary inspection. 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. Contributions from riverine discharges and bather densities need to be scaled, based on local knowledge of hydrological conditions.

A case study is provided in Box 4.6 to illustrate the approach.

BOX 4.6 CASE STUDY (PART 1)

The following is an example of how to apply the framework guideline approach to a seawater used for body contact recreation. Historical microbiological 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 microbial water quality assessment.

1 SANITARY INSPECTION CATEGORY

(following criteria described in 4.5.1)

a) Sewage discharges (if present)—based on Table 4.9

		If present:		
Outfalls	**Present? Y / N**	**Type of sewage treatment**	**Type of outfall**	**Category**
Sewage outfalls	Y	primary	effective	low
Combined sewer overflows	N			—
Stormwater	Y		direct	very high

b) Riverine discharges (if present)—based on Table 4.10
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			—

Continued

c) Bather shedding (based on Table 4.11)

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

d) Physical characteristics of the beach
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.

e) Overall category of sanitary inspection
Very high

2 MICROBIAL WATER QUALITY ASSESSMENT

a) Describe the current monitoring programme for assessing microbial water quality.

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

b) Summarize data file(s) covering at least 5 years of monitoring (or 100 samples) for faecal index organisms—100 raw numbers are needed in order to calculate 95th percentiles. Preferably these should be the most recent data available.

n = 100
95th percentile = 276 intestinal enterococci/100 ml
Microbial Water Quality Assessment Category = C

3 COMBINED SANITARY AND MICROBIAL WATER QUALITY ASSESSMENT AND OVERALL CLASSIFICATION

This beach is rated as "poor":
Sanitary Inspection Category—Very low
Microbial Assessment Category—C

		Microbial Water Quality Assessment Category (intestinal enterococci/100 ml)				
		A ≤40	**B** 41–200	**C** 201–500	**D** >500	**Exceptional circumstances**
Sanitary Inspection Category (susceptibility to faecal influence)	Very low	Very good	Very good	Follow up[1]	Follow up[1]	Action
	Low	Very good	Good	Fair	Follow up[1]	
	Moderate	Good[2]	Good	Fair	Poor	
	High	Good[2]	Fair[2]	Poor	Very poor	
	Very high	Follow up[2]	Fair[2]	$\boxed{\text{Poor}}$	Very poor	
	Exceptional circumstances	Action				

Notes: See Table 4.12

4.6.2 *Follow-up of initial classification*

Where the sanitary inspection and water quality data inspection result in a potentially incongruent categorization in Table 4.12, 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) and additional analysis of water quality, with specific consideration given to the sampling protocol and analytical methodology.

Examples of situations that may lead to potentially incongruent assessments include the following:

- analytical errors;
- where the importance of non-point sources is not appreciated in the initial survey;
- where the sampling points are not representative of sewage influence;
- where CSOs are present on the beach but it is not appreciated that they do not discharge during the bathing season;
- where the assessment is based on insufficient or unrepresentative data; 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 connections to surface water of foul drains from domestic and other properties.

Where sanitary inspection indicates low risk but microbial water quality assessment 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 appropriate biomarkers, surveys of mammal and bird numbers etc.) may be appropriate. Confirmation that contamination is primarily from non-human sources may allow reclassification (see 4.6.4) to a more favourable grading, although care is needed here as risk will depend on the type of non-human pollution as it may still be a source of a number of important pathogens (section 4.6.5). Similarly, where microbial water quality assessment indicates a very low risk that is not supported by the sanitary inspection, consideration should be given to the sampling design, the analytical methodology used and the possibility that the sanitary inspection may have been incomplete.

4.6.3 *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 information required in Figure 4.4 for moving to the classification (or reclassification) step is incomplete. Three scenarios may be envisaged:

- where there are no data of any kind available as to the microbial water quality of the water body or its susceptibility to faecal influence (such as new developments);

- where the data available are incomplete, in respect of either the microbial water quality assessment 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 classification are insufficient.

In these circumstances, it may be necessary to issue a provisional classification (see Box 4.7). When such a step is taken, it should be made clear that the advice is provisional and subject to change. A provisional classification should be time-limited, and there should be a commitment to obtaining the necessary data to follow the steps described in Figure 4.4 to provide a definite classification as soon as possible.

BOX 4.7 EXAMPLE ACTIONS FOR PROVISIONAL CLASSIFICATION

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 microbial water quality assessment will be required; if the sampling and analytical capacities are insufficient, the most intensively used recreational water area should be selected for initial study.

At the first opportunity and in any event during the “bathing” season, take a minimum of 8–12 samples across the selected transect, ideally at about 50-m intervals (depending upon the length of the beach), 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. While laboratory results are awaited, the sanitary inspection should be completed as far as possible and arrangements made to obtain maps, plans, information on the sewer system 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 microbial water quality 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 recreational 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. This should be conducted as soon as possible, providing it is during the “bathing” season.

On the basis of the sanitary inspection and microbial water quality assessment data available after the second round of sampling, an early assessment should be made, and, if judged necessary, a time-limited provisional classification of the recreational water environment should be made and acted upon. 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 4.4 and Table 4.12 as soon as possible.

INCOMPLETE DATA

Where the data available are insufficient, in respect of either the microbial water quality 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 recreational water area or a set of analytical results with a strong trend to very high or very low values may enable a provisional classification to be made. If it is not possible to make a provisional classification, 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.

In the absence of past intestinal enterococci data it may be necessary to make use of historical records relating to another index organism, such as thermotolerant coliforms. The issue of conversion factors that may be applied for that purpose is dealt with in section 4.5.2.

If 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 4.4 may need to be embarked upon. In the event that it is necessary for beach classification to be urgently undertaken (in the absence of sufficient data), the procedure outlined above for a recreational water environment for which there are no data may be adapted accordingly.

INAPPROPRIATE EXISTING CLASSIFICATION

Where there is reason to believe that the existing classification no longer accords with changed circumstances, sufficient data need to be collected before completing the reclassification or, as in the above, it will be necessary 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 microbial water quality assessment data, steps should be taken, as set out (in section 4.6.2), to understand this. Should both the sanitary inspection data and the microbial water quality data point to a similar change in beach classification, a provisional conclusion should be drawn, but steps should be taken to obtain sufficient data for proper beach classification.

4.6.4 Reclassification, including advisories and upgrading

As water contamination may be triggered by specific and predictable conditions (e.g., rainfall), 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 upgraded to a more favourable level. A reclassification should, however, initially be provisional and time-limited. It may be confirmed if the efficacy of management interventions (e.g., advisories) is subsequently verified during the following bathing season, if the reclassification is not confirmed it will automatically revert to the original classification. This is illustrated, in Box 4.8, by a continuation of the case study introduced in Box 4.6.

BOX 4.8 CASE STUDY (PART 2)

Initial classification (see Box 4.6), on the basis of a sanitary inspection category of 'very high' and a microbial water quality assessment of 'C', was:

'Poor'.

The initial classification, however, appeared to be driven principally by the presence of occasional stormwater overflows. Subsequent investigation found that the stormwater overflow events were predictable and signage was introduced to warn bathers not to swim during rainfall and for up to 2 days following heavy rainfall. The beach was 'posted' whenever heavy rainfall had occurred.

Exclusion of the stormwater overflow changes the sanitary inspection category from 'very high' to 'low', which results in a provisional upgrading of:

'Fair (but unsuitable for 2 days after heavy rain)'.

Monitoring of the recreational water over a bathing season revealed that bathers complied with the notices not to bathe. Water quality sampling showed that after 2 days the microbial quality returned to normal levels. Reanalysis of microbial water quality data using the water quality to which users were exposed found a 95th percentile of 185, resulting in a final classification of:

'Good (but unsuitable for 2 days after heavy rain)'

The local authority intends to remove the source of stormwater overflow in the expectation that on completion the advisory can be removed and the beach classified as:

'Good'.

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. The real-time prediction of faecal index organism concentrations at recreational compliance points has been achieved using two principal approaches. The first uses background conditions to calibrate a statistical model, typically based on the relationships of multiple predictor variables, such as:

- preceding rainfall;
- wind direction;
- tides and currents;
- visible/modelled plume location;
- solar irradiance (and turbidity of water); and
- physicochemical parameters of water quality.

The alternative approach is the construction of a nearshore hydrodynamic model linked to a water quality model predicting concentrations of faecal index organisms (Falconer et al., 1998). Both approaches offer potential for real-time prediction of faecal pollution changes for protection of public health through timely management interventions. As such, some of these parameters could be considered for analysis at control points (see Table 4.6). Control points are those points that can be monitored to provide information to management so that management actions can have an impact on risk (section 4.3.3).

4.6.5 Exceptional circumstances

While no general guidance concerning risks during exceptional circumstances is provided here (for instance as guideline values), there is a need to make provisions to enable their identification and management (see Chapter 13 and Table 13.3). Examples could include sewer breaks, extreme floods or rainfall events with a return period of more than five years. Public health authorities should be engaged in the definition of water quality standards or appropriate action triggers 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 intersectoral action and will also often include local government, facility operators, user groups and so on. Public health authorities may be required to interpret the relevance of specific pathogens or outbreak events, 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 is associated with 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. Although there are no documented cases of transmission through swimming, 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/100 ml (approximately equivalent to 1000 intestinal enterococci/100 ml).

- 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.
- 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 in sewage and sewage-polluted recreational 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.

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 microbial water quality 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.

4.6.6 Monitoring and auditing

Monitoring and auditing include visual inspection of potential sources of contamination in a catchment, water sampling and verification of control points. Examples of control points include rainfall measurement in the catchment, municipal sewage discharge points, treatment works operation, combined sewer overflows and illegal connections to combined sewers.

Following initial classification, all recreational water environments 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 inspection category has occurred over several years, the sanitary inspection category was "Very low" or "Low" and the microbial water quality assessment is stable and based on at least 100 samples, microbiological sampling can be reduced to a minimum of five samples per year to ensure that no major changes go unidentified. 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"), a greater annual microbiological sampling programme is recommended (Table 4.13).

TABLE 4.13. RECOMMENDED MONITORING SCHEDULE

Risk category identified by sanitary inspection	Microbial water quality assessment	Sanitary inspection
Very low	Minimum of 5 samples per year	Annual
Low	Minimum of 5 samples per year	Annual
Moderate	Annual low-level sampling 4 samples *x* 5 occasions during swimming season Annual verification of management effectiveness Additional sampling if abnormal results obtained	Annual
High	Annual low-level sampling 4 samples *x* 5 occasions during swimming season Annual verification of management effectiveness Additional sampling if abnormal results obtained	Annual
Very high	Minimum of 5 samples per year	Annual

4.7 Management action

There are two main elements to consider in respect of management actions, classification of recreational water locations and short-term information that reflects changes in conditions. Good-quality public information in near-real time about the recreational water environment, through, for example, public health advisories, is particularly important to enable the public to make informed choices about if and where to use recreational water areas. Long-term management, on the other hand, might also be aimed at encouraging pollution abatement and prevention.

4.7.1 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 assess the impact of those advisories in discouraging water contact. This approach has the benefit of protecting public health and, in many circumstances, provides potential both to improve the classification of a location through low-cost measures and to enable safe use of areas for long periods that might otherwise be considered inappropriate for recreational use (see section 4.6.4).

Some locations will consistently have very poor water quality due to the proximity of sewage discharges; others will have intermittently poor water quality due to pollution that may be rare or impossible to predict. Still other sites will have episodic, but possibly predictable, deterioration in water quality, such as that driven by weather 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 outbreaks or endemic occurrence of potentially serious illness that may be spread by recreational water exposure (see section 4.6.5 and Table 13.3). In cases where locations are known to have consistently very poor microbial water quality, an appropriate management action may be to permanently discourage its use as a recreational water area by, for example, fencing, signposting, moving the location of car parks, bus stops and toilets, and so on (Bartram & Rees, 2000 chapter 9).

4.7.2 Pollution prevention

Recreational waters are often polluted by sewage and industrial discharges, combined sewer overflows, diffuse source pollution from agricultural areas and urban runoff. This section describes abatement and remediation measures available for water quality improvement.

1. *Direct point source pollution abatement*

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

For nearshore discharges of large urban communities, where effluent may come into contact with recreational water users, tertiary treatment with disinfection will provide the greatest health benefits and a sanitary inspection category of 'very low' (see Table 4.9), although public health risks will vary depending on the operation and reliability of the plant and the effectiveness of disinfection.

2. *Intermittent pollution abatement*

Runoff via drainage ditches, combined sewer overflows, etc. is predominantly "event-driven" pollution that may affect recreational water areas for relatively short periods after rainfall. Combined sewer and stormwater overflows, which are built into most sewerage systems where the effluent "combines" with rainfall, may present the greater health risk, because water users may be exposed to diluted untreated sewage. Where the sewer does not receive surface water after rainfall, the "uncombined" raw sewage overflows present a direct health risk, contact with which should be avoided.

The best option is to have separate collection systems for sewage and rain/stormwater. Although treatment is an option for combined sewer overflows often the treatment plant cannot cope with the quantity of the sewage, or the effectiveness of the treatment is lowered due to a change in the "quality" of the sewage.

Other pollution abatement alternatives for CSOs include:

- retention tanks that discharge during non-recreational water use periods. These are costly and may be impractical for large urban areas, although examples do exist (e.g., Barcelona);
- transport to locations distant from recreational areas via piped collection systems or effective outfalls; and

- disinfection (ozone, chlorine, peracetic acid or UV), which may be not be effective against all hazards.

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. An alternative adopted is the development and application of management programmes that minimize recreational use during event-driven pollution episodes.

Reuse of wastewater for agricultural, groundwater injection/infiltration or other purposes may eliminate health risks for recreational water areas. However, during event periods, such as heavy rainfall, recycled materials may be carried into waterways.

3. *Catchment pollution abatement*
Upstream diffuse pollution, point source discharges, pathogen accumulation and remobilization from stream sediments and riverine discharges to coastal recreational areas may be significant pollution sources that present a challenge to pollution abatement (Kay et al., 1999). Major sources of pollution should be identified and a catchment-wide pollution abatement programme developed. Multi-agency and interdisciplinary cooperation among health and environmental control agencies, local authorities, users, polluters, etc. assist in effective programme development (integrated management approaches are outlined in 1.7.2). The role of the agricultural sector in generation and remediation of pollution loadings is often crucial.

4.7.3 Enforcement of regulatory compliance

Regulatory compliance enforcement has limitations as the principal tool for the protection and improvement of microbial quality of recreational waters, although the power of closure or threat of closure may be a powerful driver for improvement. The two principal limitations concern responsibility for cause of failure and the nature of intervention.

Where a recreational water use location fails a regulatory standard, it may be difficult to define responsibility for failure. In many locations, a number of sources will contribute to overall pollution, 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. They may themselves be recipients of multiple pollution loads. Approaches to regulatory compliance enforcement that depend upon identifying and requiring change of a single discharge/pollution source "responsible" for failure may therefore be problematic.

It may be appropriate to base regulatory compliance on the obligation to act. Thus, there could be a requirement to immediately consult the public health authority and to inform the public as appropriate on detection of conditions potentially hazardous to health and uncharacteristic of the location. There could also be a general requirement to strive to ensure the safest achievable bathing conditions, with measures to be taken in order to improvement classification, including pollution control.

4.8 References

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