Development of a Decision-Support Tool to Support the Implementation of Fecal Coliform BMAPs in the Hillsborough River Watershed

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Acronym List

AWWA	American Water Works Association
BMAP	Basin Managemet Action Plan
CAFO	Concentrated Animal Feed Operation
CDC	Center for Disease Control
CEU	Council of the European Union
CSS	Contaminant Source Survey
CWA	Clean Water Act
EP	European Parliament
EPA	Environmental Protection Agency
EPCHC	Environmental Protection Commision of Hillsborough County
EU	European Union
FDACS	Florida Department of Agriculture and Consumer Services
FDEP	Florida Department of Environmental Protection
FDOT	Florida Department of Transportation
GI	Gastrointestinal
GIS	Geographic Information System
IWR	Impaired Water Rule
MWQA	Microbial Water Quality Assessment
NPDES	National Pollution Discharge Elimination System
NRC	National Research Council
SSO	Sanitary Sewer Overflow
SWFWMD	Southwest Florida Water Management District
TBEP	Tampa Bay Estuary Program
TMDL	Total Maximum Daily Load
WHO	World Health Organization

1. Purpose

The purpose of this project is to develop a decision-support tool — conceptually similar to the "decision matrix" that is currently used by the Tampa Bay Estuary Program (TBEP 2006a) to assess water quality in Tampa Bay — to guide management actions that are carried out to implement the Basin Management Action Plans (BMAPs) that are currently being finalized to address fecal coliform impairments and their associated total maximum daily loads (TMDLs) in the Hillsborough River watershed.

The decision-support framework used in the project is based on technical approaches and resource management strategies recommended by the National Research Council (NRC 2000, 2004), World Health Organization (WHO 2000, 2003), European Union (EP/CEU 2006) and U.S. EPA (EPA 1986, 2007). In addition to fecal coliform levels it incorporates other relevant information, such as the presence and relative magnitudes of human fecal contamination and other potential sources of human pathogens within BMAP management areas that are currently classified as impaired due to elevated fecal coliform levels. Because the impaired portions of the Hillsborough River watershed include or discharge to water bodies which are used for public recreation and/or as sources of potable water supply, the project focuses on these two designated uses and potential exposure routes for the purpose of managing water quality conditions to help reduce human health risks posed by waterborne pathogens.

If the approach proves to be workable and cost-effective in the Hillsborough River watershed, it may be applied (with appropriate modifications and extensions to include other beneficial uses) to other impaired waters throughout the state. In addition to its primary use in the fecal coliform TMDL/BMAP process, it is also intended to provide a technical framework that could potentially help water quality managers address other indicators of waterborne health risks in watersheds and water bodies that are used for recreational purposes and as shellfish harvesting areas and sources of potable water supplies.

2. Approach

Development of the decision-support tool followed a collaborative, consensus-based process. It was done in conjunction with the Hillsborough River BMAP Working Group and Stakeholder Group, which included representatives from the Florida Department of Agriculture and Consumer Services (FDACS), Florida Department of Environmental Protection (FDEP), Florida Department of Transportation (FDOT), Southwest Florida Water Management District (SWFWMD), Tampa Bay Estuary Program (TBEP), Tampa Bay Water (TBW), Hillsborough County Health Department (Florida Department of Health), Hillsborough County City/County Planning Commission, Hillsborough County Planning and Growth Management Department, Hillsborough County Public Works Department, the Environmental Protection Commission of Hillsborough County (EPCHC), the University of South Florida, the cities of Tampa, Temple Terrace and Plant City, the U.S. Environmental Protection Agency, environmental organizations, area landowners, and representatives from the private sector.

3. Background

Pursuant to the federal Clean Water Act (CWA), the State of Florida has adopted water quality criteria for fecal coliform bacteria (summarized in Appendix A) in order to reduce human health risks in cases where waterborne pathogens could potentially be present in water bodies that are used for recreation, shellfish harvesting, or potable water supply. In most cases, however, the fecal coliform bacteria that are detected in surface water samples are not human pathogens. Instead, they are indicator organisms that are used as a warning signal that water may have been contaminated by fecal material (from humans or other warm-blooded vertebrates), and may therefore contain disease-causing organisms that pose a risk to public health. The actual pathogens causing reported disease outbreaks associated with recreational waters in the U.S. during the years 1985 through 1998, and the frequency and magnitude of those outbreaks, are listed in Table 1.

Table 1. Reported disease outbreaks associated with recreational waters in the United States, 1985-1998. (Source: WHO 2003)

Etiological Agent	Number of Cases	Number of Outbreaks
Shigella spp.	1,780	20
Escherichia coli 0157:H7	234	9
Leptospira sp.	389	3
Giardia lamblia	65	4
Cryptospiridium parvum	429	3
Norwalk-like viruses	89	3
Adenovirus 3	595	1
Acute gastrointestinal infections (no causative agent identified)	1,984	21

Unfortunately, a considerable amount of research has shown that fecal coliform bacteria are not particularly accurate indicators of the presence of human pathogens in surface waters. For example, Wade et al. (2003) carried out a meta-analysis of the usefulness of fecal coliforms as predictors of the risk of gastrointestinal (GI) illness associated with recreational water use. Based on the results of 27 previously-published epidemiological studies, Wade et al. (2003) concluded:

> "This review… supports the recommended move away from the use of fecal coliform… as an indicator because there was no evidence that risk of GI illness increased at levels above the previously proposed [*i.e., pre-1986 EPA*] guideline value. In fresh water, *E. coli* was superior to enterococci at predicting illness, and the *E. coli* guideline level was supported, because exposure below presented no significant risk, whereas exposures above were associated with an elevated and statistically significant increased risk of GI illness."

> "We found that enteroviruses, which have been suggested as specific indicators of human contamination… were strongly associated with GI illness. They may, however, be impractical for use as water quality indicators because they are not easily cultivated in environmental samples..."

> "Based on the epidemiologic studies conducted to date, it is evident that no single indicator can predict illness consistently in all environments at all times, perhaps because of the wide array of pathogens that have been associated with GI illness in recreational water environments as well as natural variability in pathogen–indicator associations. For example, both bacterial and viral indicators of water quality may correlate poorly with the occurrence of protozoan parasites such as *Cryptosporidium parvum*, a leading cause of freshwater outbreaks of GI illness…"

Other issues affect the use of fecal coliforms and other indicator bacteria in tropical and subtropical climates. Rose et al. (2001), in a recent study conducted in the Tampa Bay area, provided the following summary of this issue:

> "Risks to swimmers using polluted beaches have been a major issue associated with the setting of ambient water quality standards and discharge limits to recreational sites. Public health concerns in recreational waters in the tropics and subtropics differ from those of cooler waters. Prevention of disease depends on the use of appropriate fecal indicators. However, the finding that the most widely

used fecal contamination indicator, fecal coliforms and more specifically *E. coli*, grow naturally on vegetation in warm climates clearly brings into question whether these or other indicators developed for temperate climates are applicable in Florida and other southeastern areas… In recent years, total and fecal coliform bacterial indicators have not been able to consistently indicate the persistence of pathogens, especially viruses in surface waters."

Similar points have been made, at the national and international levels, in reports published by the National Research Council (NRC 2000, 2004), the World Health Organization (WHO 2000, 2003), the European Union (EP/CEU 2006) and the U.S. Environmental Protection Agency (EPA 2007), which have pointed out a number of limitations affecting the accuracy of fecal coliforms and other existing indicators in estimating human health risks (e.g., Appendix B). Fujioka and colleagues (1985, 1997, 1999, 2001) have reported a number of instances in tropical waters in which none of the commonly used bacterial indicators have proven adequate. Given these limitations, it is evident that fecal coliform concentrations alone should not be relied upon by FDEP or its local partners when developing BMAPs to address microbial water quality and its potential impacts on human health, or when evaluating the effectiveness of the management actions that are taken to implement those BMAPs. Additional information regarding the dominant source(s) of the fecal coliform bacteria that are detected in surface water bodies (e.g., whether the source(s) are humans, pets, livestock, birds, other wildlife, soil, aquatic sediments, or vegetation), and the risks those sources pose to human health, will need to be collected and assessed to ensure that appropriate management actions are identified, prioritized, and carried out through the BMAP process.

The shortcomings of the existing bacteriological indicators are well known within the public health and water quality management communities, and intensive work is being carried out by a number of organizations to develop a cost-effective suite of indicators that provide more accurate estimates of human health risks than reliance on fecal coliforms alone. Currently, however, no single indicator or analytical method has been found that possesses the major attributes — such as greater accuracy in estimating risk, reasonable cost, and feasibility for day-to-day use by laboratory personnel in local monitoring programs — that would be needed to justify abandoning the existing bacteriological indicators and developing new water quality criteria based on alternatives (e.g., Fujioka 1997, Wade et al. 2003, EPA 2007). Until reliable, cost-effective alternatives or supplementary measurements are found and adopted, local management programs will need to continue using the existing indicators and other available information as effectively as possible in their efforts to reduce human health risks associated with waterborne pathogens.

In the case of recreational waters (i.e., Class III waters in Florida's classification system), a conceptual framework for prioritizing and managing health risks from waterborne pathogens and other factors has been developed by the WHO (2000, 2003) and applied in a number of countries around the world. Some elements of that conceptual framework are shown in Figures 1 and 2. Because monitoring of indicator organisms such as fecal coliform bacteria provides only limited information regarding health risks, the WHO framework also relies on information provided by annual "sanitary inspections", which are defined as "a search for, and evaluation of, existing and potential microbiological hazards that could affect the safe use of a particular stretch of recreational water or bathing beach" (WHO 2000). Under the WHO approach—which is commonly referred to as the "Annapolis protocol" due to its initial development at an international conference held in Annapolis, Maryland in the late 1990s—the information provided by sanitary inspections and water quality monitoring data are combined to provide a graded, risk-based assessment of a given water body, as summarized in Figure 2.

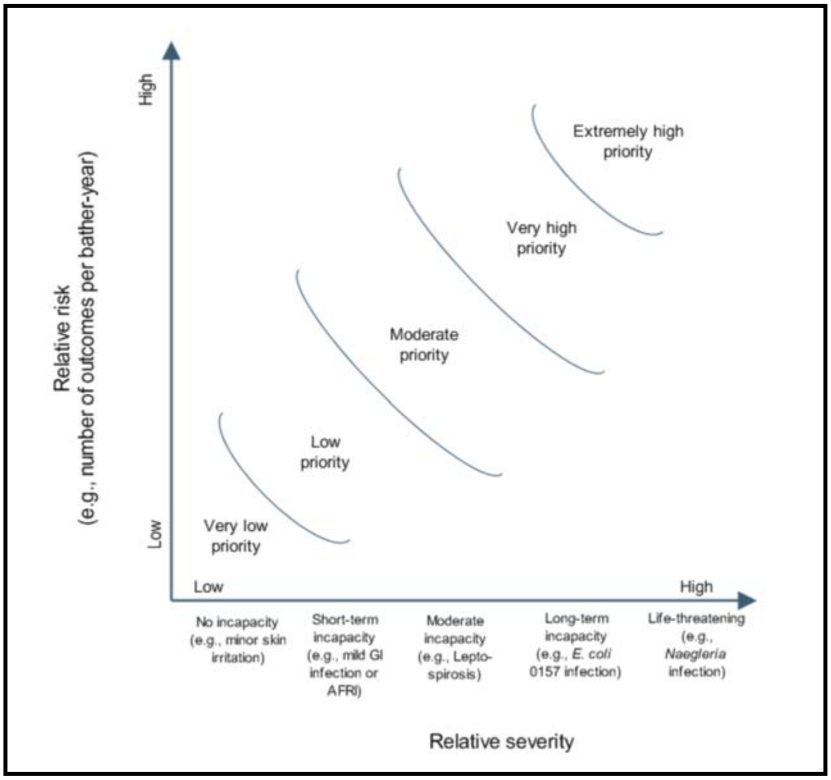

Figure 1. Schematic approach for prioritizing and managing health-related and other risks associated with recreational uses of surface water bodies.
(Source: WHO 2003)

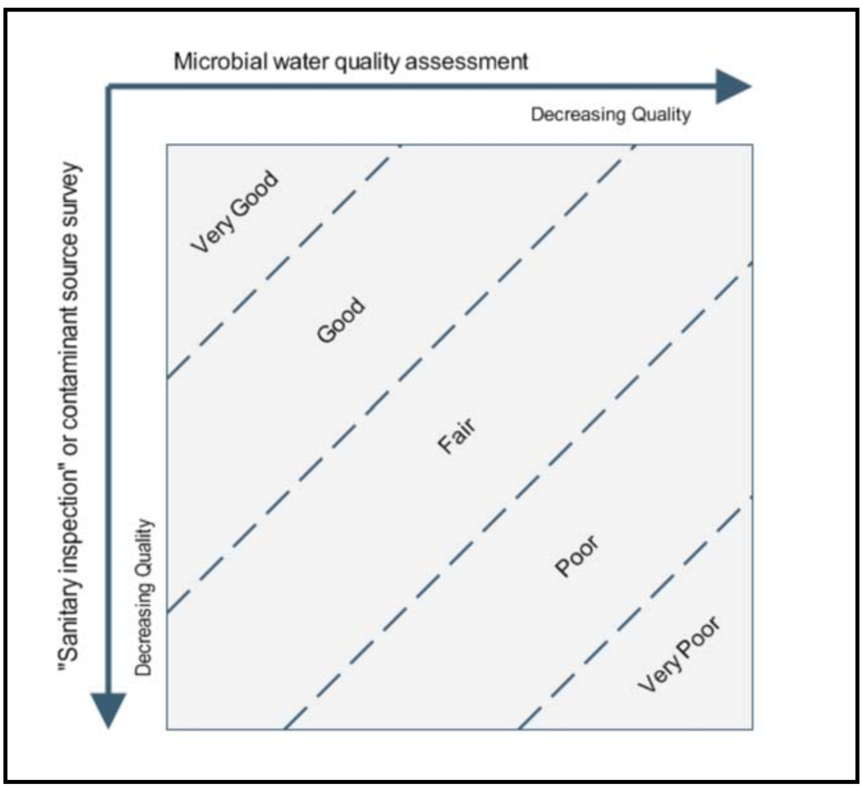

Figure 2. World Health Organization (WHO) water body classification matrix based on water quality monitoring data and sanitary inspections.
(Sources: WHO 2000, 2003)

4. Recent Management and Policy Recommendations

4.1. For Recreational Waters

In March, 2007, the U.S. EPA convened a national panel of experts to address known issues, identify information gaps, and review potential approaches that could be used by EPA to develop alternative microbial water quality criteria for water bodies that are used for recreational purposes. The panel selected three potential approaches for doing so, based on methodological frameworks currently recommended by the EPA (1983, 1984, 2004, 2006), WHO (2003), and European Union (EU) (2006). The three approaches were selected because they are based on peer-reviewed epidemiological research, are currently in use at the national level in one or more countries, and meet the following benchmarks (EPA 2007):

- They are health-based and linked to health effects demonstrated in epidemiology studies;
- They are compatible with the requirements of the federal Clean Water Act; including water quality assessment for public notification at beaches in a timely manner, assessment for impaired waters listings, TMDL development and implementation, and development of National Pollution Discharge Elimination System (NPDES) permits;
- They are scientifically defensible for application in a wide variety of geographic locations and climatic conditions, including fresh and marine waters, and temperate, subtropical, and tropical waters;
- They are sufficiently robust and flexible so that they can be configured to protect the public health of those exposed to recreational water impacted by sewage effluent, concentrated animal feed operation (CAFO) contaminated runoff, non-point sources (e.g., non-CAFO agricultural sources, urban runoff) and waters not impacted by anthropogenic sources;
- They are sufficiently robust and flexible to provide regulators the ability to protect susceptible (sensitive) subpopulations such as children and immunocompromised individuals;
- They are linked with laboratory analytical methods that are reliable, robust, and provide reproducible results; and
- They have the potential to equally protect primary contact recreation in freshwaters, marine waters, temperate, subtropical, and tropical waters, and to provide equal protection for those exposed via primary contact recreation to waters potentially impacted by effluent, urban runoff, and/or non-point source runoff.

The panel provided the following summary of the three approaches (EPA 2007):

4.1.1. World Health Organization (2003) Approach

As summarized by EPA (2007), "the WHO has been concerned with health issues associated with recreational water environments for many years and has published several influential reports that represent a well accepted view among international experts (Prüss, 1998). The WHO approach provides a basis for standard setting in light of local and regional circumstances, such as the nature and seriousness of local endemic illness, exposure patterns, and competing health risks that are not associated with recreational water exposure."

"The WHO approach is based on the perspective that recreational water quality and protection of public health are best described by a combination of sanitary inspection and microbial water quality assessments (the WHO [2003] Guidelines use enterococci as the fecal indicator of choice; see Table [2]). This approach considers possible sources of pollution in a recreational water ("sanitary inspection category" in Table [2]), as well as observed levels of fecal pollution ("microbial water quality assessment category" in Table [2]), and combines them into a five-level classification scheme for recreational water environments. To date, the classification system has been used primarily to "grade" recreational waters and to provide an assessment for regulatory compliance purposes. This approach however, also could be adapted for other CWA §304(a) applications such as NPDES permitting and TMDL development."

"The microbial water quality assessment criteria are based on a banded system, where the band divisions are equivalent to a risk of acquiring gastrointestinal (GI) illness for (A) <1 case in 100 exposures, (B) <1 case in 20 exposures, (C) <1 case in 10 exposures, and (D) >1 case in 10 exposures. The 95th percentile value was selected as an appropriate descriptor of the microbial probability density function because it is easily understood to be the probability of encountering polluted water and focuses on water quality that is likely to cause illness (i.e., greater probability of illness associated with increasing density of human sources of fecal pollution). The WHO levels of risk for the bands described above were selected based on a series of science policy decisions in consultations with numerous international experts and were intended to be reasonable for both the developed and developing world…"

Table 2. World Health Organization (WHO) classification matrix summarizing the "Annapolis protocol", which integrates microbial water quality and sanitary inspection categories. (Source: WHO 2003)

<table>
<tr><td colspan="2" rowspan="2"></td><td colspan="5">Microbial water quality assessment category (95th percentile intestinal enterococci/100 mL)</td></tr>
<tr><td>A
≤40</td><td>B
41-200</td><td>C
201-500</td><td>D
>500</td><td>Exceptional circumstances[3]</td></tr>
<tr><td rowspan="5">Sanitary Inspection Category (susceptibility to fecal influence)</td><td>Very low</td><td>Very good</td><td>Very good</td><td>Follow up[1]</td><td>Follow up[1]</td><td rowspan="6">Action</td></tr>
<tr><td>Low</td><td>Very good</td><td>Good</td><td>Fair</td><td>Follow up[1]</td></tr>
<tr><td>Moderate</td><td>Good[2]</td><td>Good</td><td>Fair</td><td>Poor</td></tr>
<tr><td>High</td><td>Good[2]</td><td>Fair[2]</td><td>Poor</td><td>Very Poor</td></tr>
<tr><td>Very High</td><td>Follow up[2]</td><td>Fair[2]</td><td>Poor</td><td>Very Poor</td></tr>
<tr><td colspan="2">Exceptional circumstances[3]</td><td colspan="4">Action</td></tr>
</table>

Notes:

1 Implies non-sewage sources of fecal indicators, which should be verified

2 Indicates possible episodic contamination (e.g., rainfall- and runoff-driven), which should be investigated further, including sampling during and following storm events

3 **“Exceptional circumstances” relate to known periods of higher risk, such as sanitary sewer overflows (SSOs) in the vicinity of recreational water bodies. Under such circumstances the matrix may not represent risk/safety accurately**

“The sanitary inspection category is intended to classify the risk of illness caused by fecal pollution in a recreational water body, although human fecal pollution will tend to drive the overall sanitary inspection category derived for an area. WHO experts believe that the three most important sources of human fecal contamination of recreational water environments for public health purposes are typically sewage, riverine discharges, and direct contamination from bathers. Sanitary inspections are required to address those sources as well as others, and inspections should take on a tiered approach, dependent on the level of perceived risk and its uncertainty. For example, if human and domestic fecal pollution is considered low based on land uses, but fecal indicator counts are relatively high, further exploration of the source(s) and their relative risks would be recommended. This higher level of examination (tier) may utilize more expensive methods and approaches and further cycles (tiers) of investigation as necessary. Based on the results of the sanitary inspections, recreational waters are ranked (from very low to very high) with respect to evidence for the degree of influence of fecal material.”

4.1.2. European Union (2006) Approach

As summarized by EPA (2007), “the EU Bathing Water Directive (7/EU/EEC; dated February 15, 2006) is currently being translated by Member States for implementation (EP/CEU, 2006). The Directive establishes separate numerical microbiological criteria for fresh (inland) and marine (coastal and transitional) bathing waters for EU Member States (Table [3]). The numerical values are based on epidemiological studies reported by Kay et al. (1994) and Wiedenmann et al. (2006) — the former was used by WHO in formulating their Guidelines (Kay et al., 2004; WHO, 2003).”

Table 3. Numerical microbiological water quality assessment classification for fresh (inland) and marine (coastal and transitional) bathing waters for European Union member states. (Sources: EP/CEU 2006; EPA 2007).

Inland (Fresh) Waters			
Indicator	**Excellent**	**Good**	**Sufficient**
Intestinal enterococci (CFU/100 mL)	200*	400*	360**
E. coli (CFU/100 mL)	500*	1,000*	900**
Coastal and Transitional (Marine) Waters			
Indicator	**Excellent**	**Good**	**Sufficient**
Intestinal enterococci (CFU/100 mL)	100*	200*	200**
E. coli (CFU/100 mL)	250*	500*	500**

Notes: * = based on 95th percentile evaluation; ** = based on 90th percentile evaluation to reduce the risk of statistical anomalies when using a small data set, which also allows lower limit values for enterococci and *E. coli* densities in inland waters to be classified as sufficient versus good microbiological water quality.

4.1.3. Environmental Protection Agency (1986) Approach

EPA (2007) provided the follow summary of the agency’s earlier (1986) approach: “In the late 1970s and early 1980s, EPA conducted public health studies evaluating several organisms as possible indicators, including total and fecal coliforms, *E. coli*, and enterococci. The studies showed that enterococci and *E. coli* are the best predictors of GI illness (gastroenteritis) in sewage effluent-impacted freshwaters, while enterococci were the best predictor in sewage-impacted marine waters. Gastroenteritis describes a variety of diseases that affect the GI tract and are rarely life-threatening; self-limiting symptoms include nausea, vomiting, stomachache, diarrhea, headache, and fever. Based on these studies, EPA published a criteria document, *Ambient Water Quality Criteria for Bacteria – 1986*, recommending the use of these bacterial indicators in ambient water quality criteria values for the protection of primary contact recreation (US EPA, 1986). Table [4] summarizes the *Water Quality Standards for Coastal and Great Lakes Recreation Waters Rule* (US EPA, 2004) that requires States and Tribes to adopt the 1986 AWQC for Bacteria."

"States and Tribes generally define their designated use of 'primary contact recreation' to encompass recreational activities that could be expected to result in the ingestion of, or immersion in, water, such as swimming, water skiing, surfing, or any other recreational activity where ingestion of, or immersion in, the water is likely. EPA derived standards that implied an acceptable excess illness probability of 0.8% in swimmers exposed in freshwater and 1.9% in swimmers exposed in marine waters. EPA's 1986 bacteria criteria document indicates the illness rates are 'only approximate' and that the Agency based the 1986 values that appear in Table [4] on these approximations."

"EPA's 1986 bacteria AWQC document provides geometric mean densities as well as four different single sample maximum values (representing values below which an increasing percentage of single values are expected to fall if the geometric mean of samples from the water body is equal to the geometric mean criteria). The 1986 bacteria AWQC document categorizes the single sample maximum values based levels of beach usage as follows: 'designated bathing beach' for the 75% (most conservative) confidence level, 'moderate use for bathing' for the 82% confidence level, 'light use for bathing' for the 90% confidence level, and 'infrequent use for bathing' for the 95% confidence level. The lowest confidence level corresponds to the highest level of protection."

Table 4. Summary of Environmental Protection Agency's (EPA) 1986 recommended water quality criteria for bacteria and 2004 Rule.
(Sources: EPA 2004, 2007)

Indicator	Swimming-associated gastroenteritis rate per 1,000 swimmers	Geometric Mean	Single sample maximum allowable density (CFU/100 mL)			
		Steady state geometric mean indicator density	Designated beach area	Moderate: full body contact recreation	Lightly used: full body contact recreation	Infrequently used: full body contact recreation
Freshwater						
Enterococci	8	33	61	78	107	151
E. coli	8	126	235	298	409	575
Marine Water						
Enterococci	19	35	104	158	276	501

"In the 1986 AWQC context, single sample maximum criteria are water quality assessment tools that provide a sense of when the water quality in a water body is not consistent with the AWQC. Insights based on single observations are very difficult because of the expected variability of fecal indicators. For instance, if the long-term geometric mean concentration of enterococci in the water at a marine beach is 35/100 mL and the log standard deviation is 0.4, then there is an 18% chance that the concentration of enterococci in a single sample would be over 158/100 mL. The higher the single sample maximum, the lower the probability that a single sample exceeding that value would occur as part of the normal random variability of samples (EPA, 2006)."

"Since publication of the 1986 criteria, many States have expressed concern that the current fecal indicator/illness rate relationships identified in the epidemiology studies leading up to the 1986 criteria are not appropriate or representative of all U.S. waters. For example, States have concern that the most appropriate indicator in tropical waters may be different than in temperate waters, and that appropriate levels of indicators may be different in waters where human fecal waste predominates animal waste. Other identified issues are as follows:

- Lack of clear, timely, and flexible guidance regarding use of the single sample maximum values and differing risk levels;
- No EPA-approved analytical methods for use in wastewater for the indicator bacteria;
- Lack of data to correctly assess the applicability of the 1986 bacteria criteria to flowing waters; and
- Lack of data to quantify the risk associated with contributions from nonhuman sources of fecal contamination as well as lack of flexibility to adjust the criteria for water bodies that do not receive human sources of fecal contamination."

4.2. For Potable-Supply Source Waters

Effective watershed management and source water protection are key components in the "multiple barrier" approach that is widely prescribed for the protection of drinking water supplies in the United States (AWWA 1997, EPA 1997). Because a number of the bacteriologically-impaired segments of the Hillsborough River watershed discharge to a municipal drinking water reservoir, management of pathogen loads that are contributed from those areas to the reservoir is a topic that should be addressed through the BMAP process.

A recent National Research Council (2000) review of this topic, although focused on watershed management issues that have arisen in the network of regional reservoirs that provides potable water supplies to New York City, includes a number of technical and policy-related points that appear relevant to the Hillsborough River watershed:

- The most commonly detected human pathogens associated with ingestion of water include several groups of enteric and aquatic bacteria, enteric viruses and enteric protozoa (summarized in Appendix C);
- Data collected nationally on reported waterborne disease outbreaks, from 1920 to 2000, suggest that a shift has occurred in the microorganisms responsible;
- Recognized disease outbreaks during the first half of the 20^{th} century were caused primarily by bacterial agents – most frequently by *Salmonella typhi* and *Shigella* sp.;
- Since the 1970s, recognized outbreaks have been caused predominantly by enteric protozoa such as *Giardia* or *Cryptosporidium* (when a pathogen has been identified), or by viral agents;
- This shift may be due, in part, to the greater resistance of protozoa to chlorination;

- Most waterborne pathogens are removed or inactivated by conventional drinking water treatment processes such as coagulation and sedimentation, filtration, and disinfection;

- Enteric viruses and bacteria are particularly susceptible to disinfection by free chlorine and ozone;

- Enteric protozoa are relatively resistant to chlorine disinfection, but are typically removed by the physical processes of coagulation and flocculation followed by filtration;

- Unfortunately, these treatment processes do not guarantee complete removal of pathogens from drinking water, as evidenced by recent outbreaks of waterborne disease associated with conventionally-treated municipal water supplies;

- These disease outbreaks have generally resulted from (a) source contamination and the breakdown of one or more of the treatment barriers or (b) contamination of the distribution system; and

- Although treatment has reduced the incidence of waterborne disease in the U.S. since the 19^{th} century, waterborne pathogens continue to pose a significant threat to public health in this country;

The NRC (2000) report also highlights the following points regarding the sources and relative human health risks associated with different types of potential pathogens in drinking water reservoirs:

- Many bacterial pathogens (e.g., *Salmonella, Shigella, Vibrio*) are known to be waterborne, and may emanate from both human and nonhuman sources. However, they are at least as sensitive to disinfection with chlorine as are coliforms. Therefore, for disinfected drinking water systems, bacterial pathogens in source water are considered less significant than some other (less sensitive) microbial pathogens as potential health threats;

- There are many human enteric viruses that may be present in surface water supplies, such as rotaviruses, coxsackieviruses, and echoviruses. Human wastes (including wastewater discharges) are the most frequently documented sources of human enteric viruses in surface waters. Viruses are more sensitive to disinfection than are cysts of *Giardia* and oocysts of *Cryptosporidium*. Thus, like bacterial pathogens, the presence of viruses in water supplies that are disinfected is of comparatively less concern than the presence of protozoan pathogens;

- Two protozoa – *Giardia lamblia* (now named *Giardia intestinalis*) and *Cryptosporidium parvum* – form stages known as cysts and oocysts, respectively, that are resistant to disinfection with chlorine;

- *G. lamblia* cysts occur in human wastewater and in animal wastes, and some animals (e.g., beavers and other aquatic animals) are believed to serve as reservoirs for strains that are pathogenic to humans. Conclusive evidence that animal-derived cysts have actually caused human disease outbreaks has not yet been found. However, it is clear that aquatic animals, domestic dogs and cats, and cattle are potential sources of measurable cysts in surface waters;

- Knowledge of the life history of *Cryptosporidium* in water is less developed than that of *Giardia*, and the taxonomy and human pathogenicity of potential subspecies/strains of *C. parvum* are topics of ongoing research;

- The literature indicates that domestic animals, livestock and wildlife may serve as reservoirs of *C. parvum*, with infected calves and lambs excreting large amounts of oocysts;

- Human *C. parvum* isolates have been found to infect calves, lambs, goats, pigs, dogs, cats, mice and chickens. Isolates from calves have been found to infect humans, and there is also epidemiological evidence of transmission from cats or pigs to humans.

Based on this information it appears that, with respect to source water protection issues, the fecal coliform decision support tool for the Hillsborough River watershed should place additional emphasis on the detection and management of potential *Giardia* and *Cryptosporidium* sources and loads to the Hillsborough River Reservoir and the Tampa Bypass Canal, the two surface water bodies in the watershed that are used as sources of potable water supplies. Because bacteria and viruses continue to cause waterborne disease outbreaks in the U.S. (CDC 2004), however, sources of these pathogens should not be neglected.

5. Responding to Data Gaps and Uncertainty: A Phased Monitoring Approach

The National Research Council (NRC 2004) recently conducted a review of microbial water quality monitoring methods that summarized the shortcomings of existing indicators and explored potential solutions. Given the increased understanding of microbiology that has developed at the molecular level in recent years, the report projects that technological advances are likely to occur within the next decade or so that will provide cost-effective methods for monitoring new indicators—perhaps including pathogens themselves—and will allow managers to "rapidly and correctly identify when water used for recreational or drinking water purposes is contaminated with microorganisms that are pathogenic to humans" (NRC 2004).

Until those improved monitoring methods are developed and can be performed regularly at reasonable cost, a phased monitoring approach was recommended to help reduce the regulatory and resource management uncertainties brought about by the shortcomings of the existing monitoring tools (NRC 2004). The approach consists of three monitoring levels (routine sampling, expanded sampling, and microbial assessment), which are summarized in Figure 3.

At the first level (Level A), routine or "screening level" monitoring using existing indicators is conducted by local, state or federal water quality monitoring programs. The purpose of this monitoring is to provide an early warning of potential health risk, or a change from background conditions that could lead to unacceptable risk levels. The most important indicator attributes for use at this monitoring level are speed, low cost, logistical feasibility, applicability, and sensitivity. Because of the known drawbacks of existing indicators, significant management decisions should usually not be made based on this level of information "unless the indicator concentrations are extreme or supported by ancillary information" (NRC 2004).

In situations where routine/screening level sampling indicates a potential water quality problem, the second level of monitoring (Level B) is implemented to test for the presence of a public health risk. If this level produces evidence that a substantial risk may be present, management responses could include actions such as beach closures or boil-water orders. Level B monitoring typically includes expanded sampling with the same indicators used in Level A, to confirm the initial results and provide information on possible gradients in indicator concentrations. Ancillary information from on-the-ground surveys — which are equivalent to “sanitary inspections” in the WHO (2003) terminology and to the “contaminant source surveys” (CSS) used in FDEP’s fecal BMAP implementation process — can also be used to check for sewer system leaks, malfunctioning septic systems, illicit discharges, livestock waste entering streams, or similar potential fecal sources in upstream areas. In some situations, such as continued high levels of fecal indicators in the absence of an identifiable source, more specific indicators (including pathogens themselves) could potentially be measured at this stage. Given the currently high cost of many of these methods, however, they would presumably not be used on a routine basis (NRC 2004).

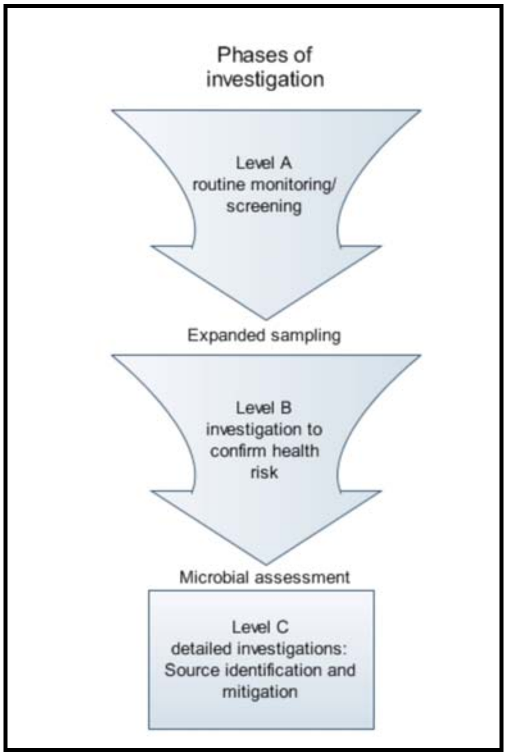

Figure 3. Phased monitoring approach recommended by the National Research Council (NRC). (Source: NRC 2004)

Because the studies carried out in Level B are intended to assess human health risk, the most important attributes for any additional indicators included at this stage are transport and survival characteristics that are similar to human pathogens, quantifiability, and effectiveness in measuring pathogen presence, viability or infectiousness (NRC 2004). Although more expensive than basic screening-level monitoring, a variety of new and emerging technologies such as quantitative polymerase chain reaction (qPCR), microarrays, and viral enumeration/detection by culture or PCR may be helpful at this level, particularly as methods are validated and refined.

The third recommended monitoring phase (Level C) adds focused studies whose objective is to “determine sources of microbial contamination so that health risk can be abated through a variety of engineering and policy solutions” (NRC 2004). In some cases expanded monitoring has been used to search for spatial gradients in indicator or pathogen concentrations. Methods based on molecular signatures have also proven helpful in some situations (NRC 2004). Important indicator attributes at this level of study include quantifiability and specificity for particular fecal sources.

The methods applied in Level A and B studies are in use today in many local environmental or health department laboratories. However, a number of the analytical techniques used in some Level B and most Level C studies have not yet been standardized and may only be available through research laboratories (NRC 2004). The responsibility for study costs may thus need to shift as the process moves forward, with parties potentially responsible for the operation of contamination sources becoming more involved in the funding process during the latter stages of Level B and much of Level C (NRC 2004).

6. A Potential Model for the Fecal Basin Management Action Plan Implementation Process: The Tampa Bay Estuary Program Water Quality "Decision Matrix"

As noted earlier, the TBEP (2006a) has developed a decision-support tool (or "decision matrix") that is applied annually to:

- track changes in water clarity and chlorophyll-*a* concentrations in Tampa Bay;
- compare annual mean water clarity and chlorophyll-*a* values to target levels that have been adopted to support the achievement of the program's seagrass restoration goals; and
- guide management responses to situations in which targets are not being achieved.

The TBEP decision matrix incorporates a number of elements that are desirable in a water quality management program, such as:

- the use of locally relevant water quality indicators and well-defined, quantitative management targets;
- the use of data from ongoing water quality monitoring programs, including annual evaluations of the data to detect exceedances of target values; and
- the adoption of a well-defined resource management process for responding to target exceedances when they occur.

It thus provides a conceptual model that could be used, with appropriate modifications, to develop a decision-support tool for addressing bacteriological impairments in the Hillsborough River watershed that would be consistent with the State's existing regulatory requirements, the "Annapolis protocol" recommended by the WHO (2000, 2003), and the phased monitoring approach recommended by the NRC (2004).

Poe et al. (2006) provide the following summary of the TBEP decision matrix approach:

> "Water quality targets have been adopted by the TBEP Management and Policy Committees for the four mainstem segments of Tampa Bay. The Tampa Bay Estuary Program has developed a tracking process to determine if water quality targets are being achieved (Janicki et al., 2000). The process to track the status of chlorophyll-*a* concentration and light attenuation involves two steps. The first step utilizes a decision framework to evaluate differences in mean annual ambient conditions from the established targets. The second step incorporates the results of the decision framework into a decision

matrix leading to possible outcomes dependent upon the magnitude and duration of the events in excess of the target (Janicki et al. 2000)."

"The tracking process is used not only to determine if there are differences between ambient conditions and targets, but also to determine the size of the differences and how long the conditions exist. The first step of the tracking process is presented in [Figure 4]. When mean ambient chlorophyll-*a* concentrations are less than the target, there is no cause for concern, as represented by Outcome 0 in [Figure 4]. When mean ambient chlorophyll-*a* concentrations are greater than target values, however, the size of the difference and the duration of the difference are considered. Small differences for short time periods result in Outcome 1, while large differences for short time periods and small differences for long time periods result in Outcome 2. In the most severe condition, when large differences exist for long periods, the framework results in Outcome 3."

"The second step of the tracking process involves combining the outputs from the decision frameworks for chlorophyll-*a* concentration and light attenuation in a decision matrix to provide direction for management responses when targets are exceeded. The decision framework shown in [Figure 4] for chlorophyll-*a* concentration is the same as that for light attenuation."

"The decision matrix incorporating the outcomes for chlorophyll-*a* concentration and light attenuation is shown in [Table 5]. When outcomes for both chlorophyll-*a* concentration and light attenuation are good, as represented by Outcome 0 for both, a condition exists in which targets are being met, and so no management response is required. This condition is signified by the green cell in [Table 5]. When conditions are intermediate, as signified by the yellow cells in [Table 5], differences from the targets exist for either or both chlorophyll-*a* concentration and light attenuation. These conditions may result in some type of management response. When conditions are problematic, such that the outcomes for the parameters fall within the red cells of [Table 5], stronger management responses may be warranted. The types of management actions resulting from the decision matrix are classified by color into three categories, shown following [Table 5]."

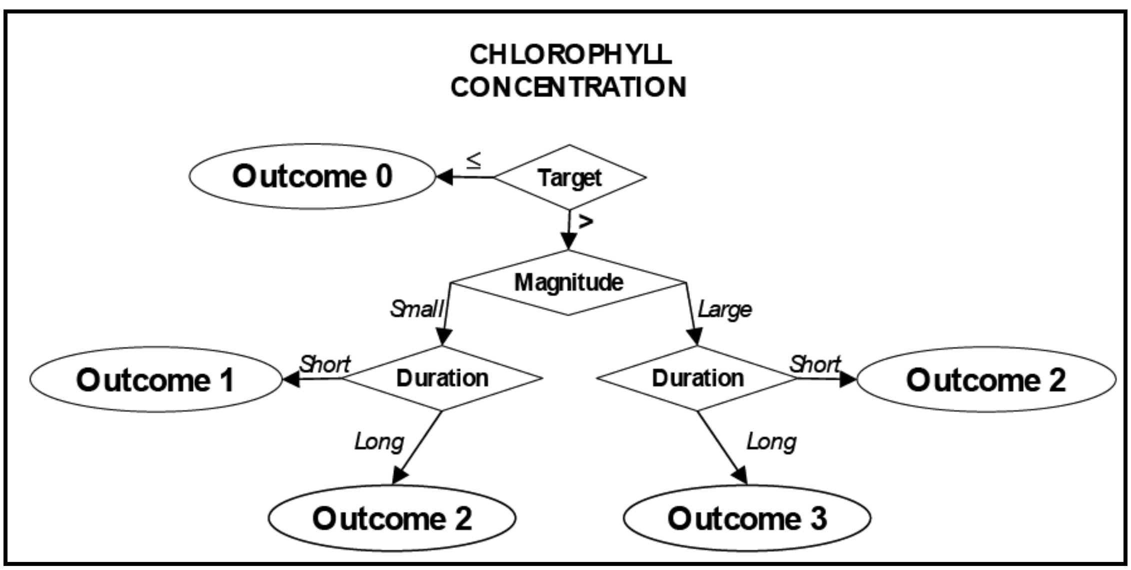

Figure 4. Decision framework used to identify water quality outcomes in Tampa Bay Estuary Program "decision matrix" for chlorophyll-*a*.
(Source: Poe et al. 2006)

Table 5. Decision matrix identifying appropriate categories of management actions in response to various outcomes of the monitoring and assessment of chlorophyll-*a* and light attenuation data.

CHLOROPHYLL	LIGHT ATTENUATION			
↓	Outcome 0	Outcome 1	Outcome 2	Outcome 3
Outcome 0	GREEN	YELLOW	YELLOW	YELLOW
Outcome 1	YELLOW	YELLOW	YELLOW	RED
Outcome 2	YELLOW	YELLOW	RED	RED
Outcome 3	YELLOW	RED	RED	RED

- **Green**: "Stay the course"; partners continue with planned projects to implement the Tampa Bay Comprehensive Conservation and Management Plan (CCMP; TBEP 2006a). Data summary and reporting occurs via the Baywide Environmental Monitoring Report (e.g., TBEP 2006b) and annual assessment and progress reports (e.g., Poe et al. 2006).

- **Yellow**: Technical advisory committee (TAC) and Management Board on caution alert; review monitoring data and loading estimates; attempt to identify causes of target exceedances; TAC report to Management Board on findings and recommended responses if needed.

- **Red**: TAC, Management and Policy Boards on alert; review and report by TAC to Management Board on recommended types of responses. Management and Policy Boards take appropriate actions to get the program back on track.

7. A Potential Framework for Addressing Fecal Coliform Impairments

An analogous decision-support framework and classification matrix can be developed to guide management actions addressing fecal coliform impairments. The Annapolis protocol recommended by the WHO (2003; summarized in Table 2), and the phased monitoring approach recommended by the NRC (2004; summarized in Figure 3), appear to offer the best means of doing so, given the well-known limitations of the existing bacterial indicators. The WHO (2003) and NRC (2004) approaches acknowledge the limitations of the existing indicators, and use a weight-of-evidence approach to help compensate for those limitations. They use two independent categories of information — bacterial indicator data to identify locations with potential fecal contamination, combined with site-specific surveys to identify and classify indicator sources on the basis of their potential human health risks — to prioritize and guide management actions.

Ways in which these approaches could be developed and applied to impaired WBIDs in the Hillsborough River watershed are described in detail in Sections 7.1 through 7.5 below. As a preliminary overview, the process involves the following steps, which are summarized in flowchart form in Figure 9:

- First, microbial water quality conditions within each WBID are categorized based on fecal coliform concentrations observed in the available monitoring data. (If existing monitoring programs do not provide sufficient data to characterize fecal coliform concentrations, additional monitoring should be conducted to ensure that adequate data will be available for future assessments);

- For characterization purposes, each monitoring station within a WBID is assigned to a microbial water quality assessment (MWQA) category. As shown in decision-tree form in Figure 8, the MWQA categories are symbolized as letter grades (A through E) reflecting how frequently the State's fecal coliform criterion of 400 CFU/100 mL is exceeded at a given site. Because sites with higher frequencies of criterion exceedances also tend to exhibit higher overall concentrations of fecal coliforms and enterococci (see Figures 5 through 7), MWQA categories A through E also represent progressively higher indicator organism concentrations and increasing levels of potential human health risk;

- Next, for sites at which more than 10% of the samples exceed the State's 400 CFU/100 mL criterion, contaminant source surveys (CSS) are carried out in order to identify the types of probable sources that could contribute to the elevated bacterial concentrations occurring at the site and to characterize their potential human health risks;

- Following the concepts outlined in the Annapolis protocol, each surveyed site is placed in a CSS category (ranging from "very low" to "very high" levels of potential risk) reflecting the types of probable bacterial sources found in the vicinity of the site and the estimated likelihood that those sources pose human health risks;

- Following the phased monitoring concept recommended by the NRC (2004), the intensity of CSS investigation that a site receives is based on its microbial water quality assessment (MWQA) classification. That is, sites that exhibit more frequent (and higher magnitude) exceedances of the State's 400 CFU/100 mL fecal coliform criterion (e.g., sites in MWQA categories C, D or E) are subject to more intensive CSS investigations than sites exhibiting less frequent (and lower magnitude) exceedances;

- Once the MWQA and CSS analyses are completed, each site receives a two-part classification (summarized in Table 7), based on the MWQA and CSS categories into which it has been placed;

- As with the Tampa Bay water quality decision matrix discussed earlier, these classification outcomes (which are color-coded in Table 7 to indicate the estimated potential for human health risk) would be reported in an annual progress report. This report would provide policy-makers and the public with a summary of the management actions that have been implemented in the impaired WBIDs and the changes in water quality that have occurred there.

7.1. Using Bacterial Indicator Data to Develop Microbial Water Quality Assessment Categories

As shown in Table 2, the WHO (2003) Annapolis protocol uses observed concentrations of "intestinal enterococci" to classify locations into MWQA categories. (For an explanation of the intestinal enterococcci indicator and a comparison to the enterococci and fecal coliform indicators that are used more commonly in the United States, see NRC (2004)). The particular MWQA categories developed by the WHO are not directly applicable to fresh water bodies in Florida, for two reasons:

- The epidemiological studies on which the WHO categories were based were carried out in northern temperate marine waters, primarily in the United Kingdom, and cannot be assumed *a priori* to reflect human health risks in subtropical freshwater or estuarine environments (NRC 2004); and

- The State of Florida does not have numerical criteria for the intestinal enterococci indicator, making the WHO (2003) MWQA categories difficult to integrate into Florida's existing water quality management system.

However, the conceptual approach underlying the WHO (2003) MWQA categories has been thoroughly reviewed and is technically sound (e.g., NRC 2004, EPA 2007). The approach can be applied to WBIDs in the Hillsborough River watershed (and potentially other Florida waters) by developing a comparable range of MWQA categories based on fecal coliforms, the bacterial indicators for which the State has adopted water quality criteria.

7.1.1. Florida's Fecal Coliform Criteria and the Impaired Waters Rule

The six WBIDs within the Hillsborough River watershed that are currently involved in the BMAP process are all Class III waters, whose designated uses include primary contact recreation and the provision of fish and wildlife habitat. Through its existing water quality standards (Chapter 62-302.250, F.A.C.), Florida has adopted a fecal coliform criterion for Class III waters of 400 CFU/100 mL (see Appendix A). Under the State's "impaired waters rule" (IWR; Chapter 62-303 F.A.C), water bodies are designated as impaired if more than 10% of their fecal coliform samples exceed this criterion.

Two general approaches have been evaluated in the technical literature as potential methods for defining exceedances of these types of numerical water quality criteria (e.g., Lin et al. 2000, Smith et al. 2001):

- A "raw scores" approach, in which any site at which more than 10% of the samples have fecal coliform counts greater than 400 CFU/100 mL would be designated as impaired; and

- A statistical approach, based on the binomial distribution, which asks whether the percentage of samples exceeding the 400 CFU/100 mL criterion is significantly greater than the 10% threshold.

Both approaches have positive and negative attributes, but from a statistical perspective the binomial approach has an overall advantage due to its ability to quantify and manage both "type I" and "type II" error rates.

In the TMDL/BMAP framework a type I error would occur if a water body or WBID was incorrectly designated as impaired for some water quality constituent, when it fact it met the applicable criterion and should have been designated as unimpaired. A type II error would occur if a water body or WBID was incorrectly classified as unimpaired, when it fact it failed to meet the criterion and should have been designated as impaired (Smith et al. 2001). Both types of error can potentially lead to substantial costs, by causing management resources to be allocated inefficiently and through the economic and other costs that are incurred if environmental resources and public health are not adequately protected.

The raw scores approach does not recognize the existence of type I or type II error, but in practice it tends to have a higher type I error rate and a lower type II error rate than the binomial approach at any given sample size (Smith et al. 2001). Regarding the minimum sample sizes needed for impairment designations, Smith et al. (2001) note that: "given the information routinely used in an assessment, the binomial method should replace the raw score approach when sample sizes are greater than 20. With sample sizes smaller than 20, neither the raw scores or the binomial method adequately control the error rates."

FDEP uses the binomial approach (see Chapter 62-303.420, F.A.C.) to determine whether WBIDs are impaired for fecal coliforms. This use of the binomial approach is explained in detail by Lin et al. (2000), who provide additional background information on the statistical computations involved. Using the approach, individual WBIDs are designated as impaired or unimpaired based on the total number of fecal coliform samples (*N*) collected during a defined impairment evaluation period and the number of those samples (*x*) that exceed the 400CFU/100 mL criterion. If the percentage of samples exceeding the criterion,

$$\hat{P} = 100 * x / N$$

is significantly greater than 10%, based on the binomial test applied at an alpha level of 0.10, the WBID is designated as impaired (Chapter 62-303.420, F.A.C.; Lin et al. 2000). By setting alpha (the "significance" level or expected type I error rate of the test) at 0.10, rather than at the lower 0.05 level that is traditionally used in laboratory research experiments, the expected type II error rate is reduced for a given sample size (Smith et al. 2001). The type II error rate can also be reduced, at any given alpha level, by increasing the sample size.

An application of this approach to fecal coliform data from a number of Hillsborough County water bodies is shown in Figure 5. The data used in this example were provided by the Environmental Protection Commission of Hillsborough County (EPCHC) which, since 2001, has performed monthly measurements of fecal coliforms from 39 inland (primarily fresh, but some occasionally brackish) sampling locations that are distributed throughout the county. These records, collected at a regular frequency using the same field sampling protocols and a single analytical laboratory, offer a relatively large data set that can be used to evaluate bacteriological water quality conditions and potential MWQA categories on a county-wide basis.

Figure 5 shows the relationship that exists between geometric mean fecal coliform concentrations and the proportion of samples exceeding the 400 CFU/100 mL criterion at each of these EPCHC stations during the years 2001 through 2007. Stations at which significantly more than 10% of the samples exceeded the 400 CFU/100 mL criterion (shown as closed circles in Figure 5) exhibited higher geometric mean fecal coliform levels than stations that did not significantly exceed the criterion (shown as open circles), and stations with the highest exceedance percentages also tended to have the highest geometric mean fecal coliform counts.

In addition to fecal coliform counts, the EPCHC monitoring program has also measured enterococci levels at these stations on a monthly basis since 2001. Relationships between geometric mean enterococci levels and the percentage of samples that exceeded the 400 CFU/100 mL fecal coliform criterion are also summarized, at the site level, in Figure 5.

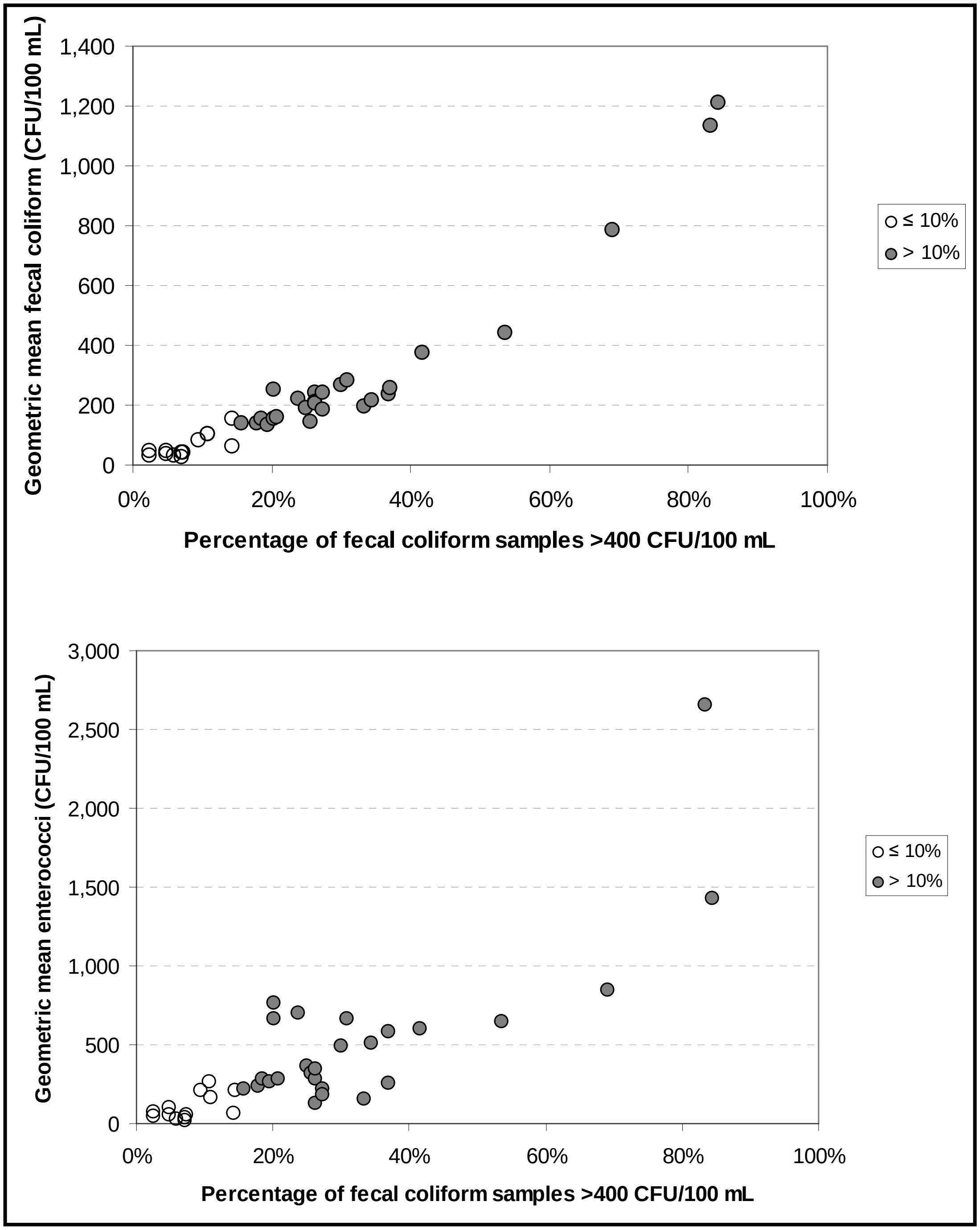

Figure 5. Relationships between geometric mean fecal coliform (upper panel) and geometric mean enterococci (lower panel) concentrations and the percentage of samples in which fecal coliform levels exceeded the 400 CFU/100 mL fecal coliform criterion (*x*-axis) at 39 stations monitored by Environmental Protection Commission of Hillsborough County (EPCHC) during the years 2001 through 2007. Stations at which the percentage of samples exceeding the criterion is significantly greater than 10%, based on the binomial test, are shown as filled circles. Each point represents one monitoring station. (Data source: EPCHC)

7.1.2. Using the 400 CFU/100 mL Fecal Coliform Criterion to Identify Microbial Water Quality Assessment Categories

Based on the relationships shown in Figure 5, the percentage of samples exceeding the 400 CFU/100 mL fecal coliform criterion appears to offer a practical method for assigning monitoring locations to MWQA categories, as called for under the Annapolis protocol. Because increasing fecal coliform and enterococci concentrations have been shown to be associated with increasing (although highly variable) human health risks in a number of epidemiological studies (e.g., Prüss 1998, WHO 2003, NRC 2004), the relationships shown in Figure 5 indicate that MWQA categories based on the percentage of samples exceeding the 400 CFU/100 mL fecal coliform criterion will also reflect site-to-site variations in potential health risks.

In order to keep the MWQA scoring process as simple as possible, the *x*-axis in Figure 5 can be divided at a small number of "break points," and each sampling station can then be assigned to a MWQA category based on the exceedance frequency group into which it falls. One obvious break point is at the 10% criterion exceedance level, which is used by FDEP to classify WBIDs as impaired or unimpaired under the impaired waters rule (Chapter 62-303.420, F.A.C.). Visually (see Figures 5 through 7), additional natural break-points can be selected at exceedance frequencies of 30%, 50% and 75%, producing the five MWQA categories summarized in Table 6.

Table 6. Proposed microbial water quality assessment (MWQA) categories, based on the percentage of samples exceeding the State's 400 CFU/100 mL fecal coliform criterion. "Break points" separating the MWQA categories are at exceedance frequencies of 10%, 30%, 50% and 75%.

MWQA Category	Break Point (percentage of samples exceeding the 400 CFU/100 mL fecal coliform criterion)	Range of exceedance frequencies (percentage of samples exceeding the 400 CFU/100 mL fecal coliform criterion) included in category
A	≤ 10%	0% to 10%
B	> 10%	>10% to 30%
C	> 30%	>30% to 50%
D	> 50%	>50% to 75%
E	> 75%	>75% to 100%

For consistency with the IWR methodology, the binomial test can be carried out (at an alpha level of 0.10) to identify the EPCHC sampling locations where exceedances of the 400 CFU/100 mL criterion are significantly greater than the 10%, 30%, 50% and 75% break point levels described above. An application of this approach is shown in Figure 6, which is identical to Figure 5 but identifies the individual EPCHC stations based on their MWQA category assignments. Stations were assigned to MWQA categories based on the 10%, 30%, 50% and 75% break points, using the binomial test to identify exceedances of these break points that are statistically significant at an alpha level of 0.10.

In addition to the station-by-station view shown in Figure 6, it is also helpful to pool the data (across stations) within each of the proposed MWQA categories, and consider variations in fecal coliform and enterococci concentrations that occur within and between the proposed categories. This view of the EPCHC monitoring data is shown, in box-and-whisker format, in Figure 7.

Due to the extreme variability of bacterial counts in ambient monitoring data, with values at a single station often ranging over several orders of magnitude during a multi-year sampling period (e.g., see

Figures D-1 and D-2 in Appendix D), it is often difficult to plot minimum and maximum values on a single graph in a readily viewable format. This was the case with the data shown in Figure 7. As a result the 10^{th} and 90^{th} percentile values were substituted in place of the minimum and maximum values in the box-and-whisker plots shown in the figure.

Based on the concentrations of indicator bacteria observed within and between the proposed MWQA categories (Figures 6 and 7), the categories appear to provide the level of among-site discrimination of microbial water quality conditions that is needed for use in the BMAP implementation process. The primary purpose of the MWQA categories is to provide managers a tool that can be used to prioritize monitoring locations for follow-up investigations, and determine the level of effort that should be expended in those investigations, in order to evaluate and address potential human health risks. From a broader (e.g., regional) water quality management perspective, it appears that the proposed MWQA categories could also be used as a screening-level tool when prioritizing watersheds and their WBIDs for BMAP development. Watersheds or WBIDs with numerous sites in MWQA categories E and D could be given the highest priorities for BMAP development and implementation. If sources posing potential public health risks are detected in these areas, they in turn could be identified as high priorities for remedial action to reduce or eliminate the apparent risks.

The proposed approach for assigning sites to MWQA categories that is described above is summarized, in decision-tree form, in Figure 8. In addition to the five MWQA categories (A through E) described above, a sixth group — denoted as “insufficient data”— has also been included in Figure 8 to address situations in which the available data are not sufficient to allow microbial water quality conditions at a site to be evaluated. As noted earlier, Smith et al. (2001) reported that neither the raw scores nor the binomial approach appear to provide adequate protection against type I or type II errors at sample sizes smaller than 20 – 25. To minimize this concern, we would suggest that the “insufficient data” category include sites with <30 samples over the time period of interest (e.g., the previous five years).

Because this is the pilot application of the proposed MWQA categories, which have not yet been tested for their performance over time or in other geographic locations, the proposed category break-points should be viewed as provisional and subject to possible revision as additional information becomes available through the BMAP implementation process.

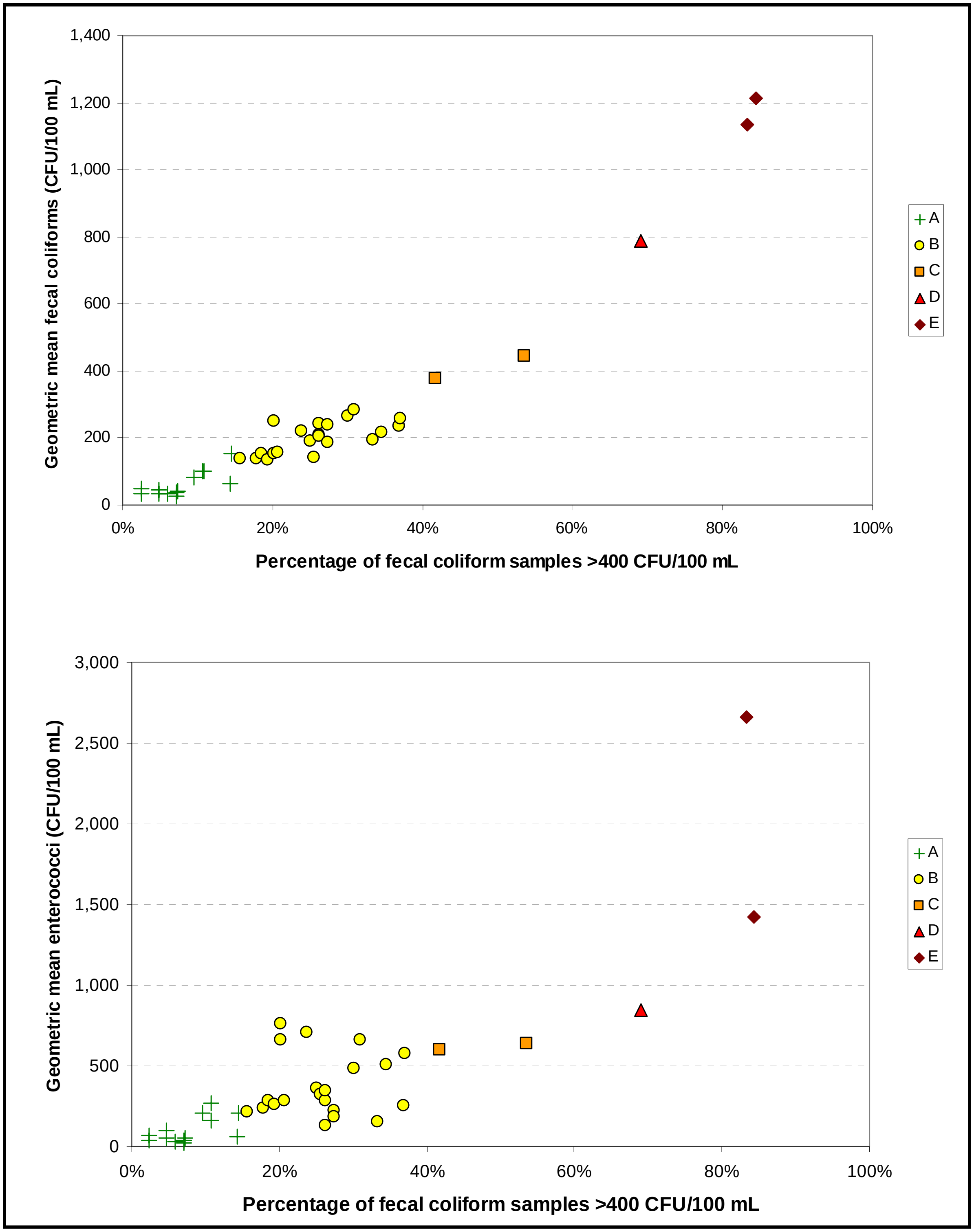

Figure 6. Geometric mean fecal coliform (upper panel) and geometric mean enterococci (lower panel) concentrations at the Environmental Protection Commission of Hillsborough County (EPCHC) monitoring stations shown in Figure 5. Microbial Water Quality Assessment (MWQA) categories A through E were defined using the exceedance frequency break points of 10%, 30%, 50% and 75% described in Table 6. For consistency with the Impaired Water Rule (IWR), the binomial test was used to test the significance of break point exceedances and assign monitoring stations to MWQA categories. (Data source: EPCHC)

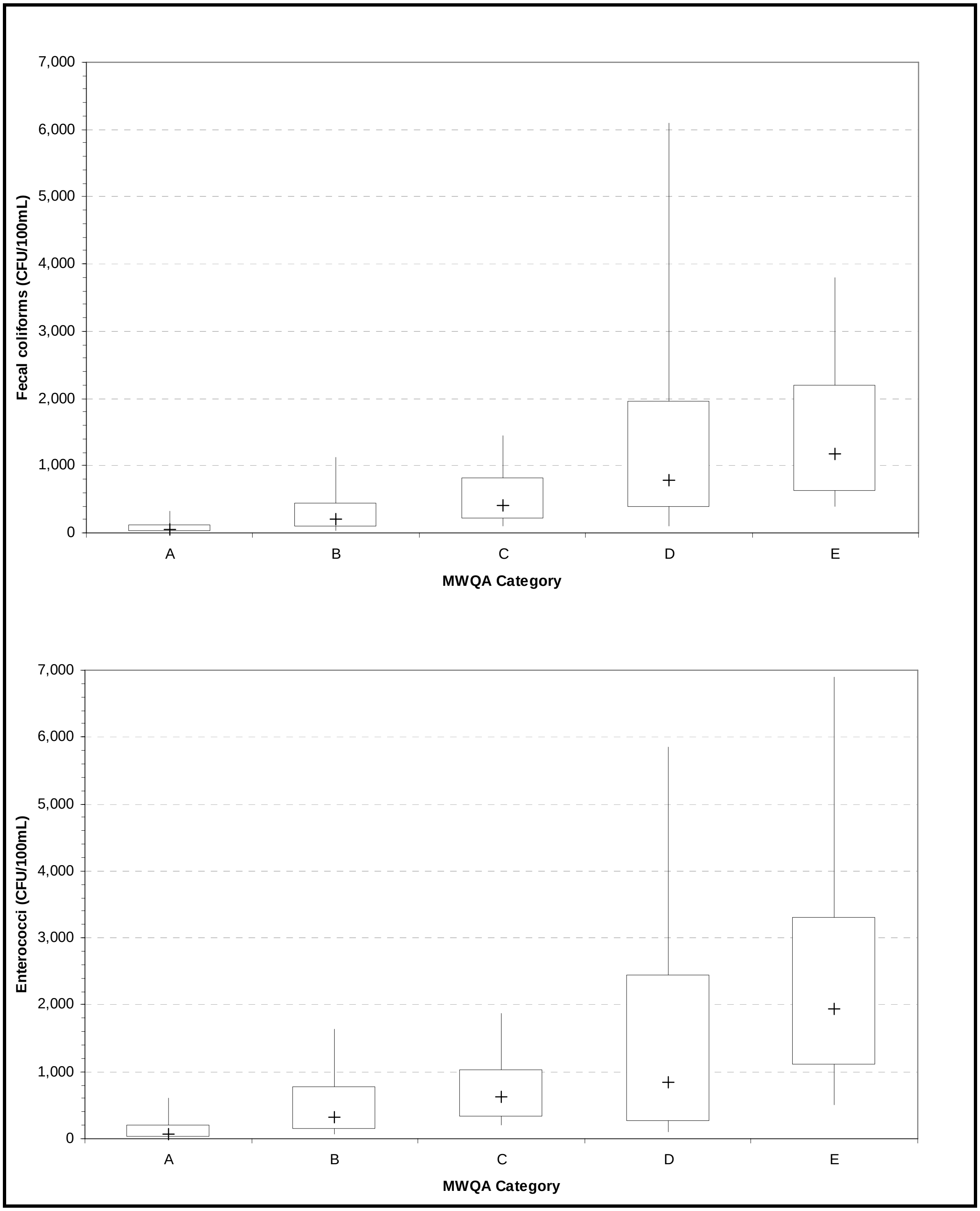

Figure 7. Box–and-whisker plots (showing geometric mean, interquartile range, and 10th and 90th percentile values) summarizing variability of fecal coliform and enterococci concentrations within and among the proposed microbial water quality assessment (MWQA) categories listed in Table 6, based on the Environmental Protection Commission of Hillsborough County (EPCHC) monitoring stations shown in Figures 5 and 6.
(Data source: EPCHC)

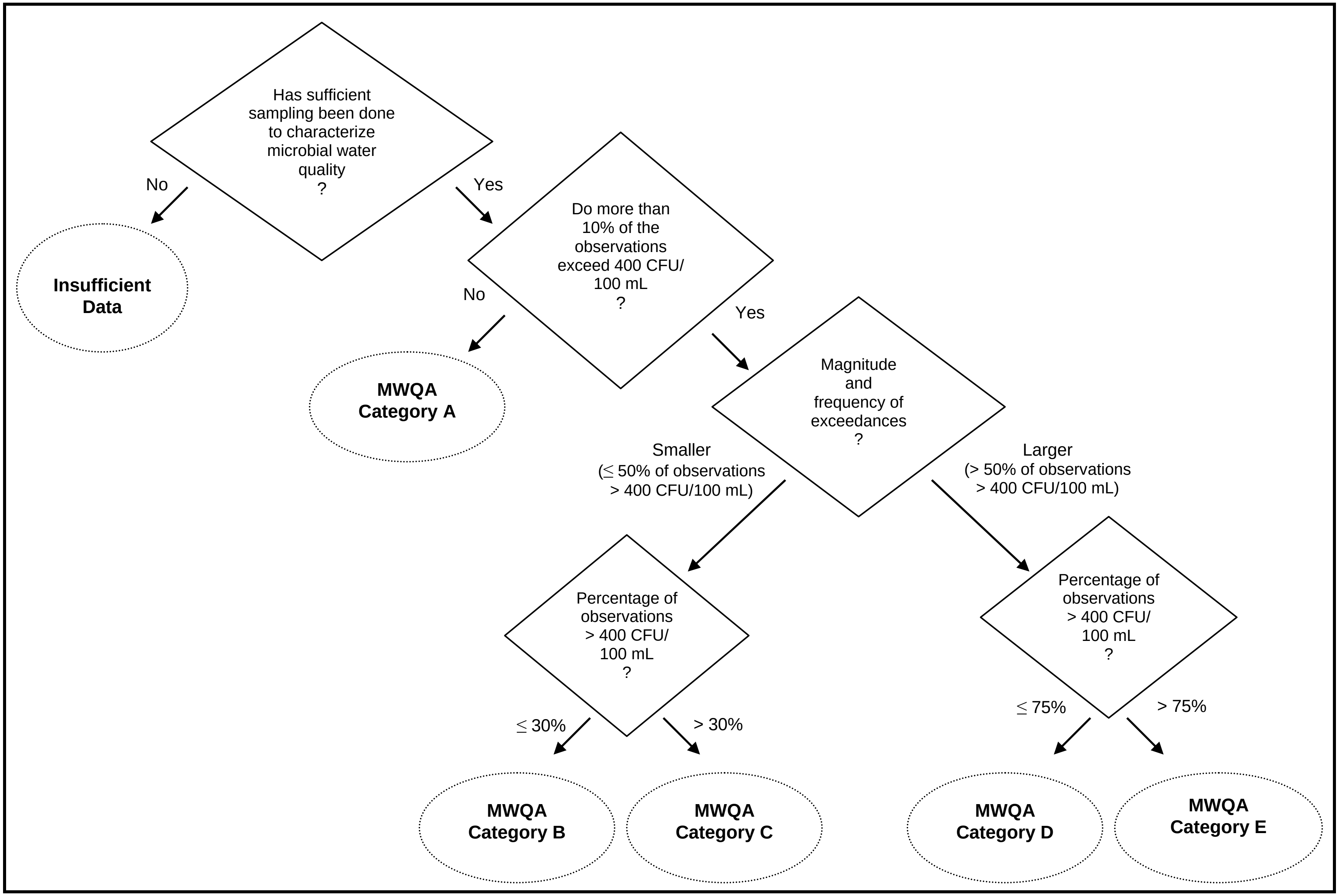

Figure 8. Decision tree for assigning monitoring locations to microbial water quality assessment (MWQA) categories based on observed fecal coliform concentrations. MWQA categories A through E are based on the percentage of samples at a given site that exceed the State's 400 CFU/100 mL fecal coliform criterion, using exceedance frequency break points of 10%, 30%, 50% and 75% as described in Section 7.1.2.

7.2. Incorporating Information from Contaminant Source Surveys and Other Sources

In addition to the bacterial indicator data considered in the previous section, another key element of the WHO (2003) Annapolis protocol and the NRC (2004) phased monitoring approach is the use of independent information — provided by CSS — as a cross-check for evaluating potential human health risks at monitoring locations where elevated indicator levels are observed. The additional information provided by CSS is used to identify the probable sources of the elevated indicator levels that occur at these sites, and determine if one or more of those sources fall into categories (e.g., inadequately treated sewage, livestock waste) that represent likely public health risks, or into other categories (e.g., wildlife, soils, sediments, vegetation) that appear to pose much lower risk.

Following the WHO (2003) approach, information from contaminant source surveys can be added by including a second axis in the decision-support framework, as shown in Table 7. Based on field experience from a number of areas in Florida, the project team recommends that the CSS assessment categories listed in the table be defined as follows, based on the likelihood of fecal contamination that would pose human health risks:

1. **Very Low**: No visual evidence of potential sources of human pathogens; natural environment; no or minimal anthropogenic land uses; wildlife present (any density)

2. **Low:** Low density agricultural and residential sources, including pets, livestock (without direct access to surface waters), or poultry operations; residences on septic systems

3. **Moderate**: Urban stormwater sources (including pet waste) present; well-functioning wastewater infrastructure (both sewer and septic); episodic/low volume sanitary sewer overflows (SSOs) reaching surface waters; moderate-density livestock with little direct access to surface waters; Class A residual and/or septage spreading areas may be present

4. **High**: Major stormwater outfalls present; history of failing wastewater infrastructure (central sewer or onsite systems); episodic or chronic/high volume SSOs reaching surface waters; concentrated livestock without direct access to surface waters; residual/septage spreading (Class B)

5. **Very High**: Current failing wastewater infrastructure; chronic/high volume SSOs reaching surface waters; concentrated livestock with direct access to surface waters; evidence of direct sewage inputs (e.g., confirmed illicit discharges)

For consistency with the WHO (2003) approach, an additional category ("exceptional circumstances") is also included in Table 7 to account for acute situations, such as sewer line breaks, that are known to be associated with increased health risk and would prompt immediate remedial action.

Within the classification matrix (Table 7), putative water quality conditions associated with chronic contamination sources are indicated by color, ranging from green (best water quality) to purple (poorest water quality). As a general rule, it is anticipated that the MWQA categories, based on fecal coliform measurements, and the CSS categories, based on assessments of local contaminant sources, will provide similar ratings of microbial water quality at a given site. However, because of the well-known variability of fecal coliform concentrations, their sensitivity to some environmental conditions (e.g., elevated salinity), and their frequently low correlation with human fecal contamination or other potential sources of enteric human pathogens, it is not unusual for monitoring locations to receive different water quality classifications based on their MWQA scores and initial CSS assessments (WHO 2003). In situations where the fecal coliform counts are elevated due to environmental sources such as wildlife, sediments, soils or vegetation, for example, the MWQA scores may lead to substantially poorer water quality ratings than the CSS assessments. In other situations, significant fecal contaminant sources may be present but not be detected through the fecal coliform data (e.g., because the discharge is episodic), and information from the CSS may provide a more accurate assessment of the human health risks present at the site. The cells colored gray in Table 7 represent these cases. As indicated in the notes to Tables 2 and 7, management responses need to have sufficient flexibility to address these situations and managers should be prepared to conduct additional investigations in cases where the two sources of information appear to be providing conflicting information.

Table 7. A potential classification matrix, based on the World Health Organization (WHO 2003) Annapolis protocol approach, using a combination of fecal coliform measurements (represented by the Microbial Water Quality Assessment [MWQA] group) and contaminant source survey (CSS) information to rank recreational sites based on the apparent likelihood of human health risk. (See text for explanation of contaminant source survey [CSS] assessment categories.) (Source: Modified from WHO 2003)

		MWQA group (based on binomial assessment of frequency of 400 CFU/100 mL fecal coliform exceedances)					**Exceptional Circumstances (e.g., sewer line break)** [c]
		A (≤ 10%)	B (>10% - 30%)	C (>30% - 50%)	D (>50% - 75%)	E (>75%)	
Contaminant source survey (CSS) assessment category (likelihood of fecal contamination posing human health risks)	1. Very Low	A1	B1	C1[a]	D1[a]	E1[a]	**Immediate Action**
	2. Low	A2 [b]	B2	C2	D2[a]	E2[a]	
	3. Moderate	A3[b]	B3	C3	D3	E3	
	4. High	A4[b]	B4[b]	C4	D4	E4	
	5. Very High	A5[b]	B5[b]	C5[b]	D5	E5	
	Exceptional Circumstances (e.g., sewer line break) [c]	**Immediate Action**					

Notes:

a) **These outcomes imply that the CSS may be providing an overly optimistic rating of water quality, or the fecal coliform sources in the area may be relatively low-risk or primarily environmental (e.g., wildlife, sediments, soils, vegetation), and the cause(s) of the discrepancy should be verified.**

b) **These outcomes imply that the fecal coliform indicator may be providing an overly optimistic MWQA rating, or the CSS may be providing an overly negative assessment, and the cause(s) of the discrepancy should be verified.**

c) **As explained by WHO (2003), exceptional circumstances involve acute situations known to be associated with higher public health risks, such as sewer line breaks and other SSOs that contaminate surface waters, which require immediate remedial action.**

7.3. Recommended Management Responses to Classification Matrix Outcomes

The sets of management actions that are recommended to identify and respond to potential human health risks associated with the MWQA categories shown in Figure 8 and the classification matrix outcomes listed in Table 7 are summarized below. For each outcome, it is assumed that any sources identified during the CSS process that appear to represent significant potential public health risks will be dealt with as expeditiously as possible by local and state authorities.

For consistency with the phased monitoring approach recommended by the NRC (2004), the following levels of CSS intensity are included in these recommended management responses:

- **Phase 1 CSS** – includes basic (screening-level) analyses using available water quality data to identify patterns and trends that may be present in bacterial indicator levels, and land use information to identify potential sources in the contributing watershed. These screening-level analyses are followed by an on-the-ground survey by knowledgeable technical personnel to verify the presence of the potential sources identified through the initial analyses, identify other sources that may have been missed by the initial analyses, and assess the potential human health risks posed by all sources found in the vicinity of the site.

- **Phase 2 CSS** – includes all elements of Phase 1, plus more intensive analysis of a broader range of available information. In addition to ambient monitoring data, information from regulatory agencies regarding permitted discharges of indicator bacteria and other constituents (e.g., discharge monitoring data collected by permitees, or data collected by the agencies for permit renewal or compliance/enforcement purposes) is collated and analyzed to provide more detailed information on permitted sources. Georgraphic Information System (GIS) and other mapping tools are used for a more detailed evaluation of factors such as land use intensity (e.g., levels of urban development or agricultural use), presence and condition of existing stormwater infrastructure, existing wastewater treatment methods (e.g., septic or central sewer), septic system failure and repair rates, reported SSOs and other leaks and spills from central sewer systems, and citizen complaints involving water quality problems, within the monitoring site's contributing watershed. If necessary, expanded spatial and temporal monitoring of indicator bacteria (and/or other water quality constituents) is also carried out to identify gradients in contamination and trace those gradients to individual sources (e.g., McDonald et al. 2006).

- **Phase 3 CSS** – includes all elements of Phases 1 and 2, plus the use of appropriate microbial source tracking (MST) tools, as necessary, to identify and characterize sources. A number of MST tools of varying cost and complexity, and their applications in identifying sources of fecal contamination, have been summarized by NRC (2004) and EPA (2005). A review of the need for validation of MST methods and strategies to accomplish this goal is presented in Stoeckel and Harwood (2007).

Before a classification matrix outcome (Table 7, Figure 8) can be calculated for a given site, sufficient monitoring data must be available to assign the site to an MWQA category. If sufficient data are not available (i.e., if the site falls into the "insufficient data" category of Figure 8), the following management actions are recommended:

- Implement a monitoring program that will provide a sufficient amount of data to characterize microbial water quality in the area of interest (e.g., collect a minimum of six bi-monthly samples per year over the next five-year period).

- Conduct a Phase 1 CSS to evaluate potential sources of human fecal contamination or enteric human pathogens in the site's contributing watershed.

- Base initial site assessment and remedial actions on CSS results, until sufficient fecal coliform data are available to assign the site to an MWQA category

- Document results in annual progress report to elected officials and the public.

Otherwise, if sufficient data are available to assign the site to an MWQA category and develop an initial classification matrix outcome (using Table 7), the following management actions are recommended:

Outcome A:

- Continue monitoring fecal coliforms.

- Continue standard permitting and compliance monitoring of potential sources; report and map all SSOs and septic system failures/repairs.

- Calculate MWQA category annually, and include result in annual progress report.

- A CSS does not appear necessary for sites in MWQA category A. However, if a CSS is conducted, and the resulting site classification is something other than A1 (i.e., A2 through A5), take follow-up action to identify and address any fecal contaminant sources that pose potential health risks.

- Document results in annual progress report.

Outcomes B1 through B5:

- Continue monitoring fecal coliforms at current, and possibly additional stations, and consider adding other indicators (enterococci and possibly *E. coli* in freshwater) if resources allow.

- Continue standard permitting and compliance monitoring of permitted sources; report and map all SSOs and septic system failures/repairs.

- Conduct a Phase 1 CSS and proceed through the classification matrix analysis (Table 7). Repeat Phase 1 CSS every 3 years.

- Take management actions to reduce fecal contamination from potential sources of human pathogens as those sources are detected and identified.

- Document results in annual progress report.

Outcomes C1 through C5:

- Continue monitoring fecal coliforms at current and additional stations, and consider adding enterococci and possibly *E. coli* in freshwater if resources allow.

- Continue standard permitting and compliance monitoring of permitted sources; report and map all SSOs and septic system failures/repairs.

- Conduct a Phase 2 CSS and proceed through the classification matrix analysis (Table 7).

- Take management actions to reduce fecal contamination from potential sources of human pathogens as those sources are detected and identified.

- Document results in annual progress report.

Outcomes D1 through D5:

- Provide public outreach notifying recreational users in areas where potential waterborne health threats appear to be present.

- Take management actions to reduce fecal contamination from potential sources of human pathogens as those sources are detected and identified.

- Perform heightened compliance monitoring of permitted sources, and take corrective action as necessary; continue to report and map all SSOs and septic system failures/repairs.

- Conduct Phase 2 and, if necessary, Phase 3 CSS to identify indicator sources. Proceed through the classification matrix analysis (Table 7). Implement corrective actions for any sources detected through these investigations.

- Report actions taken and short-term results in semi-annual (twice per year) Work Group meetings.

- Continue monitoring fecal coliforms, and consider adding enterococci and possibly *E. coli* if resources allow.

- Document results in progress report.

Outcomes E1 through E5:

- Take immediate action to provide public outreach notifying recreational users in areas where potential waterborne health threats appear to be present.

- Take immediate management actions to reduce fecal contamination from potential sources of human pathogens as those sources are detected and identified.

- Perform heightened compliance monitoring of permitted sources, and take corrective action as necessary; continue to report and map all SSOs and septic system failures/repairs.

- Conduct Phase 2 and, if necessary, Phase 3 CSS to identify indicator sources. Proceed through the classification matrix analysis (Table 7). Implement corrective actions for any sources detected through these investigations.

- Report actions taken and short-term results in semi-annual (twice per year) Work Group meetings.

- Continue monitoring fecal coliforms, and consider adding enterococci and possibly *E. coli* if resources allow.

- Document results in annual progress report.

These responses could be implemented by following the process summarized in Figure 9 for each monitoring location within an impaired WBID. The results (e.g., site-specific classification matrix outcomes, and the management actions taken in response to those outcomes) could be reported annually in a "citizen-friendly" format that allows status and trends in microbial water quality conditions to be tracked by non-technical readers, including interested citizens, elected officials and other policy-makers. If desired, a more technical report could also be produced on an annual or semi-annual basis for resource managers and other technical audiences.

Because potential sources of human pathogens would be addressed through the proposed process whenever they are identified, it is anticipated that microbial water quality conditions at sites impacted by such sources should improve over time as the BMAP process is implemented. Sites impacted by non-anthropogenic background sources that do not appear to pose significant public health risks (e.g., wildlife, sediments, soils, vegetation) would be identified as such through the CSS assessments, and may not require remedial management actions (e.g., Chapter 62-303.460(2) F.A.C.).

Given the large costs that could be incurred through extensive MST analyses, it is important that they be used judiciously. Recognizing this point, the management responses described above only make an explicit call for MST as part of a Phase 3 CSS. This would occur in cases where monitoring sites fall into MWQA groups D or E, indicating the potential for relatively high public health risks. During the BMAP implementation process, however, it is possible that cases may also arise in a Phase 1 or Phase 2 CSS in which MST analyses are needed to confirm the identity of suspected contaminant sources. These situations should be addressed on a case-by-case basis, with MST techniques applied when necessary to provide confirmation. A relatively low-cost approach that could be used in these cases, involving targeted sampling and three inexpensive MST methods, is described by McDonald et al. (2006).

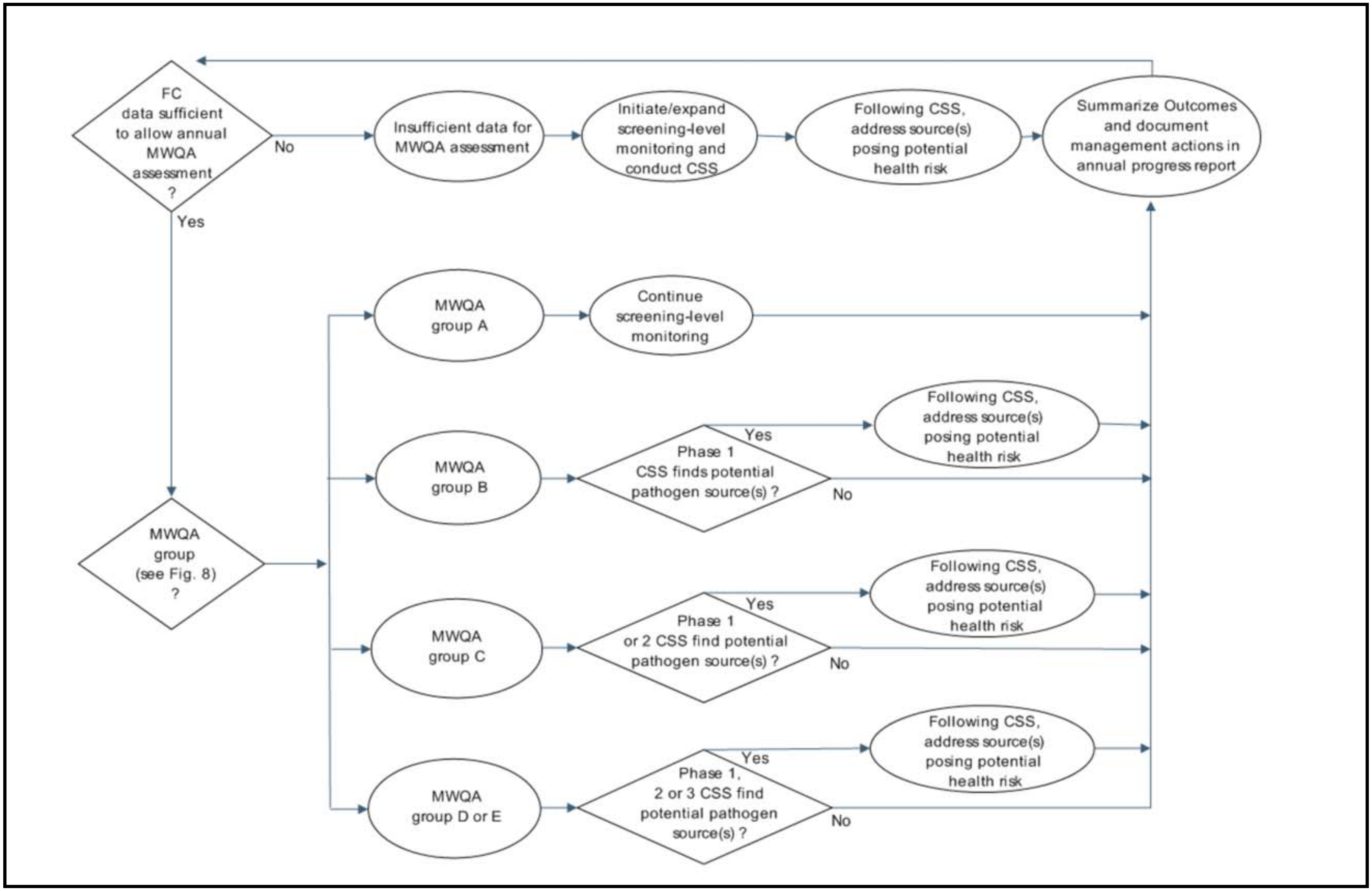

Figure 9. Potential management process for combining fecal coliform monitoring data and contaminant source survey (CSS) assessment information, and for addressing and reporting fecal coliform impairment.

7.4. An Initial Application of the Classification Matrix in the Hillsborough River Basin Management Action Plan Areas

In order to apply the proposed approach on a pilot basis to the currently-active BMAP areas in the Hillsborough River watershed, MWQA categories (Table 6) and CSS assessments and classification matrix outcomes (Table 7) were calculated and assigned to individual EPCHC monitoring stations within each BMAP area. (CSS assessments were provided through a separate MST project, funded by FDEP, which PBS&J and USF are currently conducting in the Hillsborough River watershed.)

The six WBIDs in the Hillsborough River watershed for which BMAPs are currently being developed are:

- Blackwater Creek (WBID 1482)
- Spartman Branch (WBID 1561)
- Baker Creek (WBID 1522C)
- Flint Creek (WBID 1522A)
- New River (WBID 1442), and
- Lower Hillsborough River (WBID 1443E)

Monthly EPCHC monitoring data from the years 2001 through 2007 are available for five of these six WBIDs, the exception being New River (WBID 1442) where EPCHC monitoring was initiated only recently. MWQA categories for the nine monitoring stations located within these five WBIDS, calculated by applying the methods described in Section 7.1.2 to the EPCHC fecal coliform data, are summarized in Table 8. Within the table, monitoring sites are ranked on the basis of their geometric mean (2001 – 2007) fecal coliform counts.

None of the nine monitoring sites showed extremely high fecal coliform levels during the 2001 – 2007 period, and all sites fell within MWQA categories A (≤ 10% of samples exceeding 400 CFU/100 mL) or B (>10% and ≤ 30% of samples exceeding 400 CFU/100 mL) during the period (Table 8). Each of the monitoring stations in the lower Hillsborough River (WBID 1443E) was in MWQA category B, as were the stations on Baker Creek at Thonotosassa Road (WBID 1522C) and Flint Creek at U.S. Highway 301 (WBID 1522A). In terms of land use in the contributing watersheds, the lower Hillsborough River WBID is within a highly urbanized portion of the City of Tampa that appears to be impacted by stormwater runoff and other urban contaminant sources, including SSOs. The Baker Creek and Flint Creek WBIDs are in rural portions of unincorporated Hillsborough County and contain a mix of agricultural and low-density rural residential-associated sources. Blackwater Creek is also primarily located in rural areas of Hillsborough County, although low-, medium- and high-density population neighborhoods and associated sources are present in the upstream portion of the watershed within Polk County.

Based on the combination of MWQA categories and initial CSS assessments assigned to these stations, their classification matrix designations fell within outcomes A2, B2, B3 and B4, as summarized in Table 8. Recommended management responses for sites falling within these outcome categories were described earlier, in Section 7.3.

As an aside, water resource and public health managers should also be aware that the two Lake Thonotosassa stations (EPCHC stations 118 and 135) are located in areas that are known to be subject to frequent, high-density cyanobacterial ("blue-green algae") blooms (SWFWMD 2003). The EPCHC monitoring program does not analyze samples for cyanobacterial toxins, and it is not known if cyanotoxins occur at the sites. However, if cyanotoxins are present at the sites (e.g., during or immediately following blooms), they could potentially pose an independent (non-fecal) source of health risk during periods when they occur at sufficiently high concentrations (e.g., WHO 2003). Guidelines for human exposure to some categories of cyanotoxins have recently been recommended by the WHO (2003), and are currently being evaluated by the U.S. EPA, but to date no water quality criteria have been adopted in the U.S. for these constituents. This is an example of beneficial ancillary information that can be provided to resource managers as a result of conducting a CSS, in which a wide range of monitoring data and other information is examined to gain a better understanding of underlying water quality conditions at each evaluated site.

Returning to fecal contamination issues, it is interesting for comparative purposes to consider the sampling locations shown in Table 8 in the context of the fecal coliform and enterococci counts reported by the EPCHC for all 39 of its long-term inland stations that were monitored during the 2001 – 2007 period. Those values are summarized in Appendix D. Within this larger group of stations, five locations, all located outside the Hillsborough River watershed, would be assigned to MWQA categories C, D or E based on the frequency of exceedance of the 400 CFU/100 mL fecal coliform criterion (Appendix D). Those locations, particularly the ones falling within MWQA categories D or E, appear to be high priorities for further investigation and potential management action.

Interestingly, a Bullfrog Creek location (in the vicinity of EPCHC station 132) was identified by Rose et al. (2001), through MST analyses, as an area in which abundant biochemical indicators of human fecal contamination were detected in surface water samples. As of yet, no CSS or MST studies appear to have been done for other category C, D or E sites listed in Appendix D. The information shown in Appendix D provides an example of ways in which the classification matrix approach (Table 7) could be used, at a watershed or regional scale, to prioritize sites for further investigation and possible management action to improve water quality.

Table 8. Microbial water quality assessment (MWQA) categories, geometric mean fecal coliform and enterocci values, and initial classification matrix outcomes for stations within active Basin Management Action Plan (BMAP) areas that were monitored by the Environmental Protection Commission of Hillsborough County (EPCHC) during the years 2001 through 2007. MWQA category assignments are based on the binomial method described in the text and the decision tree shown in Figure 8. Bacterial indicator organism (fecal coliform and enterococci) values are in units of CFU/100 mL. Within the table, sites are arranged in order of increasing geometric mean fecal coliform values. (Data source: EPCHC)

Location	EPCHC Station No.	WBID No.[1]	Geometric Mean Fecal Coliform Count	Geometric Mean Enterococci Count	MWQA Category	CSS Category	Classification Matrix Outcome
L. Thonotosassa middle	135	1522A	33	42	A	2	A2[a]
L. Thonotosassa at Flint Creek	118	1522A	47	69	A	2	A2[a]
Hillsborough River at US 301	108	1482	102	159	A	2	A2[a]
Blackwater Creek at SR 39	143	1482	157	284	B	2	B2
Flint Creek at US 301	148	1522A	191	360	B	2	B2
Hillsborough River at Columbus Ave	137	1443E	195	154	B	3	B3
Hillsborough River at Rowlett Park	105	1443E	209	128	B	3	B3
Baker Creek at Thonotosassa Rd	107	1522C	216	509	B	2	B2
Hillsborough River at Sligh Ave	152	1443E	240	183	B	4	B4[a]

Notes:

a) **As indicated in Table 7, these outcomes imply that the fecal coliform indicator may be providing an overly optimistic MWQA rating, or the CSS may be providing an overly negative assessment, and the cause(s) of the discrepancy should be verified.**

7.5. Additional Questions and Issues

The proposed BMAP decision support process outlined above has been discussed with the Hillsborough River BMAP Working Group, which was organized by the FDEP and the Tampa Bay Estuary Program and receives technical support from FDEP contractors and local government agency staff. That group has identified the following questions, issues and recommendations that its members would like to see addressed, either through the present project (if they fall within its scope) or in follow-on projects that are carried out during later phases of BMAP implementation:

1. The BMAP process should include a method for designating sites that do not meet the State's 400 CFU/100 mL fecal coliform criterion, but which are impacted only by non-anthropogenic environmental sources (e.g., wildlife, soils, aquatic sediments) that do not appear to represent significant public health risks. (A section of the impaired waters rule — Chapter 62-303.460[2], F.A.C. — notes this issue and may offer a mechanism for addressing the Working Group's concern.)

2. The process should be expanded to include other designated uses (e.g., Class I and II waters), in addition to the approach proposed here for Class III waters.

3. For Class III waters, it may also be helpful to incorporate a method for prioritizing sites based on the magnitude and intensity of the recreational use they receive, in addition to the prioritization approaches proposed here based on MWQA and CSS categories.

4. An additional decision support process may need to be developed for bacteriologically-impaired Class III waters that are used as designated bathing areas. Such areas are monitored (and, when necessary, posted to discourage public use) by the Florida Department of Health and its affiliated County Health Departments.

5. The minimum number of fecal coliform samples needed to assign sites to MWQA categories when applying the decision tree (Figure 8) and classification matrix (Table 7) needs to be defined, if a working definition has not already been provided through the IWR. (As noted in the current report, the project team would recommend that a minimum of 30 samples, collected at approximately bi-monthly intervals over a period of five years, be considered for this purpose.)

6. Potential approaches for assigning MWQA categories at tidally-influenced monitoring sites — which exhibit freshwater salinities during some sampling events and estuarine (or marine) salinities during others — need to be considered. The very different survival times that fecal coliforms and other indicator bacteria may have at these locations, depending on salinity characteristics of the site during the period immediately preceding a sampling event, complicate efforts to assign meaningful MWQA categories (or MWQA categories that are comparable to freshwater sampling stations) to these sites.

7. The impacts of sediment-associated fecal coliforms and other indicator bacteria on water quality and potential human health risk need to be better defined. When sediments are resuspended by storm events or other disturbances, indicator organism levels frequently become elevated. However, the implications for human health are poorly understood in this case because the bacteria may be surviving and growing in the sediments. Furthermore, it is unclear whether survival characteristics of pathogens are similar to those of the indicator bacteria in the sediments.

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Appendices

Appendix A. Florida Water Criteria for Fecal Coliform Bacteria (62-302-250 F.A.C.)

Florida water quality criteria for fecal coliform bacteria
(62-302.250 F.A.C.)

Parameter	Units	Class I	Class II	Class III: Fresh	Class III: Marine	Class IV	Class V
(6) Bacteriological Quality (Fecal Coliform Bacteria)	Number per 100 ml (Most Probable Number (MPN) or Membrane Filter (MF))	MPN or MF counts shall not exceed a monthly average of 200, nor exceed 400 in 10% of the samples, nor exceed 800 on any one day. Monthly averages shall be expressed as geometric means based on a minimum of 5 samples taken over a 30 day period.	MPN shall not exceed a median value of 14 with not more than 10% of the samples exceeding 43, nor exceed 800 on any one day.	MPN or MF counts shall not exceed a monthly average of 200, nor exceed 400 in 10% of the samples, nor exceed 800 on any one day. Monthly averages shall be expressed as geometric means based on a minimum of 10 samples taken over a 30 day period.	MPN or MF counts shall not exceed a monthly average of 200, nor exceed 400 in 10% of the samples, nor exceed 800 on any one day. Monthly averages shall be expressed as geometric means based on a minimum of 10 samples taken over a 30 day period.		

Appendix B. Relative Merits of Selected Indicators of Sewage Contamination

The relative merits of selected indicators of sewage contamination.
(Source: WHO 2000)

Indicator	Advantages	Disadvantages
Faecal streptococci/ enterococci	Marine and potentially freshwater human health indicator More persistent in water and sediments than coliforms Faecal streptococci may be cheaper than enterococci to assay	May not be valid in tropical waters due to potential growth in soils
Thermotolerant conforms	Indicator of recent faecal contamination	Possibly not suited to tropical waters due to growth in soils and waters Confounded by non-sewage sources (e.g. *Klebsiella* spp. in pulp and paper wastewaters)
E. coli	Potential freshwater human health indicator Indicator of recent faecal contamination Potential for typing *E. coli* to aid in sourcing faecal contamination Rapid identification possible if defined as β-glucuronidase-producing bacteria	Possibly not suited to tropical waters due to growth in soils and waters
Sanitary plastics	Little training of staff required and immediate assessment can be made for each bathing day Can be categorised	May reflect old sewage contamination and thus be of little health significance Subjective and prone to variable description
Preceding rainfall (12, 24, 48 or 72 h)	Simple regressions may account for 30-60% of the variation in microbial indicators for a particular beach	Each beach catchment may need to have its rainfall response assessed Response may depend on the period before the event
Sulphite-reducing clostridia[1]	Inexpensive assay with H_2S production Always in sewage impacted waters Possibly correlated with enteric viruses and parasitic protozoa	Enumeration requires anaerobic culture May also come from dog faeces May be too conservative an indicator
Somatic coliphages	Standard method well established Similar physical behaviour to human enteric viruses	Not specific to sewage May not be as persistent as human enteric viruses May grow in the environment
F-specific RNA phages	More persistent than some coliphages Standard ISO method available Host does not grow in environmental waters below 30°C	WG49 host may lose plasmid (although F-amp is more stable) Not specific to sewage Not as persistent in marine waters
Bacteroides fragilis phages	More resistant than other phages in the environment and similar to hardy human enteric viruses Appears to be specific to sewage ISO method recently published	Because numbers in sewage are lower than for other phages and most do not excrete this phage, it is of limited value for small populations Requires anaerobic culture
Faecal sterols	Coprostanol largely specific to sewage	Requires expensive gas

	Coprostanol degradation in water similar to die-off of thermotolerant coliforms A ratio of 5β: 5α stanols > 0.5 is indicative of faecal contamination; i.e. a ratio coprostanol: 5α-cholestanol of > 0.5 indicates human faecal contamination, while C_{29} 5β(24-ethylcoprostanol): 5α stanol ratio of > 0.5 indicates herbivore faeces Ratio of coprostanol: 24-ethylcoprostanol can be used to indicate the proportion of human faecal contamination, which can be further supported by ratios with faecal indicator bacteria (Leeming *et al*., 1996)	chromatography (about US$ 100 per sample) Requires up to 10 litres of sample to be filtered through a glass fibre filter to concentrate particulate stanols
Caffeine	May be specific to sewage, but unproven to date Could be developed into a dipstick assay	Yet to be proven as a reliable method
Detergents	Relatively routine methods available	May not be related to sewage (e.g. industrial pollution)
Turbidity	Simple, direct and inexpensive assay available in the field	May not be related to sewage; correlation must be shown for each site type
Cryptosporidium[2]	Required for potential zoonoses, such as *Cryptosporidium* spp., where faecal indicator bacteria may have died out, or not present	Expensive and specialised assay (e.g. Method 1622, US EPA); human/animal speciation of serotypes is not currently defined

[1] *Clostridium perfringens*

[2] Animal-sourced pathogens

Appendix C. Illnesses Caused by Microbial Agents Acquired by Ingestion of Water (Source: NRC 2000)

Illnesses Caused by Microbial Agents Acquired by Ingestion of Water
(Source: NRC 2000)

Agent	Source	Incubation Period	Clinical Syndrome	Duration
Viruses:				
Astrovirus	human feces[1]	1-4 days	Acute gastroenteritis	2-3 days; occasionally 1-14 days
Enteroviruses (polioviruses, coxsackieviruses, echoviruses)	human feces	3-14 days (usually 5–10 days)	Febrile illness, respiratory illness, meningitis, herpangina, pleurodynia, conjunctivitis, myocardiopathy, diarrhea, paralytic disease, encephalitis, ataxia	Variable
Hepatitis A	human feces	15-50 days (usually 25-30 days)	Fever, malaise, jaundice, abdominal pain, anorexia, nausea	1-2 weeks to several months
Hepatitis E[2]	human feces	15-65 days (usually 35-40 days)	Fever, malaise, jaundice, abdominal pain, anorexia, nausea	1-2 weeks to several months
Norwalk-like viruses	human feces	1-2 days	Acute gastroenteritis with predominant nausea and vomiting	1-3 days
Group A rotavirus	human feces[1]	1-3 days	Acute gastroenteritis with predominant nausea and vomiting	5-7 days
Group B rotavirus[2]	human feces[1]	2-3 days	Acute gastroenteritis	
Bacteria:				
Aeromonas hydrophila	fresh water		Watery diarrhea	Average 42 days
Campylobacter jejuni	human and animal feces	3-5 days (1-7 days)	Acute gastroenteritis, possible bloody and mucoid feces	1-4 days occasionally >10 days

Protozoa:				
Balantidium coli[2]	human and animal feces	Unknown	Abdominal pain, occasional mucoid or bloody diarrhea	Unknown
Cryptosporidium parvum	human and animal feces	1-2 weeks	Profuse, watery diarrhea	4-21 days
Entamoeba histolytica[2]	human feces	2-4 weeks	Abdominal pain, occasional mucoid or bloody diarrhea	Weeks to months
Cyclospora cayetenensis[2]	human feces	1 week average	Watery diarrhea, profound fatigue, anorexia, weight loss, bloating, abdominal cramps, nausea	Weeks if untreated
Giardia lamblia	human and animal feces	5-25 days	Abdominal pain, bloating, flatulence, loose, pale, greasy stools	1-2 weeks to months and years
Algae:				
Cyanobacteria (*Anabaena* spp., *Aphanizomenon* spp., *Microcystis* spp.) [2]	Algal blooms in water	A few hours	Toxin poisoning (blistering of mouth, gastroenteritis, pneumonia)	Variable
Helminths:				
Dracunculus medinensis[2] (Guinea worm)	larvae	8-14 months (usually 12 months)	Blister, localized arthritis of joints adjacent to site of infection	Months

[1]Animal strains of these viruses are not believed to be pathogenic for humans.
[2]Waterborne infections in the U.S. are rare or undocumented.

non-O1[2]

Yersinia enterocolitica	animal feces and urine	2-7 days	Abdominal pain, mucoid, occasionally bloody diarrhea, fever	1-21 days average 9 days

Appendix D. Microbial Water Quality Assessment Categories for Other Environmental Protection Commission of Hillsborough County Monitoring Stations

Table D-1. Criterion exceedance frequencies, microbial water quality assessment (MWQA) category assignments, and geometric mean fecal coliform and enterocci concentrations at tributary stations monitored by the Environmental Protection Commission of Hillsborough County (EPCHC) during the years 2001 through 2007. Microbial Water Quality Assessment (MWQA) group assignments are based on the binomial method (see text), applied using the decision tree shown in Figure 8. (Data source: EPCHC)

Location	Sta. No.	No. Samples (N)	% >400 CFU/ 100 mL ($\hat{P}$)	MWQA category	Geometric mean fecal coli. (CFU/ 100 mL)	Geometric mean enterococci (CFU/ 100 mL)
Turkey Creek at SR 60	111	77	84%	E	1,212.0	1,426.0
Bullfrog Creek at Symmes Rd	132	78	83%	E	1,134.8	2,658.9
Sweetwater Creek at Hillsborough Ave	104	84	69%	D	786.4	849.9
Delaney Creek at 36th Ave	138	84	54%	C	443.3	642.2
Delaney Creek at US 41	133	84	42%	C	376.7	596.8
Little Manatee River at CR 579	140	84	31%	B	283.6	664.9
English Creek at SR 60	154	83	30%	B	266.6	487.7
Mill Creek at I-4	149	81	37%	B	258.0	578.7
Little Manatee River at US 301	113	84	20%	B	250.8	664.0
Rocky Creek at Hillsborough Ave	103	84	26%	B	242.6	285.2
Hillsborough River at Sligh Ave	152	84	27%	B	239.8	182.8
Bullfrog Creek at US 41	144	84	37%	B	233.7	254.7
Little Manatee River at CR 674	129	84	24%	B	218.5	704.0
Baker Creek at Thonotosassa Rd	107	84	35%	B	216.5	508.7
Hillsborough River at Rowlett Pk	105	84	26%	B	208.7	127.7
Rocky Creek at Waters Ave	141	84	26%	B	205.8	343.4
Hillsborough River at Columbus Ave	137	84	33%	B	195.5	153.7
Flint Creek at US 301	148	84	25%	B	191.2	359.8
Double Branch Creek at Hillsborough Ave	101	84	27%	B	185.1	222.6
Blackwater Creek at SR 39	143	82	21%	B	156.9	284.2
Trout Creek at CR 581	145	65	18%	B	152.9	281.8
Sweetwater Creek at Anderson Rd	142	84	20%	B	152.2	764.8
Turkey Creek at Durant Rd	151	82	26%	B	142.0	322.6
Alafia River South Prong at Bethlehem Rd	139	83	16%	B	138.8	215.7
Alafia River at Bell Shoals Rd	114	84	18%	B	136.4	240.7
Cypress Creek at CR 581	120	72	19%	B	132.9	261.9
Alafia River at US 301	153	83	14%	A	152.7	209.7
Hillsborough River at US 301	108	83	11%	A	101.7	159.4
Alafia River North Prong ab confluence	115	84	11%	A	100.4	267.3
Alafia River South Prong ab confluence	116	84	10%	A	80.6	207.3
Channel A at Hillsborough Ave	102	84	14%	A	63.2	63.1
L. Thonotosassa at Flint Creek	118	82	2%	A	46.8	68.5
Hillsborough River at Fowler Ave	106	84	5%	A	46.6	96.4
TBC at Fowler Ave	146	82	7%	A	40.5	51.0
Palm River at US 41	109	84	7%	A	39.1	36.4
Little Manatee River at US 41	112	84	5%	A	34.6	55.1
L. Thonotosassa middle	135	82	2%	A	32.9	41.7
TBC at MLK Blvd	147	84	6%	A	31.9	31.0
Palm River at SR 60	110	84	7%	A	25.4	21.8

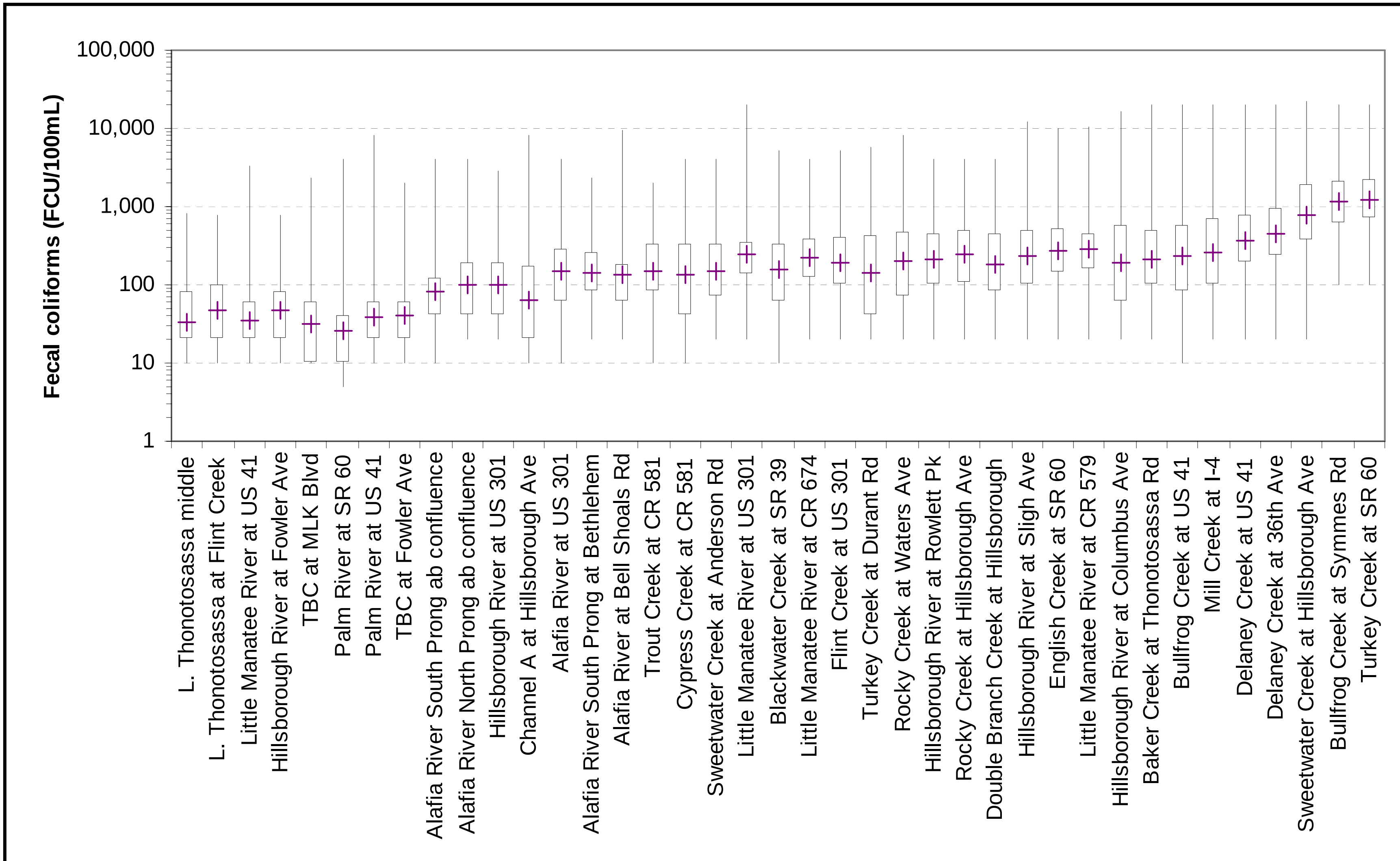

Figure D-1. Box–and-whisker plots (showing minimum, maximum, interquartile range and geometric mean) of fecal coliform counts (CFU/100 mL) observed at 39 Environmental Protection Commission of Hillsborough County (EPCHC) inland monitoring stations during the years 2001 – 2007. Stations are ranked based on the percentage of samples exceeding the State's 400 CFU/100 mL fecal coliform criterion. (Data source: EPCHC)

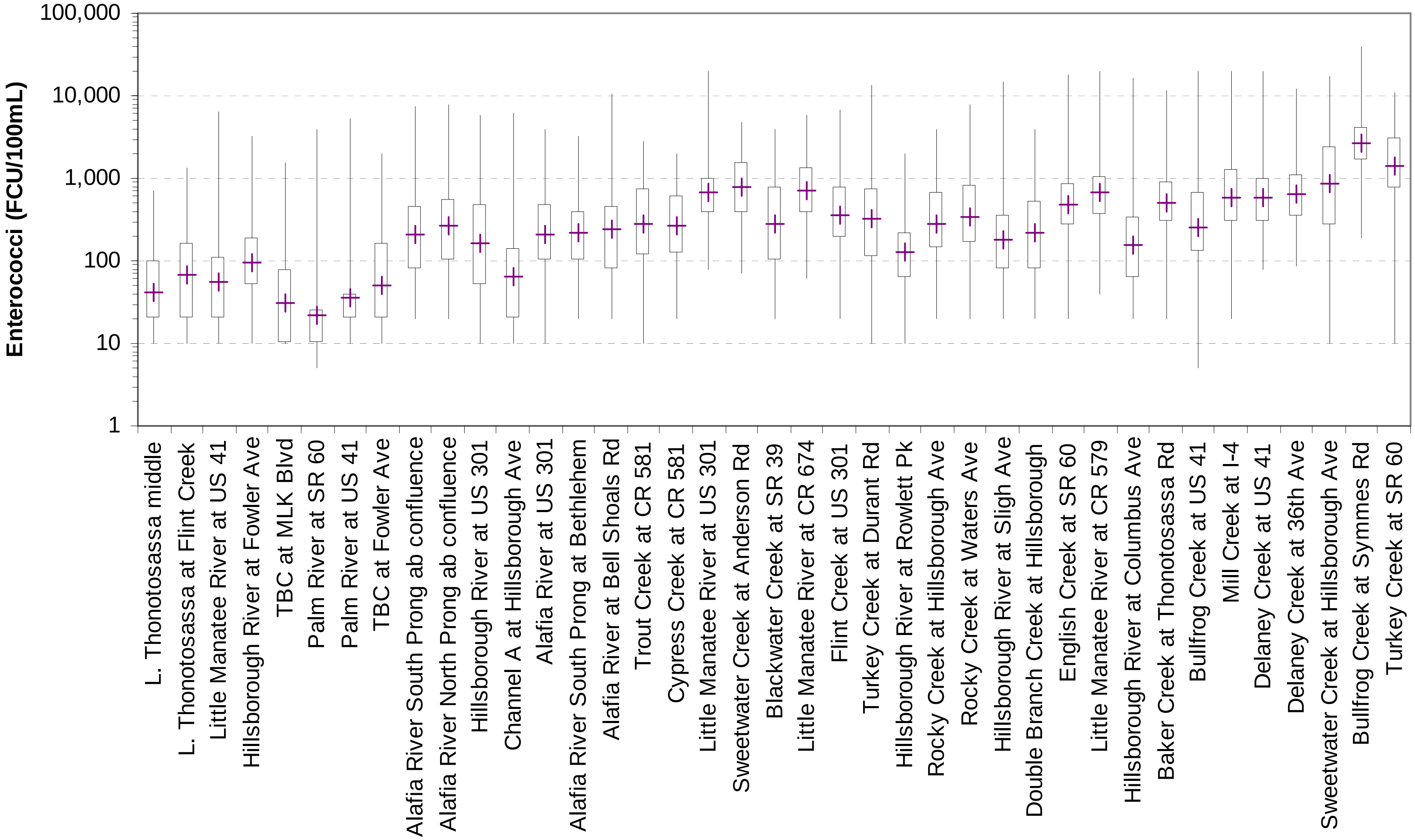

Figure D-2. Box–and-whisker plots (showing minimum, maximum, interquartile range and geometric mean) of enterococci counts (CFU/100 mL) observed at 39 Environmental Protection Commission of Hillsborough County (EPCHC) inland monitoring stations during the years 2001 – 2007. Stations are ranked based on the percentage of samples exceeding the State's 400 CFU/100 mL fecal coliform criterion. (Data source: EPCHC)

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Ambient Water Quality Criteria for Bacteria - 1986

Bacteriological Ambient Water Quality Criteria
for Marine and Fresh Recreational Waters

U.S. Environmental Protection Agency

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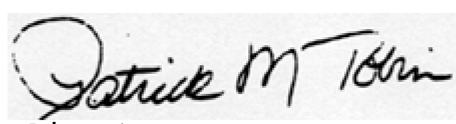
FOREWORD

Section 304(a)(1) of the Clean water Act of 1977 (P.L. 95-217) requires the Administrator of the Environmental Protection Agency to publish criteria for water quality accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare which may be expected from the presence of pollutants in any body of water, including ground water. This document is a revision of proposed criteria based upon a consideration of comments received from other Federal agencies, State agencies, special interest groups, and individual scientists. The criteria contained in this document supplements previously published EPA bacteriological criteria in Quality Criteria for Water (1976).

The term "water quality criteria" is used in two sections of the Clean Water Act, section 304(a)(1) and Section 303(c)(2). The term has a different program impact in each section. In section 304, the term represents a non-regulatory, scientific assessment of ecological and public health effects. The criteria presented in this publication are such scientific assessments. Water quality criteria associated with specific ambient water uses when adopted as State water quality standards under section 303 become enforceable maximum acceptable levels of a pollutant in ambient waters. The water quality criteria adopted in the State water quality standards could have the same numerical limits as the criteria developed under section 304. However, in many situations States may want to adjust water quality criteria developed under section 304 to reflect local environmental conditions and human exposure patterns before incorporation into water quality standards. It is not until their adoption as part of the State water quality standards that the criteria become regulatory.

The bacteriological water quality criteria recommended in this document are based on an estimate of bacterial indicator counts and gastrointestinal illness rates that are currently being accepted, albeit unknowingly in many instances, by the States. Wherever bacteriological indicator counts can consistently be calculated to give illness rates lower than the general estimate, or when the State desires a lower illness rate, indicator bacteria levels commensurate with the lower rate should be maintained in State water quality standards.

Guidelines to assist the States in modification of criteria presented in this document, in the development of water quality standards, and in other water-related programs of this Agency, have been developed by EPA.



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BACTERIOLOGICAL AMBIENT WATER QUALITY CRITERIA FOR MARINE AND FRESH RECREATIONAL WATERS

Introduction

Federal water quality criteria recommendations were first proposed in 1968 by the National Technical Advisory Committee (NTAC) of the Department of the Interior (1). The microbiological criterion suggested by the NTAC for bathing waters was based an a series of studies conducted during the late 1940's and early 1950's, by the United States Public Health Service, the results of which were summarized by Stevenson in 1953 (2). The studies were conducted at bathing beaches located on Lake Michigan at Chicago, Illinois; on the Ohio River at Dayton, Kentucky; and on Long Island Sound at Mamaroneck and New Rochelle, New York. All of the studies followed a similar design. Two beaches with different water quality were selected at each location except at the Dayton location where a beach with high quality water could not be found. A large public swimming pool was used as a substitute. Each location was chosen because, in addition to beaches having suitable water quality, there was a large residential population nearby that used the beaches. Cooperating families used a calendar system which allowed then to record their swimming activity and illnesses on a daily basis for the entire summer. Gastrointestinal, respiratory, and other symptom such as skin irritations were recorded. The water quality was measured on a routine basis using total coliform bacteria as the indicator organism.

The results of the Lake Michigan beach study indicated that there was no excess illnesses of any type in swimmers at beaches that had median coliform densities of 91 and 180 per 100 ml over a swimming season when compared to the number of illnesses in the total study population. The water quality similarity at the two Chicago beaches was unexpected since previous experience had indicated that there was a difference in water quality at the beaches. A second method of analysis compared the illness observed in the week following three days of high coliform density with that observed following swimming on three days of low coliform density. The analyses showed that there was a significantly greater illness rate in individuals who swam on the three days when the geometric mean coliform density was 2300/100 ml when compared to the illness in swimmers who swan on the three days when the geometric mean coliform density was 43 per 100 ml. A difference was not observed when the geometric mean coliform density per 100 ml on high and low days was 732 and 32 respectively. Data from the Ohio River study indicated that swimmers who swam in water with a median coliform density of 2300 coliform per 100 ml had an excess of gastrointestinal illness when compared to an expected rate calculated from the total study population. No other associations between swimming and illness were observed. The results of two marine bathing beach studies showed no association between illness and swimming in water containing 398 and 815 coliforms per 100 ml.

The coliform water quality index used during the USPHS epidemiological studies was translated into a fecal coliform index in the mid-‘60’s by using the ratio of fecal coliforms to coliforms, at the location on the Ohio River where the original study had been conducted in 1949. The NTAC committee suggested that the change was necessary because fecal coliforms were more fecal specific and less subject to variation than total coliforms which were greatly influenced by storm water runoff. About 18% of the coliforms were found to be fecal coliforms and this proportion was used to determine that the equivalent of 2300 coliforms per 100 ml, the density at which a statistically significant swimming-associated gastrointestinal illness was observed, was about 400 fecal coliforms per 100 ml. The NTAC suggested that a detectable risk was undesirable and, therefore, one-half of the density at which a health risk occurred, 200 fecal coliform per 100 ml, was proposed. The NTAC also suggested that the use of the water should not cause a detectable health effect more than 10% of the time. Thus, the recommended criterion for recreational waters was as follows:

> "Fecal coliforms should be used as the indicator organism for evaluating the microbiological suitability of recreation waters. As determined by multiple-tube fermentation or membrane filter procedures and based on a minimum of not less than five samples taken over not more than a 30-day period, the fecal coliform content of primary contact recreation waters shall not exceed a log mean of 200/100 ml, nor shall more than 10 percent of total samples during any 30-day period exceed 400/100 ml.”

This criterion was recommended again in 1976 by the USEPA (3), even though it had been criticized on a number of issues. Henderson (4) published one of the earliest critiques of the recommended criterion. He noted the paucity of epidemiological data in support of any numerical ceilings based on fecal indicators and criticized the one proposed as to the poor quality of the data base, the derivation of the specific limits and the indicator system used.

Moore (5) objected to the selection of only part of the data from the Lake Michigan study to develop the 200 fecal coliforms per 100 ml recreational water criterion. He observed that opposite findings in the Lake Michigan studies were ignored. He pointed out that the inclusion of all illnesses reported during the week after a bathing episode made the association of these ailments with the bathing episode tenuous, and that there was no way of knowing whether the incidence of skin irritations in bathers who swam on clean days was compared to the frequency of diarrhea in those who swam an other days, because all the illnesses reported were lumped together.

Cabelli et al. (6) suggested other weaknesses in the USPHS study design which would have precluded the identification of swimming-associated, associated, pollution-related illnesses if, in fact, they occurred. They pointed out that "swimming" was "poorly defined and that it was unknown whether or not study participants who said they had been swimming actually immersed their bodies, much less their heads, in the water. This short coming and the use of the calendar method for recording "swimming" episodes and illnesses also was criticized as precluding the inclusion of beachgoing but nonswimming control groups in the studies. Moreover, the use of the calendar approach with nearby residents and the day-to-day variability in the pollution levels at the beaches increased the probability of a given individual's exposure to different levels of pollution during the incubation period of the illness.

The deficiencies in the study design and in the data used to establish the 200 fecal coliforms per 100 ml criterion were noted by the National Academy of Science - National Academy of Engineers Committee in their 1972 report which stated that they could not recommend a recreational water quality criterion because of the paucity of epidemiological information available (7).

The fecal coliform indicator used to measure water quality under the current system has also been faulted because of the non-fecal sources of at least one member of the fecal coliform group. For example, thermotolerant <u>Klebsiella</u> species have many sources. They have been observed in pulp and paper mill effluents (8,9), textile processing plant effluents (10), cotton mill wastewaters (11), and sugar beet wastes (12), in the absence of fecal contamination.

The Environmental Protection Agency, in 1972, initiated a series of studies at marine and fresh water bathing beaches which were designed to correct the perceived deficiencies of the Public Health Service studies. One goal of the EPA studies was to determine if swimming in sewage-contaminated water carries a health risk for bathers; and, if so, to what type of illness. If a quantitative relationship between water quality and health risk was obtained, two additional goals were to determine which bacterial indicator is best correlated to swimming-associated health effects and if the relationship is strong enough to provide a criterion.

<u>Study Design</u>

The marine studies were conducted at bathing beaches in New York City, New York, Boston, Massachusetts, and at Lake Pontchartrain, near New Orleans, Louisiana. Two beaches were selected at each site, one that received very little or no contamination and the other whose water quality was barely acceptable with respect to local recreational water quality standards. In the Now York City and Boston Harbor studies, the “barely acceptable” beaches were contaminated with pollution from multiple pointsources, usually treated effluents that had been disinfected.

The freshwater studies were conducted on Lake Erie at Erie, Pennsylvania and on Keystone Lake outside of Tulsa, Oklahoma. The “barely acceptable” beaches at both sites were contaminated by effluents discharged from single point-sources.

The epidemiological surveys were carried out on weekend days and individuals who swan in the midweeks before and after a survey were eliminated from the study. This maximized the study populations; allowed the water quality measurements for a single day to be specifically associated with the corresponding illness rates, and permitted the grouping of days with similar water quality levels and their corresponding study populations. The design of the epidemiological survey portion of the study has been described elsewhere (13,14). Specific steps taken to correct the deficiencies of earlier studies were noted earlier.

In the initial phases of the overall study, multiple indicators of water quality were used to monitor the water. This was done because it was not known which indicator of water quality might show a quantitative relationship with swimming-associated health effects. This unique approach resulted in the selection of the best indicator based on the strength of the statistical relationship between the water quality indicator and a swimming-associated health effect.

Each participant was querried at length about any illness symptoms, their date of onset and the duration of the symptoms. The symptoms were grouped into four general categories, gastrointestinal, respiratory, eye, ear and nose, and "other". Gastrointestinal symptoms included vomiting, diarrhea, stomachache and nausea. Sore throat, bad cough and chest colds comprised the respiratory symptom, and runny or stuffy nose, earache or runny ears and red, itchy or watery eyes were considered symptomatic of eye, ear or nose problems. Other symptoms included fever greater than 100° F, headache for more than a few hours or backache.

All of the symptoms were self-diagnosed and therefore subject to variable interpretation. The potential for misinterpretation was minimized by creating a new symptom category called highly credible gastrointestinal symptoms. This symptom category was defined as including any one of the following unmistakable or combinations of symptoms: (1) vomiting, (2) diarrhea with fever or a disabling condition (remained home, remained in bed or sought medical advice because of the symptoms) and (3) stomachache or nausea accompanied by a fever. Individuals in this symptom category were considered to have acute gastroenteritis.

Data Base for Marine and Fresh Water Criteria

The results of the marine Bathing Beach Studies have been reported by Cabelli (15) and those of the freshwater studies have been described by Dufour (16). In general, those symptom categories unrelated to gastroenteritis usually did not show a significant excess of illnesses at either of the paired beaches at each study location. Moreover, the significant swimming-associated rates for gastroenteritis were always observed at the more polluted of the paired beaches at each study location. Table 1 shows the number of occasions when significant swimming-associated gastroenteritis was observed at barely acceptable and relatively unpolluted marine and fresh water beaches. Statistically significant swimming-associated gastroenteritis rates were not observed at any of the relatively unpolluted beaches. The occurrence of a statistically significant excess of swimming-associated gastroenteritis in swimmers who bathed at beaches that were, by selection, more polluted is indicative that there is an increased risk of illness from swimming in water contaminated with treated sewage, i.e., both swimming-associated and pollution related. This finding, which was observed at both marine and fresh water locations was important because it placed in proper perspective the relationship between water contaminated with treated sewage and health risks for swimmers. This association was not very well defined in the earlier USPHS studies. The only evidence that sewage-contaminated water carried a risk for gastroenteritis in those studies was observed at the Ohio River beach where swimmers had an excess of gastrointestinal illness when the median coliform density in the water was 2300 per 100 ml. This was counter to the results found at freshwater beaches in Chicago and at marine beaches on Long Island Sound where swimmers had no more gastrointestinal Unless than nonswimmers even when days of "high" and "low" coliform densities were selected. Therefore, other than the occasional association of an outbreak of disease with swimming (17), the data from Cabelli (15) and Dufour (16) are the only available evidence linking sewage contaminated water with a health risk for bathers.

Although the association of illness in swimmers using bathing water contaminated by treated sewage is an important aspect of the process for developing recreational water quality criteria, it is the establishment of a quantitative relationship between the two variables that provides a useful relationship for regulating water quality. A part of this process is the development of suitable methods for measuring the quality of the water.

A comprehensive discussion of microbial water quality indicators is beyond the scope of this document, even as the basis for the selection of those examined in the epidemiological studies. The reader is referred for this to the reports of the studies (15,16) and to reviews on the subject (18,19). The examination of a number of potential indicators, including the ones most commonly used in the United States (total coliforms and fecal coliforms), was included in the studies. Furthermore, the selection of the best indicator was based on the strength of the relationship between the rate of gastroenteritis and the indicator density, as measured with the Pearson Correlation Coefficient. This coefficient varies between minus one and plus one. A value of one indicates a perfect relationship, that is, all of the paired points lie directly on the line which defines the relationship. A value of zero means that there is no linear relationship. A positive value indicates that the relationship is direct, one variable increases as the other increases. A negative value indicates the relationship is inverse, one variable decreases as the other increases. The correlation coefficients for gastroenteritis rates as related to the various indicators of water quality from both marine and fresh bathing water are shown in Table 2.

The data from the three years of the New York City study were analyzed in two ways. The first was by grouping trial days with similar indicator densities from a given swimming season and the second was by looking at each entire summer. The results from both analyses are shown in Table 2. For either type of analysis, enterococci showed the strongest relationship to gastroenteritis. _E_. _coli_ was a very poor second and all of the other indicators, including total coliforms and fecal coliforms showed very weak correlations to gastroenteritis. Enterococci and _E_. _coli_ were used in subsequent studies including the freshwater trials. Fecal coliforms also were included in subsequent studies because of their status as an accepted basis for a criterion.

The freshwater studies were analyzed only by summer. The correlation coefficient for E. coli was slightly greater than that for enterococci, however, statistical analysis indicated that the two values were not significantly different. Fecal coliforms, on the other hand,, had a correlation coefficient that was very similar to that observed for fecal coliforms from the marine data analyzed by summer. The freshwater studies confirmed the findings of the marine studies with respect to enterococci and fecal coliforms in that the densities of the former in bathing water showed strong correlation with swimming associated gastroenteritis rates and densities of the latter showed no correlation at all. The similarities in the relationships of E. coli and enterococci to swimming associated gastroenteritis in freshwater indicate that these two indicators are equally efficient for monitoring water quality in freshwater,, whereas in marine water environments only enterococci provided a good correlation. The etiological agent for the acute gastroenteritis is probably viral (20). The ultimate source of the agent is human fecal wastes. E. coli is the most fecal specific of the coliform indicators (21); and enterococci, another fecal indicator, better emulates the virus than do the coliforms with respect to survival in marine waters (22).

Basis of Criteria for Marine and Fresh Recreational Waters

Cabelli (15) defined a recreational water quality criterion as a “quantifiable relationship between the density of an indicator in the water and the potential human health risks involved in the water's recreational use." From such a definition, a criterion now can be adopted by a regulatory agency, which establishes upper limits for densities of indicator bacteria in waters that are associated with acceptable health risks for swimmers.

The quantitative relationships between the rates of swimming associated health effects and bacterial indicator densities were determined using regression analysis. Linear relationships were estimated from data grouped on the basis of summers or trials with similar indicator densities. The data for each summer were analyzed by pairing the geometric mean indicator density for a summer bathing season at each beach with the corresponding swimming-associated gastrointestinal illness rate for the same summer. The swimming-associated illness rate was determined by subtracting the gastrointestinal illness rate in nonswimmers from that for swimmers. These two variables from multiple beach sites were used to calculate a regression coefficient, y-intercept and 95% confidence intervals for the paired data. in the marine studies the total number of points for use in regression analysis was increased by collecting trial days with similar indicator densities from each study location and placing them into groups. The swimming-associated illness rate was determined as before, by subtracting the nonswimmer illness rate of all the individuals included in the grouped trial days from the swimmer illness rate during these same grouped trial days. The grouping by trial days with similar indicator densities approach was not possible with the freshwater data because the variation of bacterial indicator densities in freshwater samples was not large enough to allow such an adjustment to be made.

For the saltwater studies the results of the regression analyses of illness rates against indicator density data was very similar using the "by summer" or "by grouped trial days" approaches. The data grouped by trial days will be used here because of the broader range of indicator densities available for analysis. Table 3 shows the results of the marine and fresh water bathing beach studies conducted from 1973 through 1982. These data were used to define the relationships between swimming-associated gastroenteritis and bacterial indicator densities presented below.

The methods used to enumerate the bacterial indicator densities which showed the best relationship to swimming-associated gastroenteritis rates were specifically developed for the recreational water quality studies. The membrane filter procedure for enumerating enterococci was developed by Levin et al. (23). Evaluation of the method using fresh and marine water samples indicated that it detects mainly Streptococcus faecalis and Streptococcus faecium. Although these two species were thought to be more human specific than other Streptococci, they have been found in the intestinal tract of other warm-blooded animals such as cats, dogs, cows, horses and sheep.

E. coli were enumerated using the membrane filter procedure developed by Dufour et al. (24). Evaluation of this method with marine and fresh water samples has shown that 92 to 95% of the colonies isolated were confirmed as E. coli.

These membrane filter methods have successfully undergone precision and bias testing by the EPA Environmental Monitoring and Support Laboratory. The test methods are available in the EPA Research and Development report, EPA-600/4-85/076 Test Methods for Escherichia coli and Enterococci in Water by the Membrane Filter Procedure.

Recommendations on Bacterial Criteria monitoring

Several monitoring situations to assess bacterial quality are encountered by regulatory agencies. The situation needing the most rigorous monitoring is the designated swimming beach. Such areas are frequently lifeguard protected, provide parking and other public access and are heavily used by the public. Public beaches of this type were used by EPA in developing the relationship described in this document.

Other recreational activities may involve bodies of water which are regulated by individual State water quality standards. These recreational resources may be natural wading ponds used by children or waters where incidental full body contact occurs because of water skiing or other similar activities.

It is EPA's judgement that the monitoring requirements for these various recreational activities are different. For the public beaches, more frequent sampling is required to verify the continued safety of the waters for swimming, and to identify water quality changes which might impair the health of the public. Increasing the number of samples improves the accuracy of bacterial water quality estimates, and also improves the likelihood of correct decisions on whether to close or leave open a beach.

Waters with more casual and intermittent swimming use need fewer samples because of the reduced population at risk. Such sampling may also be used in establishing trends in the bacterial water quality so that the necessary improvements in the sanitary quality can be identified before disease risks become acute.

The following compliance protocol is one recommended EPA for monitoring recreational bathing waters. It is based on-the assumption that the currently accepted risk level based on the QCW recommendation has been determined to be appropriate and that the monitoring methods, i.e., bacterial enumeration techniques are imprecise, and environmental conditions, such as rainfall, wind and temperature will vary temporally and spatially. The variable nature of the environment, which affects the die-off and transport of bacterial indicators, and the inherent imprecision of bacterial enumeration methods, suggests an approach that takes these elements into account. Noncompliance with the criterion is signaled when the maximum acceptable geometric mean is exceeded or when any individual sample exceeds a confidence limit, chosen accordingly or to a level of swimming use. The mean log standard deviation for *E. coli* densities at the nine freshwater beach sites that were studied was about 0.4. The mean log standard deviation for enterococci in freshwater samples was also about 0.4 and in seawater samples it was about 0.7. These two values, 0.4 and 0.7 will be used in calculations associated with the proposed monitoring protocol and upper percentile values.

It is recommended that sampling frequency be related to the intensity of use of the water body. In areas where weekend use is substantial, weekly samples collected during the peak use periods are reasonable. In less heavily used areas perhaps bi-weekly or monthly samples may be adequate to decide bacterial water quality. In general, samples should be collected during dry weather periods to establish so-called "steady state" conditions. Special studies may be necessary to evaluate the effects of wet weather conditions on waters of interest especially if sanitary surveys indicate the area may be subject to storm water effects.

The water samples are collected in sterile sampling containers as described in *Standard Methods for the Examination of Water and Wastewater* (25).

Development of Recommended Criteria Based on E. coli/Enterococci

Currently EPA is not recommending a change in the stringency of its bacterial criteria for recreational waters. Such a change does not appear warranted until more information based on greater experience with the new indicators can be accrued. EPA and the State Agencies can then evaluate the impacts of change in terms of beach closures and other restricted uses. EPA recognizes that it will take a period of at least one triennial review and revision period for States to incorporate the new indicators into State Waster Quality Standards and start to accrue experience with the new indicators at individual water use areas.

EPA's evaluation of the bacteriological data indicated that using the fecal coliform indicator group at the maximum geometric mean of 200 per 100 ml, recommended in Quality Criteria for Water would cause an estimated 8 illness per 1,000 swimmers at fresh water beaches and 19 illness per 1,000 swimmers at marine beaches. These relationships are only approximate and are based on applying ratios of the geometric means of the various indicators from the EPA studies to the 200 per 100 ml fecal coliform criterion. However, these are EPA's best estimates of the accepted illness rates for areas which apply the EPA fecal coliform criterion.

The *E. coli* and enterococci criteria presented in Table 4 were developed using these currently accepted illness rates. The equations developed by Dufour (16) and Cabelli (15) were used to calculate the geometric mean indicator densities corresponding to the accepted gastrointestinal illness rates. These densities are for steady state dry weather conditions. The beach is in noncompliance with the criteria if the geometric mean of several bacterial density samples exceeds the value listed in Table 4.

Noncompliance is also signalled by an unacceptably high value for any single bacterial sample. The maximum acceptable bacterial density for a single sample is set higher than that for the geometric mean, in order to avoid unnecessary beach closings based on single samples. in deciding whether a beach should be left open, it is the long term geometric mean bacterial density that is of interest. Because of day-to-day fluctuations around this mean, a decision based on a single sample (or even several samples) may be erroneous, i.e., the sample may exceed the recommended mean criteria even though the long-term geometric mean is protective, or may fall below the maximum even if this mean is in the nonprotective range.

To set the single sample maximum, it is necessary to specify the desired chance that the beach will be left open when the protection is adequate. This chance, or confidence level, was based on Agency judgment. For the simple decision rule considered here, a smaller confidence level corresponds to a more stringent (i.e. lower) single sample maximum. Conversely, a greater confidence level corresponds to less stringent (i.e. higher) maximum values. This technique reduces the chances of single samples inappropriately indicating violations of the recommended criteria.

By using a control chart analogy (26) and the actual log standard deviations from the EPA studies, single sample maximum densities for various confidence levels were calculated. EPA then assigned qualitative use intensities to those confidence levels. A low confidence level (75%) was assigned to designated beach areas because a high degree of caution should be used to evaluate water quality for heavily used areas. Less intensively used areas would allow less restrictive single sample limits. Thus, 95% confidence might be appropriate for swimmable water in remote areas. Table 4 summarizes the results of these calculations. These single sample maximum levels should be recalculated for individual areas if significant differences in log standard deviations occur.

The levels displayed in Table 4 depend not only on the assumed standard deviation of log densities, but also on the chosen level of acceptable risk. while this level was based on the historically accepted risk, it is still arbitrary insofar as the historical risk was itself arbitrary. A detailed protocol is available* which shows how to determine the confidence level associated with any illness risk of interest, once a maximum has been established for single samples. The protocol also indicates how the confidence level approach can be applied to multiple sample geometric means. In Table 4, the limit for the measured geometric mean is determined directly from the regression equation relating illnesses to bacteriological density, without any "confidence level" allowance for random variations in the geometric mean of several samples.

Limitations and Extrapolations of Criteria

The limitations of Water Quality Criteria based on swimming-associated health effects and bacterial indicator densities have been addressed by Cabelli (18). Briefly, the major limitations of the criteria are that the observed relationship may not be valid if the size of the population contributing the fecal wastes becomes too small or if epidemic conditions are present in a community. In both cases the pathogen to indicator ratio, which is approximately constant in a large population becomes unpredictable and therefore, the criteria may not be reliable under these circumstances. These two considerations point out the importance of sanitary surveys and epidemiological surveillance as part of the monitoring program.

The presence of these indicators, in rural areas, shows the presence of warm blooded animal fecal pollution. Therefore, EPA recommends the application of these criteria unless sanitary and epidemiological studies show the sources of the indicator bacteria to be non-human and that the indicator densities are not indicative of a health risk to those swimming in such waters. EPA is sponsoring research to study the health risk of nonpoint source pollution from rural areas on the safety of water for swimming. Definitive evidence from this study was not available at the time of preparation of this criterion, but will be incorporated into subsequent revisions.

Relationship with the Criterion contained in Quality Criteria for Water (QCW)

The 1976 QCW criterion contained recommendations for both swimming and shellfish harvesting waters. This criteria recommendation is intended as a modification to the earlier criterion. Nothing in this criterion is intended to supersede the QCW recommendations concerning the bacterial quality of shellfish waters. EPA is currently co-sponsoring, with the National oceanic and Atmospheric Administration, research into the

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application of the enterococci and _E_. _coli_ indicators for assessing the quality of shellfish harvesting waters. The Food and Drug Administration is also reviewing the results of these studies. A change to the new indicators may be forthcoming if the studies show a correlation between gastrointestinal disease and the consumption of raw shellfish from waters with defined densities of the new indicators. However, these studies have not sufficiently progressed to justify any change at this time. Thus, the recommendations in QCW for shellfish waters must remain unchanged.

The QCW recommendations for swimming waters were based on fecal coliform. Data submitted to EPA during the public comment period showed that within sane beaches, a correlation could be shown between _E_. _coli_ densities and fecal coliform densities. such a site-specific correlation is not surprising because _E_. _coli_ is part of the fecal coliform group. However, the EPA tests show that no general correlation exists across different beaches. Therefore, EPA believes that the newly recommended indicators are superior to the fecal coliform group. Therefore, EPA strongly recommends that states begin the transition process to the new indicators. While either _E_. _coli_ or enterococci may be used for fresh waters, only enterococci is recommended for marine waters.

Table 1. Relationship Between Significant Swimming-Associated Gastroenteritis and the Degree of Pollution at Marine and Fresh Water Bathing Beaches.

	Beach Water Quality	
	Barely Acceptable	Relatively Unpolluted
No. Trials	17	8
No. Trials with Excess Illness in swimmers[1]	7	0
% Trials with Excess Swimmer Illness	41	0

[1]Difference between swimmer and nonswimmer illness rates during a trial period statistically significant at $p < 0.05$ level

TABLE 2. Correlation Coefficients for Swimming-Associated Gastroenteritis Rates Against Mean Indicator Densities at marine and Fresh Water Bathing Beaches

Type of Water	Indicator	Correlation Coefficients	
		Data by Summers	Data by Grouped Trials[1]
Marine[2]			
	enterococci	.75	.96
	E. coli	.52	.56
	Klebsiella	.32	.61
	Einterobacter/Citrobacter	.26	.64
	Total Coliform	.19	.65
	C. perfringens	.19	.01
	P. aeruginosa	.19	.59
	Fecal Coliforms	-.01	.51
	A. hydrophila	-.09	.60
	V. parahemolyticus	-.20	.42
	Staphylococci	-.23	.60
Fresh[3]			
	enterococci	.74	
	E. coli	.80	
	Fecal Coliforms	-.08	

[1]Groups of trials (days) with similar mean indicator densities during a given summer
[2]Data from trials conducted at New York City beaches 1973-1975 (Reference 18)
[3]Data from Reference 19

TABLE 3. Summary of Mean Indicator Density--Swimming--Association Gastroenteritis Rates From Trials of All U.S. Studies

Type of Water	Location	Beach[1]	Year	E. coli Density	Enterococcus Density	Number Swimmers	Number illnesses	Number Nonswimmers	Number Illnesses	Gastroenteritis Rate nor- 1000
Marine	N.Y.C.	RW	1973		21.8	484	30.4	197	15.2	15.2
		CI			91.2	474	46.4	167	18	28.4
			1974		3.6	1391	7.6	711	4.2	3.4
					7.0	951	10.5	1009	6.9	3.6
					13.5	625	16.0	419	2.4	13.6
					31.5	831	18.1	440	-	18.1*
			1975		5.7	2232	8.8	935	19.3	-0.5
					20.3	1896	14.8	678	7.4	7.4
					154	579	34.5	191	-	34.5*
	Lake Pontchartrain	L	1977		44	874	32	451	11.1	20.9*
					224	720	31.9	456	8.8	23.1*
					495	895	35.8	464	6.6	27.2*
		L	1978		11.1	1230	36.6	415	14.5	22.1*
		F			14.4	248	44.3	303	23.1	21.2
		L			142	801	42.4	322	15.5	26.9*
	Boston Harbor	RE	1978		43	697	23	529	11	12
		N			7.3	1130	33	1099	28	5
		RE			12.0	222	41	376	13	28*
Fresh										
	Lake Erie	A	1979	23	5.2	3020	17.2	2349	14.9	2.3
		B		47	13	2056	19.5	2349	14.9	4.6
	Keystone take	E		138	38.8	3059	20.6	970	15.5	5.1
		w		19	6.8	2440	20	970	15.5	0.5
	take Erie	A	1980	137	25	2907	16.5	2944	11.7	4.8
		B		236	71	2427	26.4	2944	11.7	14.7*
	Keystone take	E		52	23	5121	13.5	1211	8.1	5.2
		w		71	20	3562	11.2	1211	8.1	3.0
	Lake Erie	B	1982	146	20	4374	24.9	1650	13.9	11.0*

[1]RW = Rockaways, CI = Coney Island, L = Levee Beach, F = Fontainbleu Beach, R = Revere Beach, N - Nahant Beach, A = Beach 7, B = Beach 11, E = Washington Irving Cove Beach, W = Salt Creek Cove--Keystone Ramp Beaches
*Indicates swimmer-nonswimmer illness rate difference significant at p = 0.05 level

TABLE 4. CRITERIA FOR INDICATOR FOR BACTERIOLOGICAL DENSITIES

	Acceptable Swimming Associated Gastro-enteritis Rate per 1000 swimmers	Steady State Geometric Mean Indicator Density	Single Sample Maximum Allowable Density			
			Designated Beach Area (upper 75% C.L.)	Moderate Full Body Contact Recreation upper 82% C.L.)	Lightly Used Full Body Contact Recreation upper 90% C.L.)	Infrequently Used Full Body Contact Recreation (upper 95% C.L.)
Freshwater						
enterococci	8	33 (1)	61	78	107	151
E. coli	8	126 (2)	235	298	409	575
Marine Water						
enterococci	19	35 (3)	104	158	276	501

Notes:

(1) Calculated to nearest whole number using equation:

$$(\text{mean enterococci density}) = \text{antilog}_{10} \frac{\text{illness rate/1000 people} + 6.28}{9.40}$$

(2) Calculated to nearest whole number using equation:

$$(\text{mean } \textit{E. coli} \text{ density}) = \text{antilog}_{10} \frac{\text{illness rate/1000 people} + 11.74}{9.40}$$

(3) Calculated to nearest whole number using equation:

$$(\text{mean enterococci density}) = \text{antilog}_{10} \frac{\text{illness rate/1000 people} - 0.20}{12.17}$$

(4) Single sample limit = antilog$_{10}$

$$\left| \begin{array}{l}\log_{10} \text{ indicator} \\ \text{geometric mean} \\ \text{density/100 ml}\end{array} + \left\{ \begin{array}{l}\text{Factor determined from} \\ \text{areas under the Normal} \\ \text{probability curve for} \\ \text{the assumed level of} \\ \text{probability}\end{array} \times \begin{array}{l}(\log_{10} \text{ standard} \\ \text{deviation})\end{array} \right\} \right|$$

The appropriate factors for the indicated one sided confidence levels are:

75% C.L. - .675
82% C.L. - .935
90% C.L. - 1.28
95% C.L. - 1.65

(5) Based on the observed log standard deviations. During the EPA studies: 0.4 for freshwater *E. coli* and enterococci; and 0.7 for marine water enterococci. Each jurisdiction should establish its own standard deviation for its conditions which would then vary the single sample limit.

EPA Criteria for Bathing(Full Body Contact) Recreational Waters

Freshwater

Based on a statistically sufficient number of samples (generally not less than 5 samples equally spaced over a 30-day period), the geometric mean of the indicated bacterial densities should not exceed one or the other of the following:[1]

_E_. _coli_	126 per 100 ml; or
enterococci	33 per 100 ml;

no sample should exceed a one sided confidence limit (C.L.) calculated using the following as guidance:

designated bathing beach	75% C.L.
moderate use for bathing	82% C.L
light use for bathing	90% C.L.
infrequent use for bathing	95% C.L.

based on a site-specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.4 as the log standard deviation for both indicators.

Marine Water

Based on a statistically sufficient number of samples (generally not less than 5 samples equally spaced over a 30-day period), the geometric mean of the enterococci densities should not exceed 35 per 100 ml;

no sample should exceed a one sided confidence limit using the following as guidance:

designated bathing beach	75% C.L.
moderate use for bathing	82% C.L.
light use for bathing	90% C. L.
infrequent use for bathing	95% C. L.

based on a site-specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.7 as the log standard deviation.

Note (1) - Only one indicator should be used. The Regulatory agency should select the appropriate indicator for its conditions.

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EPA 823-R-07-006

REPORT OF THE EXPERTS SCIENTIFIC WORKSHOP ON CRITICAL RESEARCH NEEDS FOR THE DEVELOPMENT OF NEW OR REVISED RECREATIONAL WATER QUALITY CRITERIA

Airlie Center
Warrenton, Virginia
March 26-30, 2007

U.S. Environmental Protection Agency
Office of Water
Office of Research and Development

June 15, 2007

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DISCLAIMER

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Appendix B includes a complete list of all persons who attended the workshop and their affiliations and contact information, including workshop experts, EPA resource personnel, and meeting support personnel (note takers, meeting facilitator, logistics staff).

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TABLES AND FIGURES

ACRONYMS

7Q10	the lowest streamflow for 7 consecutive days that occurs on average once every 10 years
AWQC	ambient water quality criteria
ASABE	American Society of Agricultural and Biological Engineers
ATP	adenosine triphosphate
BEACH Act	Beaches Environmental Assessment and Coastal Health Act of 2000
BMP	Best Management Practices
CAFO	concentrated animal feeding operation
CDC	Centers for Disease Control and Prevention
cfu	colony forming unit
CSO	combined sewer overflow
CWA	Clean Water Act
DNA	deoxyribonucleic acid
DU	designated use
EHEC	enterohemorrhagic *E. coli*
EPA	U.S. Environmental Protection Agency
EU	European Union
FDA	U.S. Food and Drug Administration
FFU	focus forming units
GC/MS	gas chromatography-mass spectrometry
GI	gastrointestinal
GIS	geographic information systems
GM	geometric mean
HACCP	Hazard Analysis Critical Control Point
HEV	hepatitis E virus
HIV/AIDS	human immunodeficiency virus/acquired immune deficiency syndrome
HPLC	high performance liquid chromatography
HSPF	Hydrological Simulation Program-Fortran
ILSI	International Life Sciences Institute
IMS	immunomagnetic separation
ISO	International Organization for Standardization
L	Liter
mL	Milliliter
MPN	Most Probable Number
MST	microbial source tracking
NASBA	nucleic acid sequence based amplification
NEEAR	National Epidemiological and Environmental Assessment of Recreational Water Study
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NPS	non-point source program
NWS	National Weather Service
PCR	polymerase chain reaction
pfu	plaque forming unit

POP	probability of precipitation
POTW	publicly owned [wastewater] treatment works
QMRA	quantitative microbial risk assessment
qPCR	quantitative polymerase chain reaction
RCT	randomized controlled trial
RDS	relative standard deviation
RMSE	root mean square error
RNA	ribonucleic acid
SAFE	Swimming Advisory Forecast Estimate
SCCWRP	Southern California Coastal Water Research Project
SD	standard deviation
SETAC	Society of Environmental Toxicology and Chemistry
SRC	sulphite-reducing clostridia
SSM	single sample maximum
SSO	sanitary sewer overflow
STEC	shiga-toxin producing *E. coli*
SWMM	Storm Water Management Model
TMA	transcription-mediated amplification
TMDL	total maximum daily load
UAA	Use Attainability Analysis
U.K.	United Kingdom
URI	upper respiratory illness
U.S.	United States
USGS	U.S. Geological Survey
UV	ultraviolet light
WQS	Water Quality Standards
WHO	World Health Organization (United Nations)

INTRODUCTION

Purpose of the Workshop

Since the U.S. Environmental Protection Agency (hereafter EPA or the Agency) last published recreational water quality criteria in 1986, there have been significant advances, particularly in the areas of molecular biology, microbiology, and analytical chemistry. EPA believes that that these new scientific and technical advances need to be factored into the development of new or revised Clean Water Act (CWA) Section 304(a) criteria for recreation. To this end, EPA has been conducting research and assessing relevant scientific and technical information to provide the scientific foundation for the development of new or revised criteria. The enactment of the Beaches Environmental Assessment and Coastal Health (BEACH) Act of 2000 (which amended the CWA) required EPA to conduct new studies and issue new or revised criteria, specifically for Great Lakes and coastal marine waters.

From March 26 through 30, 2007, EPA convened a group of 43 national and international technical, scientific, and implementation experts from academia, numerous states, public interest groups, EPA, and other federal agencies, at a formal workshop to discuss the state of the science on recreational water quality research and implementation.

The purpose of the workshop was for EPA to obtain individual input from members of the broad scientific and technical community on the "critical path" research and science needs for developing scientifically defensible new or revised CWA §304(a) recreational ambient water quality criteria (AWQC) in the near-term. Near-term needs were defined as specific research and science activities that could be accomplished in 2 to 3 years so that results are available to EPA in time to support the development of new or revised criteria. The new or revised criteria, which would be available from EPA in roughly 5 years (2012), must be scientifically sound, protective of the designated use, implementable for broad CWA purposes, and when implemented, provide for improved public health protection. (See Appendix A for the full charge to the experts.) The Agency wants to develop this new or revised criteria in a highly participatory framework within the next 5 years based on the best available science.

Workshop Design

The *Experts Scientific Workshop on Critical Research and Science Needs for the Development of New or Revised Recreational Water Quality Criteria* was designed to be similar in organization and format to the Society of Environmental Toxicology and Chemistry (SETAC) Pellston Workshops, where technical experts in a particular subject area are invited to participate and evaluate current and prospective environmental issues. A Pellston-type workshop typically brings together between 40 to 50 technical experts from academia, business, government, and public interest groups. Experts are semi-sequestered for up to a week to facilitate focused discussions and individual and collaborative writing of a draft summary report by the end of the workshop. Subject leaders are then responsible for consolidating, editing, producing, and distributing the final (formal) workshop proceedings.

Participant Affiliation Balance

In addition to U.S. and international experts drawn from academia, public interest groups, and numerous state and other federal agencies, EPA selected several experts from within EPA to serve in the workgroups (see Appendix B for participant list). The 43 experts serving in 7 subject areas were supported by a total of 9 EPA resource personnel, 10 note takers, 3 logistics contractors, and a professional facilitator. The proper balance between EPA presence and outside experts was crucial for keeping the discussions on track with EPA's needs from the workshop while providing ample opportunity for the external experts to voice their opinions and intellectually explore topics of interest to EPA.

Agenda Overview

The workshop began on Sunday evening, March 25, 2007, with a logistics meeting for the workgroup chairs, EPA staff, and note takers. The plenary sessions on Monday served to orient participants regarding CWA §304(a) AWQC and EPA's needs from the workshop discussions and these proceedings. Monday afternoon the seven workgroups met for the first time to discuss interpretation of the charge questions (Appendix A). On Tuesday, all workshop participants met in a plenary session, which was followed by workgroup sessions throughout the day. The agenda facilitated and encouraged the workgroups to meet with each other to discuss common and overlapping issues. At the end of the day the workshop participants met again in plenary to hear report-outs from each workgroup chair that described their progress for the day.

Because the seven workgroup topics have many overlapping issues, it was important for the groups to communicate as needed so they could both stay informed of and build on each other's discussions. In addition to several joint breakout sessions, the workgroup chairs also shared all of their meals to discuss ongoing progress. On Wednesday, the workshop participants met once again in a plenary session to discuss overall progress followed by workgroup breakout sessions where each group continued discussions and began writing a draft workgroup report. The workgroups continued writing on Thursday. Friday morning, each workgroup turned in a 10 to 20 page draft report and their respective chairs provided an overview of each report regarding the major themes discussed and critical research needs in a final plenary session.

Seven Workgroup Topics

The seven workgroup topics are presented in seven chapters in this report. The relationships between these and other topics are graphically represented in Figure 1. In Figure 1 shaded boxes correspond to the seven workgroups. The alternatives boxes in Figure 1 refer to various possible indicators that a toolbox approach could provide for each of the CWA applications. The charge questions helped the workgroups to define the scope of their discussions. The experts were asked to provide their individual insights on the state of the science as well as critical path research that could be completed by EPA in the next 2 to 3 years. A short description of each workgroup and the tasks EPA asked them to discuss follows.

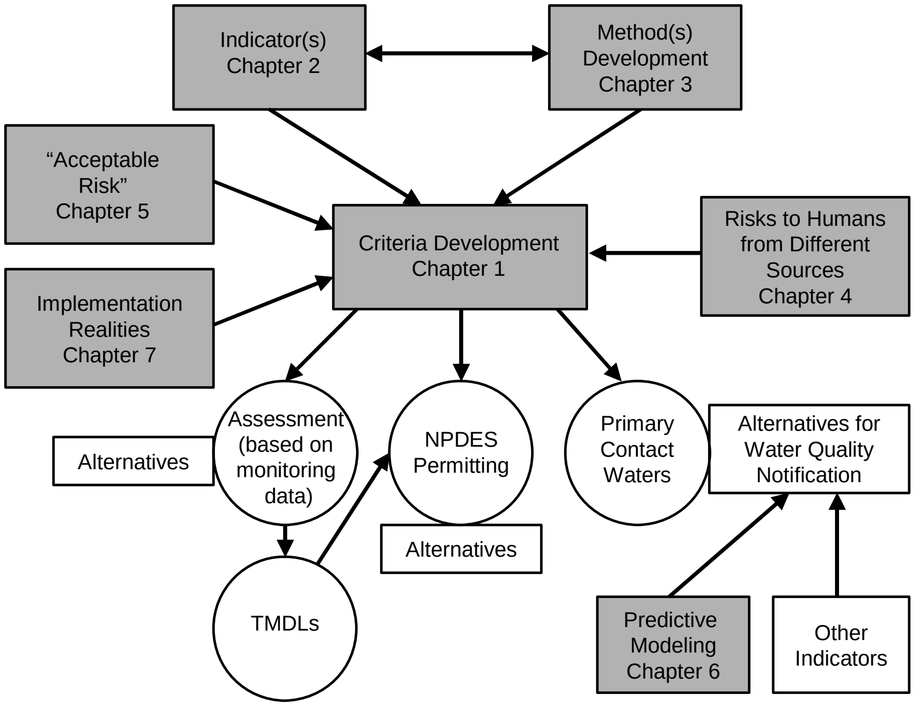

Figure 1. Flow Diagram of How the Workgroup Components Contribute to the Development of New or Revised Recreational Water Quality Criteria.

1. **Approaches to Criteria Development** – focus on a toolbox approach as well suggest other potential approaches for new or revised criteria development.
2. **Pathogens, Pathogen Indicators, and Indicators of Fecal Contamination –** discuss the strengths and limitations of indicators of fecal contamination, pathogen index microorganisms, and specific pathogens for development of new or revised recreational AWQC.
3. **Methods Development –** discuss methods for quantifying indicators and pathogens, such as culture-based methods, molecular-based methods (e.g., quantitative polymerase chain reaction [qPCR]), and faster culture-based methods and their applicability for AWQC.
4. **Comparing Risks to Humans from Different Sources** – discuss the relative risks of illness to humans in waters contaminated with human fecal material versus animal fecal material.
5. **Acceptable Risk** – discuss the level of risk to various populations that would be associated with numeric AWQC. EPA was interested in the science necessary to inform the policy decision regarding the target risk range and the process through which the policy decision could be reached.
6. **Modeling Applications to Criteria Development and Implementation** – discuss predictive modeling approaches and their potential applications in implementation of AWQC.

7. **Implementation Realities –** identify and consider factors that influence implementation of criteria for each of the CWA uses (beach monitoring and notification, development of National Pollutant Discharge Elimination System [NPDES] permits, assessments to determine use attainment, and development of total maximum daily loads [TMDLs].

Background

Clean Water Act §304(a) Recommended Criteria

What are EPA's Recommended §304(a) Criteria?

CWA §304(a) AWQC are (typically) expressed as numeric concentrations of pollutants. These are essentially the numbers that EPA recommends that States and Tribes adopt in setting their own Water Quality Standards (WQS) to protect waters for specified designated uses. State and Tribal WQS, once approved by EPA, are the effective standards used in CWA regulatory and non-regulatory programs. Figure 2 provides an overview of CWA WQS.

States and Tribes classify waters by their designated use,[1] which includes "primary contact recreation." States and Tribes typically define primary contact recreation to encompass recreational activities that could be expected to result in the ingestion of, or immersion in, a waterbody (such as swimming, water skiing, surfing, or any other recreational activity where ingestion of, or immersion in, the water is likely).

CWA §304(a):

- AWQC often are described as concentrations in the water column and generally have a time and duration component.
- AWQC could be expressed as an annual average concentration that should not be exceeded; a daily value or seasonal concentration that should not be exceeded; or a value that should not be exceeded, on average, more than one time every 3 years (for acute aquatic life criteria).
- AWQC are often associated with EPA-approved analytical methods. This is partly because without EPA-approved methods to measure concentrations in effluent, States are reluctant to adopt criteria in WQS that are then used in NPDES permits (see more below).

States typically adopt the recommended criteria into their WQS (i.e., regulations promulgated using state rulemaking processes [similar to Federal regulation development]).

What do States do with these EPA-recommended Numbers and how are they used by States?

Increasingly, because of the dynamics of State rulemaking processes and public and regulated community involvement, States are reluctant to adopt EPA's recommended criteria unless the

[1] CWA designated use (DU) classifications are narrative statements describing appropriate intended human and/or aquatic life and other quality objectives for waterbodies.

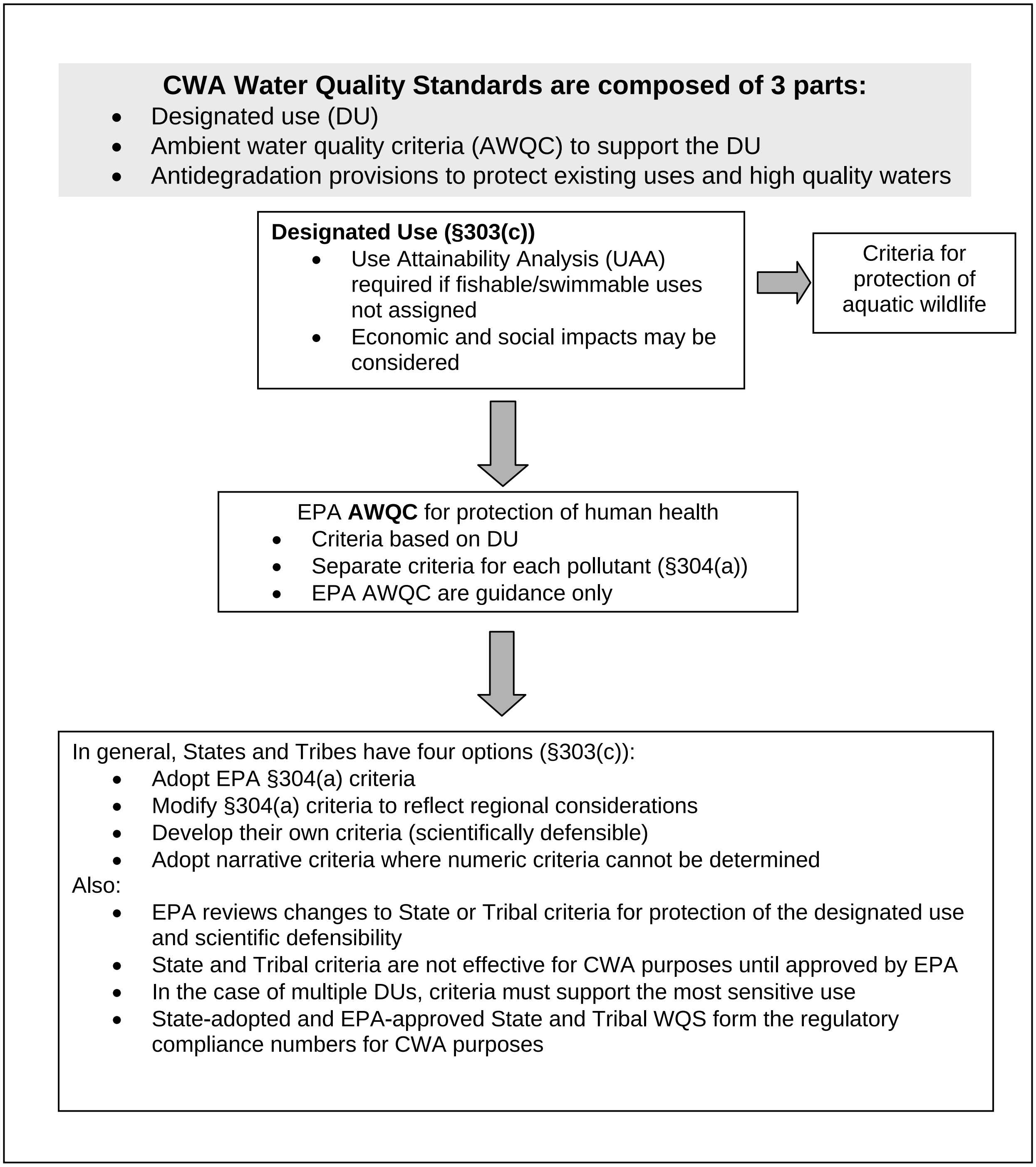

Figure 2. Clean Water Act: Water Quality Standards Overview.

underlying science supports the desired environmental result *and* the criteria can be implemented for all aspects of their CWA Programs.

Under CWA §304(a)(9), EPA is required to publish new or revised water quality criteria for pathogens and pathogen indicators (including a revised list of testing methods, as appropriate) for the purpose of protecting public health in coastal recreation waters. Coastal recreation waters are marine and Great Lake waters designated by States for use for swimming, bathing, surfing, or similar water contact activities. Under CWA §303(i)(1)(B), States are then required to adopt new or revised WQS for those pathogens and pathogen indicators for which EPA's new or revised criteria have been developed. States must submit these standards to EPA for approval or disapproval. EPA approves the standards if they are scientifically defensible and protective of the designated use.

Once approved, State WQS become effective for CWA purposes. This means that the State-adopted §304(a) criteria become regulatory standards and are used for several different CWA purposes, including the following:

- **§303(d) listings.** Under §303(d) of the CWA, States prepare lists of waters that are impaired and need TMDLs; States develop the lists every 2 years and submit them to EPA for approval. If States determine that waters are not meeting applicable water quality standards (whether from point or non-point sources of pollution), States are to identify those waters as "impaired" under §303(d).
- **TMDL calculations** for impaired waters must be prepared to implement the applicable State WQS.
- **NPDES permits,** which are issued after State WQS are in place for a pollutant, must have discharge limits as stringent as necessary to meet such WQS. EPA's analytical methods are often used to measure compliance with permit limits.
- **Public Notification at Beaches.** Under the BEACH Act of 2000, eligible coastal and Great Lakes States may apply for and receive BEACH Act grants for their beach monitoring and public notification programs. Those States use their recreational contact WQS to determine whether to close an area for swimming or issue a swimming advisory.

Toolbox Approach

EPA's recommended AWQC have to be applicable at a national level. A toolbox approach is under consideration because of the potential for greater flexibility in selecting situationally-appropriate indicators/methods and increased options for implementation, which is desirable for nationally applicable criteria. A toolbox allows for the use of varied techniques and approaches to achieve public health protection.

A preliminary working definition of the Toolbox approach for recreational water quality criteria might be the following:

> The toolbox approach is a set of potential microbiological (i.e., a microbe plus a specified enumeration method) and/or physico-chemical assays that could be employed alone, or in certain combinations, to protect and restore the recreational use of waters. The contents of the toolbox (the "tools") would be used by State public health and water quality agencies for beach advisory/closing program purposes and for all other Water Quality Standard related regulatory purposes under the CWA. The level of risk (or public health protection) would be the same regardless of which tool is used.

Although the toolbox concept allows a context for considering feasibility and applicability of different indicator and method combinations in developing new or revised recreational criteria under CWA §304(a), it is critical that there is an understanding of the relationship among the different methodologies for proper implementation of the criteria. For example, if EPA recommended one type of indicator for one set of uses (e.g., culturable enterococci) and also recommended the use of a DNA-based method (e.g., enterococci qPCR) for other uses, then there would have to be an understanding of the meaning of those multiple measures (i.e., linkage) in the context of the overall CWA §304(a) program. Without a clear understanding of the linkage and context of different methods the entire "toolbox" concept becomes unmanageable from a regulatory perspective.

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CHAPTER 1
APPROACHES TO CRITERIA DEVELOPMENT

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1.1 Benchmarks for Criteria Development

The workgroup was charged with answering 21 questions and providing a range of alternatives for the development of new or revised national recreational ambient water quality criteria (AWQC; see Section 1.5 for summary response). The following six potential approaches that could be used or adapted for an approach to develop new or revised criteria were initially discussed: (1) EPA's 1986 approach, (2) World Health Organization (WHO), (3) European Union (EU), (4) Hazard Analysis and Critical Point Analysis (HAACP), (5) Heal the Bay's Beach Report Card, and (6) EPA's Air Quality Index. The workgroup members concentrated the discussions on the three approaches that were deemed most appropriate for consideration in the context of Clean Water Act (CWA) Section 304(a) ambient water quality criteria (AWQC), namely, the WHO approach (with possible modifications), the EU approach (adopted 2006), and a modified version of EPA's 1986 criteria. Before the workgroup defined the approaches and determined the potential application of the three alternative approaches, workgroup members agreed that it was critical to identify desirable attributes or benchmarks for the criteria. The benchmarks or attributes that were identified are summarized below.

1. The criteria are health-based. The workgroup demonstrated a preference that the criteria be as directly as possible anchored to health effects demonstrated in epidemiology studies.
2. The criteria should demonstrate utility for and be compatible with all of the CWA §304(a) criteria (as amended by the Beaches Environmental Assessment and Coastal Health Act of 2000 [BEACH Act]) needs, including water quality assessment for public notification at beaches in a timely manner, assessment for impaired waters listings, development of total maximum daily load (TMDL) development and implementation, and development of National Pollution Discharge Elimination System (NPDES) permits.
3. The criteria should be scientifically defensible for application in a wide variety of geographical locations (climatic conditions), including fresh and marine waters, and temperate, subtropical, and tropical waters.
4. The criteria be sufficiently robust and flexible so that they can be configured to protect the public health of those exposed to recreational water impacted by sewage effluent, concentrated animal feed operation (CAFO) contaminated runoff, non-point sources (e.g., agriculture [non-CAFO], urban runoff) and waters not impacted by anthropogenic sources.
5. The criteria should be sufficiently robust and flexible so that they can be configured to provide regulators the ability to protect susceptible (sensitive) subpopulations such as children and immunocompromised individuals. Commonality was found among workgroup members that protecting the health of children was of paramount concern.
6. The criteria are associated (linked) with analytical methods that are reliable, robust, and provide reproducible results.
7. The criteria should protect primary contact recreation in freshwaters, marine waters, temperate, subtropical, and tropical waters equally. Similarly, the criteria should provide equal protection those exposed to effluent, urban runoff, and/or non-point source runoff impacted waters via primary contact recreation.

The workgroup members agreed that all seven of the above attributes are critical considerations for criteria development. In assessing the potential application of each of the proposed alternatives, it is important to keep in mind that criteria applied to these alternatives are assumed to be consistent with all of the above attributes (or at least most of them) before the final frameworks and criteria are developed. The likelihood that some of these attributes will not be met in the near-term seems to make the WHO or EU approaches more suitable for implementation.

The workgroup expressed the opinion that EPA should release the new or revised criteria and implementation guidance concurrently to provide clarity to States on how the criteria should be used for regulatory and public notification needs.

1.2 Integration of Workshop Components

A summary of the interactions between the various subject areas addressed in this workshop is presented in Figure 3. In Figure 3 shaded boxes correspond to the seven workgroups. The alternatives boxes in Figure 3 refer to various possible indicators that a toolbox approach could provide for each of the CWA applications. Briefly, the Pathogen/Pathogen Indicator workgroup proposes indicators that may have utility for criteria development (see Chapter 2). In doing so, they consulted with the Methods Development workgroup members (see Chapter 3) to assure that validated methods are or could be available and usable for the implementation of the proposed parameter. Different methods have different specificities for identifying whether the source of fecal contamination is human- or animal-based. The Comparing Risks workgroup provided information on the relative risks to human health from different sources of fecal contamination (Chapter 4). Once identified, the pathogen/pathogen indicator and the associated method are used during the criteria development process. Another critical component in the criteria development process is the identification of a risk level. Information from the Acceptable Risk workgroup (see Chapter 5) on how to develop "acceptable risk" thresholds is used in this context during the criteria development process. The Modeling workgroup discussed how predictive modeling can be used to inform criteria approaches and to provide information on water quality notification (Chapter 6). Once these pieces were integrated, an initial check was conducted against the suggestions and concerns of the Implementation Realities workgroup (see Chapter 7) members to help ensure that the potential for criteria development does not conflict with actual "on the ground" implementation.

As discussed in the Introduction to these proceedings, recreational AWQC are used for a number of purposes. First, these criteria are used to make assessment determinations under CWA §305(b) and §303(d).[2] Within this regard and depending on the framework, a number of alternate indicators or methods may be used to assist in making the determination as to the overall quality of a waterbody and the compliance with the underlying criteria. Second, these criteria are used to determine permit limits for NPDES permit holders and for TMDL purposes. Finally, these criteria are used to determine the acceptability of the water for direct primary contact recreation. Conceptually, alternative indicators, including models, could also be used for these purposes.

[2] http://www.epa.gov/owow/tmdl/tmdl0103/

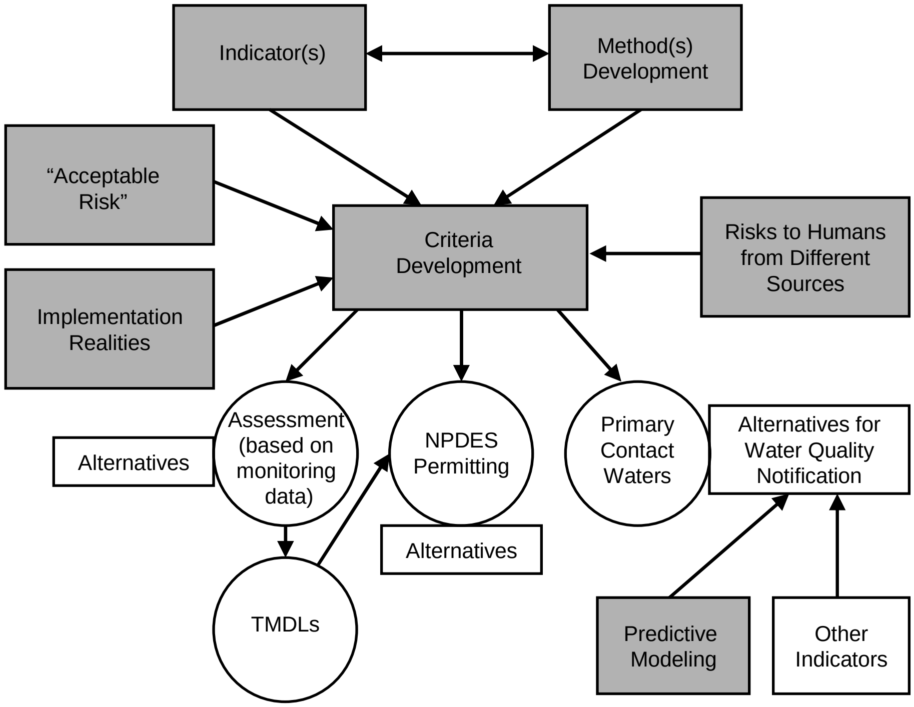

Figure 3. Flow Diagram of How the Workgroup Topics Contribute to the Development and Implementation of New or Revised Recreational Water Quality Criteria.

1.3 Summary of Currently Available Water Quality Criteria Setting Approaches

The three alternatives that were identified by the workgroup were a modified WHO approach, the EU approach, and a modified version of the EPA 1986 approach because all of these approaches are largely based on peer reviewed epidemiology studies and some version of each of these frameworks are in use currently in at least one country.

Workgroup members generally agreed that all three criteria development approaches are feasible providing the criteria meet the becnhmarl/attributes listed above. Although the workgroup briefly discussed other approaches such as the EPA's Air Quality Index, HAACP, and Heal the Bay's Beach Report Card approaches, none of these approaches were deemed to be appropriate for the desired purposes for a variety of reasons, including lack of applicability for criteria development.

1.3.1 WHO Approach for Water Quality Criteria Setting

The WHO has been concerned with health issues associated with recreational water environments for many years and has published several influential reports that represent a well accepted view among international experts (Prüss, 1998). The WHO approach provides a basis for standard setting in light of local and regional circumstances, such as the nature and seriousness of local endemic illness, exposure patterns, and competing health risks that are not associated with recreational water exposure.

The WHO approach is based on the perspective that recreational water quality and protection of public health are best described by a combination of sanitary inspection and microbial water quality assessments (the WHO [2003] Guidelines use enterococci as the fecal indicator of choice; see Table 1). This approach considers possible sources of pollution in a recreational water ("sanitary inspection category" in Table 1), as well as observed levels of fecal pollution ("microbial water quality assessment category" in Table 1), and combines them into a five-level classification scheme for recreational water environments. To date, the classification system has been used primarily to "grade" recreational waters and to provide an assessment for regulatory compliance purposes. This approach however, also could be adapted for other CWA §304(a) applications such as NPDES permitting and TMDL development.

The microbial water quality assessment criteria are based on a banded system, where the band divisions are equivalent to a risk of acquiring gastrointestinal (GI) illness for (A) <1 case in 100 exposures, (B) <1 case in 20 exposures, (C) <1 case in 10 exposures, and (D) >1 case in 10 exposures. The 95th percentile value was selected as an appropriate descriptor of the microbial probability density function because it is easily understood to be the probability of encountering

Table 1. WHO Classification Matrix for Integrating Microbial Water Quality as Measured by Enterococci Density with Sanitary Inspection Category.

		Microbial Water Quality Assessment Category (95th 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]	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]	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.

SOURCE: WHO, 2003.

polluted water and focuses on water quality that is likely to cause illness (i.e., greater probability of illness associated with increasing density of human sources of fecal pollution). The WHO levels of risk for the bands described above were selected based on a series of science policy decisions in consultations with numerous international experts and were intended to be reasonable for both the developed and developing world. The expectation in the United States is that the "acceptable risk" (see Chapter 5) levels would be similar or more protective than the risk levels adopted by other developed countries. The methodology used to derive the 2003 WHO Guideline values is summarized in Appendix C.

The sanitary inspection category is intended to classify the risk of illness caused by fecal pollution in a recreational waterbody, although human fecal pollution will tend to drive the overall sanitary inspection category derived for an area. WHO experts believe that the three most important sources of human fecal contamination of recreational water environments for public health purposes are typically sewage, riverine discharges, and direct contamination from bathers. Sanitary inspections are required to address those sources as well as others, and inspections should take on a tiered approach, dependent on the level of perceived risk and its uncertainty. For example, if human and domestic fecal pollution is considered low based on land uses, but fecal indicator counts are relatively high, further exploration of the source(s) and their relative risks would be recommended. This higher level of examination (tier) may utilize more expensive methods and approaches and further cycles (tiers) of investigation as necessary. Based on the results of the sanitary inspections, recreational waters are ranked (from very low to very high) with respect to evidence for the degree of influence of fecal material.

1.3.2 EU Approach for Water Quality Criteria Setting

The EU broadly adopted the 2003 WHO Guidelines in formulating the 2006 Bathing Water Directive. A summary of the European Commission Directive is provided in Appendix D. The approach incorporates the following fundamental elements:

- The EU starts with the WHO risk assessment framework, but does not include the sanitary inspection category information for the purposes of recreational water classification. Instead, it uses only the microbial water quality assessment information to characterize the probability of exposure to human pathogens.
- The EU approach used the WHO microbiological criteria for marine waters and applied the same risk assessment framework to new epidemiological data to derive standards for fresh recreational waters.
- The EU approach allows sample discounting. Under discounting, numeric excursions above the water quality standards that are predicted and/or measured do not count against the waterbody for compliance determination (i.e., such values are discounted from the data set prior to calculation of the 95th percentile, but only 15% of scheduled samples can be so discounted). Sample discounting is allowed when a predictive model, source reduction plan, and communication management system are in place to inform the public about short-term pollution events derived during predictable conditions (e.g., rainfall).

The EU Bathing Water Directive (7/EU/EEC; dated February 15, 2006) is currently being translated by Member States for implementation (EP/CEU, 2006). The Directive establishes separate numerical microbiological criteria for fresh (inland) and marine (coastal and transitional) bathing waters for the 24 EU Member States (Table 2). The numerical values are based on epidemiological studies reported by Kay et al. (1994) and Wiedenmann et al. (2006)—the former was used by WHO in formulating their Guidelines (Kay et al., 2004; WHO, 2003).

Table 2. Numerical Microbiological Water Quality Assessment Classification for Fresh (Inland) and Marine (Coastal and Transitional) Bathing Waters for the 24 EU Member States.

Inland (Fresh) Waters			
Indicator	Excellent	Good	Sufficient
(Intestinal) enterococci (cfu/100 mL)	200*	400*	360**
E. coli (cfu/100 mL)	500*	1,000*	900**
Coastal and Transitional (Marine) Waters			
Indicator	Excellent	Good	Sufficient
(Intestinal) enterococci (cfu/100 mL)	100*	200*	200**
E. coli (cfu/100 mL)	250*	500*	500**

Notes: * = Based on a 95th percentile evaluation; ** = Based on a 90th percentile evaluation to reduce the risk of statistical anomalies when using a small data set, which also allows lower limit values for enterococci and *E. coli* densities in inland waters to be classified as sufficient versus good microbiological water quality.
Source: Adapted from EP/CEU (2006).

1.3.3 EPA 1986 Water Quality Criteria Setting

In the late 1970s and early 1980s, EPA conducted public health studies evaluating several organisms as possible indicators, including total and fecal coliforms, *E. coli*, and enterococci. The studies showed that enterococci and *E. coli* are the best predictors of GI illness (gastroenteritis) in sewage effluent-impacted freshwaters, while enterococci were the best predictor in sewage-impacted marine waters. Gastroenteritis describes a variety of diseases that affect the GI tract and are rarely life-threatening; self-limiting symptoms include nausea, vomiting, stomachache, diarrhea, headache, and fever. Based on these studies, EPA published a criteria document, *Ambient Water Quality Criteria for Bacteria – 1986*, recommending the use of these bacterial indicators in ambient water quality criteria values for the protection of primary contact recreation (US EPA, 1986). Table 3 summarizes the *Water Quality Standards for Coastal and Great Lakes Recreation Waters Rule* (US EPA, 2004) that requires States and Tribes to adopt the 1986 AWQC for Bacteria.

States and Tribes generally define their designated use of "primary contact recreation" to encompass recreational activities that could be expected to result in the ingestion of, or immersion in, water, such as swimming, water skiing, surfing, or any other recreational activity where ingestion of, or immersion in, the water is likely.

EPA derived standards that implied an acceptable excess illness probability of 0.8% in swimmers exposed in freshwater and 1.9% in swimmers exposed in marine waters. EPA's 1986 bacteria criteria document indicates the illness rates are "only approximate" and that the Agency based the 1986 values that appear in Table 3 on these approximations.

Table 3. Summary of EPA's 1986 Recommended Water Quality Criteria for Bacteria and 2004 Rule

Indicator	Swimming-Associated Gastroenteritis Rate per 1,000 Swimmers	Geometric Mean	Single Sample Maximum Allowable Density			
		Steady State Geometric Mean Indicator Density	Designated Beach Area	Moderate: Full Body Contact Recreation	Lightly Used: Full Body Contact Recreation	Infrequently Used: Full Body Contact Recreation
Freshwater						
Enterococci	8	33	61	78	107	151
E. coli	8	126	235	298	409	575
Marine Water						
Enterococci	19	35	104	158	276	501

Source: US EPA (2004).

EPA's 1986 bacteria AWQC document provides geometric mean densities as well as four different single sample maximum values (representing values below which an increasing percentage of single values are expected to fall if the geometric mean of samples from the waterbody is equal to the geometric mean criteria). The 1986 bacteria AWQC document categorizes the single sample maximum values based levels of beach usage as follows: "designated bathing beach" for the 75% (most conservative) confidence level, "moderate use for bathing" for the 82% confidence level, "light use for bathing" for the 90% confidence level, and "infrequent use for bathing" for the 95% confidence level. The lowest confidence level corresponds to the highest level of protection.

In the 1986 AWQC context, single sample maximum criteria are water quality assessment tools that provide a sense of when the water quality in a waterbody is not consistent with the AWQC. Insights based on single observations are very difficult because of the expected variability of fecal indicators. For instance, if the long-term geometric mean concentration of enterococci in the water at a marine beach is 35/100 mL and the log standard deviation is 0.4, then there is an 18% chance that the concentration of enterococci in a single sample would be over 158/100 mL. The higher the single sample maximum, the lower the probability that a single sample exceeding that value would occur as part of the normal random variability of samples (US EPA, 2006).

Since publication of the 1986 criteria, many States have expressed concern that the current fecal indicator/illness rate relationships identified in the epidemiology studies leading up to the 1986 criteria are not appropriate or representative of all U.S. waters. For example, States have concern that the most appropriate indicator in tropical waters may be different than in temperate waters, and that appropriate levels of indicators may be different in waters where human fecal waste predominates animal waste. Other identified issues are as follows:

- lack of clear, timely, and flexibile guidance regarding use of the single sample maximum values and differing risk levels;
- no EPA-approved analytical methods for use in wastewater for the indicator bacteria;

- lack of data to correctly assess the applicability of the 1986 bacteria criteria to flowing waters; and
- lack of data to quantify the risk associated with contributions from nonhuman sources of fecal contamination as well as lack of flexibility to adjust the criteria for water bodies that do not receive human sources of fecal contamination.

1.3.4 Summary of Proposed Criteria Development Approaches

Workgroup members developed a summary of the three proposed criteria development approaches, including strengths and limitations (Tables 4a and 4b).

Table 4a. Summary of Proposed Criteria Development Approaches: Strengths and Limitations

Criteria Approach	Science Supporting Approach	Strengths	Limitations
World Health Organization (WHO, 2003)	Fleisher et al., 1996 Kay et al., 1994, 2004 WHO, 1999 Wyer et al., 1999	• Flexible • Most comprehensive of available methods • Adopted by other countries • Incentives for beaches to upgrade • Allows more site appropriate protection of health	• Sanitary inspection component is qualitative; not quantitative • Greatest data needs • Would need to adapt potentially complex system to wide range of conditions in U.S. • Potential implementation issues
European Union (EP/CEU, 2006)	Fleisher et al., 1996 Kay et al., 1994, 2004 Wiedenmann et al., 2006 WHO, 1999, 2003	• Flexible • Relatively straightforward • Incentives for beaches to upgrade • Adopted by other EU Member States	• Discounting system has no direct precedent in the U.S. • Would need to devise robust and acceptable discounting scheme • Potential implementation issues
Current U.S. Criteria (US EPA, 1986)	US EPA, 1983, 1984	• Relatively straightforward • Currently in place in most states, new implementation issues less likely • Fewest data requirements	• Allows less flexibility • Single sample max (75^{th} percentile) has been criticized from implementation perspective • Credibility concerns in many parts of the U.S.

Table 4b. Summary of Three Proposed Criteria Development Approaches: Benchmarks

Criteria Approach	Criteria Attribute	Approach Compatible with Attribute
World Health Organization (WHO, 2003)	Health-based	Yes
	CWA §304(a) applications	Most challenging – unclear how different grades for beaches would be interpreted with respect to impaired waters; for example, TMDLs would need to be considered.
	Geographic variability	Not with current indicator, ongoing research could fill gaps
	Point vs. non-point	No, epidemiological data would be needed
	Multiple subpopulations	Could be, but in current configuration children not analyzed separately
	Uniform risk across waterbody types	Yes
	Linked to method that is validated	Yes currently, but will also depend on future indicators
European Union (EP/CEU, 2006)	Health-based	Yes, but differential risks from different sources of fecal contamination is not included, thus, this approach is less health-based than WHO approach
	CWA §304(a) applications	Yes, but challenging for same reasons as WHO approach
	Geographic variability	Not with current indicator, ongoing research could fill gaps
	Point vs. non-point	No
	Multiple subpopulations	Could be, but in current configuration children not analyzed separately
	Uniform risk across waterbody types	Yes
	Linked to method that is validated	Yes currently, but will also depend on future indicators
Current U.S. Criteria (US EPA, 1986)	Health-based	Yes, but concern about single sample standard, also concerns that differential risks from different sources of fecal contamination are not included
	CWA §304(a) applications	Yes
	Geographic variability	No
	Point vs. non-point	No
	Multiple subpopulations	No
	Uniform risk across waterbody types	No, fresh and marine recreational waters have different “acceptable risks”; this could be addressed in new or revised criteria
	Linked to method that is validated	Yes currently, but will also depend on future indicators

1.3.5 Other Approaches Considered

As noted previously, the workgroup considered a number of other frameworks and approaches that might be applicable to criteria development, including the following:

- Hazard Analysis and Critical Control Point Principles (HACCP);
- Heal the Bay Beach Report Card; and
- EPA Air Quality Index.

The EU, EPA (1986) criteria, and the WHO approaches are already being used for the intended purpose, either in the United States or other countries. The other possible approaches listed above have not been applied in a regulatory framework for proposed water regulation and would need to be thoroughly assessed to determine their utility or applicability to derive recreational water quality criteria. The workgroup members felt that it was beyond their ability to conduct such an assessment at this time. One workgroup member noted that the Heal the Bay approach was never intended for use in all regulatory purposes and would not be recommended for such.

1.4 Summary of Critical Issues to be Resolved in Applying Available Water Quality Criteria Approaches

No matter which recreational water quality criteria development approach is selected, a number of research needs have to be met before criteria development can reach completion. Additional epidemiological studies that take into account marine waters, subtropical and tropical waters, urban runoff, and non-point sources of contamination will need to be completed in the next 2 to 3 years to provide the health effects data necessary if nationally applicable are to be developed. Further testing of quantitative polymerase chain reaction (qPCR) methods to detect enterococci and/or any additional proposed indicators under the conditions listed above also is critical. The epidemiological studies should also include (1) culture-based methods in addition to molecular methods for enterococci; (2) culture and molecular-based methods for *E. coli* in fresh water studies because national freshwater criteria and numerous States currently use *E. coli* in recreational criteria and including *E. coli* would maintain a level of consistency with the existing CWA §304(a) guidance; and (3) sensitive subpopulations to the extent feasible, including children at a minimum.

Other research gaps that can be filled in the next few years include, but are not limited to, fate and transport of molecular-based indicator organisms in wastewater treatment plants and in the ambient aquatic environment. Workgroup members expressed a significant concern about the issue of conservation of measurable genetic material throughout the treatment process because of the regulatory ramifications in the NPDES, water quality assessment, and TMDL programs of moving to molecular-based criteria.

Another research need is for effective predictive models for beach water quality forecasting to notify the public of the potential health risks of recreational water contact. The current use of single sample assessments using culture-based methods has proven to be largely ineffective for public notification of beaches purposes because of the time required for sample processing (i.e., sample transportation to a laboratory, 18 to 24 hour incubation time, and time required for results to reach and be evaluated by a decision maker). In addition to their development, the models need to be adequately field verified and calibrated. Ideally, regional models can be developed, but if predictive models can only be developed on a site specific basis over the next 3 years, the data needed to develop, field verify, and calibrate the models should not be cost prohibitive to collect. At a minimum, recreational beach managers should consider a simple, predictive rainfall model to more effectively protect public health.

Workgroup members emphasized that a sanitary investigation[3] approach to characterize drainages to primary contact recreational waters would prove useful for at least the WHO criteria development framework. A simple to implement, quantitative-based sanitary investigation, in conjunction with the health risk data from the proposed additional epidemiology studies, may enable the development of source specific risk parameters for criteria development. To clarify expectations for these surveys, a standardized and relatively simple approach would need to be developed that includes fecal bacteria source characterization (publicly owned [wastewater] treatment works [POTWs], storm drain outfalls, CAFOs, on-site wastewater treatment systems [“septic systems”], agriculture, etc.) on a drainage-wide basis, distance of sources to primary contact recreational waters, flow, developed area in the drainage, and the frequently high variability in water quality from day to day. Additional sanitary investigation components such as source identification and source tracking[4] may not need to be implemented unless there is a need in the regulatory process to implement a TMDL or to protect the public health of swimmers at chronically polluted beaches.

The following summary assumes that all of the approaches encompass and achieve the benchmarks outlined in Section 1.1 to the extent feasible.

1.4.1 Summary of Application of WHO Approach for U.S. Criteria Setting

The general framework described by the WHO (2003) would be applicable to U.S. criteria setting in the near-term given that the following research is conducted and science policy decisions are made:

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[3] This is similar to Canada’s “Environmental Health and Safety Assessment” in Appendix A of *Guidelines for Canadian Recreational Water Quality* (MNHW, 1992). Although the WHO (2003) uses the term “sanitary inspection,” some workgroup members expressed concern that use of that specific term or the related term “sanitary survey” might imply adoption of all the protocols for sanitary inspections/surveys from other contexts. Thus, the term “sanitary investigation” was selected for use in these proceedings to minimize preconceived assumptions regarding the nature of the sanitary investigation and is used to refer to a quantitative approach to gauge watershed susceptibility to fecal influence. However, “sanitary inspection” is used when the WHO approach is described.

[4] Although there is not universal acceptance of definitions for microbial source tracking and microbial source identification, the Methods workgroup discussions assumed the following working definitions: source identification is determination of the type of animal (sometimes human versus nonhuman, sometimes more specific) that produced the fecal contamination. It does not include determining where in the watershed that material came from, but it does suggest what to look for upstream. Source tracking is determination of the actual source of fecal matter, such as a leaking pipe, a septic system, or a cow pasture. It typically involves using some of the marker techniques associated with source identification, but not necessarily. Source tracking can also be achieved through extensive spatial sampling with existing indicators or (for example) through use of dye tablets in septic systems.

1. Analyze epidemiological data to determine the values of water quality that correspond to the identified levels of "acceptable risk" for the indicator of fecal contamination using the selected method(s).
2. Identify a suitable indicator of fecal contamination or suite of fecal indicators (particularly for subsequent tiers of investigation). This information needs to be epidemiologically based.
3. Identify "acceptable risk" levels. Choosing an "acceptable risk" level is a policy decision that is informed by science (e.g., epidemiology studies). See Chapter 5 for a discussion of the process through which an "acceptable risk" level could be chosen.
4. Derive a quantitative sanitary investigation category rather than a qualitative process; also, the sanitary investigation should be standardized nationwide.
5. Statistically validate the linkages between different indicator/method combinations for different CWA §304(a) purposes to facilitate translation between the various indicator/methods.
6. Consider and develop a recreational water quality reclassification scheme, if appropriate. If such a reclassification scheme is appropriate, a management system would be necessary to facilitate implementation of beach advisories and to ensure informed choice regarding beach use.
7. Develop a public information management system and a beach signage provision. The purpose of these programs would be to represent bathing water characteristics derived from a "bathing water profile" and historical water quality.
8. Institute a monitoring program to acquire bathing water quality data for numerical compliance assessment purposes.
9. Release CWA §304(a) criteria guidance and associated implementation guidance concurrently.

To apply the WHO (2003) approach for future criteria setting, the following issues will need to be considered in detail and expanded:

1. Develop a process to determine how waterbodies get listed as impaired.
2. Determine the appropriate number of categories for microbial and sanitary investigation categories.
3. Possibly change several qualitative determinations in the framework (i.e., very good, good, fair) to less descriptive terms (i.e., Category I, Category II, etc.).
4. Develop a process for categorization of NPDES dischargers (consideration for default to most restrictive category).
5. Determine how to use different indicator/method combinations for CWA §304(a) applications and translate to each other to ensure equivalent levels of protection.
6. Determine whether health risks from nonhuman fecal sources are substantially different than from human sources.
7. Determine what is the most appropriate metric for expressing the water quality criteria (geometric mean, upper percentile, a combination of those and/or other)

8. Determine how to make water quality public notification decisions (this is likely a function of the indicator/method combination[s] that are employed and the strength of a predictive model).
9. Develop a well described and vetted quantitative sanitary investigation guidance; here the workgroup members suggested a tiered approach that allows for varying levels of effort based on likely benefit from the assessment (high and low risk should be easier to assess [i.e., beaches downstream from POTWs or urban catchments would be high risk, and beaches downstream of catchments with 100% natural sources would be low risk]). Although completion of the sanitary investigation does not need to be required, surface waters would default to the most restrictive criteria until such time as a completed investigation provides justification for changing the applicable criteria.
10. Develop a well described and vetted recreational water quality reclassification scheme.

1.4.2 Summary of Application of EU Approach for U.S. Criteria Setting

The general framework described by the EU (EP/CEU, 2006) would be applicable to U.S. recreational water quality criteria setting given the same research and science policy decisions as described above for the WHO except (1) a classification scheme based on a quantitative sanitary investigation would not be necessary because the sanitary inspection category is not used to determine the beach classification, and (2) it would not be necessary to determine whether health risks from nonhuman sources of fecal contamination are substantially different than from human sources, because the beach classification is based on microbial densities only..

To apply the EU approach for future criteria setting the same issues described above will need to be considered in detail and expanded, with the following exceptions:

1. Reform the microbial categories to fit U.S. waters, do not include the “sufficient” category of EU Directive EEC/7/2006.
2. Determine if a discounting scheme is necessary and appropriate (e.g., elimination of monitoring data for compliance purposes), and if so, then there is a need to determine how to make it most protective of public health.

1.4.3 Summary of Application of EPA 1986 Approach for U.S. Criteria Setting

The current EPA (1986) framework described previously would be applicable to new or revised U.S. criteria development with the following modifications:

1. Develop additional indicators and analytical methods that would be applicable to tropical and temperate waters and also for use in wastewater.
    a. Base additional indicators and methods on health risks (i.e., occurrence would be correlated with rates of illness from epidemiological studies).
    b. Ensure that the revised criteria framework specifies the appropriate indicator/methods combination for the various waters.

2. Consider more timely methods for beach monitoring and water quality notification. Currently, there is no scientific evidence supporting beach water quality determinations based on, at best, day-old (culture-based) data.
    a. If molecular-based methods are used, then fate and transport data for that indicator using that method would be needed.
    b. If molecular-based methods are limited to beach monitoring and water quality notification, then these methods must be linked somehow to the methods used for the other CWA purposes. Currently, very limited data are available for this purpose.
    c. If predictive modeling is used in water quality notification programs, the models need to be adequately field-verified and calibrated.
3. Risk threshold
    a. Any final recommendation for CWA §304(a) criteria must be health-based and derived from the available epidemiological data.
    b. If a single sample criteria is used, it should be of similar stringency to any other measure used (e.g., geometric mean) and the single sample criteria should account for the expected frequency of exceedance (e.g., if the single sample criteria is based on a 95th percentile, a 5% exceedance should be allowed without invoking compliance ramifications).
    c. Consider risk to sensitive subpopulations (e.g., children) in the determination of the risk threshold.
    d. The risk of illness should be the same for swimmers in all types of waters (i.e., marine, fresh, temperate, tropical, etc.) exposed to all types of fecal contamination sources (e.g., point, non-point).
    e. Secondary contact recreation waters:
        i. Acquire data to show health risks associated with limited, but defined levels of contact and/or incidental exposure.
        ii. Data can be from epidemiological studies or estimated using quantitative microbial risk assessment (QMRA).
        iii. Develop a more accurate descriptor of what constitutes secondary contact.
4. CWA §304(a) AWQC recommendations and associated implementation guidance should be released concurrently.

1.5 Summary of Response to Workgroup Charge Questions

See Appendix A for the complete (original) charge questions.

1. ***What approaches exist currently for setting limits of pollutants that may be relevant for developing nationally recommended recreational water quality criteria? Consider approaches used for other kinds of pollutants in water, in other environmental media, and by other countries as well as approaches being implemented by States. What are the pros and cons of each of these approaches?***

- European Union Revised Bathing Water Directive 2006/7/EC

- Hazard Analysis and Critical Control Point Principles
- Heal the Bay Beach Report Card
- EPA Air Quality Index
- EPA *Ambient Water Quality Criteria for Bacteria – 1986*
- WHO *Guidelines for Safe Recreational Water Quality Environments. Volume 1 Coastal and Fresh Waters*

The EU (EP/CEU, 2006), (US EPA) 1986 criteria, and the WHO (2003) approaches are already being used for the intended purpose, either in the United States or other countries. The other possible approaches listed above have not been applied in a regulatory framework for proposed water regulation and would need to be assessed to determine their utility or applicability to derive new or revised recreational water quality criteria.

2. *Which of these approaches is most applicable and appropriate for developing nationally recommended recreational water quality criteria in the near-term? Why is this approach on balance considered the most applicable and appropriate?*

Workgroup members identified the following critical benchmarks for water quality criteria development:

- Be applicable to human health effects;
- Fulfill the needs of Clean Water Act (CWA) and meet the associated regulatory purposes (monitoring, permitting, total maximum daily loads [TMDLs], and §303(d));
- Address geographic variability (i.e., tropical, subtropical, and temperate regions);
- Address potential differences between point and non-point sources of fecal contamination and associated risk;
- Consider risks to susceptible subpopulations, primarily children; and
- Be based upon methods that are reliable and reproducible.

Based on these benchmarks, workgroup members further identified three approaches for further consideration—European Union Revised Bathing Water Directive 2006/7/EC (EP/CEU, 2006), EPA *Ambient Water Quality Criteria for Bacteria – 1986*, and the 2003 World Health Organization *Guidelines for Safe Recreational Water Quality Environments. Volume 1 Coastal and Fresh Waters.* Table 4a summarizes the advantages and disadvantages of each approach and is provided in Section 1.3.4.

3. *For those approaches identified as applicable and appropriate, what is the science that supports the approach? Is that science sufficient and of adequate quality?*

Epidemiological research identified to support the best selected approaches was:

- European Union Revised Bathing Water Directive (2006/7/EC; EP/CEU, 2006)
  - Fleisher et al. (1996)

    - Kay et al. (1994)
    - Weidenmann et al. (2006)
    - Wyer et al. (1999)
- EPA *Ambient Water Quality Criteria for Bacteria - 1986*
    - US EPA (1983)
    - US EPA (1984)
- World Health Organization 2003 *Guidelines for Safe Recreational Water Quality Environments. Volume 1 Coastal and Fresh Waters*
    - Fleisher et al. (1996)
    - Kay et al. (1994)

All members of the workgroup agreed that the research reports listed above support the respective approaches but some members questioned whether the research identified above was adequate to meet all of the identified benchmarks. They also agreed that additional epidemiological and modeling work needed to be performed in order to successfully implement any of the approaches above for future new or revised recreational water quality criteria development in the United States.

4. ***Are there any critical research and science needs that should be addressed in developing or selecting an appropriate approach? Can this research be completed n time to be used in criteria development in the near term?***

The workgroup members identified the following research and science needs to support the suggested approaches.

- Information on the geographic applicability of fecal indicators for assessing health risks at tropical and subtropical fresh and marine recreational bathing areas impacted by point and non-point sources of fecal contamination (see Chapters 2, 3, and 4; research on sensitive subpopulations should also be incorporated into this need [see Chapter 5]);
- Ability to discriminate between human and nonhuman sources of fecal contamination;
- Information on sources of runoff (e.g., concentrated animal feeding operations [CAFOs]) from both marine and fresh recreational waters;
- How much water are bathers ingesting while swimming?; and
- Fate and transport of indicators (and pathogens) in the aquatic environment.

Workgroup members also identified the following possible long-term research needs:

- Comparison of prospective cohort and randomized control trial epidemiological studies;
- Identification of pathogens (viruses, bacteria, or parasites) responsible for GI illnesses at bathing beaches;

- Health impacts following exposures over multiple days;
- Significance of non-GI illnesses (dermal, aural, nasal); and
- Comparison of severity of illnesses related to exposure to human and animal (domestic and wildlife) fecal contamination (see Chapter 4).

Although workgroup members identified these long-term research needs there were some differences of expert opinion on the essentiality of these needs. In conjunction with these research needs, workgroup members also noted the necessity to clarify the objectives of environmental health assessments (sanitary investigations) and microbial source tracking methods.

5. ***Is a "toolbox" approach appropriate for developing new or revised recreational criteria in the near-term? Why or why not?***

The Approaches to Criteria Development workgroup members interpreted the concept of a toolbox approach differently. Some members believed that shifting from the current (US EPA, 1986) criteria approach to either the WHO (2003) or EU (EP/CEU, 2006) model approach would constitute a type of toolbox approach. For example, the sanitary investigation as used within the WHO approach could be considered to be an additional tool in the implementation of the new or revised criteria. Others believed the toolbox approach meant the use of alternative or additional fecal indicators or pathogen methods. In either case, the implementation of the toolbox approach was dependent upon additional epidemiological studies being conducted that may or may not be possible within the near-term (2.5 to 3 years).

Predictive models could be an integral part of the toolbox. Models that have been both validated and calibrated are critical for accurately predicting recreational waters that exceed criteria. Improved notification via forecasting models is likely to protect public heath better then the use of single sample criteria based on current indicators measured by culture methods.

6. ***What are the pros and cons of selecting a "toolbox" approach?***

There was commonality of workgroup member opinions in regards to several of the pros and cons related to the use of a toolbox approach.

Most of the workgroup members believed that a toolbox approach would help address some of the issues with geographic variability. For example, the use of different fecal indicators that demonstrate improved indicator/illness rate relationships in subtropical or tropical waters would reduce the likelihood of these waters inappropriately being listed as impaired under the CWA. The use of some form of sanitary investigations, as within the WHO (2003) approach, would potentially allow for discounting those waters that were identified as having limited or no anthropogenic fecal loading, thereby avoiding those waters being listed as impaired inappropriately.

The cons associated with a toolbox approach were primarily related to the current lack of data on the fecal indicator/illness rate relationships for additional methods. There was also some concern expressed about the difficulty in incorporating the toolbox approach to account for the use of different indicators for different CWA §304(a) needs. There was also concern about the feasibility of establishing requisite and defensible linkages between the various indicator/method combinations that could comprise the toolbox.

7. *What are the desired features or characteristics that would make a "toolbox" approach appropriate?*

Any additional fecal indicator or pathogen measure within the toolbox would need to have proven indicator/illness rate relationships, or at a minimum, have a linkage to another indicator that does. The characteristic of being interrelated (correlated) with each other would be of particular use, especially if one was going to be used to support one aspect of the CWA §304(a) needs and the other was being used to support another §304(a) need.

The toolbox approach should support more then just one aspect of the CWA §304(a) needs. Any of the tools within the toolbox should be validated, either by predictive modeling or by correlation to other tools within the toolbox. Additionally, if a management action is initiated on the results of a particular tool within the toolbox (e.g., a beach closure based on qPCR) the follow up action should also be based upon the same tool (beach opened based on qPCR), to the extent possible.

8. *Would a "toolbox" approach achieve additional public health protection as compared to another approach? Why or why not?*

Yes, as mentioned above, the additional tools within the toolbox could potentially improve the assessment of waters (e.g., reduce the listing of tropical or subtropical waters as impaired due to the poor indicator/illness relationship for these waters) or the appropriateness of beach advisories or closures.

9. *Criteria for secondary contact recreation could be part of a "toolbox." What approaches would be appropriate for developing criteria for secondary contact recreation?*

Workgroup members defined secondary contact as limited or incidental contact. As such, workgroup members believed that the same approach could be used for waters designated as secondary contact as used for primary contact, meaning that epidemiologically-based health data could be used to define acceptable exposure limits. QMRA could also be used for these purposes to supplement available epidemiological information.

10. *What are critical research and science needs in developing or selecting an appropriate approach for secondary contact recreation?*

Additional epidemiological studies may be needed under secondary contact conditions. These epidemiological studies should address the same data needs as those proposed in support of the primary recreation criteria. Alternatively, QMRA could be used if exposure data are available.

Can this research be completed in time to be used in criteria development in the near-term?

It is possible, but unlikely given the current demands for additional epidemiological work in support of the primary contact designated use. However, a QMRA study could be conducted during this timeframe.

11. *What are the implementation considerations of the different approaches for CWA purposes (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs?*

All three approaches—the (EPA) 1986 criteria, EU (EP/CEU, 2006), and WHO (2003) approaches—would require additional epidemiological studies to implement. Given additional epidemiological data with additional indicators, it is possible that each approach could potentially be implemented and could support multiple CWA §304(a) needs. As noted above, using multiple indicators for different purposes is a cause of concern.

Are there practical considerations that could preclude, or greatly limit, the use of an approach in routine, regulatory implementation (e.g., field sampling issues, laboratory challenges, staff training, etc.)?

If future epidemiological studies do not identify additional indicator tools that would improve the indicator/illness relationship for a broader geographic range, the (EPA) 1986 model would be a much less desirable option than either the WHO (2003) or EU (EP/CEU, 2006) approach. Both the EU and WHO approaches apply a discounting scheme, so the failure of near-term epidemiological studies to identify more robust indicator tools does not preclude the implementation of these approaches for the development of new or revised recreational water quality criteria.

<u>Geographic Applicability</u>

1. Is a single criterion available that is appropriate for the diverse range of geographic conditions? Why or why not?

No. Different regions of the country have different potentials for regrowth, persistence, indicator/pathogen die off rates (UV exposure), and indicator/illness rate relationships. The literature supports the conclusion that additional indicators will be necessary to accurately identify those recreational waters that are at risk across all geographic regions of the country.

Workgroup members felt that future epidemiological studies should include additional indicators to improve the indicator/illness relationship across all geographic areas of the United States.

2. Is a toolbox approach appropriate for different geographical conditions? Why or why not?

Yes, for the reasons noted in the response to Question #1 above.

3. What would a "toolbox" that addresses geographical differences look like?

The toolbox might include alternative or additional indicators that better predict, either individually or in combination, the indicator/illness relationship. Alternatively, the toolbox might include environmental health and safety assessments (sanitary investigations) that allow for the discounting of waters that appear as impaired based upon the indicator results, but for which the impairment judgments are not supported by demonstrable impacts (elevated indicators from wildlife or sediment sources only). The toolbox approach also could be used to allow different indicators and be used for different CWA §304(a) purposes.

4. What are critical research and science needs in developing or selecting an approach that will appropriately factor-in diverse geographical conditions?

Additional epidemiological studies are needed that provide improved indicator/illness rate relationships for all regions of the United States. These additional epidemiological studies should focus on recreational waters that are under a variety of potential pathogen sources (e.g., sewage, urban runoff, non-point sources, non-anthropogenic sources). Where possible, the various potential sources of pathogens should be considered within a single epidemiological study rather than each being considered in separate studies. This might be possible by examining waters that have varying sources depending upon rainfall or climatic conditions. For example, California beaches that have urban runoff sources during wet weather but no known point or non-point sources during dry weather.

To pursue the 2003 WHO approach, a quantitative environmental health and safety assessment (i.e., sanitary investigation) tool would have to be developed in order to support the categorization of recreational waters as to their risk of potential fecal contamination. To have greater confidence in the WHO model, research is needed to determine if the notion that fecal contamination from non-anthropogenic sources is of lesser human health risk that anthropogenic sources.

Expression of Criteria

1. Given the diverse needs of the CWA programs and the overarching goal of protecting and restoring waters for swimming, what protection is provided by establishing a 30-day "average" value as the criteria?

There was some commonality of workgroup member opinion on this issue. Several members felt that the criteria would best be expressed as a geometric mean and/or a standard deviation or

95th percentile. Several members believed that these values would have to be site specific in order to be protective. If formatted correctly, an average value is as protective as any other single measure.

What additional protection (if any) is provided by a daily or instantaneous maximum value?

The added value provided by an instantaneous maximum value is dependent upon the indicator/illness rate association. Short-term variability associated with the current indicators limit the usefulness of single sample maximum values; however, using qPCR or other non culture-based methods may improve the utility of single sample values. Additional epidemiological data is needed to assist in making this determination. One problem is that the formulation of the current single sample maximum is such that it is more stringent than the geometric mean, and this has caused substantial confusion among States.

From a scientific standpoint, is one measure better scientifically than another for particular purposes (e.g., mean value for purposes of identifying waters and daily maximum for beach monitoring and notification purposes)? Why?

It depends on how the new or revised criteria are derived and the assumptions made about the variability in water quality. There is some scientific merit to the continued use of single sample maximum values for some CWA purposes. This is of particular interest with respect to public health protection; however, there was not agreement among the workgroup members on this point.

2. What are pros and cons of expressing the criteria differently for the various CWA program needs?

As currently used, single sample maximum values are not effective for beach monitoring purposes. This may change somewhat if the shift from culture-based methods to non culture-based methods improves the issues with variability and indicator/illness rate relationships.

There is potential to use single values, whether culture- or non-culture based, in predictive models for beach monitoring. The geometric mean, standard deviation, or 95th percentiles show promise for multiple CWA programs. So long as there is data from epidemiological studies that demonstrates the one expression of the criteria is equally as protective as another, there would be no problem with using different expressions of the criteria for different CWA purposes.

3. What are the implications of instantaneous or daily values for public health protection? If we don't currently have a good understanding of this, what are the critical research and science needs to answer the questions?

Currently there is very little data to support the use of instantaneous or daily values derived from culture-based methods from Day 1 to predict the need to post or close recreational waters on Day 2. There are recent epidemiological data (Wade et al., 2006) that indicates that qPCR (non culture)-based indicator methods may be effective for same day notification at some beaches.

Group members identified several critical research needs that would potential improve the utility of an instantaneous or daily value (see Chapter 1).

4. If EPA were to set criteria at a mean concentration over 30 days and not recommend a single sample maximum, do we understand the illnesses that could occur on a single day (where the level would still lead to compliance with the 30-day average)?

In general, workgroup members agreed that the probability of illness on any single day is not understood. Some reasons for this are as follows:

- the variance in water quality in any particular water body could be significantly different than the criteria from which it was derived;
- the variability of indicator could change between beaches and temporally; and
- if the exposure-response curve is based on a geometric mean, the interpretation of a single sample is difficult.

With research, a single sample maximum could be used to estimate the probability of illness on a particular day.

5. If the science is "not there," what are the critical research and science needs to answer this question?

Critical research and science needs centered on expanding epidemiological studies based on qPCR (and other indicators and methods) to other situations (e.g., example marine beaches or non-point pollution sources).

6. What are the implementation considerations for CWA purposes of failing to address (and addressing) differences geographically in the criteria and failing to include (and including) a single sample maximum value for (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs? Are there practical considerations that could preclude, or greatly limit, the usage in routine, regulatory implementation (e.g., field sampling issues, laboratory challenges, staff training, etc.)?

To address this question it was separated into the following two components: (1) failing to address geographical differences, and (2) failing to include a single sample maximum for various purposes identified in the CWA.

(1) Failure to recognize that the current criteria may not be applicable to tropical and subtropical beaches could present an unacceptable risk to bathers in these recreational waters.

(2) Single sample maximum NPDES permits – compliance tools such as NPDES permits require single sample maximums and need to continue that approach.

Regarding States' designated use attainment, single sample maximum values are not necessarily required and in fact may not be necessary. These types of criteria could be useful for beach monitoring and water quality notification. Regarding TMDLs, single sample maximum values do not seem applicable for TMDLs for indicator bacteria.

1.6 Concluding Remarks

The workgroup members did not specifically prefer one criteria approach over another primarily because it is believed that additional data that will become available from epidemiological studies within the next 2 to 3 years and how those data will inform the criteria development process is not yet known. For example, it is not known what new information will be available based on new indicators and/or methods and how that information might inform a sanitary investigation component of a WHO-based criteria approach.

Further, it was the opinion of workgroup members that there may be differences in the ability to implement each of the three approaches if all of the various criteria attributes are not met. For example, workgroup members felt that the 1986 EPA approach with a new or different indicator and/or method would not be satisfactory if most or all of the criteria attributes were not met. Workgroup members also felt that the same set of circumstances may not preclude the use of the WHO or EU approach. With respect to the use of a toolbox-based approach where different indicators are used for different CWA §304(a) applications, workgroup members expressed concern about the feasibility of developing health-based linkages (as described above) as would be required by either the WHO or EU approaches. Thus, at this time, no workgroup member was definitively able to recommend one approach over another; however, the workgroup members agreed that the choice of approaches must be deferred pending the outcome of ongoing and near-term research.

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CHAPTER 2
PATHOGENS, PATHOGEN INDICATORS, AND INDICATORS OF FECAL CONTAMINATION

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2.1 Application of Microbial/Biomarker Parameters

The charge of the Pathogen, Pathogen Indicators, and Indicators of Fecal Contamination workgroup was to identify critical research and science needs in the development of new or revised criteria for recreational waters, including total maximum daily load (TMDL) implementation, and National Pollutant Discharge Elimination System (NPDES) implementation using microbial and chemical indicators. The discussions were limited to constituents for which methods are currently available or expected to be available within the next 3 years and focused around the following four issues:

1. Fecal matter indicators (as surrogates for gastrointestinal [GI] and non-GI illnesses);
2. Pathogens and their index organisms (GI and non-GI illnesses);
3. Application of fecal indicators, pathogen index organisms, and pathogens in combination for criteria development; and
4. Application of all the above for all categories of waters, climatology, and geographical considerations.

Currently, implementation of ambient water quality criteria (AWQC) for the four Clean Water Act (CWA) applications require monitoring fecal bacterial indicators to assess the degree to which the water is contaminated with sewage and sewage-borne pathogens with respect to the accepted risk for exposure. Development of the existing (US EPA, 1986) AWQC for recreational waters were based on epidemiological studies that related concentrations of fecal indicator bacteria at recreational waters impacted primarily by point sources of human sewage.

Since development of the currently used 1986 AWQC, research has shown that this narrow health effects-based standard (i.e., epidemiological studies at beaches with point sources of human sewage) is limited in that it does not take into account differences in geographical conditions, ecology of microorganisms, and varying sources of fecal indicator bacteria. In this regard, the expected relationship between illness and indicator organism densities would be high if the source of contamination is human sewage, moderate if the source was a mixture of human and animal feces, or lower if the source is the result of replication of the indicator bacteria in the environment, such as in soil, sediments, storm drains, or on plants or aquatic vegetative matter. Initially, replication of fecal indicator bacteria was reported in tropical areas (e.g., Hawaii, Guam, Puerto Rico) but has now been documented in subtropical areas such as south Florida and even temperate areas (Great Lakes States). A further but untested complication in interpreting fecal indicator bacteria results may arise due to different rates of pathogen inactivation in the environment relative to fecal indicators across different geographic and climatic regions.

It is for the above reasons that experts in the field of microbial water quality generally agree that the principles of microbial ecology must be considered in water quality assessment. Understanding and applying these principles requires an assessment of the sources of fecal contamination, selection of the appropriate methods used to assess these sources, a connection between the intended AWQC application, and the fecal and/or pathogen indicator or pathogen measured and an analysis of that indicator’s fate and transport. Because of this understanding, workgroup members suggested a tiered assessment of a watershed, starting with traditional fecal indicators (conservative measures), and progressing to select a suite of indicators of contamination (providing source specificity and contaminant load information). A characterization of contaminant inputs through a sanitary investigation of the watershed and the waterbody being assessed should be undertaken, specifically assessing hazardous events, such as rainfall-induced runoff or wastewater treatment failure. Key information would pertain to a cataloguing of point sources (e.g., sewage effluent) and non-point sources (e.g., animals, runoff, on-site septic systems, environmental regrowth) so that a comparative risk assessment can be made based on the concentrations of standard (traditional) monitoring fecal indicators (i.e., *E. coli*, enterococci) and the expected presence of human pathogens. This initial assessment should assist in understanding the relationship between the contamination and epidemiological studies (indicator levels and risks of illness) mentioned elsewhere in these proceedings. In order to select appropriate indicators, a tiered toolbox approach was preferred by workgroup members rather than promoting use of one particular indicator over another.

2.2 Tiered Toolbox Monitoring Approach

An initial cataloguing of fecal pollution sources should include a review of existing monitoring data and a sanitary investigation to assess contaminant levels and sources that impact a given recreational water site. Based on that information, the indicator used in monitoring or the predictive modeling tool most appropriate for each CWA AWQC application and contamination source would be selected for the situation. Water quality assessment for each recreational water site should begin with the simplest analyses and assessment and move on to the most appropriate (specific or targeted) indicator for that site or purpose. More refined tools to differentiate between human, domestic animal, or environmental sources of fecal contamination could subsequently be used if deemed necessary.

If a sanitary investigation determines that fecal pollution is human or animal origin, then *E. coli* or enterococci could be used in the tier one water quality assessment because many pathogens can be expected to multiply in human and animal intestines. If the source of the "fecal" indicator organisms is determined to be from the environment (i.e., from growth in soil/sand, sediment, or water), then *E. coli* and enterococci may be inappropriate because most pathogens are not capable of environmental multiplication. As a result, the monitoring for this tier would need to be a fecal organism/chemical that does not amplify in the environment, such as spores of *Clostridium perfringens* or male-specific (F+) coliphages measured by culture- or molecular-based methods, specific members of the *Bacteriodes* bacteria measured by a molecular method, or use of a chemical indicator of fecal material.

For a subsequent tier of monitoring, infectivity and/or molecular methods could be used for specific groups of pathogens such as, for bacterial pathogens (shiga-toxin producing *E. coli* [STEC] or *Salmonella*), for protozoa (*Cryptosporidium* or *Giardia*), and for representative human sewage-borne viruses (enteroviruses, adenoviruses, polyomaviruses, or noroviruses).

Location-specific data should be archived for potential use in future predictive modeling that might allow for management of site-specific fecal contaminants. Finally, if possible, archiving samples for further characterization and national comparison of new indicators, and/or pathogens and their respective methods would be advantageous assuming a national repository database and sample archive facility could be established.

Several non-GI illnesses have been associated with recreational uses of water but these are not addressed by monitoring for fecal indicator microorganisms or chemicals because the etiological agents for these waterborne diseases come from non-sewage sources. Examples include animal urine (*Leptospira* spp.), shedding from human skin (*Staphylococcus aureus*), or microorganisms that are naturally present in freshwater environments (*Aeromonas hydrophila*, *Naegleri fowlerii*, *Legionella pneumophila*). Further, several human pathogenic *Vibrio* species (*V. cholerae*, *V. vulnificus*, *V. parahaemolyticus*, *V. alginolyticus*) are indigenous to marine and brackish waters. Because reliable indicators have not been developed for non-GI etiological agents, the best approach to address aquatic non-enteric pathogens is to characterize the aquatic conditions that increase the risk for these pathogens. For pathogenic *Vibrio* spp., this includes saline waters of warmer temperature and waters that contain high levels of nutrients.

2.3 Parameters for Hazardous Event Pollution Monitoring

The first approach to investigate a hazardous event (sewage discharge, rainfall impact, etc.) would be to assay for fecal indicators (appropriate for a climatic/geographic area of concern, see Section 2.4). The primary indicators of fecal contamination are *E. coli* or enterococci; however, based on the classification from an initial sanitary investigation, alternatives may include *Clostridium perfringens* or F+ coliphages (dependent on a robust method being confirmed). These must be demonstrated to relate to a possible health outcome (see Section 2.4). When information is required on source characterization, then additional microbial indicators (see Section 2.5) are generally preferred over chemical biomarkers (see Section 2.7).

Focused sampling during and after higher risk periods is important when information from the sanitary investigation (which may include system models) is used to predict such risk. For these applications, the context of the likely pathogen group(s) should dictate the type of indicator to assay. For example, for rainfall in an area possibly impacted by on-site septic systems, viruses are considered the most mobile pathogen group so use of virus model organisms, such as the F+ coliphages, would be informative. For sites contaminated by concentrated animal feeding operations (CAFOs), reasonable pathogen index tests include shiga-toxin producing *E. coli* or *Cryptosporidium*. This approach assumes that some background level of the targeted group is known for the area of concern (see Section 2.8, research needs).

2.3.1 Microbiological Parameters

The Beaches Environmental Assessment and Coastal Health (BEACH) Act of 2000 requires States with coastal or Great Lakes recreational waters to adopt the current (US EPA, 1986) criteria for *E. coli* and enterococci. In November 2004, EPA promulgated a final rule that put federal standards in place for the 21 coastal states that had not adopted the 1986 criteria or established criteria as protective of human health as EPA’s 1986 criteria. However, these federal criteria apply only to coastal states and Great Lakes waters. In many cases, a fecal coliform-based standard still applies to many states having only inland waters. It is important to note that the results of the epidemiological studies used to generate the 1986 criteria for coastal and Great Lakes waters may not be directly applicable to all inland waters.

2.4 Traditional Fecal Indicators (Coliforms and Enterococci)

Consideration of the environmental context for which these traditional indicators are used is critical to the interpretation of the results. For example, wastewater treatment/disinfection may be effective in reducing the number of these traditional fecal indicators but ineffective in reducing/inactivating some pathogens of concern (Blatchley et al., 2007). Some industrial treatment systems may contain/enable replication of high number of "fecal indicators," which are not necessarily associated with fecal sources (Degnan, 2007; Gauthier and Archibald, 2001). Ambient water and soils in tropical environments may be conducive to the growth of environmental strains of *E. coli* and enterococci (Fujioka and Byappanahalli, 2001). A similar situation was found in temperate Australian waters (Ashbolt et al., 1997; Barnes and Gordon, 2004; Davies et al., 1995) and these indicators have also been found to persist in U.S. beach sand/sediments (Whitman et al., 2006).

The range of strains of fecal indicators identified by traditional culture-based methods may differ from those identified by enzyme-based and quantitative polymerase chain reaction (qPCR [molecular])-based methods. Further, the strains more associated with fecal matter are not differentiated from the environmental strains by all of these methods. There are commercially available systems that can aid in the discrimination of strains; however, less expensive typing kits do not accurately provide such discrimination for environmental strains. It is important to note then when quantifying fecal indicator organisms, different methods target different strains. For example, cells stressed by wastewater disinfection processes may be enumerated using MPN (Most Probable Number) methods but excluded by methods enumerated by colony forming unit (cfu) methods. When current qPCR methods are used, both viable and non-viable cells are detected. In addition, the number of gene targets may vary per cell and therefore do not provide comparable information to culture-based results.

E. coli

Of the traditional fecal indicators (see also Appendix E, Text Box E-1), only *E. coli* has been shown in epidemiological studies to consistently relate to health outcomes for freshwater recreational water users (Cabelli et al., 1982; Wade et al., 2003; Wiedenmann et al., 2006). In marine/estuarine waters, *E. coli* is more readily inactivated than enterococci and appears to correlate less well to health risk than enterococci for saline water environments.

Subtyping of different strains of *E. coli* (library-dependent microbial source tracking methods) appears to be very site-specific if useful at all. Thus, it is not generally suggested as an effective way forward to separate environmental sources of *E. coli* from fecal sources across the United States (see Section 2.5).

Enterococci

The enterococci are the major group of fecal indicators that have a clear link to GI illness and upper respiratory disease in bathers in marine and fresh recreational waters (Kay et al., 2004).

There are, however, several shortcomings in the use of current methods for enterococci. Most importantly, there is a range of different *Enterococcus* spp. detected by current methods. Based on unpublished Californian studies (Stephen Weisberg, SCCWRP, personal communication, 2007), greater fecal specificity may result from specific identification and enumeration of *E. faecalis* or *E. faecium* or molecular-based methods targeting specific genes within these species (e.g., ribosomal RNA or enterococcal cell surface-associated protein and its gene *Esp*) (Lehner et al., 2005; Liu et al., 2006). However, no robust method is currently available that readily provides such information, nor has this concept been verified at other U.S. recreational water sites (Anderson et al., 1997).

2.5 Alternative Fecal Indicators

2.5.1 Bacteria

Clostridium perfringens

C. perfringens is a member of the sulphite-reducing clostridia (SRC), which are spore-forming anaerobic bacteria excreted in human and animal fecal matter, but unlike other SRC, do not appear to grow in the aquatic/soil environment. These bacteria have been used as fecal indicator organisms for decades. Australian and North Carolina studies show *C. perfringens* levels in humans comparable to levels found in dog and feral pig feces, but low levels in cattle, sheep, horses, and birds (Leeming et al., 1998; Mark Sobsey, University of North Carolina, Chapel Hill, personal communication, 2007). Importantly, because *C. perfringens* does not appear to grow in aquatic/soil environments, it has potential to be useful as a fecal indicator for tropical environments such as in Hawaii where growth of *E. coli* and enterococci in soil/sand, sediment, and water make those indicator organisms less useful (Byappanahalli and Fujioka, 1998; Hardina and Fujioka, 1991; Roll and Fujioka, 1997). For example, in ambient streams in Hawaii, concentrations of fecal coliforms, *E. coli*, and enterococci consistently exceed recreational Water Quality Standards due to contribution by extra enteric sources (Hardina and Fujioka, 1991, Luther and Fujioka, 2004). Thus, monitoring inland and coastal waters for *C. perfringens* provides reliable data for sewage contamination and is used by the Hawaii State Department of Health to confirm a sewage contamination event (Fujioka and Byappanahalli, 2001).

The presence of *C. perfringens* (spores) in water, therefore provides evidence of existing human/urban fecal contamination, which may reflect either recent or historical fecal contamination from humans or animals. Although methods have been available for some time, confirmation of a robust and consistent method approach should be developed. For example, the advantages of heat-treating samples (or not) to remove background vegetative cells and induce spore germination remains unclear.

The environmental resistance of *C. perfringens* spores has both advantages and disadvantages in their application as a fecal indicator, pathogen indicator, and as an indicator of wastewater treatment efficacy. Collectively, these make *C. perfringens* spores better indicators of persistent and treatment-resistant pathogens, such as *Cryptosporidium* oocysts (resistance to chlorine) and adenoviruses (resistant to UV radiation). However, they can be so persistent in the environment that they may not indicate the presence of pathogens coming from recent (contemporary) fecal contamination.

Recent studies on the partitioning of *C. perfringens* and other fecal indicator microbes in environmental waters, such as *E. coli* and coliphages, indicate differences in the extent of their association with settleable particulate matter (Characklis et al., 2005; Krometis et al., 2007). To date, limited data have been collected on any potential relationship between *C. perfringens* counts and recreator health outcomes (see Section 2.8).

Bacteroides

Bacteroides spp. are members of the normal microbiota of warm blooded animals and studies have shown them to be among the most prevalent genera in feces (Holdeman et al., 1976). Because they are strict anaerobes that grow in the GI tract of humans and animals, they do not survive for long periods of time under aerobic conditions (Kreader et al., 1998). However, their survival under different redox potential conditions (e.g., sediments) has not been thoroughly studied. Recent research based on molecular methods has demonstrated that some isolates may be strictly associated with human feces (Walters et al., 2007). If this is the case, these microorganisms also have the potential to be used for microbial source tracking (MST) applications.

Studies have indicated human versus bovine specificity in certain 16S rRNA genes therefore, 16S rRNA *Bacteroides* genes have been used as an index of human or animal contamination in Europe and the United States. The ability to differentiate sources of fecal contamination is very attractive when it comes to determining risk as a result of exposure via recreational waters. The molecular methodology has been shown to be robust and applicable in the United States and Europe, though it remains to be seen if this robustness holds across temperate versus tropical or subtropical zones of the world. Some results from Hawaii and Europe indicate that these methods may be useful under those climatic conditions (Betancourt and Fujioka, 2006; Seurinck et al., 2006). Either way, it is unclear whether quantification of human/animal fecal loads will be consistent or indeed possible using these molecular-based methods.

Though data from molecular techniques have shown that there is specificity in the human versus animal strains, the fact that both human and animal feces contain a diverse population of *Bacteroides* spp. may limit the usefulness of some detection methods. Methods that focus on one target may have reduced sensitivity as a result of the lower concentrations of a specific *Bacteroides* strain. Data have shown that *Bacteroides* spp. does not survive for long periods of time in the environment; thus, *Bacteroides* detected by qPCR in ambient waters includes a high percentage of inactivated microorganisms. The fact that qPCR detects both live and dead organisms needs to be considered when data are applied in different contexts (e.g., different AWQC applications). That is, qPCR detection is linked to the time the nucleic acid remains within the cell without being degraded. EPA data have demonstrated that the DNA remains undegraded for up to 20 days (Kevin Oshima, USEPA, Office of Research and Development, personal communication, 2007) in the inactivated unlysed cells. This may be equivalent to the survival of some enteric pathogens under environmental conditions. Thus, the presence of *Bacteroides* may have possible use as an indicator of health effects. Because the concentration of *Bacteroides* spp. in feces is much higher than other fecal bacteria, once the persistence of PCR-detected types is better understood, it may also be useful for TMDL applications, although this possibility needs further evaluation.

The molecular methodology for the detection of general and human-specific *Bacteroides* spp. is already being tested and has proven to be robust (Gawler et al., 2007; Walters et al., 2007). Thus, if detection methods are validated in the United States there is an excellent opportunity for short-term advances in quickly adapting the use of this alternate indicator for the rapid analyses of recreational waters and fecal source identification.

2.5.2 Bacteriophages

Coliphages

Bacteriophages (viruses) that infect *E. coli* and possibly other closely related coliform bacteria are called coliphages. There is a long history of research documenting the possible uses of phages as indicators of fecal contamination (Grabow et al., 1998). Coliphages were first proposed as indicators of the presence of *E. coli* bacteria and are taxonomically very diverse, covering the following six virus families: three families of double-stranded DNA viruses (*Myoviridae*, *Styloviriae*, *Podoviridae*), two families of single-stranded DNA phages (*Microviridae* and *Inoviridae*), and one family of single-stranded RNA viruses (*Leviviridae*).

Coliphages that infect via the host cell wall of *E. coli* are called somatic coliphages (including families *Myoviridae*, *Styloviriae*, *Podoviridae*, and *Microviridae*). Male-specific (also called F+) coliphages (*Inoviridae* and *Leviviridae*) infect by attaching to hair-like appendages called F-pili protruding from the host bacterium surface.

Somatic phages have been explored as fecal, treatment efficacy, and health effects indicators. However, little is known about the specificity of their occurrence in human or animal feces. Furthermore, their considerable taxonomic diversity and the lack of readily available and convenient methods to distinguish or specifically detect the different groups has made it difficult to determine which, if any, are effective fecal, treatment efficacy, or health effects indicators. In a recent study by Colford et al. (2007), somatic coliphages were not predictive of human health risks from bathing in marine recreational water largely impacted by non-point sources of fecal contamination. Furthermore, there is very little information on the sources and ecology of the somatic coliphages, especially for the different taxonomic groups. With rare exceptions, they are detected as a broad group with no effort to identify specific taxonomic groups or relate or attribute these different taxonomic groups to specific sources of human or animal fecal contamination or possibly non-fecal environmental sources.

Male-specific coliphages have been studied extensively as fecal indicators and for water/wastewater treatment/disinfection efficacy. Furthermore, F+ RNA coliphages can be distinguished genetically (via nucleic acid detection methods) or antigenically (via immunological methods), into four distinct subgroups: I, II, III, and IV. There is reasonably good evidence that Groups II and III are associated primarily with human fecal waste and that Groups I and IV are associated primarily with animal fecal waste (Furuse et al., 1975; Hsu et al., 1995; Osawa et al., 1981) in the United States. Male-specific coliphages have been included in some epidemiological studies of recreational water. In the recent study by Colford et al. (2007) at a marine recreational water site impacted primarily by non-point source fecal contamination, F+ coliphages were the only microbial indicator whose levels were associated with risks of swimming-associated illness.

Strengths of Coliphages as Indicators

Advantages of both somatic and F+ coliphages as fecal indicators include their (1) presence in relatively high concentrations in sewage; (2) relatively high persistence through wastewater treatment plants, compared to typical bacterial indicators like *E. coli* and fecal coliforms (coliphages may behave similarly to human viruses during wastewater treatment); and (3) ability to be detected in relatively small (100 mL) to medium (1,000 mL) volumes of fecally contaminated water.

Coliphages can be detected by relatively simple, affordable, and robust culture methods—several of which have been standardized and collaboratively tested as EPA, EU, and ISO (International Organization for Standardization) water methods. However, the EPA methods for somatic and F+ coliphages have been fully validated only for groundwater and not for ambient surface waters or wastewaters. Recent research also describes a rapid, simple, and affordable method to detect and group infectious F+ coliphages by short-term (3-hour) enrichment culture, followed by quick (<1 minute) detection of positive cultures by a simple immunological (particle agglutination) method scored by simple visual examination (Love and Sobsey, 2007). The method can be conducted in an MPN format to quantify concentrations of the different F+ coliphage groups (F+ DNA and F+ RNA Groups I, II, III, and IV).

These findings indicate that robust, simple, rapid, and low-cost F+ coliphage methods could be implemented within the 2 to 3 year time frame if correlations to health targets are observed in epidemiological studies. It would be valuable if water samples from upcoming EPA and SCCWRP (Southern California Coastal Water Research Project; see also Appendix F) marine recreational water epidemiological studies are collected and archived for analysis by these emerging qPCR methods once they are fully developed and validated. In addition, research is suggested to compare the performance of methods for rapid coliphage detection by short-term enrichment-particle agglutination and qPCR and to consider the advantages and disadvantages of these two methods for application to recreational water quality monitoring.

Limitations of Coliphages as Indictors

Although effective methods are available to recover, detect, and quantify coliphages, limitations and unsolved problems with these methods remain. The single agar layer method (EPA Method 1601) for enumeration of coliphages by counting plaques is limited to sample volumes of about 100 mL. Analyzing larger volumes is cumbersome and consumes considerable materials, such as Petri plates. Although the enrichment culture-spot plate method can be used to conveniently analyze sample volumes of up to 1 L, the method makes it more difficult to resolve coliphage mixtures when more than one type of coliphage is present in the enriched sample volume. In some cases, one coliphage will grow faster and to a higher concentration. This makes it difficult to detect and isolate minority coliphages that grow more slowly and to lower concentrations. However, detection of all of the different coliphages present as a mixture in enriched sample is possible by either nucleic acid or immunological (particle immunoagglutination) methods.

The ecology of both somatic and F+ coliphages remains poorly documented and inadequately understood. Information is lacking on bacterial host range, sources, occurrence, and behavior (survival, transport, and fate) in different geographical regions having different climates (temperate, subtropical, and tropical) and in waters and wastewaters of different microbial quality.

F+ coliphages can also be detected by molecular-based methods, including conventional and qPCR methods, according to recent studies. Careful review of these studies suggests that there may be deficiencies in the ability of these qPCR methods to detect the broad range of F+ DNA and F+ RNA coliphages and their subgroups. Nevertheless, research is now in progress to further improve F+ RNA qPCR by developing and performance-validating primer sets for all four genogroups of F+ RNA coliphages (Stephanie Friedman, EPA Environmental Effects Research Laboratory Laboratory, personal communication, 2007). Reliable methods have not been developed for genetic analysis and characterization of different somatic coliphage taxonomic groups.

Very few studies have been conducted to evaluate F+ coliphages as predictive indicators of human health risks from recreational use of water. The most extensive study was conducted by Colford et al. (2007). That study showed no health relationship for somatic coliphages, but a weak relationship for F+ coliphages examined by two different assay methods—an MPN version of EPA Method 1601 (enrichment-spot plate method) and EPA Method 1602 (saline agar layer plaque assay). However, these methods have not been performance characterized and fully validated for use in fresh and marine recreational waters according to EPA collaborative study protocols. Additional studies of this type are needed to clarify their potential criteria uses.

Bacteroides phages

Bacteroides phages, viruses that specifically infect *Bacteroides* spp., have been tested as indicators of fecal material in Spain and more recently in the U.K. The former used a method (bacterial host) that was tested in some labs in the United States but further efforts were not made as a result of the perceived difficulty in dealing with anaerobic methodology. Attempts to use the *B. fragilis* strain VPI 3625 showed low occurrence of these phages in the United States (Chung and Sobsey, 1993). Spanish data initially supported the use of *B. fragilis* HSP40, which is specific to phages that only occur in human feces. More recent British work indicated human specificity and high phage counts for a newer Spanish host *Bacteroides* (GB-124), thus providing the opportunity for determining human fecal contamination and virus transport using a rapid and inexpensive phage method (Ebdon et al., 2007).

Strengths of Bacteroides Phages as Indicators

The methods for the detection of *Bacteroides* spp. phages are inexpensive and their presence indicates human fecal contamination. In addition, there is research that indicates specific

Bacteroides hosts are susceptible to phages that are possibly useful for MST, which would be beneficial for its use for CWA §304(a) criteria (Chung and Sobsey, 1993; Ebdon et al., 2007).

Limitations of Bacteroides Phages as Indicators

The diversity of phages including their specificity for human host strains is not yet well characterized over a range of locations. This type of data could be easily obtained in 2 to 3 years, but if it is discovered that there is wide variability in their validity for MST, then their attractiveness for use in national AWQC would be reduced. Many laboratory personnel may not have the experience required to work with anaerobic microorganisms; however, little additional laboratory equipment would be required. Because detection methods have not been standardized in the United States, it would likely take several years to develop standardized methods for enumeration of *Bacteroides* phages in water samples.

2.5.3 EU Project Summary of Tracers

Several microbes and chemicals have been considered as potential tracers to identify fecal sources in the environment. However, to date, no single approach has been shown to accurately identify the origins of fecal pollution in all aquatic environments. In a European multi-laboratory study, different microbial and chemical indicators were analyzed in order to distinguish human fecal sources from nonhuman fecal sources using wastewaters and slurries from diverse geographical areas across Europe. Twenty-six parameters, which were later combined to form derived variables for statistical analyses, were obtained by performing methods that were achievable in all the participant laboratories and include the following: enumeration of fecal coliform bacteria, enterococci, clostridia, somatic coliphages, F+ RNA phages, bacteriophages infecting *Bacteroides fragilis* RYC2056 and *Bacteroides thetaiotaomicron* GA17, and total and sorbitol-fermenting bifidobacteria; genotyping of F+ RNA phages; biochemical phenotyping of fecal coliform bacteria and enterococci using miniaturized tests; specific detection of *Bifidobacterium adolescentis* and *Bifidobacterium dentium*; and measurement of four fecal sterols. A number of potentially useful source indicators were detected (bacteriophages infecting *B. thetaiotaomicron*, certain genotypes of F+ bacteriophages, sorbitol-fermenting bifidobacteria, 24-ethylcoprostanol, and epicoprostanol), although no one source identifier alone provided 100% correct classification of the fecal source. Subsequently, 38 variables (both single and derived) were defined from the measured microbial and chemical parameters in order to find the best subset of variables to develop predictive models using the lowest possible number of measured parameters. To this end, several statistical or machine learning methods were evaluated and provided two successful predictive models based on just two variables that provided 100% correct classification—(1) the ratio of the densities of somatic coliphages, and phages infecting *Bacteroides thetaiotaomicron* to the density of somatic coliphages and (2) the ratio of the densities of fecal coliform bacteria and phages infecting *B. thetaiotaomicron* to the density of fecal coliform bacteria. Other models with high rates of correct classification were developed but they required higher numbers of variables (Blanch et al., 2006).

2.6 Pathogens and Pathogen Indicators

Many beach regulators and scientists believe that there are significant opportunities to utilize specific pathogens or pathogen indices to better understand or characterize potential health risks from recreational exposures. Some reasons for not doing so, however, remain—especially that pathogen numbers are generally significantly lower and more variable than fecal indicator organisms. Nonetheless, pathogens could be utilized to accurately determine risks as there have been a number of studies that define actual human dose-response from oral exposures such as may be encountered during swimming. Enteric pathogens are found in raw and even treated sewage so there is merit in using them in water quality monitoring to assess the risks from exposure. Also, it is possible that an entire "class" of pathogen risks can be determined by the presence of an "index pathogen" representing that group. The current capabilities of molecular methods to detect, identify, and enumerate pathogens has increased regulators' and stakeholders' interest in seeing these applied to ambient water quality monitoring to better protect public health.

There are a number of criteria related capabilities that may be provided by use of specific pathogen or index pathogen monitoring, such as the following: (1) determination of specific pathogen residuals from sewage discharges, the data from which could then be used to conduct quantitative microbial risk assessment (QMRA) studies to assess relative levels of public health concern at a beach; (2) establishment of "model" pathogens and index pathogens that could be used to assess risks from new or reemerging pathogens (an example would be the use of a virus model to assess the recreational risks from avian influenza [H5N1] because this virus can be released from infected human and animal feces [especially waterfowl] and can directly or indirectly contaminate recreational waters); and (3) determination of levels of pathogens that can subsequently be used in QMRA studies to inform decision making relative to whether or not a beach should be closed or reopened after a closure.

There are currently two approaches to pathogen detection, identification, and enumeration, (1) the traditional culture-based techniques that are especially useful in determining viability of the sampled materials; and (2) the molecular-based methods (PCR, antibody-based, and metabolic-based) that generally cannot distinguish between viable and non-viable pathogens, but which may be quite useful in further differentiating or speciating pathogens in water samples. The culture-based methods are useful for recreational waters in that they can determine if there is a viable disease risk from exposure while the molecular methods may not be capable of discerning viability.

Moreover, the culturable isolate can be further characterized for the presence of human virulence genes and compared to clinical isolates in waterborne disease outbreaks. In contrast, molecular-based methods may not be capable of discerning viability although the presence of virulence genes can also be assayed by molecular methods. Because molecular methods do not recover the entire microorganism, further characterization of that microorganism is limited.

Specific tracking of host sources using molecular techniques for pathogens can be very useful in setting TMDLs, as it can help identify the source of the pathogen and its magnitude. Recent improvements in molecular science applications have brought about a capability to simultaneously sample and evaluate large numbers of pathogens (e.g., microarray technology). Microarray technology still requires high concentrations of pathogens for detection. However, ambient waters generally contain pathogen levels below the limits of detection and are unevenly distributed in the water matrix. Thus, research is needed to determine how to best apply these advanced technologies for characterizing enteric and non-enteric disease contaminants, their levels, and potential risks associated with their presence in recreational waters.

Workgroup members expressed some concerns about using either specific pathogens or pathogen class indices as a first tier monitoring requirement for infectious disease risks in a recreational water setting. First, pathogens are typically present in low concentrations in treated sewage, receiving waters, and also in recreational waters; therefore, high volumes of water need to be sampled, which is time consuming, costly, and contributes to analytical variability. Second, pathogen presence is typically sporadic in a community as many waterborne diseases may not be endemic, but are rather transient/episodic so they do not represent a constant contaminant source of fecal pathogens to monitor. Third, there is a variable component in terms of fecal contributions from humans and various animal sources in ambient waters that may have an impact on determining recreational exposure risks. Typically, a number of the bacterial pathogens (e.g., toxigenic *E. coli*, *Campylobacter*) are found in both humans and animals, but there may be differences in strain virulence or infectivity potential from different sources. Likewise, there are a number of protozoan pathogens that cross-infect animal species and humans (*Giardia* spp. and *Cryptosporidium parvum*). On the other hand, human enteric viruses have a much more limited host range and except for a potential few (e.g., hepatitis E virus [HEV]), animal sources of enteric viruses are not a major public health concern in recreational waters. Lastly, it is important to note that at any given time only a small portion of the human population may be infected and excreting any specific pathogen or index pathogen. Thus, large wastewater treatment systems may always contribute a small level of pathogens of concern while septic systems or small treatment systems may not have enough contribution from the infected population to ensure that those effluents would contain specific pathogens of concern to use as a routine measure of contamination—even if the disease organisms are endemic in the population. Also, many types of pathogens are associated with a seasonality or periodicity to their occurrence in a given population.

It is reasonable to use specific pathogens or their index organisms (or model organisms) in a toolbox or tiered approach to monitoring if considered as other than as a first tier measure of fecal contamination. In a toolbox approach, the determination of the presence and concentration of specific pathogens or their index organisms could be useful to characterize risks once it has been determined that there is a trigger level of fecal contamination at a given recreational water site. Dose-response data for a number of the primary pathogens from oral exposures is available and these data would help more narrowly define exposure risks for a detected pathogen. Because of the costs, time for analysis results, and expertise needed to test specific pathogens or index organisms, these measurements would be the last set of measurements applied to monitoring of recreational sites for determining potential sources. The specific pathogen monitoring tools for other AWQC applications (e.g., TMDLs) could allow States to determine sources and concentrations of the pathogens for particular upstream contamination events. Also, pathogens could be incorporated into future NPDES permit limits and be used in the future to assess wastewater treatment plant discharges for specific pathogens of concern downstream and to provide a better understanding of the efficacy of treatment and disinfection processes.

2.7 Chemical Biomarkers of Fecal Contamination

Various shortcomings have been identified in relying solely on indicator bacteria or pathogen/index microorganisms for CWA criteria uses. Methods for MST in aquatic environments have been developed and discussed above that distinguish animal from human sources in the United States and in Europe (Blanch et al., 2006). However, for some specific tiered approaches in sanitary investigations, certain chemical biomarkers of sources may provide timely or higher resolution information in fecal source tracking. Some of the most promising are discussed below.

2.7.1 Fecal Sterols

The most commonly known fecal sterol, coprostanol (5β-cholestan-3β-ol), is largely produced in the digestive tract of humans and dogs by microbial hydrogenation of cholesterol (Leeming et al., 1996). The term "sterols" is generally used for all sterols and stanols (i.e., "fecal sterols") and is also a more specific term denoting a steroidal alcohol with at least some degree of unsaturation.

Two pathways have been proposed for the biotransformation of cholesterol to coprostanol, one in the gut and the other in natural sediments. The α-configured form (cholestanol) is the most thermodynamically stable of the reduction products and is found ubiquitously in the environment; whereas coprostanol is largely of fecal origin, but some reisomerization can yield low levels in natural sediments. Both forms are easily resolved by gas chromatography-mass spectrometry (GC/MS) analysis.

An important advance in using these fecal sterols has been the realization that it is critical to measure both the ratios and absolute concentration of at least four of these related compounds to attribute fecal source contributions between humans, herbivores, and birds (Ashbolt and Roser, 2003). Coprostanol alone has never really been embraced as an indicator for sewage pollution because its presence is not considered as indicative of a health risk due to multiple sources and low level environmental production in sediments.

The fecal sterol biomarker technique offers many diagnostic and quantitative advantages when used in conjunction with traditional techniques for detecting sewage pollution. When careful data interpretation is undertaken, fecal sterol analysis, although expensive and complex, has resolved problems of source attribution in urban and rural environments not possible with use of traditional fecal indicator bacteria and coliphage assays (Roser and Ashbolt, 2007).

2.7.2 Caffeine

Caffeine has been extensively examined as a tool for assessing human influence on aquatic systems. Although caffeine is metabolized when consumed, a small amount (<10%) of ingested caffeine remains intact when excreted (Peeler et al., 2006). Most work in the past decade has focused on heavily polluted systems and efficiency of caffeine removal in sewage treatment plants, although with improvements in techniques and the lowered detection limits, the scope of application has broadened to include stream, wetland, estuarine, and groundwater systems.

A major disadvantage is that caffeine is often present in the urban environment from numerous plant species debris as well as from human "dumping" of coffee wastes. Further, the current methods used (specific extraction and GC/MS analysis) are relatively complex and expensive. Nonetheless, based on the recent work of Peeler et al. (2006) in southwest Georgia, caffeine appears immediately below wastewater discharge sites and within towns, but not in rural watersheds. Overall, aquatic concentrations of caffeine are typically less than for fecal sterols, but caffeine tends to stay in solution, whereas the sterols associate with fine particulates.

2.7.3 Optical Brighteners and Other Sewage Markers

Recent sewage contamination may be readily identified in waters by the presence of ammonium, turbidity/particle counts, phosphate, odor, and a range of organics present. Depending on the sensitivity and AWQC applications, some of these analytes may provide value in fecal source identification.

One relatively inexpensive and sensitive fecal source identification method is fluorometry (Hartel et al., 2007). Fluorometry identifies human fecal contamination by detecting optical brighteners (also called fluorescent whitening agents) in water. Optical brighteners are compounds added primarily to laundry detergents, and because these brighteners emit light in the blue range (415 to 445 nm), they compensate for undesirable yellowing in clothes (Kaschig, 2003). In the United States, 97% of laundry detergents contain optical brighteners (Hagedorn et al., 2005). Because household plumbing systems mix effluent from washing machines and toilets together, optical brighteners are associated with human sewage in septic systems and wastewater treatment plants. However, in order to use optical brighteners to detect human fecal contamination properly, they must be combined with use (counts) of fecal indicator bacteria. For example, effluent from a wastewater treatment plant contains optical brighteners, regardless of how effective the treatment processes have been at removing or inactivating pathogens. Thus, data on the presence of optical brighteners without accompanying data on viable fecal indicators does not provide information on the potential health risk from pathogens.

However, results of studies that have combined fluorometry with counts of fecal bacteria have been contradictory. Although various reports have documented a strong fluorescent signal and high numbers of fecal enterococci, cases of no correlation between fluorometry and counts of fecal bacteria have also been reported (Hartel et al., 2007). One key confounder has been the presence of organic matter that fluoresces and interferes with fluorometry. Yet, this interference can be reduced by adding a 436-nm emission filter to the fluorometer, which may reduce background fluorescence by over 50%. As long as the fluorometer used is equipped with a 436-nm filter, it appears that targeted fecal indicator sampling combined with fluorometry can be a relatively inexpensive method for identifying human fecal contamination in water.

In summary, chemical biomarkers appear to have niche applicability for those with the resources and expertise to use them and where such biomarkers are advantageous, such as where other less expensive MST options have shown to be unsatisfactory or provide ambiguous results.

2.8 Research Needs

2.8.1 Near-term (1 to 3 Years)

1. Validate the range and species or sub-species diversity qPCR assays identify, and how they may relate to health outcomes for recreational exposures (also using archived epidemiological study material) (**high priority**).
    a. Example priority list of organisms: enterococci, *Bacteroides*, *C. perfringens*, *E. coli*, F+ RNA coliphages, and somatic coliphages
2. Investigate the potential for speciation of enterococci to identify fecal-specific (preferably human) from environmental strains, then apply results to future MST and epidemiological studies (**high/medium priority**).
3. Ensure that archived samples (collected from epidemiological/specific studies) are suitably sorted and stored (to maintain their integrity) for future viability as well as molecular-based method comparison or validation studies for candidate indicators/methods (**high priority**).
4. Validate *C. perfringens* (SRC) assay's robustness over a range of water and sediment sample characteristics and correlate health effects relationships to this indicator (**high priority**).
5. Determine if there are *Bacteroides* analytical targets that are human-specific and validate their use over a range of geographic areas, diverse populations, climates, and water quality conditions to correlate levels to health targets (**high priority**).
6. Conduct health and epidemiological studies with as wide a range of microorganisms (indicators/MST organisms) as possible to identify risk correlations for a range of pathogens/indicators (including bacteriophages) from various nonhuman sources; at a minimum would include *E. coli*, enterococci, enterococci-qPCR, coliphages, *Bacteroides*-PCR, *C. perfringens*; where possible, *Bacteroides* phage GB-124, enterohemorrhagic *E. coli* (EHEC); and check for absence of human Norovirus-qPCR, adenovirus-qPCR, Pan-enterovirus-qPCR, polyoma viruses (**high priority**).
7. Conduct health and epidemiological studies with microorganisms from nonhuman sources such as *Leptospira* spp. in fresh and *S. aureus* and pathogenic *Vibrio* spp. in marine recreational waters and determine appropriate indicators for these pathogens (**medium priority**).
8. Conduct epidemiological studies incorporating the measurement of pathogens of interest (along with indicators) as monitoring tools in sewage in order to determine the correlations of the occurrence of these pathogens to indicators, and to better understand their association with diseases at downstream recreational locations. For instance, while it is strongly suggested that enteric viruses are major contributors to illness from swimming, there have not been prospective epidemiological studies to actually support this association. Use serology (also consider collecting saliva and possibly fecal

samples) to help identify the etiological agents from sewage that are impacting on recreational water sites (**high priority**).

   a. Conduct similar studies in recreational waters (above refers to studies in sewage) (**medium priority**).

9. Systematically identify and evaluate more reproducible, accurate, and cost effective methods to sample and identify priority pathogens or their index organisms (including the total adenoviruses, [e.g., Groups A-F and adenovirus 40/41], but also JC virus, and Norovirus) in ambient waters (**medium priority**).
10. Determine if there are any appropriate sewage associated bacterial pathogens that can adequately serve as an index of any of the currently known sewage-borne bacterial organisms to use on a more routine basis in recreational water criteria. For example, determine if monitoring recreational waters for *Salmonella* spp. bacteria and phages of *Salmonella* can fulfill the criteria of a pathogen index for sewage-borne bacterial pathogen can be developed (**medium/low priority**).
11. Conduct microbial fate and transport studies to determine relationships between traditional and new fecal indicators, index pathogens, and priority pathogens in treated effluents and in downstream recreational waters to compare and validate their applicability for specific criteria uses (**high/medium priority**).

2.8.2 Longer-term Research Goals

The research below may take longer than 2 to 3 years of research to complete. These are *not* presented in order of priority.

1. Review archived samples to look for trends in evolution of viruses (new or cyclic re-emergence of viruses) and the efficacy of current indicator targets used by molecular methods for health based correlations.
   a. Develop predictive models to understand the conditions that promote the emergence or re-emergence of new pathogens.
2. Continue to conduct additional epidemiological studies on non-point sources of fecal contamination and assess illness relationships to pathogen/indicators.
3. Continue to conduct sewage surveillance for pathogens as a means of public health surveillance and informing pathogen monitoring programs for CWA purposes.
4. Develop robust method for speciation of enterococci with a view to identify fecal-specific (preferably human) from environmental strains; then apply to future MST and epidemiological studies (assuming initial studies suggest that this should be explored further).
5. Conduct studies on beaches to characterize the usefulness of total adenoviruses (Groups A-F), adenovirus 40/41, JC virus, and Norovirus to meet recreational water quality criteria purposes.
6. Conduct health/epidemiological studies to identify a range of pathogens/indicators from various nonhuman sources of fecal contamination.

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CHAPTER 3
METHODS DEVELOPMENT

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3.1 Introduction

The Methods Development workgroup focused on addressing the following four key questions:

1. What are the attributes and criteria for deciding whether a new method or indicator is ready for adoption by EPA?
2. What kinds of studies are necessary to quantify those attributes?
3. Are there any new indicators/methods for which those studies have been conducted and that are ready for adoption?
4. What studies (or modifications to planned studies) are most critical for EPA to implement in the next 3 years to support adoption of new methods/indicators in a criteria development framework?

A critical starting point for the workgroup members was recognition that the evaluation of methods and/or indicators needs to be considered in context of the Clean Water Act (CWA) applications in which they would be used. The following five primary uses were identified by workgroup members:

- Routine beach monitoring to support public health warning notification systems;
- Routine beach monitoring data to support total maximum daily load (TMDL) decisions;
- Rapid methods to track the progress of a sewage spill as it moves downstream or downcoast to improve the beach closure determinations;
- Compliance assessments conducted at the terminus of National Pollutant Discharge Elimination System (NPDES) discharge pipes; and
- Trends assessments to determine whether water quality conditions at a site are changing over time.

The workgroup focused on the first two applications beacause members felt that they are most relevant to EPA's desire to redefine their current recreational water quality criteria. However, several workgroup members also recognized the relevance of the other applications so a short section is included (see Section 3.7) that illustrates the similarities and differences in the method evaluation process for these other CWA uses.

For water quality notification systems, two principal issues were identified that need to be addressed. The first is that current laboratory measurement methods require up to 24 hours to enumerate indicator bacteria. Contaminated beaches remain open during this processing period, but indicator bacteria may already have returned to acceptable levels by the time laboratory results are available and warning signs are posted. Continued advances and improvements in molecular- and immunological-based techniques provide new opportunities for measuring bacteria more rapidly. Although current (traditional/standard) methods rely on bacterial growth and metabolic activity, these new methods allow direct measurement of cellular attributes, such as genetic material or surface immunological properties. By eliminating the necessity for a lengthy incubation step, some of these methods have the potential to provide results in less than 4 hours, enabling managers to take action to protect public health (i.e., post warnings or close beaches) on the same day that water samples are collected. This assumes that samples can be processed at the beach or that the time required for transportation to a laboratory is brief. For same day posting to be achieved, the results of the tests also have to be delivered to and evaluated by beach managers in a timely manner.

The second issue workgroup members identified is that present standards used to evaluate recreational water quality data are based on a "one-size-fits-all model," relying on use of a single indicator (e.g., enterococci at marine beaches) and a single standard for all recreational waters. There is growing recognition that enterococci measured on the beach may derive from many sources, including humans, domesticated animals, indigenous wildlife (including shore and migratory birds), and regrowth in sand, sediments, or on biofilms. The health risk to humans varies depending on which of these sources is responsible for the measured enterococci. As such, existing warning systems do not provide an equal level of health risk protection at all beaches. Moreover, the costly cleanup processes associated with the TMDL programs are not necessarily focused on the beaches that represent the greatest public health risk. There are additional concerns that cleanup activities, and associated costs, are being targeted at beaches where enterococci concentrations that exceed standards result from natural sources and processes.

EPA could consider two means of adjusting their criteria framework to address one-size-fits-all concerns. The first adjustment is to develop additional indicators to replace, or to augment in a tiered fashion, the existing enterococci indicator as it is now used at marine beaches (US EPA, 1986). These new indicators would be more specific to human sources and better related to human health risk than the existing indicator.

The second potential adjustment is to adopt a framework similar to that of the World Health Organization (WHO, 2003), in which watershed characterization studies are used to adopt site-specific standards. These site-specific standards would be based on perception of health risk resulting from the types of fecal sources in the watershed and the proximity of those sources to the beach. The Methods workgroup members felt strongly that source identification methods needed to be a key tool in characterizing risk and that further evaluation of source identification methods needed to be conducted if they are to be used in this context.

This chapter is organized around describing the approach that would be used for assessing methods/indicators in the following three contexts: (1) replacement of existing methods with more rapid methods, (2) replacement of existing indicators with those that are more specific to human sources of fecal contamination, and (3) determination of source identification methods that can be used to characterize risk in the development of site-specific standards. Within each section, the adequacy of evaluations of methods/indicators is discussed and the most immediate research activities that would provide the greatest benefit to EPA for modifying monitoring and/or indicators within the next 5 years are highlighted.

3.2 Classes of Indicators

The evaluation of methods is a critical element in bringing new technology to the measurement of water quality. Current evaluation protocols were developed for cultural methods for enumerating bacterial indicators of fecal contamination. The evaluation usually included method attributes regarding the performance of the method, such as specificity, accuracy, and precision. Further evaluation that addressed how the method performed in and between laboratories included multi-laboratory testing that determined how robust a method might be (i.e., how poorly can the method be performed and still produce useful results?). The question that arises is whether the current protocols for evaluating membrane filter culture-based methods are suitable for evaluating new methods that are being proposed for measuring water quality. Some of the new or alternative methodologies that are available for testing water quality include molecular-based methods, such as quantitative polymerase chain reaction (qPCR), nucleic acid sequence based amplification (NASBA), and transcription-mediated amplification (TMA). These methods amplify nucleic acid sequences to high levels such that they can be easily detected. Other methods use antibodies to which fluorescent compounds are attached. The fluorescent-tagged antibodies then attach to specific microbes and are "counted" in a flowcytometer. The preceding methods "count" dead and live bacteria and thus differ significantly from currently used quantitative cultural methods.

Some recent methods do measure viable microbes in an indirect manner. For example, enzyme-based methods measure substrate utilization employing compounds that fluoresce when metabolized by specific bacteria. Comparison of the fluorescence to a standard curve allows a "count" to be established. Another method measures adenosine triphosphate (ATP) using a bioluminescence measuring instrument to determine the amount of ATP that is produced only from viable bacteria.

In the current context, there are indicators available or in late stages of development that are ready for evaluation to determine if they are appropriate for use in routine beach monitoring. Some can be measured with the technology described above while others can be measured with currently available methodologies.

Leading candidates are indicators and detection methods that can be used to replace current culture-based indicators of water quality (i.e., enterococci and *E. coli*). For instance, nucleic acid sequences from enterococci have been used to measure the density of enterococci in bathing beach water. Some aspects of the performance of this method have been completed. As described above, enterococci have also been quantified using a fiber optic/fluorescent antibody detection method, an enzymatic/substrate method, and a method that measures ATP. None of the latter methods have been evaluated with respect to either their performance characteristics or for robustness. Similarly, molecular-based methods that measure viruses (e.g., adenoviruses) that might replace currently used indicators of recreational water quality have not had their performance evaluated. If these indicators are shown to be effective in their performance, they will be candidates for use in epidemiological studies to determine how well their densities in recreational waters relate to swimmer health.

Another class of microbes and other analytes are related to identifying the source of fecal contamination that might affect beach microbial water quality. Other markers include genes such as the *Esp* gene from enterococci, which might be specifically associated with human feces; male-specific (F+) coliphage that can indicate whether water has been contaminated by humans or animals; and chemical markers such as optical brighteners, caffeine, coprostanol, and urobilin that may be associated with human use or are the end-products of human metabolism (see also

Chapter 2). Optical brighteners are measured quite easily with a spectrofluorometer, while caffeine, coprostanol, and urobilin require more complex instrumentation, such as a high performance liquid chromatography (HPLC) instrument. Measuring genomic markers is less complicated and does not require a thermocycler to perform a PCR test. The varied nature of these source identification markers may require modification of the performance evaluation criteria to accommodate the different characteristics of these source specific analytes. For instance, the range of applicability and practicality may be more important than the accuracy and precision characteristics of these chemical or genomic source identification approaches.

The last class of indicators that may be ready for evaluation as indicators of fecal contamination are those that may have been rejected previously, for whatever reason, but should be considered again because of the availability of new information about their occurrence in water or because of new methods for their detection. Other potential indicators may be candidates because they are species within a group indicator, such as the enterococci and clostridia, and individual species may better indicate the quality of a waterbody. It is likely that this class of indicator will fit well into the current paradigm for characterizing microorganisms that might be used for routine recreational water quality monitoring.

3.3 Evaluating New Methods for Existing Indicators

Workgroup members felt that after a method passes defined performance criteria, it must be evaluated for its application as an equivalent (or superior) water quality tool compared to the current assays. An example of when this approach might be used is the transition from culture-based enterococci detection to detection by a rapid (molecular-based) assay.

The workgroup identified two major approaches to conduct this evaluation, (1) determining the relationship to health risks based on epidemiological studies or (2) establishing equivalency to an existing water quality tool.

3.3.1 Health Risk Evaluation

Workgroup members felt that determining the relationship to health risk is the best approach to evaluating a new method. An epidemiological study that can associate human risk with a new method is the preferred approach. The new detection method will ideally show an improved relationship to illness and will therefore be more protective of public health than the current approach that relies on indicator detection. Likewise, if the new method offers other improvements over the existing method (e.g., more rapid, less costly, etc.), then its relationship with human health should be at least as good as the current indicator.

The health risk evaluation should also be used when the target of the new method differs significantly from the current system. For example, a culture-based enterococci assay does not measure the same thing as a polymerase chain reaction (PCR)-based assay, which detects DNA rather than culturable (viable) cells. In these cases, a direct comparison of methods (as described below) may not be appropriate or possible.

3.3.2 Establishing Equivalency between New and Standard Methods

The equivalency validation approach assumes that for methods with similar targets (e.g., viable cells), the performance of the new method can be compared to that of the existing method without the need to determine health risk directly. Given the cost and time involved in large scale epidemiological studies, the equivalency approach can be performed for many new methods. The EPA should determine how dissimilar the method targets can be and still be evaluated by this approach. For example, cellular activity-based assays (e.g., immunomagnetic separation and ATP bioluminescence [IMS/ATP]) and membrane-filtration assays both measure viable cells, even though the end points are different. The workgroup members suggested that this activity-based assay is similar enough to be evaluated through equivalency validation. This level of flexibility is important because of the limited number of epidemiological studies that can be carried out in the near- or long-term.

The current EPA (2003) protocol, *EPA Microbiological Alternate Test Procedure (ATP) Protocol for Drinking Water, Ambient Water, and Wastewater Monitoring Methods*, provides a suitable vehicle for performing these evaluations. The EPA recommends approval of a proposed method if it is similar or better than the approved method (the "gold standard") for 80% of the matrices tested. Currently, only culture-based methods can be included as an alternate test procedure; therefore, consideration should be given by the EPA on the comparability of other methods (as mentioned above).

Along these lines, California has adopted equivalency validation between methods with different targets (i.e., culture-based versus PCR-based). This protocol, *Beta Testing of Rapid Methods for Measuring Beach Water Quality* (SCCWRP, 2007), provides guidelines for comparing between methods. Similar to EPA, this validation compares method performance between multiple sample types and laboratories and also sets acceptable variability between results at 0.5 log (based on within method variability previously reported [Griffith et al., 2006; Noble et al., 2003]). Additionally, the precision should be equal to or better than for the existing methods.

Many workgroup members felt that EPA's protocol is too prescriptive because it disallows applications for methods that are not culture-based. In the future, for example, should the IMS-ATP test be found to have a health risk-based association, EPA should consider allowing its comparison to culture-based methods since both assay for live organisms, albeit not exactly via the same mechanism (membrane-filtration colonies or Most Probable Number [MPN] results versus ATP occurrence). By the same token, workgroup members felt that the California protocol was too relaxed in that genetic methods were compared to culture-based methods for the purposes of acceptance of the former. Because these methods do not measure the same targets (DNA versus membrane-filtration colonies), this was perceived as comparing "apples to oranges" as the criteria for making such comparisons are not yet well established.

3.4 Performance Criteria

Regardless of which of the two evaluation approaches is chosen (health risk-based or method equivalency-based), performance criteria for the method should be completed, and preferably before using the method in an epidemiological study to obtain heath risk-based association data.

Workgroup members consider the following to be the major parts of performance criteria: repeatability, accuracy, specificity, sensitivity, robustness, range of applicability, and practicality. These performance criteria are summarized below.

Repeatability asks the question: if a test is repeated, will the results be the same? Note, this does not take into account the degree of error with regard to how well the test does at identifying its target (accuracy). For example, if a person is throwing darts at a target, repeatability is the measure of how often the darts hit a specific place. Repeatability does not measure whether one hits the center of the target or not—that is accuracy (see more below). Repeatability is sometimes referred to as precision and can be expressed both on an absolute scale (i.e., standard deviation) and on a relative scale (i.e., relative standard deviation [RSD]). The RSD (sometimes referred to as coefficient of variation) is calculated as the standard deviation divided by the mean, expressed as a percent. For the purpose of summarizing data, both standard deviations and RSDs should be calculated. Generally, RSDs are most appropriate for summarizing precision when variability increases as concentration increases. To provide an indication of the effect of multiple matrices on precision, standard deviations should be calculated separately for each matrix as well as for the method over all matrices. In addition to within and among matrix/matrices for repeatability, it is important to test intra- (within lab) and inter-laboratory (among labs) repeatability to ensure consistency.

Accuracy measures the degree to which the method identifies its target. It is defined as the degree of agreement between an observed value and an accepted reference value. Accuracy includes random error (precision) and systematic error (recovery) that are caused by sampling and analysis. Using the above dart example, this would be the number of times that the dart hits the “bulls-eye.”

Specificity includes the false positive and false negative rates. The false positive question asks if the method is significantly more likely or less likely to detect non-target organisms or other sample constituents that would be reported as the target organism by the analyst when compared to the reference method. To assess whether the false positive rates are significant, replicates known to contain non-target organisms that could be falsely identified as the target organism should be analyzed. The determination that the samples do not contain the target organism should be based on a third independent standard method. For example, if the target organism is cultured *E. coli*, the test should be used against, at a minimum, other enterobacteria, and, depending on what the test is, potentially Gram positive organisms as well. If the test is for genetic material, then the primers and probes should be tested against GenBank to look for potential false positives from non-*E. coli* species with the same sequences. Specificity also asks the false negative question regarding whether the new method is significantly more or less likely to exhibit non-detections for samples with the target organism or to exhibit results that are biased low when compared to the reference method. To assess whether the false negative rates are significantly different between methods, replicates known to contain target organisms should be analyzed. As in false positive studies, the determination that the samples do not contain the target organism should be based on a third independent standard method. For example, if the target organism is genetic material from *E. coli*, then a method for culturable *E. coli* can be used.

If the culture method is able to detect *E. coli*, then the genetic method should, in general, also detect *E. coli*.

Estimates of false positive and negative rates as percentages can be calculated as follows:

1. false positive rate = # false positives/(# of true negatives + false positives) × 100%; and
2. false negative rate = # false negatives/(# true positives + false negatives) × 100%.

The sensitivity of a test is the analytical detection limit of the test (the smallest amount detectable using the method). For chemical methods, the sensitivity may be defined as the minimum amount of a particular component that can be determined by a single measurement with a stated confidence level. Generally, these refer to instrument analysis; thus, it is the lowest quantity of a substance that can be distinguished from the absence of that substance (a blank). For microbial methods, sensitivity is the limit of detection of a particular method. In general, methods are not used at this level since confidence around that level is lower and more subject to user error.

The robustness of a test is the degree to which the method can perform in the presence of incorrect inputs or stressed conditions. More simply, how poorly can a method perform and still produce useful results? For example, does the method perform as intended in the hands of a semi-novice user (e.g., a qPCR method performed by a person familiar with molecular-based methods including PCR but not qPCR)? If the test is for cultured microorganisms, can it detect stressed organisms in ambient waters (e.g., the EPA *E. coli* methods have a 2-hour resuscitation step at a lower temperature for stressed organisms)? Robustness is not a measurable attribute per se but must be considered and applied for overall method performance.

The range of applicability should also be considered as it answers the question: is the test reliable on a nationwide basis (e.g., does it work equally well in temperate and tropical climes, in the Great Lakes and other inland waters, etc.), in the presence of inhibitors (e.g., turbidity, alkalinity, organics [humic acids]), and in a variety of matrices (e.g., sewage, septic tanks, urban runoff, agricultural waste, known animal sources)? In general, the range of applicability does not apply to matrices other than the one for which the test was designed; that is, a recreational water quality method should not be expected to perform equally well for sewage sludge. Like robustness, this is not a measurable attribute but must be considered and applied for overall method performance.

Workgroup members felt that practicality should also be considered when considering a method. This issue is largely addressed in Chapter 7 (Implementation Realities workgroup). However, four main issues were considered important enough to be mentioned here—capital cost, training cost, per sample cost, and additional sampling requirements. Capital costs include the upfront costs such as equipment purchase and the actual space required for the test. For example, when performing genetic testing, aside from the equipment needed (e.g., platform [specific machine], laminar flow hoods, dedicated pipettors), space is needed, ideally in separate rooms, for reagent preparation (material not containing any genetic materials). Space is also needed for the two types of sample preparation, those containing high target sequence DNA concentrations such as DNA standards and calibrator samples, and those containing expected low target sequence DNA concentrations (e.g., filter blanks and water samples)—the latter of which should also be in separate laminar flow hoods. Training costs are those incurred prior to routine testing so that the user can perform the test within the performance criteria of the test; these may include participation in a workshop for hands-on experience or completing a training module. The other two issues regard routine use of the test. A high per sample cost may become an issue if a large volume of tests need to be completed on a routine basis. Additional sampling is generally an effort that results from rapid testing. For example, if an early morning sample yields, after 4 hours, a positive result resulting in beach closure, it may then lead to additional sampling to determine if the beach still needs to be closed in mid-afternoon. It should be noted that many laboratories (at least in California) do not object to capital or training costs, but take issue with a high per sample cost or with additional sampling requirements.

3.5 Evaluation Process for Alternative (New) Indicators

Currently, recreational water quality is assessed with a single indicator with a single threshold (i.e., a "one-size-fits-all" approach). Under consideration is the implementation of alternative indicator(s) that are better associated with human health risk than the enterococci. These alternative indicators could theoretically replace the current standard but still be used in a one-size-fits-all approach or could be targeted for specific applications (e.g., one indicator may be best associated with risk in tropical marine waters, another in temperate marine waters, and another in freshwaters). Regardless of the final implementation, any new proposed indicator will need to be vetted through performance based standards.

The system of approving an alternative indicator will follow the same process as outlined for the assessment of any indicator or method, although there will be key differences.

- Any proposed indicator and/or method should be evaluated for the following performance characteristics:
  - repeatability (i.e., precision);
  - accuracy;
  - sensitivity;
  - specificity (false positive/false negative);
  - robustness;
  - range of applicability; and
  - practicality.
- *After* performance characteristics have been demonstrated and the indicator and associated method has been *determined to have adequate performance*, it then should be evaluated for its use and application in a water quality criteria, including:
  - relationships to health risks must be established based on epidemiological studies covering an array of beach types and/or geographic areas; and
  - because of lack of comparable standards, a new indicator cannot be evaluated based upon equivalency to an existing method.

This approach would establish the basis for alternative (new) indicators, and leads into the possibility that such indicators could also serve in a role as source identifiers.

3.6 Evaluating Source Identification Methods – Proficiency and Evaluation

When bacterial levels in recreational waters exceed adopted State Water Quality Standard, the potential risk to the public health requires local authorities to post advisories or close swimming areas, risking significant losses in local revenue. The goal of microbial source tracking (MST), as applied to U.S. waters, is to accurately identify the contributors and, if possible, the relative proportions of fecal pollution from all potential sources, or at least the major contributors. Proper use of MST can assist watershed managers in implementation of best management practices (BMPs) that can reduce fecal inputs, thereby limiting or reducing public health risk.

Two major classes of microbe-based and one class of chemical-based MST methods are currently being developed and utilized in surface waters across the world (Blanch et al., 2006; Stoeckel and Harwood, 2007). Although there has been significant progress in the MST field over the past decade, variability among performance measurements and validation approaches in laboratory and field studies has led to a body of literature that is very difficult to interpret, both for scientists and for end users (Stewart et al., 2003; Stoeckel et al., 2004). This section lists and defines/describes performance characteristics that should be uniformly applied across MST studies, although selection of which criteria from the following list to use will vary somewhat based on the target. All methods and MST projects need to include some considerations for representative sampling, sampling frequency, sample volumes required, and the number and choice of source categories. Although the use of a toolbox approach has been important in MST studies, there is a desire to develop an appropriate tiered approach to avoid costs and time from using multiple methods simultaneously. Within the MST community, and largely as a result of the method comparison studies, library-independent methods are currently the priority, while chemical-based methods appear to be desirable for rapid screening and presence-absence tests (with perhaps quantification in the future). Library-based methods still have a role in MST, but only in those circumstances where detailed information is needed, such as many TMDL-based studies.

3.6.1 Library-independent Methods (also Reported as Sample-level Classification)

Examples (not comprehensive) include both molecular approaches (*Bacteroidales*, *E. coli* toxin, *Enterococcus Esp* gene, direct measurement of source-specific viruses (polyoma, adenoviruses, enteroviruses, phages, etc.) and microbe-based approaches (*Clostridium perfringens* [alternative indicator], source-related clostridia, source-related enterococci, sorbitol fermenting bifidobacteria [human], *Rhodococcus coprophilus* [grazing animals], human-specific bacteriophages, phage typing, etc.).

Method evaluation includes the following eight performance criteria:

1. Accuracy is defined as the true positive or success rate—if a method identified the presence of the target in 98 out of 100 blind samples, the accuracy would be 98%;
2. Rates of false negatives and false positives of the target are used to describe specificity;
3. The analytical detection limit of the test is used to describe sensitivity;
4. The level of target-host specificity and the range of target-host distribution;
5. Efficiency of recovery of the target from different environments;

6. The reproducibility of analytical results, both inter- and intra-laboratory;
7. The suitability of marker detection (and/or quantification) to meet study-specific objectives; and
8. Detection of several of the above, especially #4 and #5, can be referred to as robustness.

3.6.2 Non-microbial Methods (also Called Chemical Methods)

Examples of non-microbial indicators include, but are not limited to, optical brighteners, host-derived DNA (e.g., eukaryotic mitochondrial DNA), fecal sterols/stanols, and source-specific fecal compounds such as caffeine and pharmaceuticals for humans.

The performance criteria in numbers 1 through 8 above, excluding #4 and #5, apply to non-microbial methods. For chemicals, the analytical detection limit of the test is usually applied to describe both sensitivity (#3) and the efficiency of recovery of the target from different environments (#5).

3.6.3 Library-based Methods (also Called Isolate Matching)

Examples of library-based methods include but are not limited to both molecular approaches (pulsed-field gel electrophoresis [PFGE], ribotyping, PCR with different primer sets, etc.) and phenotype-based approaches (antibiotic resistance analysis [ARA], biochemical, etc.).

The performance criteria in numbers 1 to 3 from library-independent methods (above) are applicable for library-based methods. In addition, the following four criteria apply:

1. Jackknife (also reported as holdout or cross-validation) analysis and the pulled-sample test (recently described as internal proficiency) should be done on each and every library (Stoeckel and Harwood 2007);
2. Library should shave clones removed to reduce redundancy, based on the precision of the typing method;
3. External proficiency or blind tests to determine both size and representativeness of the library should be done as the library is developed; and
4. The benefit-over-random statistic should be used when accuracy is determined, and should be performed on both the library and the external proficiency (or blind) set.

3.7 Modifications to the Evaluation Process When Indicators are used for Other Applications

Indicators are used in many different contexts. Routine beach monitoring, the most time-critical use of indicator bacteria is described extensively in other chapters of these proceedings. This section briefly addresses other (secondary) uses of indicators. Another use of indicators is as an early warning system that would provide evidence for an imminent human health risk, such as a sewage spill. They can also provide evidence of returning to acceptable ambient water quality conditions as designated by the criteria. It is important that the methods be highly specific and robust. Because of the potential for illness in exposed populations, it is extremely important that this use of an indicator be associated with great specificity and robustness. Specificity in this case refers to the ability of a method to detect an indicator with certainty that the indicator is not giving a false positive response (i.e., an organism or analyte that responds similarly to the target organisms, but is not the target organism). Similarly, target microbes that do not provide a positive response are indicated as false negatives and too many of these could result in a false sense of security that would be highly unacceptable from a public health perspective. Robustness in this case means that the method can be abused and still function properly. Methods of this type are usually used under extreme conditions where the correct result must be obtained in a very short time period.

Another use of indicators is for compliance monitoring purposes, such as monitoring sewage treatment effluent for EPA's NPDES Program. Important characteristics for indicators used for such compliance monitoring are precision and specificity. The precision is necessary because sewage treatment plants would receive a fine(s) if limits of the permit are exceeded. The specificity, both false negative and false positive responses, are important for the same reason mentioned above and may influence the way beaches are managed.

Trend assessments are used to determine whether water quality conditions at a site are changing with time. The most important characteristic is precision that contributes to the ability to detect small changes over time (i.e., whether the water quality is decreasing or improving over time). If the water quality decreases then bathing may no longer be allowed. Conversely, if the water quality improves sufficiently then bathing may be re-allowed.

3.8 Research Needs

Several lines of research should be pursued in order to implement improved methods for (1) rapid detection of current water quality indicators, (2) implementing alternative indicators that are more protective of public that the current indicators, or (3) determining source (human or nonhuman) at beaches. This set of research priorities is based upon the current state of available methods and the projected feasibility of implementation in near-term (1 to 3 years) and mid-term (2 to 5 years) or longer timeframes. Although these are listed in priority order, the workgroup members felt that they largely expand on efforts that EPA or its potential partners have already initiated and all are achievable in the next 3 years. Appendix G summarizes currently planned measurements for use in the upcoming Doheny and Malibu Beach (California) epidemiology study.

1. Systematic evaluation of performance criteria for library-independent source identification methods (for use in source characterization [i.e., human versus nonhuman fecal contamination] and in MST) (timeline: 1 to 2 years).

   Workgroup members felt that EPA should fast-track studies to evaluate the performance criteria of source-specific microbial targets.

   A series of controlled trials representing a variety of geographical areas should be conducted to evaluate promising methods. Studies should include samples spiked with known source fecal matter from multiple hosts as well as environmental samples collected from areas with known dominant sources of fecal contamination. Samples should be assayed by the test methods in several laboratories using blinded controls. These protocols would be similar to those used in the Griffith et al. (2003) studies that EPA co-sponsored approximately 5 years ago, but which need to be updated as new methods have developed and existing methods refined.

Although there are many potential methods that could be included in such studies, the workgroup members identified the following as the most important:

1. enterococci *Esp* gene;
2. *E. coli* virulence genes;
3. human enteric viruses (molecular detection);
    a. DNA-based – adenoviruses and polyomavirus;
    b. RNA-based – enterovirus and norovirus;
4. *Methanobrevibacter smithii* (*nifH* gene);
5. *Clostridium perfringens*;
6. coliphage; and
7. *Bacteroides* human-specific markers.

The last two methods are also being planned for use in EPA's upcoming (2007) health risk (epidemiological) study. The workgroup members felt that the coliphage and *Bacteroides* methods are more advanced than the others and endorses their inclusion in source identification studies.

In coordination with trials over various geographic areas, candidate methods should also be evaluated from the perspective of persistence of genetic or chemical or microbial targets in both primary and secondary habitats (sediments) over longer time periods (multi-year). Although this may be a longer term goal, eventually all methods that appear to be suitable for use regulatory or management-level decisions will need such to be examined over time periods sufficiently long so that there is confidence that the desired targets do not change, or that changes can be captured and dealt with if they do occur.

2. Evaluation of chemical indicators for human sewage (timeline: 2 to 3 years).

Several possible chemical markers of sewage have been reported and have the potential to be used in a rapid to real-time assessment of source. Coordinated studies to evaluate the performance criteria over multiple labs are needed to implement these assays.

The following analytes should be included in near term evaluation studies:

1. optical brighteners;
2. coprostanol; and
3. caffeine.

At least one multi-laboratroy evaluation study of optical brighteners is currently being developed by individual investigators (Hartel et al., 2007).

3. Continued evaluation of rapid assays for the detection of enterococci in human health risk (epidemiological) studies (timeline: 1 year and beyond).

   Rapid detection of current water quality indicators are proposed to allow same day evaluation of water quality. To implement these assays, continued evaluation of the health risk relationship is needed. For qPCR (*Enterococcus*), more epidemiological studies from a range of beach types are needed before implementation. Additionally, other rapid assays for enterococci have been developed and should be evaluated in upcoming and future epidemiological studies.

   Methods under consideration for enterococci detection include the following:

   Immediate (timeline: 1 to 2 years):
   - qPCR (detection of DNA); and
   - TMA (detection of RNA).

   Mid-term (timeline: >2 years; require additional performance evaluation):
   - IMS/ATP (detection of activity);
   - RAPTOR™ (antibody-based detection)[5]; and
   - enzymatic detection.

4. Evaluation of alternate indicator candidates in human health risk (epidemiological) studies (timeline: 1 year and beyond).

   Potential alternate indicators (i.e., to replace enterococci and *E. coli*) that have already been vetted for performance criteria should be included in any future epidemiological studies of recreational waters to determine their relationship with health risk.

   The following indicators should be evaluated within the next two years:

   - *Bacteriodales* human specific markers; and
   - F+ coliphage (antibody).

   Other candidates indicators should be added for evaluation as they meet required performance criteria (as listed above)

5. Optimization of sampling, recovery, and processing methods for efficient concentration, processing and detection of rapid, alternative or host specific indicators (Time line: 1 year and beyond).

[5] http://www.resrchintl.com/raptor-detection-system.html

Additional methods need to be optimized for source specific microbial targets. Studies should address issues such as optimization of sample volume, processing/concentration methods, and extraction/purification methods (especially for targets expected to occur at low numbers in the environment).

Furthermore, research addressing straightforward techniques to enumerate *Enterococcus faecium* and *faecalis*, rather than the larger *Enterococcus* group that is presently measured, are needed as the individual species are more likely to be associated with human sewage/feces. Performance-based criteria tests are also needed for these species.

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CHAPTER 4
COMPARING RISK (TO HUMANS) FROM DIFFERENT SOURCES

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4.1 Introduction

Fresh and marine recreational waters and beaches may be impacted by human and/or animal feces from point and non-point sources. Studies have recently been completed by EPA on assessing rapid water quality indicators and their ability to predict swimming-associated illness at freshwater beaches impacted by publicly (and privately) owned (sewage/wastewater) treatment works (POTW) systems. Similar EPA studies are currently planned (starting summer of 2007) to assess the risk of illness for people who swim in marine recreational waters impacted by POTW systems (point sources of fecal contamination). Thus, in the near future additional information should be available on risk of illness for bathers at marine beaches largely impacted by human sewage. Plans are also underway by the Southern California Coastal Water Research Project to assess swimming risks at least one marine beach that is impacted by non-point source sewage that likely contains a mixture of human and animal feces. However, there remains a paucity of data on the risk of illness for swimmers at beaches exclusively (or primarily) impacted by feces from animals. The absence of such data makes it difficult to interpret the health significance of the frequent and persistent elevated fecal indicator levels in such waters that have been attributed to animals in many locations throughout the United States.

It is widely believed that human feces pose a larger health risk than animal feces to swimmers and other primary contact recreational water users. This belief derives from the basic concept that virtually all enteric pathogens of humans are infectious to other humans, while relatively few of the enteric pathogens of animals are infectious to humans. Possible exceptions are bird flu virus and swine hepatitis E virus (HEV). Workgroup members regarded the evidence for swine HEV transmission by water to be very weak and felt that it could disregarded in terms of risk assessments during the next 2 to 3 year EPA planning period. Bird flu was discounted as a major concern for swimmers because it was felt that if an outbreak of bird flu was recognized in birds or humans in the United States, early public health recommendations would include directives for people not to swim in waters that might be impacted by bird or human feces, including chlorinated public pools.

Counterbalancing the concept that animal feces may pose a lower risk is recognition that animals do harbor many bacterial and protozoan pathogens that pose a human health hazard and that some of these pathogens, such as enterohemorrhagic *E. coli* (EHEC), can cause serious, potentially life-threatening illness in humans. In addition, animal feces are often directly deposited in freshwater that receives no treatment before reaching bathing areas. The concentration of both feces and pathogens may be sufficiently high at beach locations at various times to pose a significant health risk to swimmers.

The bottom line is that there are few data to demonstrate whether animal feces pose a lower, greater, or equivalent health risk to bathers than human feces. If there is a difference, it would be helpful to know the magnitude of that difference in order for EPA to make appropriate public health recommendations. The only way to get a better sense of the health risk for swimmers posed by animal feces is to conduct targeted studies. Some types of studies (epidemiological and quantitative microbial risk assessment [QMRA] studies) would produce quantitative estimates of risks while others (fate and transport, pathogen loads in water, etc.) would provide supporting information or stand alone qualitative information about risk.

It is recognized that there are many different types of animals and that the pathogen risks posed by feces from these animals are different. These differences, as well as the different pathways (point, non-point, fecal deposition on land versus in water, etc.) that feces reach bathing areas, have to be taken into account in weighing risk. Workgroup members approached the issue by developing Table 5 in order to rank the likely risks from different sources of fecal contamination and to help prioritize which bather/animal-fecal-risk interface studies should be undertaken first.

The initial workgroup member discussion focused on assessing the universe of pathogen sources of interest to recreational waters. Workgroup members developed a table (Table 5) in which the major sources of fecal contamination categories are in rows. The major rows are wildlife, agricultural animals, domestic animals (pets), human/sewage, and what the workgroup termed "secondary environments" (i.e., soil, sand, and sediments). The wildlife row is subdivided into aquatic birds and all others. The agricultural animals are divided into poultry and other (largely comprised of domestic livestock such as cattle, sheep, and pigs). The human/sewage is divided

Table 5. Comparing Risks (to Humans) from Different Pathogen Sources.[a]

Source	Viruses	Protozoa	Bacteria	
Wildlife				
Aquatic birds	N	L	L-M	
Other (e.g., deer)	N	M	M	#2 priority
Agricultural animals				
Poultry	N	N	M-H	
Other (e.g., cattle, sheep)	N	M	M-H	#1 priority
Domestic animals				
Pets (e.g., dogs, cats)	N	L	L	
Fecal shedding by bathers				#3 priority
Adults	L	L	L	
Children	H	H	H	
Sewage				
No treatment (combined sewer overflows)	H	H	H	
No treatment (separate storm sewer overflows)	?*	?*	?*	
Secondary treatment**	H	H	M	
Plus chlorine**	H	H	L	
Plus UV	M-H (L with increased energy)		L	
Secondary environments***	L	L	M	

[a] Does not have an explicit fate and transport component
* Risk largely depends on amount of human feces present
** Focus of most (U.S.) recreational water epidemiological studies
*** Sediment suspension and contact with beach sand

N = estimated no or negligible risk, L = estimated low risk, M = estimated medium risk, H = estimated high risk into untreated sewage, secondary treatment sewage, chlorinated sewage, and UV-treated sewage. Fecal shedding by bathers (adults and children) is considered separately.

The columns are defined by broad microorganism groups of viruses, protozoan and bacteria. By an expert opinion process (within the workgroup) each cell of the table was given a risk estimate of no (zero) or negligible risk (N), low, medium, and high (L, M, H). The types of characteristics discussed included infectious dose, numbers of pathogens per gram of stool from infected animals, implication of source in waterborne disease (extended discussion on foodborne disease and vector-borne disease), persistence and survival in the environment and finally an assumption that sources are in close proximity to a primary contact recreational area. The N, L, M, H risk designations in the table cells represent the workgroup's "best guesses" and assumed that animal feces was deposited in freshwater relatively closed to bathing sites. The workgroup did not specifically address pathogen "die-off" associated with fecal deposition on land (spring/summer temperatures resulting in pathogen drying, transport from soil to water affects on viability, etc.). It was felt that many of these types of data are available and that the table could be updated with real data at a later date as a separate project. It was recognized that updating the table with published data might change the values in one or more risk rankings of the table cells.

With rare exception, viruses are species-specific. Essentially, all enteric oral/fecally transmitted viruses that infect humans are of human origin. For all of the animal viral sources of pollution, the viral cells were given a zero or negligible risk (indicated by "N" entries in Table 5). All the human sources were given a high risk estimate with the exception of UV-treated sewage. UV-treated sewage at current levels has up to a 0.5-log reduction of viruses and hence this cell was assigned a medium risk. More energy intensive UV irradiation may provide up to a 4-log viral reduction and result in a low risk ranking. Sentinel viruses for this group include enteroviruses, hepatitis A virus, norovirus, rotovirus, and adenoviruses. The major protozoan pathogens of concern are *Giardia* and *Cryptosporidium*. Given the current knowledge of infectious dose, the long survival in the environment, many of the animal cells within the table were given a low, low-to-medium, or medium risk level. As with the viruses, all the human cells within the table were given a high risk rating with the exception of UV-treated sewage. The bacteria had similar ratings to the protozoa ranging from low-to-medium and again, the human sources were all assigned a high ranking with exception of chlorine- and UV-treated sewage that received a low risk ranking.

Bather density was divided into adults and children (recognizing that children could be divided into specific age groups) with the assumption that hygiene and accidental fecal discharges were much more likely to occur in children than adults. Thus, for adults, a low risk ranking was assigned across the columns and a high risk ranking was assigned for children.

Based on the few studies done on secondary environments, viruses and protozoa were given a low risk rating, while bacteria were given a medium rating.

In developing Table 5, workgroup members noted the following discussion points:

1. Current epidemiological literature suggests that the symptomatic profile of swimming-associated illnesses indicates primarily viral illnesses.
2. Certain pathogens such as EHEC have a low probability of occurrence but are associated with severe a health outcome.
3. Information available to the workgroup suggested that nonhuman fecal sources impacted freshwater sources more than marine water sources.
4. Combined sewer overflows (CSOs) were considered as untreated sewage.
5. Separate storm sewer overflows initially were put in the domestic animal row but subsequent discussion of recent studies suggested that they could have a human component in many communities.

In discussing the future research needs related to the development of new or revised recreational water quality criteria, the workgroup members defined the ultimate goal to be a determined quantitative risk estimate for each fecal source (row). The benchmark by which risks should be compared is the secondary and chlorine treated sewage row that is currently the focus of recently completed EPA National Epidemiological and Environmental Assessment of Recreational (NEEAR) epidemiological studies for freshwater and the planned marine water studies. The following research projects were suggested to meet that objective of determining a sound and defensible risk estimate for each row of Table 5.

4.2 Summary of Workgroup Discussions and Reflections on Workgroup-specific Charge and Questions

The charge to the workgroup was to consider the impact of waterborne pathogens from various sources, both human and nonhuman, on the health risk resulting from exposure to fecal contamination in recreational waters. Workgroup members considered the impact of the issue on beach monitoring and notification and the classification of waterways as impaired. The discussions were wide-ranging. Discussions began with the consideration of the relationship of likelihood of illness due to nonhuman sources to likelihood of illness predicted by the use of epidemiological data from human exposure to POTW-impacted waters using fecal indicators. Possible approaches to modifying the application of regulatory approach using considerations of infectivity to pathogens among species were debated. The location of fecal sources relative to the site of monitoring and the potential of animals to move off-site were also discussed. These topics are all reflected in the potential research activities proposed and discussed in this chapter.

Six charge questions were provided to the workgroup (see Appendix A) to help stimulate discussion, and to identify key issues for consideration. A brief synopsis of responses to the questions is presented below.

- *Question 1: Is setting criteria based on a treated human point source such as a POTW protective, under-protective or overprotective of other potential sources of human pathogen? Why or why not? Are there data to support this conclusion?*

Whether the criteria are protective would depend on the effectiveness of treatment in reducing the levels of pathogens and the relative reduction in indicator organisms. Secondary wastewater treatment with chlorination could provide a false sense of security for protozoa and viruses. This reflects the higher degree of effectiveness of chlorine in killing/deactivating bacteria relative to viruses and protozoa. Given that current indicators are bacteria and would be reduced to a greater extent than viruses and protozoa, low indicator levels might suggest that waters impacted by POTWs were relatively pathogen-free when they still contained a significant virus and protozoan load. Data are available to characterize the relative effectiveness of disinfection techniques across classes of waterborne pathogens and indicator organisms.

- *Question 2: Based on the "state of the science," what conclusions or assumptions are reasonable to make about risks to humans exposed to human fecal contamination, non-point source contamination from animal sources, and mixed sources (e.g., combined sewer overflows [CSOs] and (separate) storm sewer overflows)?*[6]

Workgroup members felt that it is reasonable to assume that exposure to fecal contamination from untreated human waste posed the highest risk. Treated sewage was judged to be of lower concern, although it was more similar in risk to untreated human waste than to nonhuman sources. In general, treated and untreated sewage should be treated similarly for the purposes of evaluating risk. Discussion of CSOs led to the conclusion that they should be considered similarly to untreated sewage in terms of public health concern. Although separate storm sewer overflows were initially considered to be similar to animal waste in nature, there was a recollection of data in the literature (Haile et al., 1999) noting the occurrence of a significant occurrence of human pathogenic viruses in stormwater effluent and associated health effects merits further investigation. Aquatic avian sources were considered to be of low public health concern. Other wildlife and agricultural animal (including poultry) feces were deemed to be of moderate concern.

- *Question 3: To what extent is it reasonable to apply risk estimates from POTW-influenced beaches to non-POTW beaches? Do we understand scientifically whether this would lead to overprotection? What science would be important to understanding this?*

A portion of the answer to this question is reflected in the responses to Questions 1 and 2 above. The propensity to over- or under-protect would depend upon the source of the waste impacting the site. Non-point sources that largely reflect nonhuman sources of fecal contamination would probably be overprotected by studies in POTW-impacted locations. Mixed sources or untreated human sources may be inappropriately characterized by the POTW-dominated data. The workgroup's generalizations are reflected in Table 5. Addressing the public health significance of CSOs and separate storm sewer overflows are problematic because of the site-specific nature of the extent to which they vary by site characteristics. Although the importance of dilution of pathogens and indicator organisms in runoff events was discussed, no conclusion was reached about its significance.

[6] It is important to note that the workgroup was specifically charged (see Appendix A) to address (separate) storm sewer overflows and not sanitary sewer overflows, the latter of which are often discussed in conjunction with CSOs and commonly using the acronym "SSO." For this reason, workgroup members decided to not use the acronym SSO anywhere in the chapter.

- *Question 4: Assess whether there is a possibility of overprotection due to a compounding of risks from multiple factors (such as the current definition of gastrointestinal [GI] illness [i.e., no fever]; more sensitive molecular-based methods; assuming that POTW risks = nonhuman fecal contamination source risks, etc.).*

This question was referred to the Acceptable Risk workgroup (see Chapter 5).

- *Question 5: How should EPA evaluate risk that may have a low probability of occurrence but a significant risk, if it occurs?*

This question was considered by workgroup members to be unlikely to be adequately represented by completed epidemiological studies due to the low incidence (or detection) of pathogens that are associated with severe health outcomes. However, this important public health issue might be addressed using quantitative microbial risk assessment (QMRA) methods or by using large-volume filtration in future epidemiological studies.

- *Question 6: What are the key data gaps and uncertainties needed to support criteria development in the near term?*

The research needs and their prioritization are presented in a separate section (4.4). Epidemiological studies were given a high priority, with QMRA as an important adjunct. Additional epidemiological studies were encouraged by workgroup members because the data produced directly measure outcomes of interest (e.g., GI illness) and the data produced are more directly comparable to data being obtained for human health risks at marine beaches largely impacted by human sewage. Thus, epidemiological studies were preferred to the extent that they were possible and were viewed as an anchor for QMRA studies. However, it was recognized that it may be difficult to find freshwater recreational sites with sufficient bather activity to provide adequate sample sizes for an epidemiological study. If suitable sites cannot be found, then modeling the risk using QMRA techniques based on available epidemiological information would provide quantitative risk estimates that could help with short-term decision making on health risks. Similarly, if pathogen-source combinations in Table 5 cannot be conducted, it may be possible to use QMRA to provide quantitative risk estimates.

4.3 Options for Approaches and Implementation Considerations

The considerations in the followings section are not applicable to the current U.S. approach (i.e., US EPA, 1986; see also Chapter 1) because there is no way to take into consideration the charge to this workgroup on comparing risk to humans of fecal contamination from different sources. The following considerations are applicable to both the European Union (EP/CEU, 2006) and WHO (2003) approaches to criteria development. The sanitary investigations are important for the topics discussed by this workgroup. Simultaneous use of multiple indicator organisms or a tiered approach may be necessary.

4.4 Research Needs

1. Prioritize the next generation of studies. The purpose of these studies is to (1) revisit the ratings using a more through literature review and (2) gain as much information as currently exists on the magnitude of the fecal pathogen source problem across the United States.
   a. Quantify the magnitude of difference in the risk of illness from different exposure sources (see Table 5) to see if they are different from POTW-impacted waters.
      i. Initial estimate of risk – populate the table with infectious dose data and likely number of organisms excreted in stool per gram to characterize fecal source rank.
      ii. Magnitude across the United States
         1. Number of impaired waters
         2. Number of beaches affected by the sources (number of affected bathers if available)
      iii. Identify potential fresh and marine recreational sites for each of the fecal pathogen sources (rows) for future epidemiological studies. Priority should be given to freshwater sites.
2. Identify and characterize potential sites for future epidemiological studies using the following sources of information:
   a. National Pollution Discharge Elimination System (NPDES) – provides location of all point source dischargers and their levels of discharge
   b. CWA §303(d) list and §305(b) reports
   c. Sanitary investigations and microbial source tracking to confirm site characterization
   d. Compile information (via literature review and/or site-specific) about pathogen loads in non-point source water impacted by all sources of fecal contamination (human and animal), characterizing with respect to pathogens and indicators in freshwater versus marine water.

4.4.1 Epidemiological Studies

Workgroup members agreed that epidemiological studies are the most desirable approach to define and quantify health risks to humans swimming in fecally contaminated waters. Although many epidemiological studies have been previously conducted at point source-impacted beaches, very few such studies have been published on non-point source-impacted recreational waters. The relationship between current water quality indicators and health outcomes that is currently used in regulating beaches was developed from studies at point source-impacted beaches where water quality indicator levels correlated with swimming-associated illness (US EPA, 1986). It is plausible that the relationship between water quality indicators and health is different at non-point source-impacted sites since indicator levels may be high due to animal (e.g., birds, other wildlife) or other sources that do not increase the risk of human illness. Some workgroup members felt that it is appropriate to conduct epidemiological studies at non-point source-impacted sites to better define risk and guide future regulations.

Some workgroup members noted that epidemiological studies cannot be performed in all of the various types of non-point source-impacted waters for which there is a need to know risk. In many of these types of sites, other techniques (such as QMRA) will necessarily have to be used (see Section 4.4.2). The choice of the specific sites (beaches, rivers, lakes) in which to conduct epidemiological studies could be guided by the risk rankings developed in Table 5. These rankings include the types and concentrations of pathogens present, the number of affected waters across the United States, the number of people who are exposed to such sites, and the number of sites affected by regulatory restrictions under the CWA §303(d) guidelines.

Two principal study designs have been used in prior beach epidemiological studies—the randomized controlled trial (RCT) and the prospective observational cohort. The RCT has been primarily used in European studies and the observational cohort in many countries. Workshop participants discussed the relative strengths and limitations of each study design. With respect to the issue of health risks in non-point source-impacted waters, the workgroup members actively discussed the advantages of each design and felt that each had merit. Because of the required sample size (i.e., number of swimmers) is much less for an RCT, workgroup members could envision situations in which an RCT could be employed in future non-point source epidemiological studies. Workgroup members did note that in the United States it would be more likely for such an epidemiological study to receive human subjects approval if the enrollment scheme were altered from the RCT that has been used in several European studies. In Europe, subjects are typically recruited and enrolled in the studies at sites distant from the beach and then brought to the study sites. Workgroup members discussed an alternate design for consideration in the United States; specifically, enrolling willing persons who are about to enter the water and randomizing them to either swim or not swim that day. As in all epidemiological studies, aggressive exposure measurements of the water ingested and measures of water quality (e.g., indicators of fecal pollution) to which the swimmer is exposed would be critical. In non-point source sites where adequate numbers of swimmers could be enrolled, the prospective cohort design could be used for epidemiological studies. Workgroup members felt that it would be very helpful at some point to use both study designs simultaneously on one beach. This would allow for a direct comparison of the results and help guide future epidemiological studies.

1. Epidemiological studies (**highest priority is to conduct studies at beaches impacted by different types of non-point sources of fecal contamination [see Table 5]**)
    a. Randomized control trials (for consideration at beaches with low numbers of bathers)
        i. European design should be modified for use in the United States (suggestion – randomize people about to swim into groups that will swim or not swim)
        ii. Potential problem – identifying appropriate numbers of participants may be more difficult for inland (predominantly fresh) recreational waters than marine waters
        iii. Estimated necessary sample size – 1,500 people/site
    b. Prospective observational cohort study
        i. Potential problem – identifying sufficient numbers of participants may be more difficult for inland recreational waters than marine waters

ii. Estimated necessary sample size – 5,000 to 10,000 people/site (200 to 400 people/day)

iii. Wide range of exposures needed

4.4.2 Quantitative Microbial Risk Assessment

Several workgroup members advocated for QMRA studies in developing new or revised recreational ambient water quality criteria (AWQC). In part because QMRA can be used to rank the relative risks of different situations, such as sites impacted by animal versus human fecal wastes, and where no direct epidemiological information is available. QMRA studies can also be instructive in recreational areas where such studies have already been completed.

QMRA is increasingly used to characterize risk to humans from exposure to contaminated water when engaging in "contact recreation," especially swimming, but also other forms of water contact such as water skiing. It translates the environmental occurrence of pathogens and the volume of water that individuals are exposed to into a probability of infection or illness. Inputs with known variability are described by statistical distributions from which many random samples are taken, often using a "Monte Carlo" calculation procedure, to derive a risk profile.[7]

The following four step process is used: (1) identifying the important pathogens ("hazards"); (2) determining human exposures to contaminated water, via ingestion or inhalation; (3) characterizing dose-response, using data available from clinical trials, illness surveillance, and outbreak data; and (4) mathematically characterizing the risks and communicating risks and attendant uncertainties.

For step 1, a suite of sentinel pathogenic microorganisms should be considered for each situation as they are considered to cover the range of illnesses that could arise in the United States, such as the following:

- viruses – norovirus, Hepatitis A virus, caliciviruses, enteroviruses, rotavirus, adenoviruses;
- bacteria – EHEC, *Campylobacter* spp., *Salmonella* spp., *Shigella* spp.; and
- protozoa – *Giardia* cysts, *Cryptosporidium* oocysts.

The setting for each site of interest will dictate which of these pathogens should be used. For example, a recreational site impacted only by animal wastes should not need to include viruses. Adenoviruses will need to be included where aerosols may be inhaled (e.g., by water skiers).

For step 2, information on water ingestion and exposure rates, along with duration of the recreational activity, are combined with the concentration of pathogens in the water to obtain a

[7] EPA's Office of Water has developed a "complete draft" of a Protocol for Microbial Risk Assessment based on the EPA-ILSI (ILSI, 2000) *Revised Framework for Microbial Risk Assessment* (http://www.ilsi.org/file/mrabook.pdf) and which is consistent with the chemical risk assessment paradigm. The Agency has initiated a review to insure it meets risk assessment needs for all water-based media. Contact Stephen Schaub, EPA Office of Water (see Appendix B), for information on the Protocol for Microbial Risk Assessment.

dose—all these variables being described by statistical distributions. Information on the origin, quantity, and fate and transport of wastes deposited on a land surface and into waterways is of prime importance in determining the distributions of pathogens in the water that is subsequently ingested or inhaled.

For step 3, several dose-response analyses have been reported and may be used, albeit with caution. In particular, the form of the "dose" used in a clinical trial needs to be made consistent with the form used to describe the dose ingested or inhaled.[8] Also, uncertainty in the dose-response equation, in the form of credible intervals, can be captured by the calculation process.

In step 4, risk profiles may be derived, in the form of a cumulative distribution function—this will be particularly useful for examining the risks associated with rare but highly significant illness (e.g., EHEC). This also enables uncertainty measures to be calculated. Comparing relative risks for different sites should be done by comparing risk profiles, rather than by comparing single risk "numbers."

1. QMRA provides a range of possible illnesses or risks, allows comparisons across all fecal pathogen sources (see Table 5), and number of illnesses by a modeling approach (**highest priority is to conduct assessments at beaches impacted by different types of non-point sources [see Table 5]**). There was discussion among workgroup members regarding the strengths and limitations of conducting QMRA versus epidemiological studies (see Eisenberg et al., 2006); QMRA:
    a. Is a potential alternative, adjunct, or precursor to epidemiological studies
    b. Can evaluate infection and illness
    c. Could evaluate sentinel (index) pathogens such as:
        i. Bacteria (EHEC, *Campylobacter, Salmonella, Shigell*a)
        ii. Protozoa (*Giardia, Cryptosporidium*)
        iii. Viruses (norovirus, Hepatitis A, caliciviruses, enteroviruses, rotavirus, adenoviruses)
    d. Can consider inhalation as an additional route of exposure if data are available
        iv. Adenoviruses
2. QMRA is a good way to compile information (via literature review and/or site-specific) about pathogen loads in source waters impacted only by animal sources (with an emphasis on freshwater) and to characterize pathogens and indicators.

4.4.3 Etiologic Agents

Workgroup members felt it important to emphasize that there is a glaring lack of knowledge about the incidence with which specific pathogens cause swimmer-associated illnesses at both non-point source- and point source-impacted beaches. Identification of such pathogens as the actual cause of illness in swimmers would provide important information for developing new or

[8] For example, a rotavirus clinical trial will report dose as FFU (focus forming units); there may be many virus particles for each FFU.

revised recreational AWQC (or State Water Quality Standards) to enhance the protection of public health. In order to go forward with currently available technologies, the diagnosis of viruses could be made by exclusion of bacterial and protozoan pathogens causes of illness. Additionally, such information would be essential inputs into QMRA models to be used at recreational sites (or types of sites) where epidemiological studies cannot be conducted due to expense or insufficient numbers of swimmers. Because advances in modern techniques in microbiology now make a more complete identification of specific pathogens possible, workgroup members felt that the epidemiological studies currently underway and planned provide a unique opportunity to collect specimens (stool, saliva, and/or blood) from swimmers (and non-swimmers as controls) with which to identify the responsible waterborne pathogens. Such data would be complementary to the data collected in studies of pathogen occurrence in water that are presented elsewhere in this chapter and these proceedings. Workgroup members suggested that both types of pathogen occurrence information (in humans, in water) be collected during future epidemiological studies in order to minimize cost and maximize the utility of the information.

1. Identify etiologic agents of swimming-associated illness.
2. Pilot approaches for identifying etiologic agents in planned and ongoing epidemiological studies.
3. Classify etiologic agents in ill swimmers by broad groupings (i.e., viral, bacterial, protozoan).
4. Develop and evaluate sample collection techniques (stool, salivary antibodies, blood).

All of the above could be done as an adjunct to epidemiological studies.

4.4.4 Fate and Transport

Because direct pathogen detection is not feasible on an ongoing basis, a surrogate measure relating water quality conditions to human health risk is required. When developing the appropriate indicator(s) to use in this approach, knowledge of the fate and transport characteristics of the pathogens and indicator(s), both individually and as they relate to each other is critical.

Individually, fate and transport is significant because only those pathogens that are present and viable in a given waterbody pose a potential public health risk. These pathogens are typically divided into the following three major categories: viruses, bacteria, and protozoa. Because the microbiological characteristics of each of these groups are significantly different, it is not unreasonable to assume that their fate and transport characteristics will vary (perhaps significantly) as well.

The most simplistic route of pathogen transport is direct deposition. Once the pathogen(s) (assumed to be carried in the feces of warm blooded mammals) is excreted over or in the water, the question is twofold—how long will the pathogen be viable and available (i.e., persist in the water column).

Indirect deposition of feces introduces a more complex situation. First, the fecal properties of different mammals can vary substantially. One of the primary differences (aside from pathogen and indicator density) is moisture content. That is, very “wet” feces is more likely than “dry” feces to introduce pathogens into the aquatic environment. After defecation, the distance of the feces from surface water plays an important role as well. Driven by precipitation and transported primarily via surface runoff, the pathogens are typically washed into the surface water either by sheet flow or are collected and discharged through a storm water collection system. During this transport, they are subjected to a variety of environmental factors—including, but not limited to, UV disinfection, predation, temperature—that affect the proportion that will ultimately end up in surface water in which people are recreating.

Another category of indirect deposition includes point source discharges, such as POTWs, CSOs, concentrated animal feeding operations (CAFOs), and other NPDES permitees. In addition to the issues identified above, the effect of the treatment processes that these effluents are subjected to plays a role in fate and transport of the pathogens.

Resuspension from sand, soil, or sediment (i.e., secondary environments) can also play an important role in pathogen fate and transport. There may be a reservoir of indicator(s) and/or pathogens that could be reintroduced into the water column. Additionally, regrowth of either the indicator(s) or pathogens could represent a source and/or confound the risk assessment/prediction.

Ideally, the indicator(s) chosen as the surrogate for the pathogens will have the same fate and transport characteristics of the pathogens themselves. However, since this is unlikely, it is important to know and relate the characteristics that are specific to the indicator(s) and the pathogens so that the measurement of the indicator can be correlated to the concentration of the viable pathogens in the water and ultimately to public health risk.

A number of studies have been published on the fate and transport of many waterborne pathogens and current indicator organisms. Therefore, a literature review to identify any data gaps so that additional studies may be designed and also to inform QMRA studies would also be useful.

1. Conduct fate and transport studies for indicators and sentinel (index) pathogens.
2. Conduct literature review to identify data gaps and to inform QMRA.
3. Identify indicators that have the similar fate and transport characteristics as pathogens.
4. Should include assessment of risk of pathogens and indicators being resuspended from sand, soil, and sediments (secondary environments).

4.4.5 Determine the Occurrence of Pathogens in Impacted Recreational Waters

The pathogen occurrence and pathogen concentrations in water impaired by animal feces in one or more non-point study site(s) (e.g., beach impacted by [non-CAFO] agricultural animal runoff; Table 5, priority #1) could be compared with pathogen load in planned POTW-impacted marine epidemiological studies. It is also proposed that investigators consider using high-volume, tangential-flow water filtration methods that were recently developed for assessing bioterrorism threats to drinking water. This technology was designed to simultaneously capture very low concentrations of viruses, bacteria, and parasites in 10 to 100 L of water using a single collection apparatus (filter and pump). Although the equipment and pathogen recovery methods were initially designed to work on finished drinking water, there has been additional research to adapt the process for use on raw water supplies. The raw water application of this technology may be sufficiently understood for its employment in current or planned studies within the next 2 to 3 years. If the methods have not yet been adequately evaluated for this purpose, EPA may wish to encourage fast tracking their development for use in recreational water epidemiological and related field studies. Use of the large volume filtration tools might also be helpful to assess risks associated with low probability events that have serious health consequences (e.g., EHEC).

1. Determine the occurrence of pathogens in affected waters using the high volume filtration currently being developed for counter bioterrorism purposes.

4.4.6 Bather Studies

Bathers themselves can be a source of both indicator organism and pathogens in recreational waters (Elmir et al., 2007). Workgroup members suggested the following studies to determine the magnitude of this problem and/or the conditions at recreational sites in which this would be a problem.

1. Conduct additional studies on the impact of bathers on levels of indicator organisms and as a source of infectious pathogens for other bathers.
2. Develop better tools for assessing bather density.
3. Incorporate bather density into the study design and analysis of future recreational water epidemiological studies.
4. Conduct additional studies on human shedding in a controlled setting with a focus on young children.
5. Incorporate bather contribution to indicators and pathogens in QMRA studies.

4.4.7 Additional Research (Either Short- or Long-term Depending on EPA Priority-setting)

The following research would also enhance many of the ongoing and future efforts described in this chapter and elsewhere in these proceedings.

1. Include epidemiological data in predictive modeling efforts. This would broaden the use of both epidemiologic and modeling data. Many recreational epidemiological studies collect an extensive set of environmental data. Whether this is sufficient to accomplish environmental modeling is unknown. Both modelers and epidemiologists should discuss the feasibility of this effort.
2. Develop a method for accurate exposure assessment among swimmers. Exposure assessment in terms of water contact and quantity of water swallowed or inhaled is an area of potential misclassification in observational epidemiologic studies. The following would improve exposure assessment in epidemiologic studies:

a. Develop individual sampling devices.
b. Develop methods and conduct studies to determine the quantity of water ingested and inhaled in recreational settings. Consider studying secondary recreational contact for potential comparison.

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CHAPTER 5
ACCEPTABLE RISK

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5.1 Introduction

This workgroup was primarily charged to reassess the extent to which existing microbiological criteria protect the health of swimming populations, and whether or not this is appropriate for current U.S. society. In particular, the workgroup was asked to consider the case of vulnerable (susceptible or sensitive) subpopulations and whether current levels of public health protection are sufficient for these people. The workgroup was also asked to consider whether it would be possible that improvements in recreational water quality criteria would be sufficient to improve public health protection for drinking water, recreational water, or consumption of shellfish.

Group members decided to organize the main questions under the following headings:

- Whether the term "acceptable risk" is still the most appropriate term.
- Public involvement in "acceptable risk" decisions
    - To whom should any risk from recreational water contact be "acceptable"?
    - How can we get public involvement in the decision making process over what is and what is not "acceptable"?
    - How best to communicate risk with and educate the general public about risks from recreational water.
- "Acceptable risks" to the general population
    - Whether the current methods for assessing risk from recreational water exposure are sufficient and if not, what new methods may be appropriate?
    - Whether risks differ between marine and freshwaters and whether it is "acceptable" to have different levels of protection for people bathing in these different waters.
    - Whether the current approach, based on protecting people from enteric illness is sufficient, or whether "acceptable risk" decisions need to take into account non-enteric illness.
    - Whether risks are different to people swimming in tropical, subtropical and temperate waters.
- "Acceptable risks" for vulnerable subgroups
    - Define the main vulnerabilities.
    - Determine what risks are greater in vulnerable subgroups and whether general recreational water standards are sufficient to protect these groups.
- What are the current levels of protection from existing criteria?
- Potential synergies for health protection between revised recreational water criteria and standards for drinking water and shellfisheries.

5.2 Main Conclusions and Observations

5.2.1 Whether the Term "Acceptable Risk" is Still the Most Appropriate Term

There was commonality amongst the workgroup members that the term "acceptable risk" is flawed and should be avoided during the process of creating recreational water criteria. The term "acceptable" was felt to elicit responses related to "acceptable to whom?" and had the connotation that swimmers accepted the risk and there was some level of informed decision making during the process. Although a variety of suggestions for replacing "acceptable" were elicited (e.g., tolerable, appropriate, excess, increased), no agreement on terminology was reached. However, workgroup members felt that any new term should be simple, easily understood, and inclusive rather than paternalistic in nature. Workgroup members also felt that EPA should develop a policy that includes public interaction during the criteria development process.

This approach to determining "acceptable risk" should be broadly inclusive of impacted groups (e.g., swimmers, taxpayers who pay for beaches to be open) throughout the process. This would mean that EPA's decision making and criteria development process should include information on how impacted groups would determine the level of "acceptable risk" and how those risks and the concept of protective criteria would be best communicated. This would require that EPA's criteria development process (1) be clear, transparent, and communicated to all stakeholders; (2) factor in and include input and data collected from impacted groups; (3) include a data-informed communication package to educate impacted groups when the new criteria are released; and (4) develop a plan for assisting state and local authorities with future communication of the concepts of "acceptable risk" and the meaning of beach closures and advisories to the public. Such an effort would require collaboration with sociologists and anthropologists to assess risk perception and risk communication research and apply this to development of appropriate assessment tools for determining key elements necessary for criteria development, release, and interpretation. Rapid integration of this information into ongoing EPA criteria development would be expected to build or improve partner involvement and acceptance of the new criteria.

5.2.2 Public Involvement in "Acceptable Risk" Decisions

Including public involvement in the criteria setting process would require that impacted groups are first informed about the process and then information solicited about how these groups make "acceptable risk" decisions and how tolerant these groups would be of risk associated with recreational swimming area use. Key research questions include the following: (1) What does the public understand currently? (2) What does the public think of when one uses the term "acceptable risk"? (3) How does the public interpret existing criteria and beach closures/advisories? (4) How does/should EPA communicate this risk? and (5) What level of risk would the public accept? The voluntary nature of recreational swimming needs to be clearly explained and put in context with other routinely and voluntarily accepted risks (e.g., driving to the beach, eating at local restaurants, smoking). The breadth of illness associated with swimming and types of illness to be reduced by new or revised recreational water quality criteria needs to be clear. Workgroup members felt that current criteria were not well understood by the public or beach managers so that indicator cutoff values (i.e., beach closures) connoted zero risk and "safe" water rather than an understanding of the concept of "acceptable risk." These groups should be allowed to provided input on factors used in the decision making process (i.e., reduction of illness in children being a decision point). Workgroup members appreciate that EPA will ultimately be making the decisions and setting criteria but felt that a more informed and communicative path for this decision making is critical to future acceptance of these new or revised criteria.

Workgroup members suggested that EPA conduct the following activities:

1. Begin building a transparent communication plan to inform impacted groups about ongoing criteria development.
2. Rapidly initiate studies to assess how impacted groups understand and perceive the risks associated with recreational water use and what level of voluntary risk would be "acceptable," followed by evaluation of final communication materials.
3. Develop a multi-year plan to communicate the criteria development process to impacted groups and a communication plan for educating impacted groups about the new criteria.
4. Assist state and local officials in developing data-based risk communication plans for communicating information on criteria interpretation and beach closures/advisories to the public.

5.2.3 "Acceptable Risk" Levels for the General Population

Method for Assessing Risk

Workgroup members identified epidemiological (both randomized control and prospective observational cohort designs) and quantitative microbial risk assessment (QMRA) studies as the main methods for assessing risk. Some workgroup members noted that while QMRA is widely used and relied on by EPA for drinking water applications, it does not seem to be as widely used for recreational waters (with the exception of the work done by Jeffrey Soller). To broadly evaluate the gastrointestinal (GI) illness risk associated with the numerous potential pathogens found in recreational waters, epidemiological studies were viewed as more appropriate, although workgroup members believed the EPA should investigate expanding the role of QMRA (see also Chapter 4). One distinction noted was that although epidemiological studies are good at assessing the generally common and self-limiting risks associated with swimming in fecally-contaminated waters, they are not well-suited for investigating rare but potentially severe (and potentially life-threatening) illnesses that may be associated with recreational water exposure such as enterohemorrhagic *E. coli* (EHEC). For these special cases, workgroup members felt QMRA approaches may be the best way to assess risk and address potential outbreak situations.

Other cases where QMRA could be useful would be for evaluating specific risks associated with specific waterborne pathogens (although not necessary rare) such as *Cryptosporidium*, Norovirus, and *Shigella*. A third method that has not yet been widely applied to assess risk from recreational waters is dynamic infectious disease modeling (with the exception noted above). These models are a form of QMRA, but specifically account for factors such as the immune status of the population (susceptible, infected, immune), rates of secondary transmission of illness, and other parameters.

Workgroup members also noted that epidemiological studies can identify illness, but not infections, whereas QMRA studies can predict infections, but have more uncertainties associated with translating infections into an estimation of illness. Although epidemiological studies provide valuable results, there may be some confusion in their interpretation and application; for example, most studies of recreational waters to date have been conducted at beaches with known human sources of fecal contamination and results may not apply to other sites. EPA needs to clearly explain the purpose of such studies (current, planned, and previous studies), their focus, and limitations.

Marine versus Freshwater

Workgroup members did not see any reasonable rationale for different "acceptable risk" levels in marine and fresh recreational waters. Although the current "acceptable risk" levels based on EPA's *Ambient Water Quality Criteria* [AWQC] *for Bacteria – 1986* are different for fresh and marine waters (gastroenteritis rate of 8 per 1,000 swimmers in freshwaters and 19 per 1,000 in marine waters), workgroup members believed this to be an arbitrary decision that was not well founded. Workgroup members agreed that there could be different indicators, or different levels for the same indicator across marine and fresh recreational waters, but those levels should relate to the same estimate of risk. Furthermore, justifying differences in risk to the public and stakeholders based on type of water would continue to be confusing and problematic.

There was some further discussion about how to account for differences in baseline levels of illness that could exist across locales and whether use of a relative risk scale instead of an excess (or attributable risk) scale may be a better way of addressing such differences. There is a distinct difference between doubling an absolute risk versus doubling a relative risk (see Section 5.2.5).

Enteric versus Non-enteric

Workgroup members felt that criteria based on pathogen indicator levels derived to protect against GI illness would not necessarily protect against all non-enteric illnesses, with the possible exception of certain upper respiratory illnesses (URIs) transmitted via the fecal-oral route. At least one study (Fleisher et al., 1996) observed exposure-response relationships with fecal streptococci (enterococci) and URI; workgroup members believed there was potential for pathogens causing such illnesses (e.g., adenoviruses) to be transmitted via fecally-contaminated waters. The workgroup members felt that most causes of other non-enteric illnesses (e.g., rash, earache) were most likely to be caused by environmental or naturally occurring conditions and/or pathogenic microorganisms unrelated to fecal contamination (e.g., *Naegleria* infection, non-cholera *Vibrios*) and therefore would not be explicitly controlled by criteria based on protection for GI illness (WHO 2003).

There was uncertainty about EPA's role in protecting against such illnesses, particularly those that are not anthropogenic. However, there are some risks that were unclear. For example, cyanobacteria concentrations can be influenced by nutrients and human impact, and may also be a cause of swimming-associated skin infections, respiratory infections, or long-term chronic conditions such as liver cancer (Chorus and Bartram, 1999; Fleming et al., 2002).

Workgroup members felt that earaches (*otitis externa* or "swimmers ear") were probably the most debilitating of the commonly occurring swimming associated non-enteric illnesses. However, they also felt that there was no evidence that such infections (often caused by *Pseudomonas*) were associated with fecal indicator bacteria, and therefore AWQC or State Water Quality Standards based on fecal indicators would not afford public health protection for those illnesses. Workgroup members also felt that other indicator bacteria, or other types of indicators, are not currently available to protect swimmers from most non-enteric illnesses.

Workgroup members agreed that when a beach was closed due to fecal contamination then potential non-enteric swimming associated illnesses would also be prevented, although this would be inadvertent and it is not clear how often or under what circumstances this would occur (e.g., Do currently used indicators correlate with the presence of cyanobacteria or *Pseudomonas*?).

Tropical and Subtropical versus Non-tropical Recreational Waters

Workgroup members identified the possibility that tropical and subtropical recreational waters may have to be approached differently from temperate waters because of issues such as regrowth and significant spatial or temporal variability of both indicator organisms and pathogens in the water and soils, substantially different ecosystems and climatic conditions (including heavy rains), and possibly the presence of a greater range of “exotic” pathogens. In addition, persons may experience longer term seasonal exposures in tropical and subtropical recreational waters due to the warm waters throughout the year. Finally, it is highly likely that the background rate of GI diseases is higher in tropical and subtropical populations (Payment and Hunter, 2001).

It is important to note that workgroup members believe that people in tropical and subtropical areas should not be exposed to greater health risks from exposure to recreational waters than people in more temperate areas.

Relative risk measures, unlike excess risks, express risk as a proportion of baseline risk and thus correct for varying background levels. Workgroup members discussed other ways to describe risk in place of an “acceptable risk” framework, including illnesses prevented as a result of implementing criteria (as done by the U.S. Food and Drug Administration [FDA]). Workgroup members felt that there was need for risk communication in this area so that risks are fully and accurately communicated.

5.2.4 “Acceptable Risk” Levels for Vulnerable Subgroups

Definitions

In considering vulnerable human populations with regards to the health risks from exposure to recreational water, workgroup members distinguished between two major categories of vulnerability, (1) persons at different life stages, and (2) persons with suppressed immune function.

What is Different?

Life stage connotes that for a variety of reasons, humans vary in their level of vulnerability to the health risks associated with exposure to recreational water over their life span. In particular, the discussion focused on the possible increased vulnerability of children, pregnant women (and their fetuses), and the elderly. Workgroup members felt that children are at a greater increased risk compared to all other life stages because of their behavior and possibly because of naïve immune status. Because all members of the population pass through life stages, classifying childhood as a life stage instead of simply a subpopulation strengthens the argument for explicitly considering children when developing AWQC. Regarding behavior, children probably have higher exposures; that is, they are more likely to consume both marine and freshwater. Moreover, young children have significant hand-to-mouth and fecal-oral behavior that may lead to the consumption of contaminated substances. Very young children may also be more vulnerable to pathogens in recreational waters because they have never been exposed to these pathogens previously. Of note, preliminary, unpublished data from recent studies by EPA (NEEAR; Timothy Wade, EPA Office of Research and Development, personal communication, 2007) as well as results from other published studies appear to demonstrate an increased risk of GI illness and possibly respiratory illness for children from exposure to recreational waters, although this has not yet been formally reviewed.

Pregnant women (and their fetuses) and the elderly may be at increased risk for more severe consequences from acquiring GI diseases from exposure to recreational waters. Pregnant women and their fetuses may be at greater risk from certain recreational water pathogens (e.g., coxsackie B virus associated with fetal infection when acquired close to delivery, and enterovirus associated with certain fetal malformations). Furthermore, pregnant women may be at increased risk for significant dehydration and its consequences if they do acquire a GI infection resulting from contact with recreational water. Finally, although the elderly were believed to be less exposed due to decreased high intensity swimming behavior, it might be possible that the decreased immune function associated with increasing age might make them more vulnerable to infection and illness.

Workgroup members also identified a potentially large subpopulation of persons with suppressed immune function, ranging from persons with HIV/AIDS to persons undergoing chemotherapy and using other immunosuppressive medications. Of note, a portion of the latter subpopulation could be completely unaware of their suppressed immune function. As a group, persons with suppressed immune function would be at increased risk compared to the healthy population of acquiring diseases from a range of opportunistic pathogens found in recreational waters, such as *Cryptosporidium*, *Toxoplasma*, and *Vibrio parahemolyticus*. Furthermore, persons with suppressed immune function may be at increased risk of more severe consequences from these diseases as well as from the effects of dehydration—a secondary ramification of GI diseases.

Tourists and visitors were identified by workgroup members as a unique potentially vulnerable group to increased health risks associated with exposures to recreational waters. Similar to small children, these people may be previously unexposed to the range of pathogens in a new recreational water environment, and as such, more susceptible to both acquiring the infection and disease—possibly with more severe health consequences. Given that many of these people are on vacation, they may experience greater exposure to recreational waters.[9] Further, given that significant tourist travel is to tropical and subtropical areas, there may be additional risks from a range of exotic pathogens and potentially unique ecosystem conditions found in tropical and subtropical recreational waters.

[9] Tourists may spend long periods of time in the water over several days, whereas local users may have shorter exposures that are spread further apart.

Overall, workgroup members believed that the apparent increased risk for children for acquiring GI and possibly other diseases from exposure to both fresh and marine recreational waters should drive the health risk assessment of any future recreational water criteria development efforts, assuming the current and future research continue to demonstrate their apparent increased risk. Workgroup members emphasized that future recreational water criteria set on health risks and exposures of adults would not be sufficiently protective for children. As mentioned previously, because of differences in susceptibility between adults and children (and other subpopulations as well) a given numeric criteria translates to different risk levels for each subpopulation. Therefore, it is impossible to protect adults and children equally. The workgroup members felt that data on children should be explicitly considered for deriving the “acceptable risk” level in the development of new or revised recreational water quality criteria, with the understanding that the associated risk level for adults would then be even lower.

Workgroup members felt that the increased risk to immunosuppressed people should not be an important factor in setting any future recreational water criteria because the factors associated with the increased risk of disease in this vulnerable subpopulation are not controllable or achievable through management of recreational water sites. Rather, an emphasis should be made on improved risk communication with immunosuppressed groups and health care professionals to inform them about risks associated with recreational water use and, in consultation with their health care provider, assessment of the need to avoid recreational water exposure.

5.2.5 What are the Current Levels of Protection from Existing Criteria?

It is not certain how accurate the current levels of protection are. “Magic” numbers like 8 or 19 cases of gastroenteritis in 1,000 swimmers can “take on a life of their own,” increasing the risk of distraction from the basic objective—providing best effort to protect swimmers. This provides a compelling reason for not deriving and using a single numeric value for the targeted risk for new or revised AWQC. Risk levels from preliminary unpublished data from the EPA NEEAR study seems to agree with WHO (2003) B category waters (i.e., 1 illness per 20 swimmers) (see Table 1, Chapter 1; Timothy Wade, EPA Office of Research and Development, personal communication, 2007). Pathogens associated with threshold indicator levels in current (US EPA, 1986) AWQC may differ from those in 2007; the population established in 1986 also may have different susceptibility due to differences in immunity to current pathogens in 1986 versus 2007. Aside from protection against enteric illnesses, it seems likely that enterococci levels below current standards also provide some protection against upper respiratory tract infections.

Instead of absolute levels of risk, workgroup members felt that the preventable fraction is a better measure for the level of protection. This includes information on the background level of risk against which the risk associated with recreational water use must be compared. Presence of other major exposure routes may mask any beneficial effects of lowering risks due to recreational bathing. Thus, an absolute reduction in illness from recreational water may not be reflected in a similar reduction in total cases in the community if people simply become infected by other transmission routes. On the other hand, disease reduction may be even greater if secondary cases are also prevented. Most recreational water exposures are experienced by a minority of the population who are repeatedly (chronically) exposed.

It is also possible that part of the primary contact-associated infections is caused by bather-to-bather transmission. This independent, direct fecal contamination would be unaffected by monitoring programs designed to limit sewage contamination. Further studies are needed to understand the role of bather shedding in disease transmission and microbial water quality indicator levels.

In a trade-off situation, acceptability of risk is partially determined by its source; that is, pathogen-shedding by fellow swimmers is difficult to control and may be more readily accepted than contamination by treated sewage effluent or agricultural runoff, whose risks are usually considered less acceptable. More important than trying to enforce compliance with a fixed standard level of risk, is the need to work toward continual improvement in public health associated with recreational water use.

5.2.6 Potential Synergies for Health Protection between New or Revised Recreational Water Criteria and Standards for Drinking Water Sources and Shellfish Harvesting Waters

Workgroup members considered that any change in recreational water criteria that led to improved public health protection would not negatively impact on the risks from drinking water or shellfish consumption. However, some workgroup members did express concern about any change that would encourage further recreation in waters intended to be used for drinking water production or for shellfish harvesting. When people bathe they invariably contaminate the water to some extent with potential pathogens. Such pathogens may then be concentrated within shellfish or contaminate drinking water supplies and pose a health risk to others.

5.2.7 Areas of Discord

Although workgroup members accepted that the phrase “acceptable risk” was widely used, they realized that there were difficulties in its general acceptance. However, no alternative to the phrase was thought to be “acceptable” to all workgroup members. Although the phrase “tolerable risk” is now being used more frequently internationally, it was still not tolerated by all members of the workgroup.

5.3 Research Needs

1. ***Risk perception studies to inform the risk communication strategy for the criteria rollout and focus groups to evaluate the risk communication strategy***
    a. Assess public understanding of relative versus absolute risk.
    b. Key research questions include the following: (1) What does the public understand currently? (2) What does the public think of when one uses the term “acceptable risk”? (3) How does the public interpret existing criteria and beach closures/advisories? (4) How does/should EPA communicate this risk? and (5) What level of risk would the public accept?

2. ***Define the data and conditions where a directed monitoring program would be necessary to protect against certain non-enteric (non-GI) illness.***

   a. Such research would probably require pathogen-specific studies, and a possible role for QMRA.

3. ***EPA should investigate expanding the role of QMRA, particularly for investigating rare but potentially severe (and life-threatening) illnesses that may be associated with recreational water exposure such as EHEC (e.g.,* E. coli *O157:H7).***
   a. Define data needed for the QMRA modeling for special/outbreak cases and also for background/regular situations.
   b. Engage EPA experts in QMRA in recreational water research.
   c. Explore approaches to integrate QMRA (and/or dynamic modeling) to better understand recreational risk, especially situations with rare, but potentially severe outcomes.

4. ***Conduct methodologic comparisons in tropical and subtropical recreational waters and if appropriate, conduct epidemiological studies.***

   Methodological and ecological studies need to be conducted in tropical and subtropical recreational waters because of issues such as regrowth, significant spatial and temporal variability of both indicator organisms and pathogens in the water and soils, substantially different ecosystems and climatic conditions (including heavy rains), and possibly a greater range of exotic pathogens. These studies would determine the impact of these environmental factors on the use of proposed indicators organisms to be used for monitoring and regulatory purposes. Depending on the results of these studies, assessment of the need for epidemiologic studies specifically in tropical and subtropical recreational waters should be performed. This information will be essential to determine whether the same recreational water criteria as used elsewhere in the United States are also appropriate in these waters. Information on risks in such waters will help ensure appropriate risk communication to healthcare providers, public and environmental health managers, and residents of and visitors to tropical and subtropical areas concerning the risks of tropical and subtropical recreational waters.

5. ***Ensure that future epidemiological studies obtain data on and existing studies are reviewed for risk to children.***

   Children appear to be at increased risk for acquiring GI illness and possibly other illnesses from exposure to recreational waters; therefore, workgroup members felt future recreational water criteria should be based on the health risk to children. If existing standards are deemed not to provide sufficient protection to children then additional information will be needed to establish new or revised criteria that are thought to provide sufficient protection. Such information will also be essential to provide risk information to parents and others responsible for children.

6. ***Review prior data to evaluate whether additional epidemiological studies are needed to determine the risk of severe disease to pregnant women and their fetuses, to the elderly, and to immunosuppressed individuals.***

   There is evidence that pregnant women (and their fetuses), the elderly, and immunosuppressed people may suffer more serious disease and/or more serious health consequences from recreational bathing waters. If these data show that there may be increased risks, then the incorporation of these subpopulations as specific target populations in future epidemiological studies should be considered. Information on risks in such waters will help ensure appropriate risk communication to healthcare providers, public, and environmental health managers, and these potentially increased risk groups from recreational waters.

7. ***Determine how risks in tourists and visitors differ from those in residents.***

   There is some evidence that risk may be greater for tourists and visitors than for residents local to a recreational water; thus, current estimates may underestimate the actual risk and so give inappropriately lax criteria (Payment and Hunter, 2001). Consideration should be given to the design and implementation of future epidemiological studies to address risk in tourists and visitors. It may also be possible to review data from previous studies to determine if there are increased risks to tourists. Information on risks in such waters will help ensure appropriate risk communication to healthcare providers, public, and environmental health managers, and tourists with exposure to recreational waters.

8. ***Ecology of swimming-related waterborne pathogens, including studies on the role of bather shedding on transmission of illness and microbial water quality indicators***

   Further studies are needed to understand the role of bather shedding in disease transmission and microbial water quality indicator levels. How efficiently are pathogens transmitted through swimming or bathing? This could be an experimental study, partly, augmented by epidemiology (serology, or microbial source tracking in a small study population).

9. ***How many illnesses are prevented by beach closures?***

   Studies of the number of illnesses prevented by beach closures would be primarily a modeling/statistical exercise. First, the procedures/modeling assumptions should be agreed upon. It could be done relatively easily in a QMRA-type of study.

Table 6 provides a summary of how each workgroup member ranked the above research needs in relation to overall importance (1 to 5), relevance to EPA, and estimated time needed to complete the project.

Table 6. Research Needs and Rankings from Five "Acceptable Risk" Workgroup Members.

Description	Importance					Relevance to EPA					Near- and/or Long-term				
Conduct risk perception studies to inform the risk communication strategy for the criteria rollout and focus groups to evaluate the risk communication strategy (#1)	5	5	5	5	5	5	5	5	5	5	N and L	N	N	N	N
Assess public understanding of relative versus absolute risk (#1)	1	2	1	3	3	1	1	1	3	3	N and L	N	N	N	L
Define the data and conditions where a directed monitoring program would be necessary to protect against certain non-enteric (non-GI) illness (#2)	3	3	2	3	3	3	3	2	3	3	N and L	N	L	N	L
Define data needed for the QMRA modeling for special/outbreak cases also for the background/regular situation (#3)	3	4	3	3	5	3	4	3	3	5	N and L	L	N	N	N
Engage QMRA in recreational water research (#3)	2	3	3	3	5	2	3	4	3	5	N and L	N	N	N	N
Explore approaches to integrate QMRA (and/or dynamic modeling) to better understand recreational risk, especially situations with rare, severe outcomes (#3)	4	2	4	3	4	4	2	4	3	4	N and L	L	N	N	N
Conduct future epidemiological studies in tropical and subtropical bathing waters (#4)	4	4	2	5	5	4	4	2	5	5	N	N	N	L	L
Ensure that future epidemiological studies obtain data on and existing studies are reviewed for risk to children (#5)	5	4	5	5	5	5	4	5	5	5	N	N	N	L	N
Review prior epidemiological studies to determine the risk of severe disease to pregnant women and their fetuses (#6)	2	2	1	3	5	2	2	2	5	4	L	L	L	L	L
Review prior epidemiological studies to determine the risk of severe disease to the elderly (#6)	1	2	1	3	5	1	2	2	5	4	L	L	L	L	L
Review evidence about whether or not immunosuppressed individuals are at increased risk from recreational bathing waters (#6)	4	3	1	4	5	4	3	2	5	3	L	N	N	N	N
Determine how risks in tourists and visitors differ from those in residents (#7)	4	5	2	4	4	4	5	2	4	5	N	N	L	N	L
Conduct studies on the role of bather shedding on transmission of illness and microbial water quality indicators (#8)	5	5	5	4	5	5	5	5	5	5	N	N	N	L	N
Determine the ecology of swimming-related waterborne pathogens (#8)	3	4	3	5	3	3	4	3	5	5	L	L	L	L	L
Determine how many illnesses are prevented by beach closures?(#9)	4	4	4	5	3	4	4	4	5	5	N and L	N	N	N	N

Scoring for importance: score 1 not at all important to 5 highly important
Relevance to EPA: score 1 not at all relevant to 5 highly relevant
For time: N (within next 2 to 3 years); L (within next 10 years)

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CHAPTER 6
MODELING APPLICATIONS FOR CRITERIA AND IMPLEMENTATION

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6.1 Introduction

The Modeling workgroup was charged with determining how models might be incorporated into future recreational water criteria development and implementation. Workgroup members did not explicitly consider total maximum daily loads (TMDLs) in the discussion because models are already being used in TMDLs for pathogens throughout the United States. The discussion focused on what was generally felt to be the most important novel applications of models in new or revised recreational ambient water quality criteria.

In the context of recreational water quality criteria, a perfect model would allow prediction of fecal indicators, pathogens, or risk as a function of source presence and strength relative to physical, chemical, biological, and human variables.

There is limited understanding regarding the sources of microorganisms and their fate and transport in the aquatic environment, so the use of deterministic, process-based models (see Appendix G) in criteria development and implementation is not practical for most U.S. water quality managers within the next five years (2012). Rather, **simple heuristic, statistical models that do not necessarily require an understanding of processes and mechanisms are more realistic for criteria development and implementation within the next 5 years.** This is not to say that substantial research should not go into refining understanding of sources, fate, and transport of pathogens and pathogen indicators and their spatial and temporal variability in water and sediments. Thus, workgroup members suggested that a substantial research effort go into understanding these processes in watersheds and near-shore waters as this will have profound impacts of development of future ("next generation") recreational water quality criteria (see Section 6.5).

Workgroup members saw two roles for models in the development and implementation of near-term (five years) new or revised criteria: (1) recreational water quality notification models and (2) models to support sanitary investigations (hereafter referred to as "sanitary investigation models" for simplicity). Recreational water quality notification models are already in use in the Great Lakes and have proven to be effective and popular with the public (Francy and Lis, 2007; Olyphant, 2004; Whitman, 2007). There are a handful of sanitary investigation tools and models that are accessible to recreational water managers throughout the country (e.g., DigitalWatershed, the BASINS3 system). The main focus of this chapter is water quality notification models because these are easily accessible to a wide range of recreational water managers in the near-term. However, because workgroup members viewed the sanitary investigation model as an area of near-term research activities and investigation, with possible applications in the near-term development of new or revised criteria and/or implementation, discussion of sanitary investigation models was included as well.

6.1.1 Water Quality Notification

Numerous research studies in the peer reviewed literature show that a single sample standard implemented in conjunction with assays that require incubation longer than a few hours results in less accurate management decisions (Francy and Darner, 2006; Hou et al., 2006; Kim and Grant 2004). That is, by the time results from analysis of a water sample are available and a water quality notification is issued, the microbial water quality may have changed. This is due to the inherent variability in indicator bacteria levels over timescales shorter than a day (see Figures 4a and 4b), as measured by culture-based assay, both with selective membrane-filtration media and

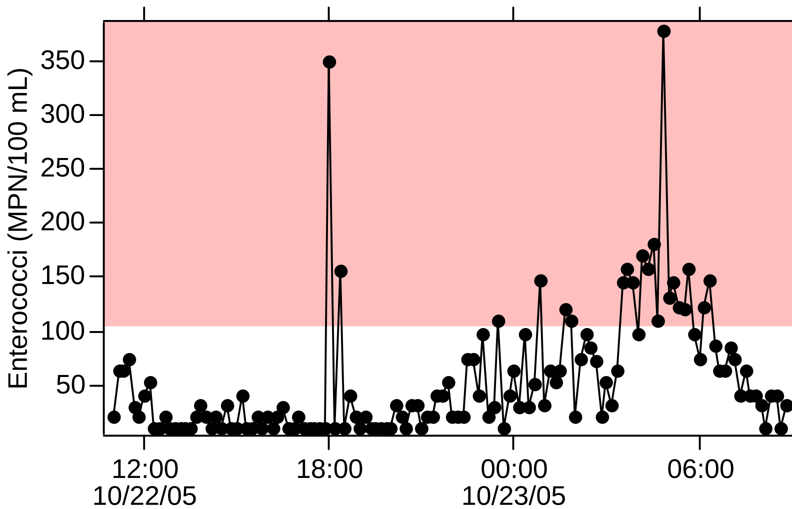

Figure 4a. Enterococci (MPN/100 mL) Sampled Every 10 Minutes at a Beach in California. (The reference background denotes the range of single sample exceedance.) SOURCE: A.B. Boehm, unpublished data (ENTEROLERT assay).

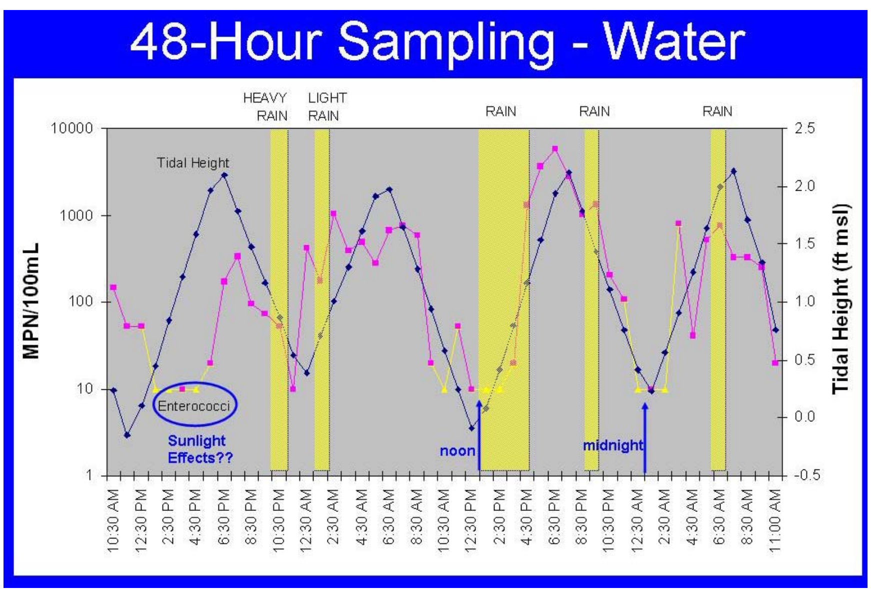

Figure 4b. Subtropical Marine Beach (Miami, Florida): 48 hours Sampling. SOURCE: Amir Abdelzaher, Samir Elmir, Lora Fleming, Kelly Goodwin, Helena Solo-Gabriele, John Wang, Mary Wright, University of Miami, personal communication, 2007.

defined substrate technologies such as Quanti-Tray (IDEXX, Westbrook, Maine). Note, variability in indicator levels as measured by nucleic acid-based assays (like quantitative polymerase chain reaction [qPCR]) has not been well characterized in the peer reviewed literature. The variability in Figures 4a and 4b is not unusual in environmental waters because "patchiness" is an inherently natural phenomena. **A water quality notification model can be used to augment monitoring data and provide more timely and accurate recreational water quality notification to better protect the public from exposure to waters not in compliance with water quality criteria or standards.**

Summary of near-term research needs (i.e., the next 2 to 3 years) specific to **water quality notification** include (see Section 6.5 for further information) the following:

1. Day-to-day water quality notifications should not be issued using a single sample standard in conjunction with a microbial assay that takes longer than a few hours due to time-lag notification errors as discussed above. Simple, heuristic or statistical water quality notification models are one way to improve water quality notification accuracy.
2. Immediate research needs include the following:
    a. Testing whether models can be used to predict health outcomes during upcoming epidemiological studies in California and in Alabama and Rhode Island and retrospectively for the Great Lake epidemiology study in the Great Lakes (Wade et al., 2006) (that is, risk = f[temperature, tides, waves, etc.]);
    b. Developing and testing simple notification models on different recreational water types with a wide range of sources and geographical locals;
    c. Exploring the feasibility of developing regional models that apply to more than one recreational water;
    d. Training recreational water managers; and
    e. Creating a user-friendly portable package for developing local models.

6.1.2 Sanitary Investigation

Quantitatively determining the potential for a waterbody to be impaired with human pathogens is essential if the European Union (EU; EP/CEU, 2006) or World Health Organization (WHO, 2003) approach to criteria development is undertaken (i.e., sanitary investigation is integrated into the criteria). This potential could be determined using a "toolbox approach" in conjunction with water quality notification models or sanitary investigation models. In the first case, the water quality notification model results can be used to learn about the factors that influence water quality in recreation waters; for example, high rainfall and wave action from a given direction and of a given height might lead to greatest impairment. The occurrence of these environmental conditions can be used to trigger sampling for "toolbox" approaches such as analyses for human-specific or bird-specific markers and human pathogens to "rule in" or "rule out" high probability of human pathogen presence. In the second case (sanitary investigation models), simple, quantitative sanitary investigation models that relate watershed attributes to probability of human pathogen impairment may be developed.

Summary near-term research needs (i.e., the next 2 to 3 years) specific to **sanitary investigation models** include (see Section 6.5 for further information) the following:

1. Simple, heuristic or statistical models that correlate watershed activities (presence of wastewater/sewage treatment plant effluents, agricultural activities, and domesticated animals) and attributes (slope, soil type, climate, soil moisture) can be used to determine the susceptibility of a waterbody to pathogen impairment.
2. Research should be conducted to better understand how watershed activities and attributes relate to pathogen presence in streams and receiving waters and include the following:
    a. factors that modulate septic tank impact on waterbodies;
    b. factors that modulate contributions of animal wastes to pathogen and pathogen loads to waterbodies;
    c. sources in urban landscapes (e.g., broken/leaky sewer pipes, combined sewer overflows [CSOs], runoff); and
    d. effect of meterological factors (e.g., rainfall, evapotranspiration, etc.) on non-point sources.

6.2 How Models are Currently Being Used

6.2.1 Sanitary Investigation Models

Sanitary investigation models that explore the relationship between land use, watershed attributes, and water quality are already in place and have been used in TMDL implementation (criteria implementation); however, they have not been specifically applied to criteria development. Creating a TMDL-like model for a waterbody prior to impairment may be viewed as proactive rather than reactive. Such models in use include deterministic models like Hydrological Simulation Program-Fortran (HSPF) and Storm Water Management Model (SWMM) for watershed loading, and CE-QUAL models for pathogen fate and transport (US EPA, 2002). Feedback from some environmental engineers and consultants who apply these models to pathogen and fecal indicator transport suggests they provide highly uncertain predictions for pathogen and indicator concentrations and fluxes (Ali Boehm, Stanford University, personal communication, 2007).

If sanitary investigation models are to be used for criteria development (i.e., prioritizing or discounting procedure for various type of sources), then models that are quantitative yet simple must be available to managers who do not have the resources to run full-scale simulations. These quantitative simple models need to relate land use activities and patterns to the likelihood of human pathogen presence. The ability to rule in or rule out the presence of human pathogen sources in a watershed would be useful to recreational water managers—especially if the EU or WHO approach to criteria development is undertaken. The relationship between land use patterns and microbial water quality has been investigated quantitatively along the California coast (Handler et al., 2006), lakes of South Carolina (Siewicki et al., 2007), North Carolina (Mallin et al., 2000), and Georgia (Fong et al., 2005; Vereen et al., 2007). In Australia, the relationship between land use and watershed attributes and pathogens and pathogen indicators has been applied to numerous catchments using what is termed "pathogen catchment budgets (PCBs)" (Ferguson and Croke, 2005; Ferguson et al., 2007). A sanitary investigation model, for example, might indicate that a completely undeveloped watershed with no agriculture has very low probability of producing runoff containing human pathogens and could potentially place a water body in a "low concern" tier in criteria similar to the EU or WHO approaches (see Tables 1 and 2, Chapter 1). Such models are being developed and used in the U.K. for criteria implementation and development (David Kay, University of Wales, U.K., personal communication, 2007; Kay et al., 2005, 2007).

6.2.2 Water Quality Notification Models

Water quality notification models that are most commonly used are simple heuristic models that relate rainfall to water quality. More complex models currently in use for informing advisory and closure decisions are exclusively statistical models that are used in conjunction with historical water quality data. The models draw on a body of past recreational water monitoring water quality data and temporally-associated physical parameters. The models are developed by assessing and exploring data for parameters that correlate most strongly with variations in water quality detected over the course of monitoring for pathogen indicators. Promising variables are selected, regression models are tested, and the models are refined on the basis of the results obtained using single variables and/or sets of variables.

Another type of "model" for water quality notification is the Heal the Bay Beach Report Card grading system (http://www.healthebay.org/brc/statemap.asp), which provides grades for water quality that are updated daily and formulated using more than one water quality measurement. Given the major uncertainty and variability in measured microbial water quality (e.g., Figures 4a and 4b), this is highly preferable compared to using a single sample to drive public water quality notifications.

One workshop participant (not from the Modeling workgroup) suggested that neural network models be used to model water quality for notification. Neural networks relate independent variables to a dependent variable non-linearly and have been used to model fecal coliforms in some waterbodies (Kumar and Jain, 2006; Neelakantan et al., 2002). However, the Modeling workgroup members agreed that neural network models would not be accessible to the majority of U.S. recreational water quality managers and public health officials in the near-term (5 years). In addition, neural network models have not been used previously for water quality notification, so they are probably not going to be useful in the near-term. They are, however, worth examining in the future.

Simple statistical models have been developed for Great Lakes and West Coast recreational waters that link fecal indicator concentration with meteorological and water quality data/information, and include the following:

- water quality and dynamic hydrologic variables (e.g., water temperature, turbidity, currents, wave height, tide level or range, lake height);
- optical property data (e.g., UV and visible irradiance, light scattering, cloud cover);
- meteorological parameters (air temperature, wind speed/direction, rainfall, pressure); and

- other factors (e.g., bird counts near a recreational water, number of swimmers in the water, video counts of swimmers and wildlife, flow/discharge from a storm drain or nearby creek).

These models have been used very successfully in three states in the Great Lakes to predict the likelihood of exceedance of the current (US EPA, 1986) indicator bacteria criteria for public water quality notification. The models have been shown to be effective in predicting indicator concentrations for compliance and for making timely public health decisions relative to recreational water advisories and beach closures.

The short-term predictions derived from these statistical models have been referred to as “nowcasting.” Nowcasting has been described in the peer reviewed literature (Boehm et al., 2007; Francy et al., 2002, 2003; Hou et al., 2006; Nevers et al., 2005). The variables that are used to correlate with indicator concentrations vary depending on the type of setting of the recreational water. Among the descriptive variables assessed to date, turbidity, rainfall, tides, and wave height have been found to be among the most highly-correlated. The success of these models has been evaluated by their effectiveness in predicting days when current EPA limits have been exceeded and comparing predictions with bacteria concentrations from monitoring on a given day.

Statistical tools such as Swimming Advisory Forecast Estimate (SAFE) and SwimCast (http://www.earth911.org/waterquality/) for Lake Michigan and nowcasting models for Lake Erie are being used to warn the public about potentially unhealthy conditions in recreational waters. Project SAFE is a statistical model used for the five recreational waters in Lake and Porter Counties that extend to the west of the Burns Ditch outfall (Ogden Dunes, West, Wells Street, Lake Street, and Marquette Beaches). These beaches are directly affected by contaminants in the Burns Ditch outfall, particularly during prevailing north wind conditions. Project SAFE models provide a far better real-time estimate of *E. coli* counts than advisories based on single sample monitoring, and are generated for the five beaches simultaneously. Similar applications are being developed for other Great Lakes recreational waters. Another instance of statistical model use is the Ohio Nowcast system. The U.S. Geological Survey (USGS) and Cuyahoga County Board of Health are implementing a pilot Nowcast project to test the use of a statistical model at Huntington Beach, Bay Village, Ohio (Francy and Lis, 2007). Nowcast was used as a tool for recreational water closure decisions for the first time in Ohio in 2006. If the testing goes well, the Nowcast model will be used in subsequent years at other Lake Erie recreational waters.

In all cases where models are being used in the Great Lakes, the modeling is being used to augment microbial water quality monitoring that is being continued as required by the *National Beach Guidance and Required Performance Criteria for Grants* (US EPA, 2002). In Lake County Illinois (SwimCast) all recreational waters are monitored each day in the morning and 5 days per week in the afternoon at locations used for obtaining data for statistical modeling. In Indiana (SAFE model) recreational waters are monitored once a week. In Ohio (Nowcast) monitoring occurs 4 days per week at most Lake Erie recreational waters; and at Huntington Beach, monitoring was increased to 7 days per week during 2006 to provide a large data set to test the accuracy of the Nowcast system.

Hou et al. (2006) and Frick and Ge (submitted) have taken other important steps in developing useful statistical tools for use in recreational water quality notifications. Currently used models are based on long time-series records because models developed from large data sets are generally considered better than models developed from smaller data sets. However, large data sets are developed over time, so this approach is "static." Because conditions at recreational waters are highly dynamic and change from year-to-year and as the season progresses, these authors' models use a dynamic approach in which the descriptive variables are updated periodically.

6.2.1 Communication of Modeled Information to the Public and Recreational Water Managers

Information on modeled projections of water quality has been communicated to the public through the use of a range of communication media and in a variety of information formats. Internet postings, radio spots, and local signs have all been employed in communicating the output of regression model-based advisories. Model outputs intrinsically include an estimate of error. This is expressed in the Nowcast program in a manner similar to the familiar weather forecast probability of precipitation (POP). That is, the likelihood of an exceedance of water quality standards for a given day is expressed as a percentage. In SwimCast, the modeled estimate of fecal indicator concentrations is provided with the average prediction and the upper and lower bounds of the 99th percent confidence limit of the projected figure. Because the value of that number to the general public is limited, a risk explanation is reported based on this statistical prediction in terms of a text description (e.g., low risk if entire confidence interval is below the single sample maximum criteria).

Information on beach water quality can be provided to the public through a tiered approach. The first tier involves communicating a red or green light; that is, simply informing the public on whether or not the recreational water is currently posted with a water quality advisory. The second tier is to provide additional information for those who desire to be more informed and could include posting the measured water quality, environmental water quality data, and the resultant numerical prediction on a website. The third tier is to provide detailed information on the Nowcast system and explain how statistical models are developed and tested, which can also be provided on a website or summarized in fact sheets distributed to the public at the recreational water. A tiered system allows the recreational water user the ability to choose their desired level of information.

Effective communication to the recreational water manager and state and local public health agency representatives is essential for acceptance of a Nowcast or similar system. Presenting the science behind statistical modeling in a simple and concise manner at periodic workshops and meetings is the first step toward gaining acceptance. Because the Nowcast system is different from conventional water quality notification systems already in place (i.e., using the previous day's measured bacterial indicator concentration), local officials may be apprehensive in accepting the new technological approach. Thus, demonstrating to local agencies that the Nowcast system provides a more accurate assessment of water quality conditions may be required before acceptance and implementation is achieved.

6.3 Advantages and Disadvantages of Modeling

The main advantage of modeling for water quality notification is that modeling can provide accurate and timely notification of water quality, whereas day-to-day monitoring cannot. Such modeling may be as simple as a heuristic model or a letter grade for recreational water. More complex models, such as those already in place in the Great Lakes, use multiple regression modeling or similar tools.

An advantage of using a simple sanitary investigation model that relates land use activities and patterns to microbial water quality is that a manager may be able to rule in or rule out the presence of human pathogen sources in a watershed to relax criteria, as is proposed in the EU (EP/CEU, 2006) and WHO (2003) approaches to criteria development.

6.3.1 Advantages of Modeling

- Statistical/regression fecal indicator estimation models are relatively easy to create for an individual with knowledge of statistics and may in some cases only require one variable to adequately describe/predict the pathogen indicator. Several government or private entities currently maintain hydro-meteorological equipment and sensors (e.g., USGS, National Oceanic and Atmospheric Administration [NOAA]) with readily accessible real-time data via the Internet, which could be used at no cost to the recreational water quality manager if deemed appropriate for the specific recreational water. Collected descriptive variables can either be continuous or categorical. Once developed and put into place, statistical models are also easy to use with minimal training required for the recreational water managers and operators.
- Predictions from a sanitary investigation model may allow managers to rule in or rule out human pathogen sources in their watershed and hence relax water quality criteria using an EU or WHO criteria approach. Land use and watershed attributes may be readily available for incorporation into such a model (e.g., Digital Watershed, see http://www.iwr.msu.edu/dw/).
- Water quality notification predictions may be made "near" real-time if required data elements (input variables) exist. This alleviates the delay currently experienced by culturable methods (18 to 24 hours for *E. coli* or at least 24 hours for enterococci). Even with the advent of rapid qPCR (molecular-based) methods, there will continue to be time associated with collection, sample preparation, analysis, and results evaluation. For example, sample preparation adds an estimated minimum of 2 hours in addition to the analysis time. In addition, only the most intensively used waterbodies will likely be monitored with a frequency that will make the best use of the timely results from the use of these methods.
- Collection and analysis delays for both culture- and non culture-based methods currently have and potentially will continue to result in false negative (Type II) advisory/closure errors (e.g., contaminated recreational waters remain open). This is due to the inherent variability of fecal bacteria densities—even over time scales as short as every 10 minutes (see Figures 4a and 4b). Statistical models created for various recreational waters in the Great Lakes have been successfully used to correctly advise/notify the recreational water user of current fecal indicator conditions. Proper public notification should result in

improved public health outcomes and is the major benefit of statistical modeling. It has been well received (instills confidence) by the public and recreational water managers and operators at currently used recreational water locations.

- For recreational waters that have daily (or multiple day per week) monitoring of a fecal indicator and other hydro-meteorological data, costs for creating a statistical model will be low relative to other monitoring/advisory costs. For many recreational waters, initial model creation will require additional water quality monitoring for fecal indicators because it is imperative that the data set on which the model is based include a full range of fecal indicator concentrations for the specific location. However, once the statistical model is created and is validated, the need for daily or weekly monitoring could be reduced, potentially reducing monitoring costs.
- Once the statistical model has been created, both the data-element collection and actual prediction can be automated using current technologies. Although automation initially increases costs (i.e., equipment and programming), personnel costs should be reduced over time.
- Many recreational waters are monitored infrequently due to economic reasons or logistical issues (e.g., difficulty of sampling on weekends). Statistical modeling, if relatively automated, will improve water quality notification activities at these locations, often during highest use days.
- When associated variables become known during model development, a deeper understanding and knowledge of the potential reasons driving increased fecal indicator concentration should assist the recreational water operator (and other interested parties) with future assessments and sanitary investigation work. Simple linear relationships can help to identify potential sources of fecal indicator bacteria (i.e., waterfowl counts versus *E. coli* measurements) and can be used to help design monitoring and microbial source tracking studies.
- Currently used statistical models are based on recreational water quality criteria and thus meet Beaches Environmental Assessment and Coastal Health (BEACH) Act of 2000 and recommended Clean Water Act (CWA) §304(a) single sample maximum allowable fecal indicator density requirements. Because previous studies have demonstrated that the currently-used bacterial indicators are statistically associated with acute GI illness, predictions based on these pathogen indicators should be protective of public health.
- Statistical models could possibly be used to forecast poor conditions at recreational waters using forecasted descriptive variables available from NOAA.
- The statistical approach is flexible and could be applied to prediction of other criteria besides the current culturable *E. coli*- and enterococci-based criteria. However, new data would be required to calibrate the models if the criteria changes and this could be a disadvantage (see more below).

6.3.2 Disadvantages of Modeling

- Because water quality notification models are based on real-time data, prediction accuracy may be diminished by poorly collected or inappropriately maintained equipment. Quality assurance and quality control procedures must be in place for all required input data elements. Recreational water managers and operators (or other individuals) must be diligent in ensuring that proper collection and data management techniques are used.

- Because current water quality notification models utilize statistical techniques, a relatively large (n = 75) and rigorous data set is required to develop the model. Both dependent and assumed descriptive variables should be collected at least 3 to 4 days/week during the recreational water season (if possible). Additionally, the data set should contain a variety of sampling events to capture temporal variability (morning and afternoon) and under both wet and dry weather conditions. It is also necessary to attempt to sample and collect the full range of fecal indicator concentrations for a specific recreational water to help ensure accurate future predictions.
- Politicians, government officials, recreational water operators and managers, and the public may be apprehensive to accept the concept or the need for a modeling-based water quality notification system. Initial support may be difficult to obtain and a local "champion" would be beneficial to advance the concept. The workgroup members noted that once a model is created and accurate predictions are demonstrated, this apprehension would lessen substantially over time.
- Statistical water quality notification models are based on previously collected data and historical associations. Unanticipated events such as sewage spills, large increases in wildlife populations, changes in shoreline from extreme weather events, or new non-point sources of fecal contamination may reduce the predictive ability of the model. If numerous under- or over-predictions occur, additional data collection activities would be warranted to determine whether the model would need to be modified.
- Statistical water quality notification models appear to be most useful at recreational water locations that have occasional but infrequent exceedances of current bacterial water quality criteria. Recreational waters with consistently low or high fecal indicator concentrations may be very difficult to model. Additionally, the need for modeling will be harder to justify as currently accepted monitoring designs may be a preferable and cheaper method.
- Simple statistical models, whether for recreational water quality notification or sanitary investigations, are generally not sufficient for use as deterministic models (e.g., bacterial fate and transport) or to provide load estimates for use in developing TMDLs.
- Current statistical water quality notification models are based on recreational water criteria and thus meet BEACH Act and recommended CWA §304(a) single sample maximum allowable fecal indicator density requirements. However, if ambient water quality criteria for bacteria change, all currently used statistical models will need to be modified to reflect and predict the new criteria. This will result in new costs in the redevelopment and modification of an existing model to incorporate the changed relationships of predictive variables to indicator concentrations. In addition, because fecal indicators are used to predict health risk, the model is only as good as the indicator used.
- There is some confusion as to whether a model output should be measured against a single sample standard and/or a 30-day geometric mean standard. Input from workshop participants revealed that these criteria are used differently around the country with monitoring data. Output from water quality notification models should be used with the

single sample standard and not the 30-day geometric mean standard because it is not clear that model outputs should be averaged for comparison with the 30-day geometric mean standard. Guidance needs to be provided on this issue if new or revised recreational water quality criteria will support the use of models.

- Because water quality is inherently variable, even over a 10-minute scale (see Figures 4a and 4b), how to collect data to develop and validate models needs to be carefully considered. In the Great Lakes, composite sampling is conducted. Guidance for any new or revised criteria that recommend models would need to address this.
- There was some concern from the Implementation Realities workgroup that recreational water advisories or closures instigated by model output would count against them for CWA §303(d) listings or other CWA applications. Guidance for any new or revised criteria that recommend models would need to address this concern.
- Models are site-specific and must be developed for various recreational sites, the same way water quality monitoring must be conducted at specific sites.
- Sanitary investigation models have not been used before for water quality criteria development.

6.4 Model Development and Evaluation

6.4.1 Initiating Model Development for Water Quality Notification

Prior to initiating statistical model development at any recreational water site, a review of all past monitoring and watershed data should be completed. In some cases, enough data may exist to analyze associations between the environmental variables and indicator densities. For example, some states and local agencies collect data on air and water temperature, rainfall, amount of algae wrack, and/or tide level during compliance water monitoring. This type of ancillary data can be used to develop preliminary models and determine if any relationships between indicators and readily available environmental variables exist. This may guide additional monitoring needs and variables to be assessed. As always should be the case, strict quality assurance and quality control practices are to be followed to ensure that a high quality data set has been or will be collected. Additionally, a good understanding of the potential sources and extent of fecal contamination should be determined to aid in choosing sample locations and frequencies. This type of information can be obtained from recently conducted sanitary investigations, historical observations from local water resource managers, and/or visits to the recreational water site.

6.4.2 Model Development for Water Quality Notification

Statistical models have relatively easy to obtain data needs. Data collection should include observations that cover the range of hydrometerological conditions that are expected to impact the recreational water. Sampling should be conducted, at the very least, by collecting at least two recreational seasons of data. A minimum of one recreational season will be necessary for model creation, while the second is used to gather additional data and for model evaluation. Water should be collected four or five times each week and the data set should contain a variety of sampling events to capture temporal variability (morning and afternoon) under both wet and dry weather conditions. It is also necessary to attempt to sample and collect the full range of fecal indicator concentrations for a specific recreational water to ensure accurate future predictions. If current monitoring is conducted on a weekly or monthly basis, serious consideration should be given to increasing data collection requirements as it will take a much longer time period (i.e., 5 years) to develop the model. Generally, a relatively large (n = 75) and rigorous data set is required to develop a water quality notification model. Recreational locations that have consistent good or bad water quality are not good candidates for statistical models. Rather, sites with mixed water quality conditions are the best candidates for statistical models. A representative sample of the waterbody (multiple point grab samples or composite samples for larger recreational areas) should be analyzed for concentrations of fecal indicator bacteria, such as *E. coli* and enterococci, determined by use of an EPA-recommended method.

The descriptive variables for each recreational waterbody will differ from site-to-site. More precise and frequent measurements may lead to better statistical models but also lead to increased costs. However, increased equipment use does lead to automated processes, greater reliability of measurements, and reductions in personnel time.

Water quality notification models use a variety of descriptive variables and all are based on statistical correlations between descriptive variables and indicator organisms. Wave height has been shown to have a positive association with fecal indicator bacteria at some beaches and thus is often included as an independent variable in water quality notification models. Wave height can be estimated visually, measured with a graduated rod, or with pressure transducers. Wave height estimates can also be obtained from an off-site external source, such as a NOAA buoy. Turbidity has also been proven to be a useful factor for use in predictive models. Turbidity can be measured with a field turbidimeter or in situ by use of a turbidity sensor. Models of marine recreational water sites may also include tides (Boehm and Weisberg, 2005; Hou et al., 2006). Insolation, a measure of solar radiation, has been shown to be a useful predictor for fecal indicator bacteria models, since fecal indicator bacteria are sensitive to sunlight (Boehm et al., 2002). Insolation can be measured using a pyranometer on site or provided by external sources (such as NOAA). Rainfall, as well as wind speed and direction, have been included in predictive models. These data can be measured in situ using a weather station or obtained from a reliable source such as operating meteorological stations, which are often located at airports (NOAA, 2007). Streamflow rates from nearby tributaries (USGS, 2007) and effluent discharge rate information from wastewater treatment plants may also be useful factors for inclusion in a predictive model. The number of birds at the recreational water might also prove useful factors for inclusion in a model. Some models presently in use in the Great Lakes for water quality notification use the amount of biological wrack or algal mats as model inputs. Overall, the factors/variables included in a model will be site-specific. A thorough review of factors that might be included in a water quality notification model is outlined in Boehm et al. (2007). Water quality notification models that are most commonly used are simple heuristic models that relate rainfall to water quality (Ashbolt and Bruno, 2003).

Two types of output may be produced by statistical models. The first and obvious output is the predicted microbial concentration and its associated confidence limits. A second output variable is the probability of exceeding an appropriate target value; for example, the probability of exceeding the single sample maximum recreational water quality criteria (Francy and Darner, 2006). Either output may be used to issue advisories or closings of a recreational water site.

6.4.3 Data Needs for Simple Sanitary Investigation Model Development

Because the sanitary investigation model has not been implemented previously for water quality criteria development, data needs are based on characteristics that are important to models used for TMDL implementation. A waterbody manager would need data on land use within a watershed, types and numbers of domesticated and wild animals, publicly (and privately) owned (wastewater) treatment works (POTW) discharges and their degree of treatment and effluent characteristics, number and types of on-site septic systems, type and age of sewage infrastructure, presence of CSO and sanitary sewer overflows (SSO) systems, soil characteristics, and watershed slope. At minimum, such a model could generate a quantitative score of “very likely” to “not probable at all” regarding the possibility of having human pathogens present.

6.4.4 Cost Estimates

There is a wide range of cost estimates for the development, validation, and maintenance of statistical model programs. For all programs, the assumption is that an indicator monitoring program is already in place for the recreational water and computer hardware and statistical software are available. The following are 3 examples of costs for statistical modeling programs for 2 recreational seasons (60 observations per season), starting from the least to most expensive programs.

1. Using existing data from other sources, such as meterological data from the National Weather Service (NWS) and wave height data from NOAA. Expenditures include data compilation and model development (200 hours of computer time).
2. Using existing meterological data from other sources, measuring turbidity, wave heights, and number of birds at the time of sample collection. Expenditures include the purchase of a turbidimeter and standards ($1,200), field measurements (30 hours), and data compilation and model development (200 hours).
3. Installing in situ site-specific instruments for measurements of wave heights, turbidity, wind direction and speed, and rainfall amounts. Expenditures include the purchase and installation of equipment (a one-time cost of $15,000 to $20,000), maintenance of equipment ($2,000/year for replacement and manufacturer calibration of equipment and 80 hours), and data compilation and model development (200 hours).

6.4.5 Understanding the Uncertainty and Measuring Success of Statistical Models

The natural complexity of environmental systems means that it is difficult to develop complete mathematical descriptions of relevant processes, including all of the intrinsic mechanisms that govern their behavior. Model evaluation is defined as the process used to generate information to determine whether a model and its analytical results are of sufficient quality to serve as the basis for decision making (CREM, 2003). Once a statistical model is constructed, it is important to describe its usefulness or success. A regression model is built using a “training” data set comprised of dependent and independent variables (Boehm et al., 2007). The ability of the model to predict the dependent variable using independent descriptive variable inputs within the training data set can be described by a root mean square error (RMSE). A coefficient of determination (R^2) can also be used and is interpreted as the percent of the variation of the independent data set described by the model. However, the workgroup members agreed that this was not the best metric for evaluating model performance. A third metric for testing the performance of a model is to examine the number of Type I and Type II errors that result. Assuming the null hypothesis is that a recreational water is in compliance with a water quality regulation and should be open to the public, a Type I error occurs when a recreational water is closed or posted with a warning when it should not be (i.e., false positive), while a Type II error occurs when a recreational water is not posted or closed when it should be based on the water quality regulation (i.e., false negative). These two types of errors can be summed to determine the total errors. The number of such errors is a function of the specific policy used by recreational water managers in making water quality notification and closure decisions.

Model evaluation must be conducted using a data set with which it was not trained before it can be applied as a predictive tool. Model evaluation is defined as the process used to generate information to determine whether a model and its analytical results are of a quality sufficient to serve as the basis for a decision (CREM, 2003). It can only be completed if an appropriate evaluation data set of independent and dependent variables not used to train the model is available. The success of a model during evaluation is described by the root mean square error of prediction, which has the same mathematical formation as the RMSE. The number of Type I and II errors, as well as the total error rate is also calculated. The model's performance is then compared with the current method for assessing recreational water quality (i.e., using the previous day's measured bacterial indicator concentration).

At a Lake Michigan recreational waterbody during 2004 (Olyphant, 2004; Pfister, 2007), swimmers were exposed to a health threat without warning on three occasions and kept out of the water when it was safe on only one occasion when a water quality model was used to make recreational water closure decisions. In contrast, swimmers would have been exposed to a health threat without warning on 19 occasions and kept out of the water when it was safe on 12 occasions if daily morning monitoring data alone had been used to notify the public of health risks.

Because every model contains simplifications, predictions derived from the model can never be completely accurate and the model can never correspond exactly to reality (CREM, 2003). After model validation (e.g., those that have been shown to correspond to field data), an additional year of data can be added to the model development process and a new model with another year of data is developed for use in subsequent years.

The information about model evaluation presented above is an overview. The peer reviewed literature should always be examined for new ideas and thoughts about model evaluation.

6.5 Research Needs

Research needs for simple, statistical models are categorized below regarding near-term activities (2 to 3 year horizon) of immediate relevance to implementation and development of new or revised rcriteria in recreational waters to long-term research activities, such as elucidation of processes affecting pathogen/indicator fate and transport, development of non-point source models for catchments or watersheds, and deterministic models for TMDL development. There were differences of opinion among workgroup members about how important TMDL model development is for the long-term for criteria development and implementation.

6.5.1 Near-term Research Needs (2 to 3 years)

There is an immediate need to conduct research for development of models that can be used for water quality notification. Statistical (or empirical) models are most promising for this purpose because they are relatively cheap and simple, and readily accessible to most recreational water quality managers (see Chapter 7). Statistical models link microbial concentrations with meteorological and water quality data/information. Recent research has lead to the development of useful statistical models for some Great Lakes recreational beaches (Francy and Darner, 2006; Francy et al., 2003; Frick et al., 2005; Olyphant, 2005; Whitman and Nevers, 2004; Whitman et al., 2006) and marine coastal beaches (Hou et al., 2006). Although these statistical models have successfully predicted criteria exceedances under a variety of environmental conditions, statistical modeling studies must be extended to a variety of other recreational waters to evaluate fully the utility of this approach.

Near-term research needs for water quality notification include the following:

1. Day-to-day water quality notifications should not be issued using a single sample standard in conjunction with a microbial assay that takes longer than a few hours due to notification errors. Simple, heuristic or statistical water quality notification models can help avoid notification errors (**all 5 workgroup members [5/5] agree**).
2. Immediate research needs include the following:
    a. Testing whether models can be used to predict health outcomes during upcoming epidemiology studies at Doheny Beach (California) and in Alabama and Rhode Island, and as well as the already completed epidemiology studies done in the Great Lakes (described by Wade et al., 2006) (**high priority [5/5]**);
    b. Developing and testing simple notification models on different recreational water types with a wide range of sources and geographical locals (**high priority [5/5]**);
    c. Exploring the feasibility of developing regional models that apply to more than one recreational water (**low priority [5/5]**);
    d. Training recreational water managers (**high priority [3/5]**, **low priority [2/5]**, there was disagreement on whether this belonged on the research list);
    e. Creating an excellent user-friendly portable package for developing local models (**high priority [5/5]**); and
    f. Developing dynamic predictive modeling methods (refers to models where variables are constantly updated over time) (**high priority [2/5]**, **medium priority [2/5]**, **low priority [1/5]**).

1. Linking statistical models to health effects. One approach would be to concurrently conduct modeling studies along with planned epidemiological studies that will be conducted by EPA and the Southern California Coastal Water Research Project during the upcoming year in California, Alabama, and Rhode Island. In addition to measurements of microbial concentrations, appropriate data for model development should be collected during the epidemiological studies (e.g., turbidity, irradiance, wind speed/direction, wave height, tides, temperature).

Another approach would be to retrospectively develop statistical models for sites of past epidemiological studies in the Great Lakes, where appropriate data relevant to statistical modeling have already been collected (or can be obtained from existing meteorological data). For example, statistical models have been developed for Huntington Beach, Ohio, and West Beach, Indiana—both of which are sites of past NEEAR epidemiological studies (Haugland et al., 2005; Wade et al., 2006).

2. Developing statistical models for different types of recreational waters. To test the feasibility of the statistical modeling approach, research is needed in recreational waters that are impacted by different sources of biological contaminants (non-point or point sources such as POTWs) and that are described by a wide range of meteorological and water quality variables. Waters that are significantly impacted by POTWs or non-point agricultural sources will be accorded the highest priority in site selection because past studies have shown that these sources are most likely to adversely affect human health. Sites located in the following regions should be considered for this research:

- West Coast (open ocean and confined beach);
- East Coast (open ocean and confined beach);
- Gulf Coast;
- inland lakes/reservoirs;
- rivers with designated primary contact recreational use; and
- tropics and subtropics.

3. Dynamic approaches to statistical modeling. Currently used models are based on long time-series records that take at least 2 years to obtain. The regression constant and coefficients are held constant when the model is used to predict (generally Nowcast) conditions. Once established, the models are changed only at the end of season to incorporate new data. Other recent research suggests that model performance may be improved by using a dynamic approach in which the descriptive variables are updated periodically with data generated within a limited recent period—usually on the order of 30 to 60 days. Using the dynamic modeling approach, the predictions of bacterial concentrations have been significantly improved (Frick and Ge, submitted; Hou et al., 2006) and the time period for model development may be reduced. An alternative approach would be the development of a sliding seasonal band of data using multi-year data from the period surrounding the date of interest. Additional research is required to refine this approach, either though use of previously obtained data sets or data obtained at sites that will be used for the first two activities (i.e., linking statistical models to health effects and developing statistical models for different types of recreational waters).

4. Communicating and training modeling techniques. Various activities can improve the communication of modeling techniques and results to the public and training recreational water managers, including the following:

- creating a user-friendly portable package for developing local models;
- training for running statistical models for recreational water managers; and
- including recreational water managers on the decision process and polling them regarding their perception of its usefulness/feasibility.

Training is an important component towards acceptance and implementation of statistical models by recreational water managers and public health agencies. In a November 2003 workshop held as part of the Great Lakes Beach Association Annual Conference in Green Bay, Wisconsin, recreational water managers expressed as a high priority the need for informed training on statistical models and other recreational water monitoring activities. Similarly, training is being provided by EPA Region 5 on statistical model and sanitary survey (investigation) development in April 2007 at the request of recreational water managers.

5. Explore the feasibility of developing regional models (e.g., for southern Atlantic coast recreational waters). At present, simple water quality notification models are site-specific. The feasibility of using a regional scale model that predicts water quality regionally, within a large waterbody, for example, should be explored.

Near-term research needs (next 2 to 3 years) for sanitary investigation models include the following:

1. Simple, heuristic, statistical/conceptual models that correlate watershed activities (e.g., presence of treatment plant effluents, agricultural activities, domesticated animals) and attributes (e.g., slope, soil type, climate, soil moisture) can be used to determine the probability of a waterbody having inputs of human pathogens (**all [5/5] agree**).
2. Research in the near-term should be carried out to better understand how watershed activities and attributes relate to pathogen presence in streams and receiving waters, including the following:
    a. factors that modulate septic tank impact on waterbodies (**high priority [1/5] medium priority [1/5]**, **low priority [3/5]**);
    b. factors that modulate contributions of animal wastes to pathogen and pathogen loads to waterbodies (**high priority [5/5]**);
    c. sources in urban landscapes such as broken/leaky sewer pipes, CSOs, stormwater and urban runoff (**high priority [5/5]**); and
    d. effect of geographical and climatic setting on non-point source delivery (**high priority [5/5]**).

6.5.2 Longer-term Research Needs (8 to 10 years)

A variety of research needs are required to be able to develop an excellent model that would allow prediction of fecal indicators or human pathogens. Additionally, important sources, and fate and transport processes will need to be elucidated. These research needs will require a longer time horizon for completion and are summarized below.

1. **Processes that affect fate and transport of pathogens and fecal indicators for incorporation into deterministic models and improving statistical models indicators.** This long-term research effort involves the development of data and descriptors of processes that are required in deterministic models that predict fate and transport effects on pathogen concentrations in recreational waters. Process information can also be used to help define appropriate variables to use in statistical models. Such research would focus on partitioning of microorganisms to suspended and bottom sediments and sands, mortality of pathogens and indicators, zooplankton grazing on fecal indicators and pathogens, and the possibility of bacteria proliferation in the environment. Some of these processes are shown in Figure 5. In addition to these processes, a better understanding of mobilization of pathogens and pathogen indicators from sources within a watershed (i.e., from animal feces) and source strength from POTWs and CSOs are needed (**high priority[5/5]**).

2. **Research on GIS layers relevant to modeling.** In order to develop viable models, recent and relevant GIS data need to be readily available and usable for models (e.g., POTW locations, recent land use categories, storm sewer locations). Digital Watershed

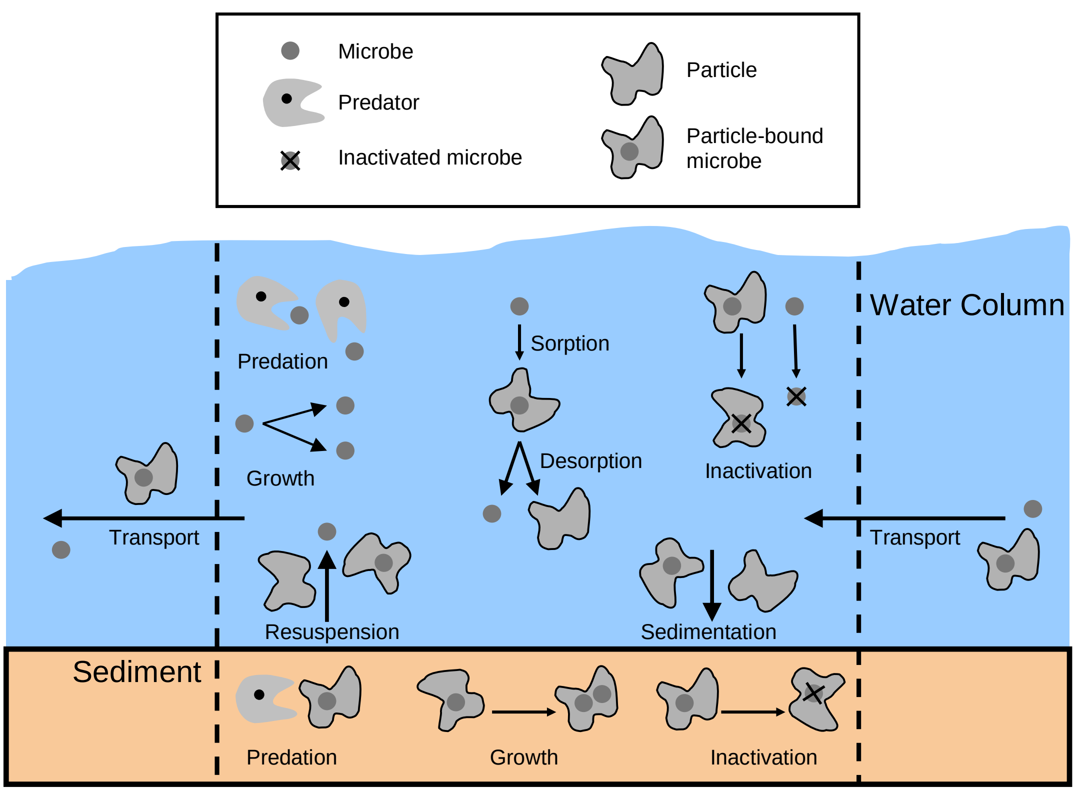

Figure 5. The Possible Fates of Microbes (Fecal Indicators and Pathogens) in Environmental Water and Sediment (the fate of nucleic acids may be different; this figure does not include those sources). SOURCE: Adapted from Olivieri et al. (2007).

is one example of a GIS-based software that can be used to provide inputs for deterministic models such as L-THIA (**high priority [1/5]**, **medium priority [4/5]**, **[1/5]** does not think this is a research need).

3. **Combining deterministic models with statistical models.** This research involves using outputs from deterministic models as inputs for statistical models that would be used for water quality notification and sanitary investigation purposes (**high priority [0/5], medium priority [5/5]**).

4. **Forecasting using statistical models.** This research will seek to expand current efforts (e.g., by Frick and Ge, submitted) to use forecasted variables (such as wind speed and direction, precipitation, wave height, and turbidity, if available) to forecast concentrations of biological contaminants in recreational waters (**high priority [2/5]**, **medium priority [1/5]**, **low priority [2/5]**).

5. **Development of deterministic models of pathogen and fecal indicators for criteria implementation and development** (**high priority [3/5]**, **medium priority [2/5]** there was concern that these would not be really used by recreational water managers and that this is already being done if resources permit).

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CHAPTER 7
IMPLEMENTATION REALITIES

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EPA requests that experts consider implementation realities when providing input to all specific and general questions throughout this document.

The Implementation Realities workgroup members were charged with providing input to EPA and the experts participating on the other six workgroups concerning the practical implications of incorporating any proposed changes to the recreational bacteria criteria into State Water Quality Standards (WQS) and subsequent impacts on existing water quality management programs. To this end, the workgroup members met frequently with and actively participated in the deliberations of the other workgroups over the course of the workshop. Implementation issues and concerns are therefore incorporated into the individual workgroup chapters throughout these proceedings. This chapter provides a summary of the major areas of concern identified by the Implementation Realities workgroup during the deliberations resulting from the workgroup's internal discussions as well as discussions with other workshop participants.

At the most basic level, the success of implementing any new initiative depends on providing resources and guidance that are adequate to accomplish the stated objectives. Where additional effort is needed, either additional resources must be obtained or existing resources must be diverted from other activities. Workgroup members attempted to evaluate resource needs as a critical component of implementing new bacteria criteria across a broad spectrum of programmatic responsibilities from conducting necessary research to educating stakeholders and gaining acceptance of the public and regulated entities of program changes, to actual impact on the day-to-day implementation of water quality management programs.

The results of the discussions are presented in three sections. First, an evaluation of the four principal program areas where recreational bacteria criteria are currently employed: (1) water quality beach notification and advisory programs; (2) National Pollutant Discharge Elimination System (NPDES) permitting, including regulation of wastewater treatment facilities, urban stormwater, and combined sewer overflow and sanitary sewer overflow (CSO/SSO) discharges; (3) monitoring and assessment programs required for compliance with Clean Water Act (CWA) §303(d) and §305(b) purposes; and (4) development of total maximum daily loads (TMDLs) for waters identified as not meeting State WQS. The second section provides an evaluation of the implementation concerns that must be addressed that relate specifically to three potential approaches for the development of new or revised recreational water quality criteria. The third and final section identifies the specific areas of research that workgroup members considered to be most critical to facilitating implementation efforts.

7.1 Application to Specific Program Areas

7.1.1 Beach Monitoring and Water Quality Notification Programs

The objective of this program is to provide accurate and timely information to the public regarding the health risks associated with participating in recreational activities at marine and freshwater beaches. Significant concerns have been expressed regarding both the accuracy and timeliness of the information currently provided.

The most pressing need for regulatory authorities who conduct beach monitoring programs is to get better information to the public as quickly as possible regarding the safety of the recreational water. There is currently a minimum 24 hour delay between the time when a water sample is collected, tested, and when the results of the test are available. Thus, decision makers only know what the quality of the bathing water was like yesterday.

"Rapid Tests"

Research in the past few years has resulted in the development of molecular-based tests that can provide results in just a few hours following the initiation of the test compared to 24 hours for the currently used culture-based tests.

Rapid tests have several benefits. They shorten the time from when an unsafe water condition occurs (an "exceedance") to when the test reveals the existence of an exceedance. This provides a capability to shorten the time it takes to post an advisory or to close the beach during unsafe conditions. The reduced test period thereby reduces the public health risk. The shorter test period also shortens the time it would take to remove the advisory and/or reopen the beach when water quality returns to a safe condition. Thus, the period of "loss of beneficial use" is also reduced. Because test results can be obtained in a shorter period, it is possible that they could be used to aid fecal pollution source identification efforts such as in identifying a problem in a specific location by enabling more samples to be analyzed in a shorter period of time.

Although there is a desire to use the new, rapid tests in beach monitoring programs, several issues related to their use must first be resolved. First and foremost, it must be shown that these new "molecular" methods provide a level of human health protection equal to or above that provided by the currently used tests. States need to know that there is a beneficial reduction in illness to justify the costs of adopting and implementing a new test methodology.

While rapid tests are sometimes referred to as "real-time" tests, they are not in fact real-time tests as there is still a delay of several hours between water sampling and test results. The public may still be exposed to potentially unsafe water for some period of time, albeit likely a shorter time period compared to current culture-based methods used to measure indicator organism levels. The rapid tests will not shorten the time required to collect water samples and deliver them to the test laboratory (typically 4 to 5 hours or longer), nor will they shorten the time required to convey test results to the appropriate authorities and the public (1 to 2 hours or more).

Many States only have the resources to sample periodically (e.g., weekly, monthly) as opposed to daily. The new tests are not likely to provide authorities with resource savings sufficient to analyze water quality more frequently. However, the ability to obtain test results faster may raise the expectation of the public or regulatory mangers that, since the tests are faster, additional samples can or should be collected and tested—even when this may not be possible due to resource constraints. Taking full advantage of the benefits associated with more rapid tests will likely require additional resources for increased monitoring.

Before any new test can be used broadly, the EPA will have to adopt and validate a standardized method for its use. State and local public health officials use the results of monitoring to make health-based decisions to close or open a beach, or to issue or lift a beach advisory. These officials need to know that the analytical method they use provides reliable results; therefore, they only endorse methods that have already been validated by EPA.

Further, to be able to bring a faster test into routine use, issues related to test equipment, training, laboratory capacity, and certification of laboratories will need to be resolved. The initial capital cost and any ongoing operation and maintenance costs need to be calculated and compared to that of the currently used tests. Regardless of how "good" the more rapid tests are, if they are too expensive, regulatory authorities may not be able to afford them.

In addition, because the test endpoints of molecular-based tests are different than culture-based tests, a new regulatory scheme may need to be adopted to accommodate the new water quality criteria. See the discussion in Section 7.1.2 for further information on this topic.

For the public and local authorities, a period of time may be required to gain "acceptance" of the new indicator.

In general, any change in current monitoring practices (e.g., sampling type, frequency, location) necessitated by a change in recreational water quality criteria will need to be carefully considered relative to benefits offered because it will involve resource issues and many implementation concerns.

Predictive Modeling

Changes in microbial indicator counts in recreational waters are typically controlled to a large extent by a variety of meteorological and water quality factors. Data for many of these factors (e.g., wind, rainfall, etc.) can be obtained in real or near-real time. By monitoring and identifying which of these factors control indicator count changes, it is possible to create "predictive models" (see Chapter 6). Such models are essentially mathematical equations that have the "controlling" meteorological and/or water quality parameters as components. A "robust" model that is validated by comparison of predicted indicator concentrations to a sufficient number of actual concentrations is able to successfully predict, within a stated degree of precision, when unsafe water conditions will exist more accurately than the currently used culture-based assays are able to do.

Predictive modeling offers great promise because it estimates when there may be a problem *prior* to the bather exposure. The use of predictive models may also reduce the need for rapid testing. Furthermore, they can be employed daily, providing information beyond that available from periodic microbial monitoring. However, it is important to note that predictive models are not themselves criteria. Predictive models are tools that can be used to evaluate compliance with criteria.

Models are only as good as the data used in their construction. If critical data are not available, a valid model cannot be developed until those data are obtained. As discussed in Chapter 6, the amount of data, especially microbial monitoring data, required to develop a predictive model within a stated confidence level may be significant. In general, model development may require significant time and resources depending primarily on the availability of data on indicator densities and associated predictive variables (e.g., antecedent rainfall, wind direction, wave height, etc.).

Currently developed predictive models appear to be site-specific. A predictive model developed for one beach or location is not likely to be usable at other beaches or locations because the effect of a predictive variable such as wind direction on indicator densities will be different at each beach. Therefore, for each "problem" beach or location, a separate model (i.e., set of predictive equations) is likely to be required.

Competing financial resources may make modeling a low priority. For example, limited funds may have to be used for higher priority tasks such as improving impaired waters (e.g., fecal source identification).

Any proposed use of modeling results for compliance purposes is likely to present implementation difficulties. Model results may not be always accepted as "proof" of a water quality standards violation because of the inherent uncertainty associated with model results. That is, regulators are likely to require actual monitoring data rather than modeling output for compliance purposes, particularly if non-compliance may lead to legal enforcement action. Regulators as well as members of the public often perceive monitoring as accurate and modeling as estimates.

Statistical models are currently used in some States to assess compliance with their water quality standards for purposes other than beach monitoring. If there is a change in the criteria (as would occur if a new indicator is adopted) then corresponding model would have to be modified, which would require additional resources.

It is important to note that modeling should not supplant routine water quality monitoring, which will always be needed to detect unanticipated events such as a sewer line break. Thus, regular monitoring provides an ongoing, direct measure of microbial water quality. Monitoring also provides data to help improve the precision of model predictions.

General Considerations

Workgroup members felt that any new or revised recreational water quality criteria need to allow for a binary (pass/fail) decision (e.g., close or not close a beach), must be a numeric, and must be based on a health risk determination for water quality notification/closure purposes. The criteria for reopening a closed beach or removing an advisory should be the same as that used for the initial closure or advisory. New or revised criteria must be expressed in a way that the authorities using the criteria are able to fully explain the criteria and their health risk basis, in a readily understandable way to the public.

New or revised criteria should have some "flexibility"; for example, there may be State-specific circumstances and the criteria will need to be able to be used in all such circumstances. At the same time, the new or revised criteria need consistency so that the public has confidence that their health is being protected.

Any new or revised criteria should be tied to a specific method unless "equivalency" of the new method to a previously used (and validated) method can be demonstrated to facilitate implementation (see Chapter 3 for further information).

The development of guidance to implement new or revised criteria should occur simultaneously with the development of the criteria. The implementing authorities will need assurance that the new criteria will be effective in ensuring that public health goals are met.

Finally, the successful implementation of new or revised criteria will very likely result in the need for increased funding for microbial source tracking (see also Chapters 2 and 3) and for beach management programs.

7.1.2 NPDES Permitting Programs

The purpose of the NPDES permitting program is to insure that point source discharges of pollutants to waters of the United States achieve the statutory required level of treatment and do not cause waters to exceed State WQS after discharge. This is accomplished by imposition of the more stringent of either technology-based or water quality-based limits on discharge quality and mandating discharge monitoring at a frequency adequate to insure compliance with permit limits and conditions.

Tiered Approach

Water quality criteria might be expressed in a tiered approach; that is, that the criteria include multiple attributes, each of which apply for a specific purpose. With respect to NPDES permits, the tiered approach should be workable as long as one attribute of the criteria is specifically developed for NPDES requirements. This would necessitate choice of a pathogen indicator that achieves NPDES needs (see more below).

In addition, NPDES effluent limits are developed with an implicit exceedence rate. NPDES permitting guidance for water quality-based effluent development is based on a wasteload allocation that is calculated based on an exposure condition that represents the upper 99th percentile of conditions (e.g., conditions occurring under rare low flows such as the 7Q10 [the lowest streamflow for 7 consecutive days that occurs on average once every 10 years]) when point source discharges have the greatest impact on water quality conditions. As a result, it is important that water quality criteria include an allowable exceedance frequency to facilitate permit limit derivation. This is particularly important for deriving permit limits for pathogen indicators in wet weather conditions because the flow conditions at the time of discharge can be extreme and represent rarely occurring situations.

Pathogen Indicators

Changes in pathogen indicators from the current ones (*E. coli* and enterococci) will significantly affect implementation, especially if the change results in a different indicator being used for TMDL modeling than for permitting or uses an indicator that cannot reflect the efficacy of wastewater (sewage) treatment practices (disinfection). At a minimum, the indicator used for

NPDES permitting needs to be sensitive to disinfection so that the permitting authority can determine that the NPDES regulated facility is adequately disinfecting its discharge. If the indicator cannot do so, then there will be a need for different indicators for ensuring the discharge achieves water quality standards and the wastewater is properly disinfected. Another way to accomplish this is to develop an approach that translates between the various indicators.

Analytical Methods

There is concern that molecular-based methods may not adequately verify that wastewater disinfection has been effective. This concern is based on research that shows the qPCR (quantitative polymerase chain reaction) signal does not decrease post-chlorination. Many State public health codes require disinfection of human waste and the analytical method used for NPDES permitting needs to be able to measure disinfection. As a result, a molecular-based method may not be suitable to fulfill all NPDES needs.

It is also important for implementation that the analytical methods be tested in a wastewater matrix and approved for use in wastewater. NPDES regulations require that effluent monitoring be conducted using either an EPA-approved analytical method or an analytical method specified in the permit. In the latter situation, the permit documentation needs to defend the use of the method. However, many States do not have the technical experience to defend analytical methods or have legal restrictions on the use of alternative methods and thus must rely solely on use of EPA-approved methods.

Resources

Many NPDES regulated dischargers conduct analysis of their wastewater on-site. The existing laboratory expertise of these dischargers may not be sufficient to conduct analyses for new pathogen indicators (e.g., molecular-based methods). The start up cost of purchasing equipment for conducting the new analyses and additional training for staff poses a resource drain for both the dischargers and the regulatory authority that must provide oversight. Should the dischargers choose to contract out their laboratory analysis, they will need to pay to ship the samples to the contract laboratories, which is also a resource drain.

Finally, many states require that laboratories be certified for analysis with certification being specific to the parameter being analyzed. Therefore, States will need to amend their laboratory certification program to include the new pathogen indicators. This is also a resource drain on States.

7.1.3 Monitoring and Assessment for CWA §303(d) and §305(b)

The purpose of this program is to provide an accounting of the condition of the Nation's waters, identify those that do not meet current State WQS for focused mitigating action, and to track progress in improving the overall quality of the Nation's water resources.

Assessment and listing based on the current ambient water quality criteria (AWQC) have disproportionately focused State resources on what are often perceived as minimal to nonexistent public health issues. States have expressed frustration at being effectively handcuffed by strict application of the criteria and the inability to adjust assessment findings based on other data indicating the health risk is significantly lower than implied by the criteria exceedance. Such factors include evidence that elevated indicator levels are not due to human sources of fecal contamination and hydrologic factors that preclude recreational exposure, such as during or immediately after high rainfall events. Areas where improvements can be made in the new or revised criteria and implementation guidance associated with the criteria includes monitoring, criteria, guidance, and (inland) flowing waters.

Monitoring

Workgroup members felt that new or revised recreational AWQC must include a clear discussion regarding linkages between an advisory/closure decision at a beach and assessment of use attainment. Beach advisories/closure decisions may, but need not necessarily, be linked to such assessments. There may be instances where beach advisories or notifications are made based on models, or special circumstances (such as sewer line breaks) that should not be counted as non-attainment for assessment purposes. In a similar vein, if the beach advisory regulations are more stringent than State WQS, the advisory in and of itself should not constitute non-attainment unless the State chooses to list that beach as impaired on that basis.

Ambient Water Quality Criteria

Alternative AWQC or methodologies that more precisely define health risk would be highly useful in improving assessments—in particular indicators of human versus nonhuman pathogens. The criteria and implementation guidance need to recognize the potentially lower risk of pathogens from nonhuman sources and provide a way for addressing and discounting pathogen and indicator data not associated with anthropogenic sources of fecal contamination.

The criteria must also be sufficiently flexible for assigning attainment of use based on limited data sets, particularly for inland waters. Often, States only collect data on a monthly, bi-monthly, or annual basis and compare these data to previously collected data to assess trends. The problem will be exacerbated for assessment purposes if new or revised criteria are adopted. It could take years to develop a statistically significant data set.

If the format of the new or revised criteria requires a specific number of samples to be collected in a set timeframe, States will be challenged as they are with the current criteria (e.g., 5 samples over a 30-day period). Criteria that allow assessment samples collected at any frequency to be statistically manipulated to the appropriate exposure frequency would allow States to maintain their current monitoring approaches while appropriately applying the criteria.

Also, for ease of State implementation, new or revised criteria need to allow for some reasonable excursion frequency. Criteria expressed as a percentile value (e.g., cannot exceed criteria more than x% of time) would provide an incentive to conduct additional sampling so as to not have the assessment rely on one or two samples and would facilitate implementation for assessment purposes.

If the European Union (EU; EP/CEU, 2006) or World Health Organization (WHO, 2003) approach for criteria development is followed, there needs to be a clear distinction between the criteria that is needed to protect human health and what is considered to be supplemental guidance. For instance, is it possible to have a "good" beach or a "very good" beach and still be considered non-impaired? Apparently, "good" meets the criteria while "very good" is a desired higher level of microbial water quality. Such discussion should be in supplemental guidance rather than in the criteria.

If a rapid method is selected as the indicator, the speed of a rapid method offers no additional benefit relative to assessment, unless the rapid method provides more precision/better protection to benefit public health. Therefore, a rapid method may offer the benefit of more rapid water quality notification, but has little positive effect on the overall assessment process that is conducted on data collected over a 2 year period.

Workgroup members expressed concerns with establishing a new or revised recreational water quality criteria linked to a sanitary investigation. If a WHO-type criteria model is chosen that includes use of a sanitary investigation to modify the criteria and allow for nonhuman sources of fecal contamination, the frequency of performing that investigation would need to be identified in assessment guidance. There was a strong preference among workgroup members that the frequency be longer than the two year assessment cycle for State's issuance of assessment information pursuant to §303(d) and §305(b) of the federal CWA. The available information for the sanitary investigation did not specify the frequency for repeating such investigations.

Lastly, for assessment purposes, there needs to be some way to translate between previously used indicators and any new indicator(s) so information from past monitoring is not lost. If a "translator" is not available, it might take several years to build up enough information to conduct a statistically valid assessment for pathogen indicators.

Guidance

If new criteria indicator/methodology combinations are adopted, issuance of guidance for implementation will be imperative. With the likelihood of rapid molecular-based test methods, sanitary investigations, and so on, guidance will need to accompany the criteria to help States understand how to apply the new or revised criteria and thus achieve State acceptance.

Flowing Waters

Flowing freshwaters (e.g., streams, rivers) present some unique challenges that have not been addressed with previous epidemiological studies of recreational waters. Therefore, if new or revised criteria include application to flowing freshwaters, consideration needs to be given to an allowance for different values/applications of the criteria to reflect the differences in hydrologic regime (e.g., extreme high flows) through one of the following:

- higher criteria that applies in extreme events; or

- changes to the use/criteria when the use is not taking place (e.g., when recreation is unlikely to occur such as during winter months or during or immediately after heavy rainfall).

Lastly, an indicator applicable to flowing freshwaters needs to be identified. As stated elsewhere in these proceedings, *E. coli* appears to be a more appropriate freshwater indicator of fecal contamination than enterococci. *E. coli* are a subset of fecal coliform bacteria while enterococci bacteria are a separate group of enteric bacteria. More recent water quality data generated using *E. coli* can be more easily compared to earlier water quality data generated using fecal coliform bacteria than can more recent water quality data generated using enterococci bacteria.

7.1.4 Total Maximum Daily Load Program

The purpose of this program is to establish the maximum pollutant load that a specific waterbody can assimilate and apportion that load among sources of that pollutant to the waterbody, leading to the development of a management plan that when fully implemented will result in reducing those loads to the extent that State WQS are achieved and maintained.

TMDLs for bacteria designed to achieve consistency with the current (US EPA, 1986) criteria are typically difficult to develop and explain to stakeholders because expressing pollutant loadings of bacteria or pathogens in terms of mass is nonsensical. Pathogens or pathogen indicators are not measured as mass but rather as cell counts (e.g., colony forming units [cfu]). Developing wasteload allocations for point sources and load allocations for non-point sources in mass units does not make sense to the vast majority of TMDL practioners and those responsible for implementing bacteria TMDLs. For this reason, alternative means of expressing loading reductions (e.g., "percent reduction," "load duration curve-based," "reference watershed" methods) have been used by many States. TMDL development for waters impaired by excessive indicator bacteria densities is further complicated in that the necessary load reductions are typically strongly linked to hydrologic factors and intermittent sources such as stormwater runoff. Establishing a static steady-state design condition, as is frequently done for other types of pollutant impairments, is not possible for bacteria due to the significant wet weather event-driven characteristics of many bacteria-impaired waters.

Workgroup members viewed criteria expressed in numerical terms as a practical necessity to implementing any revised recreational use criteria in TMDL programs due to the need to quantify loadings. Implementation realities dictate that the criteria be expressed in terms that facilitate calculation of an acceptable daily loading under a range of hydrological conditions. The criteria has to be a number (as opposed to a category/classification) to make implementation in TMDL programs feasible. The workgroup experts expressed a diversity of opinions over the benefits of a geometric mean or other statistic versus single sample maximum criteria with specified exceedance frequency for water quality assessment and TMDL purposes. Some prefer use of single sample maximum (SSM) while others prefer geometric mean largely reflecting current practice in their particular State. If the new or revised criteria are expressed as a single value, the benefits of allowing for that value to be exceeded at some stated frequency for TMDL and assessment purposes cannot be overstated. EPA should expect intense resistance from Sstates if future criteria guidance proposes criteria expressed as a "never to be exceeded" value.

An acceptable exceedance frequency is critical to facilitate design of treatment requirements and best management practices (BMPs) to implement the TMDL as well as accounting for rare extreme event-driven conditions not practical to mitigate. Providing States (and other stakeholders) with evidence that the criteria incorporate flexibility to accommodate the variability inherent in bacterial densities in natural systems would greatly facilitate acceptance and subsequent implementation efforts.

Criteria that distinguish between human and nonhuman sources of fecal contamination would also make TMDL development significantly easier. The ability to make allocation decisions would be enhanced and public acceptance of the TMDL implementation requirements would be achieved much more readily if additional confidence could be provided in estimates of source category loading. Further, the ability to adjust TMDLs based on more accurate source separation and to make allowances that "discount" the contribution of certain lower risk sources (e.g., non-anthropogenic) or sources from which the contributed risk may be lower (e.g., wildlife) would encourage States to move forward to adopt the criteria into their WQS. If the criteria or implementation protocol includes a sanitary investigation there should be guidance provided to encourage consistency in sanitary investigation methodologies among States. This guidance might be a combination of minimum expectations and general framework for what constitutes an acceptable sanitary investigation. A mandate to provide confirmation of investigation results through alternative means (e.g., microbial source tracking, use of more human-specific indicators) may also be acceptable provided the cost and technical difficulty are not prohibitive or use of this additional step is only required in selected instances where the results of the investigation are not conclusive.

7.1.5 Important Differences Between Workgroup Members as to Views/Observations

Workgroup members had a diversity of opinions over the benefits of a geometric mean-based as opposed to AWQC based on SSM for certain water quality assessment and TMDL purposes. Some preferred the use of a SSM-based standard, while others preferred the use of a geometric mean-based standard. One of the times of potential concern is when an individual sample result may be over the SSM but the data set does not exceed the geometric mean. The concern is that some event may have occurred during that time and the public could potentially be at risk; however, it is also possible that the result is a one-time occurrence and the public is not at a greater risk than at other locations that meet the geometric mean-based criteria.

7.2 Evaluation of Alternative Approaches for Criteria Development

This section describes the implementation considerations for each of the three alternative approaches for the development of new or revised recreational water quality criteria that were proposed and discussed at the workshop (see Chapter 1). Some of the concerns regarding implementation that are common to all three approaches include the following:

- level of discriminatory power/sensitivity of a method;
- if rapid method is used, difficulty in implementation in some places (e.g., holding time); and

- if site-specific epidemiological studies are needed, most States will be unlikely to fund these studies.

Many of the above concerns, as well as the concerns described in the following sections, would be eliminated if the following statements were true:

- epidemiological studies demonstrate that indicator organisms are sufficiently correlated to human health risk;
- studies provide a scientific basis for discounting risk to human health from wildlife sources of fecal contamination;
- criteria included flexibility to account for the reduced exposure (and thus, lower risk) of use at extreme conditions (e.g., high flow);
- relationships between advisories and impairments were more clearly defined in EPA guidance;
- level of disinfection necessary to provide adequate pathogen reduction/inactivation in human sewage was determined; and
- criteria applied for NPDES purposes included flexibility to account for wet weather conditions.

7.2.1 WHO Approach

The WHO approach provides a range of risk levels and accounts for differences in relative risk resulting from site-specific considerations of sources of indicator organisms based on the results of a sanitary inspection performed prior to the assessment of monitoring results. The following implementation concerns are not specific to any specific application of the WHO model, but rather reflect the general use of this approach.

The WHO (2003) approach to criteria development relies on identification of the potential for human sources of fecal contamination to impact a beach or other recreational water area. Many pathogens are host-adapted and so human fecal sources may contain many pathogens not found in feces from non-human animals (e.g., *Salmonella typhi*, *Vibrio cholerae*, *Cryptosporidium hominis*, *Entamoeba*, many viruses). Thus, it is essential to have available a reliable methodology to distinguish between human and natural sources (e.g., wildlife only) of pathogens for use of the WHO model. As part of this, the methodology should also be able to either quantify that the risk from natural sources is low or provide some way to characterize the risk from natural sources as being acceptable. It is important to characterize or quantify the risk from natural sources rather than to completely discount it because this risk needs to be included in beach advisory decisions. For example, if pathogens from sea lions pose a risk to humans, then it is important to post an advisory on a beach where sea lions reside. However, it would not be necessary to consider this risk in determining impairment because sea lions are a “natural” source and most environmental agencies would not view development of a plan to eliminate sea lions as consistent with their overall mission.

It is also important to be able to quantify the risk from domestic animals and livestock and include this risk if a WHO-based approach is pursued. Although these sources of fecal contamination are nonhuman in nature, these animals live in close proximity to humans and may carry human (zoonotic) pathogens in their feces(e.g., *E. coli* O157:H7, *Cryptosporidium parvum*; see also Strauch and Ballarini, 1994). Use of the WHO model will require including the likelihood of these sources impacting beaches and other recreational water areas. As a result, it becomes important to quantify risks of exposure to fecal material of these animals.

The WHO approach appears to be amenable for use with multiple pathogen indicators (e.g., the toolbox). If multiple pathogen indicators are used in application of the WHO model, then all the considerations related to use of both molecular and culture methods that were discussed for each CWA application above apply. In addition, if multiple WHO model tables are used, it may be advantageous to develop separate tables for lakes and flowing waters because exposure in these two situations are different.

There are several implementation issues that arise if the WHO model is applied using a qPCR analytical method. The first issue is the capacity of States and NPDES dischargers to adopt and use a qPCR method, as initially, there may be insufficient laboratory capacity to conduct the method. Specific concerns with respect to NPDES facilities are discussed in the preceding Section (7.12) on the NPDES permitting program. Additionally, it is reasonable to expect that the initial costs per sample will be substantially higher than for the currently used culture-based methods, which poses an additional cost to States and NPDES facilities.

The second implementation issue with respect to qPCR is its apparent inability to confirm that disinfection is being properly applied. As discussed previously, NPDES permits need to both assure that WQS are achieved and that State disinfection requirements are being met. If qPCR method is used to apply the WHO approach, then another indicator using culture-based methods will be needed in NPDES permits to demonstrate adequate disinfection.

Another implementation issue is the use of sanitary investigations based on the WHO approach. However, the protocols for a sanitary investigation should not be overly prescriptive to the point of making the investigation resource-prohibitive. There is a need to define the minimum elements of a sanitary investigation to ensure that it is reliable. Application of the criteria needs to invoke trust by the public. If there is too much variety in sanitary investigations, then the public will perceive that the investigations have no technical rigor and which will undermine use of the WHO model. In addition, States will need to develop the capacity to conduct sanitary investigations on every waterbody with recreational uses, which constitutes a resource burden. Finally, States need sufficient time to conduct sanitary investigations by the time the new or revised criteria are adopted into their WQS.

The WHO approach includes columns that characterize different risk (see Table 1, Chapter 1). Two of the columns include water characterizations of "very good" but are associated with different risk. The model should be applied with only one "acceptable risk" level. If there is more than one acceptable category of good, it implies there is more than one "acceptable risk" level. This makes it difficult to explain to the public, difficult to enforce, and difficult to make decisions on the lower risk level. Any further distinction between "good" and "very good" outcomes should be voluntary.

It is possible that States will issue advisories in situations that are not considered as CWA impairments. This can occur when a state public health agency wants to impose a higher degree of protection than the state environmental agency, or at beaches where there are wildlife sources that pose risk. It is uncertain how such a situation would work with the WHO model, and this would need to be developed.

The WHO model uses ranges of pathogen densities. This allows States to select which specific value to use, and thus result in inconsistencies on thresholds to close or open beaches between various states. It is much more preferable for the criteria to specify one threshold rather than a range. However, it was discussed that while the range may be difficult to implement in a regulatory fashion, it may more realistically describe the precision of epidemiological-based criteria applied to a wide range of waters coupled with the precision of indicator measurement.

Finally, it appears that empirical models of pathogen densities can be used with the WHO model, as long as one threshold is used rather than a range.

7.2.2 EU Approach

The EU approach provides defined criteria at a single risk level but allows for adjustment of the assessment result based on a sanitary investigation performed following review of monitoring results.

Like the WHO (2003) approach, the EU (EP/CEU, 2006) model uses sanitary inspections; however, unlike the WHO approach, the EU model uses the inspections to rationalize that monitoring results above the criteria levels do not indicate an elevated risk to human health. Thus, the rigor of any type of sanitary investigation that may be required for an approach based on the EU approach seems to be greater than for WHO-based approaches (i.e., requires a more detailed site assessment). A workgroup participant indicated that for some waters a desktop GIS-based methodology could constitute a sufficient sanitary survey for many bathing waters (Paul Hunter, University of East Anglia, U.K., personal communication, 2007).

As was the case for the WHO model, there are implementation concerns regarding the time and capacity for conducting sanitary investigations, and the ability to distinguish between risks from human and nonhuman sources of fecal contamination. Specifically, States will need to know how good are the techniques to distinguish between risks from human and nonhuman sources, and what is the degree of risk from nonhuman sources. Thus, the discussion of the WHO approach on these topics likewise applies to the EU approach.

As one way to implement the EU model, EPA could use a “pristine” watershed as a baseline. In this situation, EPA would look at pathogen indicator counts at baseline flows and use these values to determine how to adjust concentrations.

The EU model process presents opportunities to be more transparent to the public than the WHO approach. States could seek public involvement in determining how to conduct the sanitary investigation/discounting process.

One impediment to implementation of the EU approach is how domestic and agricultural animals are addressed. It appears that these sources can be excluded; however, these fecal contamination sources may have a potential risk to human health.

The EU model characterizes beaches using the 95th percentile of a set of microbial water quality data. This seems to prevent making short-term decisions for beach closure or reopening unless beach managers use some sort of predictive modeling. This is an implementation concern given the aforementioned (see Chapter 6) data needs of models. Not all recreational water sites can currently develop a model due to limited data. If there is no model, then decisions would likely be based on a data set over a period of time, rather than a specific data point, which would require interpretation for beach monitoring for closing or opening decisions.

Like the WHO approach, the EU approach includes columns that characterize different risk (see Table 2, Chapter 1). The model should be applied with only one "acceptable risk" level. If there is more than one acceptable category of good, it implies there is more than one "acceptable risk" level. This makes it difficult to explain to the public, difficult to enforce, and difficult to make decisions on the lower risk level. Any further distinction between "good" and "very good" outcomes will make implementation difficult in some jurisdictions.

7.2.3 Existing U.S. Model – 1986 Criteria

The existing model provides defined criteria at a single risk level but does not provide for adjustment based on other sources of information such as sanitary investigations or source identification.

The original basis for the (EPA) 1986 criteria were freshwater and marine water epidemiological studies conducted at a limited number of sites with restricted geographic extent and waterbody type (lake beaches and marine beaches). Therefore, a concern exits that single value criteria may not be applicable to all waters across the United States—for instance, inland flowing waters, tropical waters, or freshwaters under tidal influence. In the development of new or revised criteria, epidemiological data or quantitative microbiological risk assessment (QMRA) for as wide a variety of fresh and marine waters as is possible should be used.

If single value criteria are to be developed, as was the case for the 1986 criteria, it is vital to use as many indicators as necessary to best characterize the microbiological quality of the water. There is a variety of opinion as to the most appropriate indicators for fresh and marine waters. However, there is evidence that *E. coli* is the most suitable indicator for flowing freshwaters while enterococci, either by culture- or molecular-based methods, is most suitable for marine waters; however, the workgroup did not reach a common opinion on the evidence.

A major criticism of the 1986 criteria was the lack of approvable test methods for wastewater effluent. If new indicator organisms or test methods are identified for the new criteria, approved test methods must be developed for all potential needs such as NPDES permitting and ambient water quality monitoring.

The 1986 criteria provide minimal implementation guidance. Due to most States' interpretation of the criteria in their WQS, the criteria tend to be treated as requiring compliance at all times and in all waters. This interpretation has caused considerable problems in the assessment and TMDL arenas. Any new or revised criteria must include implementation guidance that allow for methods to address issues such as extreme flows and nonhuman sources of fecal contamination.

EPA needs to provide more scientific data and information to States for implementation of pathogen indicator criteria. States have concerns regarding the effectiveness of existing sewage treatment capabilities on new indicator organisms. In switching from enterococci or *E. coli* as an indicator, there is concern that disinfection designs may not meet permit limits based on the new indicator criteria. This issue needs to be addressed by EPA so that the State programs will have consistent, valid, and scientifically defensible responses when these concerns are raised during the implementation of new WQS.

7.2.4 Alternative Approaches

Two additional potential approaches to consider in the development of new or revised recreational water quality criteria include the following:

1. An alternative hybrid approach could blend the single value criteria with facets of the WHO (2003) and EU (EP/CEU, 2006) models to allow for demonstration of mitigating (or discounting) factors to be completed by a fixed date after criteria adoption. This has the advantage in preventing waters from being CWA §303(d)-listed based solely on excursions above a single value criteria. If the water was ultimately listed, it could be de-listed at a later date if it were demonstrated that mitigating factors prevented designated use attainment.
2. The largest implementation concern with the single value (EPA) 1986 criteria is regarding assessment. An alternative approach to developing new criteria could incorporate the existing 1986 criteria with the following implementation provisions:
   - a provision to discount non-compliance with the single value criteria after investigation of the contributing watershed to confirm the absence of nonhuman sources and lower risk than implied by the criteria exceedance;
   - criteria/use inapplicability during extreme high flow events; and
   - a process to exclude natural sources of fecal indicator organisms (i.e., indicators specific to human sources are not present), according to the corresponding risk to human health.

7.3 Research Needs

Research is clearly needed to provide support for implementing any alternative approach to criteria development, expression, or application. A key concern is the role research results play in the ability of State and federal regulators to explain and gain public acceptance of changes in existing CWA programs. Opportunities to leverage the value of individual research programs by employing data collection designs that may be useful to answer multiple questions should be exploited.

7.3.1 Near-term (Next 1 to 3 Years)

Beach Monitoring

1. Provide a quantitative protocol to identify the types of nonhuman sources of fecal contamination. For example, other than molecular-based fecal source identification techniques, are there methods (e.g., sanitary investigations) to track nonhuman sources such as waterfowl, dogs, horses, and other anthropogenic sources? The WHO and EU approaches to criteria development provide for "discounting" exceedances if it can be determined they are of nonhuman origin through a sanitary inspection. If risks from nonhuman sources can be adequately quantified, sanitary inspections could be used to support decision making.
2. Determine the risk from different types of nonhuman sources of fecal contamination (e.g., domestic and indigenous wildlife). Although the new or revised criteria would need to address all potential risks, a delineation of the categories of risk made available to the public would improve water quality notification and informed consent aspects of implementation. Specifically, the perceived risk associated by the public with elevated concentrations of indicators derived from indigenous sources (e.g., deer, birds) may be more acceptable than sources of domestic origin (e.g., cattle, poultry). The public may wish to make an informed decision about usage relative to specific pathogens such as enterohemorrhagic *E. coli* (EHEC) that are potentially associated with agricultural land usage.
3. Determine under what conditions a sanitary investigation would be sufficient (as opposed to microbial source tracking). This research is most important if the WHO and EU approaches are being considered.
4. Identify minimum elements that a sanitary investigation should include. Again, the focus should be on the minimum elements necessary for a reliable sanitary investigation. If the requirements for a sanitary investigation are too onerous, they will become resource-prohibitive and of minimal value. Assess the reliability, accuracy, and validity (etc.) of the various types of sanitary investigations. Without some sort of standardized investigation criteria, inconsistencies will result in the implementation of the criteria and create potential variances in health risk levels at beaches.
5. Predictive modeling offers the prospect of benefits to beach management that are sufficiently significant such that it should be explored further. An identification of data needs is required for such models. For water quality notification purposes, models should be developed and calibrated to assure a minimum confidence level.

NPDES

1. Conduct studies to develop a methodology to compare the correlation of the culture-based methods and the qPCR (molecular-based) method. Identify how or where the same level of protection can be provided, even if implementation is different. Any requirement to use non culture-based methods may have significant impacts on NPDES permit monitoring programs. Non culture-based methods may not adequately assess treatment processes or determine permit compliance.

2. Develop an improved understanding of disinfection using the different indicators. Determine how well each indicator is in measuring disinfection effectiveness, including determination of the viability of the organisms (pathogens and indicators) following various disinfection processes (e.g., chlorination, UV light).
3. Determine risks of exposure from intermittent microbial pollution discharges, CSOs, urban runoff, and concentrated animal feeding operations (CAFOs).
4. Evaluate the effectiveness and cost of stormwater and agricultural BMPs as related to pathogens and microbial contaminants. This evaluation should be made in concert with epidemiological studies/QMRA analyses that will determine the risk from different types of sources (urban and agricultural runoff, indigenous and domestic animals, regrowth).
5. Evaluate the efficacy, costs, and benefits of disinfection for the purposes of supporting eventual promulgation of a disinfection rule. It is anticipated that disinfection could eventually be promulgated as a mandatory treatment technology nationwide as it already is in many States. Specifically, research is needed to support levels of disinfection necessary to provide adequate pathogen reduction/inactivation.

Use Attainment

1. Research to determine the risk from different types of nonhuman sources of fecal contamination (e.g., domestic and indigenous) is needed to better quantify the risk from nonhuman sources so that when implemented at recreational waters, those risks are better accounted for.
2. Develop criteria or methodologies that more precisely define the health risk associated with pathogen exposure in recreational waters.

Overall

1. Conduct research so that monitoring using indicators can help to distinguish human from nonhuman sources of fecal contamination.
2. Conduct epidemiological/QMRA studies on flowing recreational waters. Current (1986) criteria were based on epidemiological studies conducted in relatively static waterbodies. Additional studies are needed to assess risks in flowing waters. This has significant implications for criteria development for inland U.S. waterways.
3. Need to better understand the health-basis for allowable exceedance frequency. Additional explanation is needed to justify percentile criteria differences between WHO, EU, and EPA (1986) criteria development approaches (e.g., use of 95^{th} or 90^{th} percentile).
4. Conduct research to better understand how to measure the impact of regrowth and persistence in sediments of indicator bacteria on water quality. The source of some problems of high pathogen indicator levels may at times be due to regrowth rather than urban runoff, animals, birds, biofilms, ocean circulation, etc.

7.3.2 Long-term (Beyond 3 Years)

NPDES

- Develop a viability assay for the viral and protozoan portion of effluent.

Overall

- Develop methodologies that are pathogen-specific.

References

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APPENDIX A: CHARGE TO THE EXPERT WORKGROUP MEMBERS

PURPOSE

The purpose of the Pellston-type[1] Experts Scientific Workshop on Critical Research and Science Needs for the Development of Recreational Water Quality Criteria is for EPA to obtain individual input from members of the broad scientific and technical community on the "critical path" research and science needs for developing scientifically defensible new or revised Clean Water Act (CWA) §304(a) recreational water quality criteria in the near-term.[2]

BACKGROUND

An important goal of the CWA is to protect and restore waters for swimming. Section 304(a) of the Act directs EPA to publish "advisory water quality guidance on the effects of the presence of pollutants in water on health and welfare." These recommendations are referred to as §304(a) criteria. Under §304(a)(9) of the CWA, EPA is required to publish water quality criteria for pathogens and pathogen indicators to protect swimmers from illnesses associated with pathogenic microbes in coastal and Great Lakes waterbodies.

In adopting new or revised water quality standards, States must adopt criteria that are scientifically defensible and protective of the use, but they have flexibility to do so by adopting EPA's recommended criteria, adopting criteria to reflect site-specific conditions, or adopting other criteria that are scientifically defensible. In the case of criteria EPA publishes under §304(a)(9), States with coastal and Great Lakes waters are required to adopt EPA's new or revised criteria for pathogens and pathogen indicators into State Water Quality Standards (WQS).

Once adopted into State WQS, water quality criteria express the desired ambient condition of the water to protect a designated use. State WQS are used for various CWA purposes or programs that identify and address the sources of pollution with the goal of attainment of the criteria, including National Pollutant Discharge Elimination System (NPDES) permits, water body assessments to determine use attainment, and development of Total Maximum Daily Loads (TMDLs). In addition, these WQS used by States in beach monitoring and water quality notification programs.

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[1] A workshop similar in organization and format to the Society of Environmental Toxicology and Chemistry (SETAC) Pellston Workshops where technical experts in a particular subject area are invited to participate and evaluate current and prospective environmental issues. A Pellston-type workshop brings together between 40 to 50 technical experts from academia, business, government, and public interest groups. Experts are sequestered for a week and expected to contribute to a summary report. Subject leaders are then responsible for consolidating, editing, producing, and distributing the workshop proceedings.

[2] Near-term requirements: in order for EPA to develop criteria in the near-term, the indicators/methods/tools upon which they are based must be currently available, have undergone scientific peer review and validation, and ready for day-to-day implementation in State public health/environmental laboratories within the next 2 to 3 years. New or revised criteria must be based on indicator/methods that are easy to use and interpret.

Historically, EPA's recommended criteria for protecting people who recreate in water have been based on fecal matter in recreational waters. In the 1960s, the federal government recommended using the indicator bacteria, fecal coliforms, as the primary contact recreational[3] criterion. In the late 1970s and early 1980s, EPA conducted public health studies evaluating several organisms as possible indicators, including fecal coliforms, *E. coli*, and enterococci. The studies showed that enterococci are a good predictor of gastrointestinal (GI) illnesses in fresh and marine recreational waters, and *E. coli* is a good predictor of GI illnesses in fresh waters. As a result, EPA published in 1986 revised criteria (*EPA's Ambient Water Quality Criteria for Bacteria – 1986*[4]*)* for primary contact recreation recommending the use of *E. coli* for fresh recreational waters (criteria set as a geometric mean of 126 colony forming units [cfu]/100 mL) and enterococci for fresh and marine recreation waters (criteria set as geometric means of 33/100 mL in freshwater and 35 cfu/100 mL in marine water). These recommendations replaced EPA's previously recommended bacteria criteria for fecal coliforms of 200 cfu/100 mL. EPA's criteria recommendations use "indicator" bacteria. Most strains of *E. coli* and all enterococci do not cause human illness (that is, they are not human pathogens); rather, they merely indicate fecal contamination, and the assumption is that pathogens co-occur with incidences of fecal contamination.

Since EPA issued its recreational criteria over 20 years ago, there have been significant scientific advances, particularly in the areas of molecular biology, microbiology, and analytical chemistry. EPA believes that these new scientific and technical advances need to be factored into the development of new or revised CWA §304(a) criteria for recreation. To this end, EPA has been conducting research and assessing relevant scientific and technical information to provide the scientific foundation for the development of new or revised criteria. The enactment of the Beaches Environmental Assessment and Coastal Health (BEACH) Act of 2000 (which amended the CWA) required EPA to conduct new studies and issue new or revised criteria, specifically for Great Lakes and coastal marine waters.

OVERALL CHARGE TO THE EXPERTS

Experts are asked to provide their individual knowledge and insight that will help EPA define the critical path research and science needs, recognizing the "state of the science" and the reality that research that cannot be completed within 2 to 3 years will not be helpful in EPA's near-term criteria development efforts. Experts should focus their efforts at this Workshop on identifying near-term research and science needs that will allow EPA to publish new or revised criteria in roughly 5 years. (While EPA understands that experts may wish to offer perspectives on research and science needs for the development of future or "next generation" criteria, this is not the primary purpose of this Workshop.) "Next generation" criteria refer to criteria EPA may publish in the longer term; that is, in approximately 10 to 15 years, pursuant to CWA §304(a)(9)(B). Section 304(a)(9)(B) directs EPA to review and, as necessary, revise the §304(a)(9) criteria 5 years after EPA publishes the initial criteria, and every 5 years thereafter.)

[3] Primary contact recreation includes activities that could be expected to result in ingestion of water or immersion. These activities include swimming, water skiing, surfing, and other activities where contact and immersion in water is likely.

[4] US EPA. 1986. *Ambient Water Quality Criteria for Bacteria - 1986*. EPA440/5-84-002. Washington, DC: US EPA.

Although not the focus of this Workshop, EPA is aware of stakeholder concerns regarding implementation issues associated with the existing (EPA) 1986 criteria and the desire on the part of some stakeholders for EPA to address these issues in the interim (i.e., before EPA publishes a new or revised recommended criteria). In recognition of these concerns, experts in the "Implementation Realities" Workgroup are encouraged to identify aspects of the 1986 criteria which have been cited as problematic, and, to the extent that these issues can be remedied through new or revised criteria, offer individual input for EPA to consider in the criteria development efforts.[5]

The new or revised criteria must be scientifically sound, protective of the designated use, implementable for broad CWA purposes, and when implemented, provide for improved public health protection. By scientifically sound, EPA means that the criteria must be based on the science and peer reviewed studies available at the time the criteria are developed. By protective of the use EPA means that the criteria must establish the desired ambient condition of the water to protect the designated use (e.g., primary contact recreation) given to the waterbody. EPA's new or revised criteria must also serve the broad purposes for which CWA criteria are intended, including beach monitoring and water quality notification programs, development of water quality based effluent limits for National Pollutant Discharge Elimination System (NPDES) permits, waterbody assessments to determine use attainment, and development of total maximum daily loads (TMDLs), where needed. Lastly, the new or revised criteria, when implemented, should also provide for improved public health protection and States must be satisfied that the underlying science is sound and that the numeric values of allowed pollutant in recreational waters will achieve the desired environmental result.

On the last day of the Workshop, the chairs for the individual breakout topic groups will provide EPA with sections of a draft Expert Report. Each of these sections will summarize the individual input provided by the experts and collected by the Chairs throughout the week's discussions. The Chairs will be asked to summarize commonalities and differences in the input provided by participants, and list out the projects and activities that the individual experts identified as critical to the development of new or revised CWA §304(a) criteria in the near-term, recognizing that research that cannot be completed in 2 to 3 years will not be useful in near-term criteria development efforts. (The workgroup chairs may also summarize any research and science needs identified by the experts for developing "next generation" criteria.)

The draft Report will include a summary of expert views on the following topics: appropriate pathogens or pathogen indicators, along with available and appropriate methods; single versus "toolbox" criteria approach; implementation issues; and most importantly, identification of critical technical issues and uncertainties that could be addressed with near-term research.

EPA contractual support will be available to the Chairs during the workshop to provide assistance in preparing the draft Report. After the workshop, EPA contractual support will be available to the Chairs to finalize their component of the Report in 1-month's time. EPA will use

[5] To the extent that experts come to some conclusion on how to better implement the 1986 criteria, EPA intends to track these issues separately in order to not depart from the primary purpose of the meeting which is to obtain input on critical research needs for the development of the near-term criteria.

the Report as it develops a critical path science plan that will guide research activities over the next 2 to 3 years.

Presented in the following sections of this document are key questions on seven major overarching issues pertaining to criteria development and implementation. A threshold issue that impacts the deliberations of all groups is whether EPA should consider a fundamental change in its approach to recommending recreational criteria; for example, switch from a single criterion in all places to a diversified toolbox or tiered approach, using multiple criteria, or several tools supporting a single criterion, or some other combination.

Break-Out Group #1: Approaches to Criteria Development (See Chapter 1)[6]

Single versus "Toolbox" Approach: A single criterion and/or method may not adequately address all CWA needs. One approach for new or revised criteria may consist of several "tools" (i.e., indicators, methods, intrinsic geographic factors, etc.) to fulfill all of the specific CWA needs. For example, it could involve using molecular methods and rainfall models for beach monitoring and water quality notification, and possibly other method-indicator combinations for other CWA uses—provided that all criteria and methods are comparable in terms of level of protection provided. For example, the definition of an impaired recreational water in terms of the number of people that would get sick when the water is not in compliance cannot differ from the illness rate that triggers a beach advisory or closing.

The following set of questions is intended to guide a robust discussion among the experts in this group. The results of this discussion will improve the understanding of the advantages and disadvantages of various approaches to criteria development.

1. *What approaches exist currently for setting limits of pollutants that may be relevant for developing nationally recommended recreational water quality criteria? Consider approaches used for other kinds of pollutants in water, in other environmental media, and by other countries as well as approaches being implemented by States. What are the pros and cons of each of these approaches?*
2. *Which of these approaches is most applicable and appropriate for developing nationally recommended recreational water quality criteria in the near-term? Why is this approach on balance considered the most applicable and appropriate?*
3. *For those approaches identified as applicable and appropriate, what is the science that supports the approach? Is that science sufficient and of adequate quality?*
4. *Are there any critical research and science needs that should be addressed in developing or selecting an appropriate approach? Can this research be completed in time to be used in criteria development in the near-term?*
5. *Is a "toolbox" approach appropriate for developing new or revised recreational criteria in the near-term? Why or why not?*
6. *What are the pros and cons of selecting a "toolbox" approach?*

[6] Because breakout group numbers do not correspond to chapter numbers in these proceedings, chapter numbers are referred to for easier reference.

7. *What are desired features or characteristics that would make a "toolbox" approach appropriate?*
8. *Would a "toolbox" approach achieve additional public health protection as compared to another approach? Why or why not? If unknown, what science would need to be completed in order to determine whether a "toolbox" approach would achieve additional public health protection?*
9. *Criteria for secondary contact recreation could be part of a "toolbox." What approaches would be appropriate for developing criteria for secondary contact recreation? Would this approach be different from that used to develop primary contact recreation criteria? Why and why not?*
10. *What are critical research and science needs in developing or selecting an appropriate approach for secondary contact recreation? Can this research be completed in time to be used in criteria development in the near term?*
11. *What are the implementation considerations of the different approaches for CWA purposes (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs? Are there practical considerations that could preclude, or greatly limit, the use of an approach in routine, regulatory implementation (e.g., field sampling issues, laboratory challenges, staff training, etc.)?*

Geographical Applicability: Options for ensuring criteria are appropriate in a diverse range of recreational waters include EPA recommending geographically different approaches, numbers, or indicators, applicable to different regions (e.g., fresh and marine waters, coastal and inland waters, tropical/subtropical and temperate waters) or types of waterbodies (e.g., lakes and flowing waters).

1. *Is a single criterion available that is applicable for the diverse range of geographic conditions? Why or why not?*
2. *Is a "toolbox" approach appropriate for different geographical conditions? Why and why not?*
3. *What would a "toolbox" that addresses geographical differences look like?*
4. *What are critical research and science needs in developing or selecting an approach that will appropriately factor-in diverse geographical conditions?*

Expression of Criteria: EPA is currently assessing the degree to which criteria should be expressed as the mean concentration over a period of time (e.g., 30 days) and/or as a daily or instantaneous maximum value.

1. *Given the diverse needs of the CWA programs and the overarching goal of protecting and restoring waters for swimming, what protection is provided by establishing a 30-day "average" value as the criteria? What additional protection (if any) is provided by a daily or instantaneous maximum value? From a scientific standpoint, is one measure better scientifically than another for particular purposes (e.g., mean value for purposes of identifying impaired waters and daily maximum for beach monitoring and notification purposes)? Why?*

2. *What are pros and cons of expressing the criteria differently for the various CWA program needs?*
3. *What are the implications of instantaneous or daily values for public health protection? If we don't currently have a good understanding of this, what are the critical research and science needs to answer these questions?*
4. *If EPA were to set criteria at a mean concentration over 30 days and not recommend a single sample maximum, do we understand the illnesses that could occur on a single day (where the level would still lead to compliance with the 30 day average)?*
5. *If the science is not there, what are the critical research and science needs to answer this question?*
6. *What are the implementation considerations for CWA purposes of failing to address (and addressing) differences geographically in the criteria and failing to include (and including) a single sample maximum value for (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs? Are there practical considerations that could preclude, or greatly limit, the usage in routine, regulatory implementation (e.g., field sampling issues, laboratory challenges, staff training, etc.)?*

Break-Out Group #2: Implementation Realities (See Chapter 7)

Although EPA wants the experts to consider implementation realities when providing input to all general and specific questions throughout this document, the following set of questions are intended to guide a robust discussion among the experts about implementation issues and how science and research could ease implementation.

1. *What are the essential implementation considerations as EPA develops new nationally recommended recreational water quality criteria for CWA purposes: (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs?*
2. *What are the major lessons learned in implementing the (EPA) 1986 criteria? What worked well and not so well? How could we avoid repeating past "mistakes" that lead to delays in adoption or difficulties in implementing these criteria?*
3. *Which approaches to criteria development have the most potential for success in implementation when new or revised criteria are adopted into State water quality standards? Why?*
4. *What are general features or characteristics that would make new or revised criteria easy to interpret and implement for states when adopted into State water quality standards? Why?*
5. *Would a "toolbox" approach be easier or more difficult to interpret and implement? What are desirable characteristics of a "toolbox" criterion from an implementation perspective?*
6. *If new or revised criteria are provided as a range of values instead of a single value, what implementation concerns are triggered (e.g., can a range of values be used when developing NPDES permit limits or TMDL calculations)?*

7. *What are critical path research and science needs that would enhance implementation of new or revised criteria in the near-term?*

Break-Out Group #3: Pathogens, Pathogen Indicators,[7] and Indicators of Fecal Contamination (See Chapter 2)

Indicator Approach: EPA previously developed criteria based on indicators of the potential presence of human pathogenic organisms; that is, based on indicators of fecal contamination. Other possible approaches such as pathogen index microorganisms and specific pathogens are discussed below.

The following set of questions is intended to guide a robust discussion among the experts toward the identification of critical research and science needs in the development of criteria based on pathogens, pathogen indicators or indicators of fecal contamination. It is essential that this group focus discussions on only those pathogens, pathogen indicators or indicators of fecal contamination where methods are ready now for day-to-day use in State public health and environmental labs or where methods will be ready for day-to-day use in these labs within the next 3 years.

A. Fecal matter indicators (as surrogates for gastrointestinal and non-gastrointestinal diseases):

1. *What are the benefits and shortcomings for continuing to implement the current fecal indicators (*E. coli *and enterococci) to meet each of the CWA §304(a) criteria uses (beach notification, TMDLs, NPDES permits, listing of impaired waters) to protect swimmers health from (a) gastrointestinal disease? (b) upper respiratory tract disease? (c) other diseases (skin, ear, eye disease)? Should other CWA §304(a) uses be tied to health outcomes?*
2. *Are there other microbial fecal indicator(s) that can be used to better meet each of the CWA §304(a) criteria uses and provide improved protection against diseases (e.g.,* Bacteroides *spp.,* Clostridium perfringens*, coliphages or other phages)? Why?*
3. *Are there any chemical biomarker fecal indicators (e.g., fecal stanols, detergents, whiteners, caffeine) that can be used to better protect public health and meet all CWA purposes than the current indicators of fecal contamination?*
4. *What critical research would improve or widen the selection of fecal indicators available for the criteria?*

B. Pathogens and their Index organisms (gastrointestinal and non-gastrointestinal disease):

1. *Would a specific pathogen or index microorganism approach present an improvement in health protection over fecal indicators for each CWA use if applied as §304(a) criteria? If yes, then see question #2. If no, what research could be done to support this*

[7] A specific pathogen belonging to a broader group of pathogens which would serve as a surrogate for the presence and/or health risks for that group (e.g., *Cryptosporidium* serving as a surrogate for all parasitic protozoa); or an indicator microorganism whose presence is correlated to the presence of a broad group of pathogens (e.g., spores of *Clostridium perfringens* serving as a surrogate for human or dog parasitic protozoa).

approach? (also for skin, upper respiratory tract, ear, eye disease criteria considerations)

2. *What are the advantages and disadvantages of this approach?*
3. *What might be the most appropriate pathogens or index organisms? Why?*
4. *What data support a dose-response relationship between a particular pathogen or its index in recreational water and any disease outcome?*
5. *The BEACH Act requires that EPA conduct research and develop new or revised water quality criteria for "Pathogens and Pathogen Indicators." The Act defines a pathogen indicator as a substance that indicates the potential for human infectious disease. How might the term "index microorganism" relate to the statutory term pathogen indicator?*
6. *What is the critical research to make the selection of pathogen/ index organisms available for the new or revised criteria and for the next generation criteria?*

Application of Alternatives: The following two sections present some possible applications of a mix of approaches that may increase the potential to improve monitoring, better express health risks from swimming exposures, and be more comprehensive in their use to meet all criteria needs and provide more efficient and cost effective procedures.

C. <u>Application of fecal indicators, pathogen index organisms, and pathogens in combination for criteria:</u>

1. *If none of the above three groups of surrogates can meet all CWA §304(a) criteria needs, is there any combination of the three that would provide an acceptable criteria approach?*
2. *What specific combined applications would have merit in meeting criteria needs?*
3. *Would the combined applications best utilize an analytical toolbox approach or a tiered analytical approach?*
4. *Would the criteria endpoint reflect a general gastrointestinal disease target or a dose response estimate base on more limited disease symptoms reflecting the metrics used?*
5. *What research is important to make the selection of combinations available for the new or revised criteria and the next generation criteria?*
6. *Can adoption of the WHO/Annapolis Protocol approach*[8] *that combines sanitary reconnaissance survey information along with microbial assessment to develop surrogates of fecal contamination (predictive modeling) on the day to manage water advisories provide improved health gains over current criteria? Are there sufficient examples of this approach to develop new/improved use of indicators/surrogates in the near term?*

[8] WHO (World Health Organization). 2003. *Guidelines for Safe Recreational Water Environments. Volume 1 Coastal and Fresh Waters*. Geneva, Switzerland: WHO.

D. Applications of all the above for all categories of waters, climatology, and geographical considerations:

1. *Will the choices of individual, combined, or tiered fecal indicators, index organisms or pathogen indicators, or pathogens selected from above be capable of working for each or all of the following:*
    a) *Freshwaters (flowing and lakes/ponds)? Marine waters? POTWs? TMDLs?*
    b) *Temperate waters? Tropical waters?*
    c) *High matrix waters (high in solids)? Special conditions?*
2. *What science or research is important in the near term to make the determination in Question D1?*

Break-Out Group #4: Methods Development (See Chapter 3)

The 1986 criteria are based on a culture method (EPA Method 1600) for the detection of fecal indicators in ambient waters. The Agency has been considering the use of newer methods, such as qPCR and faster culture-based methods, for inclusion in new or revised criteria. EPA is interested in input on what other methods or tools are available and should be considered for developing criteria/standards that would meet all CWA purposes.

The following set of questions is intended to guide a robust discussion among the experts toward the identification of critical research and science needs in the development of detection methods for the new criteria. It is essential that this group focus discussions on those methods (and pathogens, pathogen indicators or indicators of fecal contamination) that are ready now for day-to-day use in State public health and environmental labs or would be ready for day-to-day use in these labs within the next 3 years.

1. *Are there quantitative methods other than membrane-filtration/Most Probable Number (MF/MPN) methods that measure active organisms that EPA should consider for water quality criteria development?*
2. *Are there data to support other molecular methods for beach microbiological monitoring purposes? Which molecular methods are most fully developed in your view?*
3. *Are there data to support other methods targeting non-microbiological surrogates of beach fecal pollution? Which methods are most fully developed in your view?*
4. *How important is time-to-results in method selection from the perspective of public health protection?*
5. *What further work needs to be done to ensure that the qPCR method or other promising (molecular) methods are considered valid for all CWA purposes?*
6. *What are the pros and cons of the use of molecular methods in each of the CWA applications?*
7. *If some tools are available for certain CWA uses only (e.g., for beach monitoring and notification) how could other methods be "linked" to the qPCR method so that they are scientifically sound and easily implementable? If only qPCR has been validated through epidemiological studies to predict health effects, what other studies could be done to link qPCR to other methods/indicators that may be more appropriate for §304(a) uses?*

8. *Depending on the method used, how could contamination at the beach be linked to all potential fecal sources of contamination? If the source of the contamination was a treated point source, could the method be linked to the necessary source to address the contamination?*
9. *Current culture-dependent methods and qPCR are linked to health risks using epidemiological studies. How would future methods (resulting from rapid technical advances) be calibrated to health risks without new epidemiological studies?*
10. *What applications of water quality criteria would culture methods, including EPA Methods 1600 and 1603, be most suitable for and why?*
11. *What further work needs to be done to ensure that other culture methods are considered for CWA regulatory purposes? If the science is not there, what are the critical path science or research needs to be used in this aspect of criteria development in the near-term?*
12. *What new methods and analytical technologies may be useful to begin to investigate in order for these to potentially be available in the development of "next generation" criteria (i.e., 10 or more years in the future)?*
13. *Can other tools (e.g., models, sanitary surveys) be developed to enhance the insight provided by water quality indicators?*
14. *What characteristics of analytical methods are essential for the methods used in both wastewater and ambient water?*
15. *What are implementation considerations for CWA purposes (1) beach monitoring and notification, (2) development of NPDES permits, (3) assessments to determine use attainment, and (4) development of TMDLs? Are there practical considerations that could preclude, or greatly limit, the usage in routine, regulatory implementation (e.g., field sampling issues, laboratory challenges, staff training, etc.)?*

Break-Out Group #5: Comparing Risks (to Humans) from Different Sources (See Chapter 4)

New or revised criteria should be protective of waterborne organisms that are pathogenic to humans whether the source is human waste or animal waste. The following set of questions is intended to guide a robust discussion among the experts toward the identification of critical research and science needs to better understand the relationship between the risks posed by exposure to human and animal wastes in recreational waters so that this may be considered in the development of new criteria.

1. *Is setting criteria based on a treated human point source such as a publicly (or privately) owned (sewage/wastewater) treatment work (POTW) protective, under-protective or overprotective of other potential sources of human pathogen? Why or why not? Are there data to support this conclusion?*
2. *Based on the "state of the science," what conclusions or assumptions are reasonable to make about risks to humans exposed to human fecal contamination, non-point source contamination from animal sources, and mixed sources (e.g., combined sewer overflows and storm sewer overflows)?*

3. *To what extent is it reasonable to apply risk estimates from POTW-influenced beaches to non-POTW beaches? Do we understand scientifically whether this would lead to overprotection? What science would be important to understanding this?*
4. *Assess whether there is a possibility of overprotection due to a compounding of risks from multiple factors (such as the current definition of GI illness [i.e., no fever]; more sensitive molecular methods; assuming that POTW risks = non-human source risks, etc.)*
5. *How should EPA evaluate risk that may have a low probability of occurrence but a significant risk, if it occurs?*
6. *What are the key data gaps and uncertainties needed to support criteria development in the near term?*

Break-Out Group #6: Acceptable Risk (See Chapter 5)

Population to be Protected: EPA is currently reassessing the extent to which criteria protect swimming populations, including some vulnerable subpopulations (e.g., immunocompromised individuals, elderly, and children) against various types of waterborne diseases (GI and non-GI) caused by pathogens.

The following set of questions is intended to guide a robust discussion among the experts toward the identification of critical research and science needs to better understand what protections new criteria would provide and for what populations/subpopulations.

1. *Is the science there now to understand the degree and extent of protection that nationally recommended criteria for the general population would provide to vulnerable subpopulations (e.g., immunocompromised individuals, elderly, and children)? Is the science there now to understand whether nationally recommended criteria (based on the types of epidemiological studies EPA and others have conducted to date) provide protection against all types of major waterborne diseases? If not, for which subgroups, pathogens, and waterborne illnesses is the science lacking? What types of studies would be needed to answer these types of questions about the degree of public health protection provided by nationally recommended criteria?*
2. *What methodologies or approaches for assessing human health risk or hazard should EPA consider as it develops new criteria? Why?*
3. *What are the pros and cons of using GI illness rates associated with differing levels of fecal contamination as the foundation for developing nationally recommended criteria?*
4. *Is there any scientifically-based reason to establish different "acceptable" risk levels for fresh water versus marine water?*
5. *Is the phrase "acceptable risk" from the (US EPA) 1986 criteria the best terminology or should we consider other terminology (e.g., tolerable or appropriate risk level)?*
6. *What science, if any, would be helpful to EPA in making decisions about what amount and type of human illness from recreation should be considered acceptable?*
7. *What is the level of human health protection provided by the implementation of the 1986 criteria? Is it really no more than 8 to 10 GI illnesses (with fever) per 1,000 in fresh water and 19 GI illnesses (with fever) per 1,000 in marine waters, or, are we really*

protecting people from more than GI illness (with fever)? What science is needed to understand what protection is provided by the implementation of the 1986 criteria?

Protection of Humans from Drinking Water and Fish and Shellfish Consumption: EPA is currently assessing the degree to which recreational criteria can and should be developed to not only protect people from illnesses associated with recreation, but also to protect people from illness caused by drinking contaminated recreational water or consuming fish and shellfish found in contaminated recreational water.

1. *Will criteria that protect swimmers from swimming-related illnesses caused by pathogens also protect people who drink the water or eat fish or shellfish from the same water? Is the science sufficient to support a determination that recreational criteria will also protect drinking water uses and shellfish uses?*
2. *What additional science is needed to ensure that recreational criteria protect people from illnesses associated with recreation and also protect people from illnesses caused by drinking contaminated recreational water or consuming fish and shellfish found in contaminated recreational water?*
3. *Is the science there now to understand and characterize the degree of protectiveness for all these elements?*
4. *If the science is not there, what are the critical path science or research needs to address this?*

<u>Break-Out Group #7: Modeling Applications for Criteria Development and Implementation (See Chapter 6)</u>

Predictive modeling may be useful as a tool to help with the development of site-specific recreational water quality criteria, and the implementation of criteria. Presently, EPA is not considering models in its plans for new or revised criteria in the near-term. However, in recognition that some states and municipalities currently use models effectively in beach notification programs, EPA solicits input from experts regarding the potential use of models as tools to aide implementation of the new or revised criteria, and further requests input on critical research and science needs in this area for future criteria development.

1. *What potential role could estimating techniques (or models) play in criteria development? In the setting of site-specific criteria for recreational waters?*
2. *What potential role could estimating techniques (or models) play in implementing nationally recommended criteria for recreational waters?*
3. *What are advantages and disadvantages of using models, instead of direct measurement (monitoring), in water quality management? And in particular, in management of recreational waters?*
4. *What factors should be considered in integrating modeling with current monitoring regimes, or in changing monitoring regimes to include or support modeling?*
5. *What is the "state of the science" in modeling to support recreational water quality criteria development and implementation?*

6. *What model evaluation procedures are used to insure the quality of predictive models for recreational water quality?*
7. *How does uncertainty in modeling compare to uncertainty in monitoring? How can uncertainty be accurately represented and considered in risk analysis and public health decisions?*
8. *Do differences in the nature of the respective uncertainties inherent in modeling versus monitoring require different means of addressing these uncertainties? For instance, issue an advisory on the basis of modeled results, but clear the advisory only on the basis of sampling.*
8. *What models would be most useful for certain "uses" of criteria (i.e., beach notification, assessment, permitting, TMDLs)? How would modeling be used together with monitoring to cover all "uses" of criteria?*
9. *In models that are currently being used to predict levels of indicator bacteria, how are advisory/closure decisions being made using model results, and how are the results and/or the risk being communicated to the public? Do paradigms currently exist that would be applicable to the communication of modeled information on likely water quality?*
10. *Given the differences between fresh water and marine water environments in terms of physical predictive factors, what are the respective challenges of the two environments relative to developing predictive models? What are the differences in data requirements, likely effectiveness of models, and resources required to develop and implement useful models for the full range of intended purposes?*
11. *What are the critical path research and science needs EPA should pursue to further enhance the capabilities and effectiveness of models in the development/implementation of new or revised criteria? Why?*
12. *What critical path research and science needs EPA should pursue to consider modeling in the development of next generation criteria? Why?*

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APPENDIX B: PARTICIPANT LIST

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APPENDIX C: TRANSLATION OF EPIDEMIOLOGY TO DISEASE BURDEN BY WHO AND EU

In a series of five international expert consultations that took place between 1996 to 2001, the World Health Organization (WHO), together with partner organizations, including the EPA, the Commission of the European Communities, and a group of independent experts, have developed a methodology for expressing the exposure-risk relationship for recreational water. This approach is outlined in detail in Chapter 4 of the WHO's (2003) *Guidelines for Safe Recreational Water Environments. Volume 1 Coastal and Fresh Waters* (see also Kay et al., 2004). The broad framework is summarized below as a basis for burden of disease calculations.

Stated briefly, the approach is based on the following two assumptions:

1. that the statistical distribution of the fecal indicators (i.e., given a sufficiency of samples through a compliance period such as a bathing season) which predict illness in recreational waters is described by a $\log_{10}$-normal probability density function (pdf); and
2. that the pdf for any beach can be combined with the dose-response curve to produce a unique disease burden for a specific location.

Given a fixed dose-response curve, the relative disease burden (or proportion of the exposed population that becomes ill) for any beach, region or jurisdiction can be calculated from the parameters of the pdf, principally its geometric mean (GM) value (i.e., the mean of the $\log_{10}$ transformed bacterial counts) and the standard deviation (SD) of the $\log_{10}$ transformed bacterial counts. The mathematical basis of these calculations is outlined in WHO (2003), while Kay et al. (2004) and Wyer et al. (1999) provide a discussion on the impacts of different GM and SD assumptions.

Figure C-1 illustrates a theoretical pdf for any beach. The cleaner the water, the further to the left the peak of the pdf will be. Figure C-2 provides the dose response curves reported in Kay et al. (1994) that were used in deriving the standards in WHO (2003). Plot C-2a is projection of the dose-response curve beyond the actual data range of >157 (intestinal) enterococci per 100 mL. In fact, the projection of this relationship to exposures above about 150 enterococci would not be justified because the empirical data acquired during the U.K. randomized sea bathing trials was restricted to lower exposures. Figure Plot C-2b assumes that the excess probability of illness does not continue to increase as enterococci exposure increases above the levels experienced in the sea bathing trials. This was chosen as the dose-response curve in the derivation of the 2003 WHO Guidelines as a pragmatic approach. It should be recognized, however, that it may represent an underestimate of the true disease burden if the curve does not, in fact, flatten as suggested in this diagram.

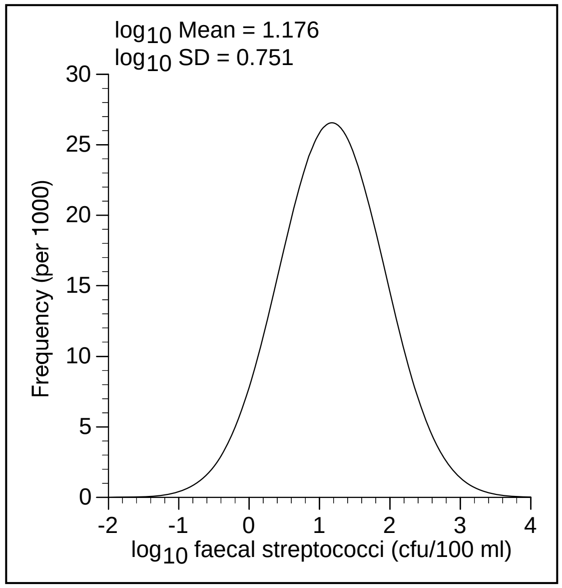

Figure C-1. A Probability Density Function of Fecal Indicator Distributions Measured at a Recreational Water Showing Probability of Exposure (Y Axis) versus Log10 Fecal Streptococci Concentration (later termed enterococci).

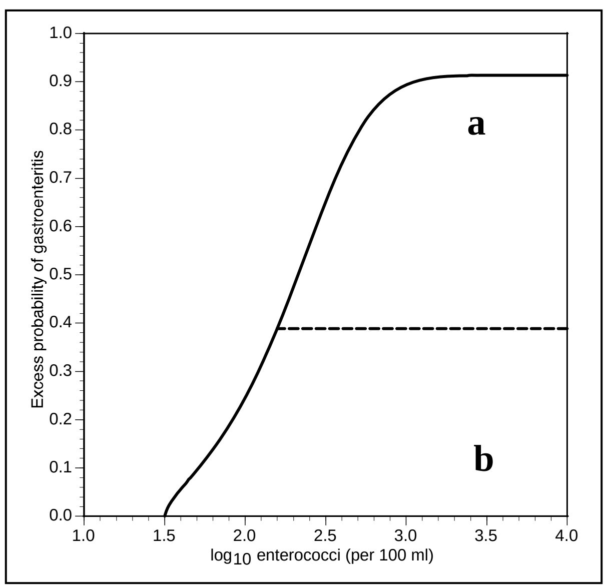

Figure C-2. The Dose-response Relationship Derived From Kay et al. (1994) (a) and the Functional Form Used to Derive the 2003 WHO Guideline Values (b). See Kay et al. (2004) for a more detailed explanation.

Figure C-3 combines the pdf of Figure C-1 with the dose-response curve of Figure C-2 to produce a relative disease burden prediction as a proportion of the exposed population. The mathematical basis of this process is provided in Kay et al. (2004).

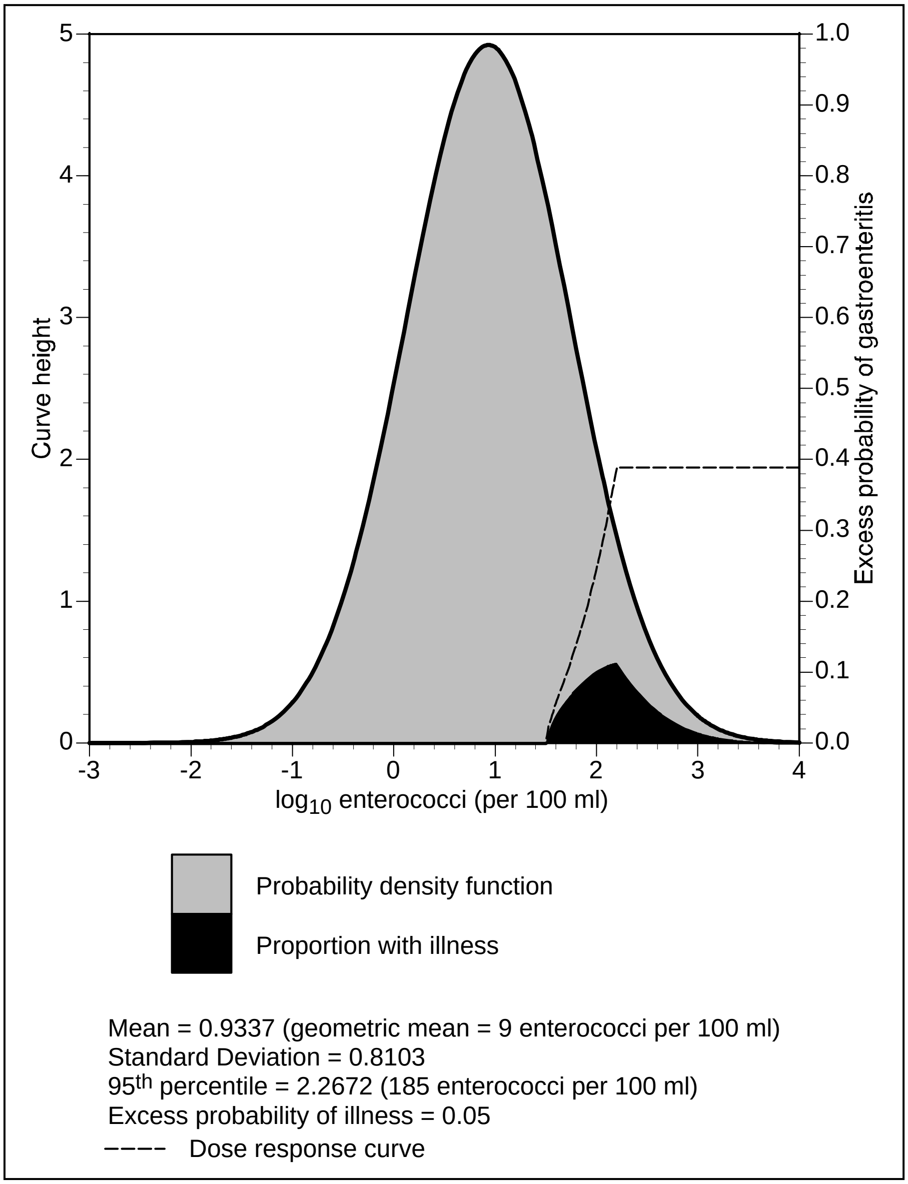

Figure C-3. Combining the Dose-response Curve and the pdf to Produce a Relative Disease Burden Assessment for any Beach or Region.

The computation of gastrointestinal (GI) illness rates in the population is accomplished as follows. The pdf is described by:

$$y = \frac{1}{s\sqrt{2\pi}} e \frac{-(c-m)^2}{2s^2}$$

Where c is the $\log_{10}$ transformed enterococci concentration, y is the normal curve height, and m is the mean enterococci concentration. The associated probability of exposure across a given range of enterococci concentration (i.e., c_a to c_b,) for a given distribution is expressed by:

$$\Phi(c) = \int_{ca}^{cb} y_{dc}$$

which is the area under the normal pdf curve between the limits c_a and c_b. The proportion of bathers with GI illness is then calculated from the area under the curve described by:

$$z = py$$

where p is the probability of GI illness (gastroenteritis) from the dose-response relationship with the upper limit set at 158 enterococci per 100 mL; that is:

$$p = 0.20102\sqrt{(c-31.9)-2.3561}$$

and z is the corresponding proportion of the normal curve height, y. The associated probability of GI illness across the range of enterococci concentrations, c_a to c_b, is then expressed by the following integral:

$$\Phi(c) = \int_{ca}^{cb} z_{dc}$$

For the WHO (2003) Guidelines derivation, the integration of these areas was performed using iterative algorithms as outlined in Khabaza (1965). The algorithm was checked against standard tabulations of the normal pdf curve (Lindley and Miller, 1968) and an accuracy of at least four significant figures was obtained over the specified range of the normal pdf.

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APPENDIX D: SUMMARY OF THE EUROPEAN COMMISSION DIRECTIVE

Revised Bathing Water Directive (EP/CEU, 2006)

The Directive sets out requirements for the following:

(a) the monitoring and classification of bathing water quality;
(b) the management of bathing water quality; and
(c) the provision of information to the public on bathing water quality.

It is meant to apply to identified European Union (EU) bathing waters used by "large numbers" of bathers which must be assessed against the criteria in Table 2 in Chapter 1 of these proceedings (using the previous 3 or 4 years of sampling data) though establishment of a sampling program to acquire data from each bathing water at locations where a bathing water "profile" suggests the greatest risk of pollution and/or the greatest numbers of bathers might be expected (Article 3.3b). Member States must monitor each bathing water in accordance with a monitoring calendar established at the start of the bathing season (Article 3.4). The monitoring calendar can be suspended during "abnormal" conditions and samples taken during "short term pollution" may be disregarded (Article 3.6) provided that Member States comply with the additional provisions outlined below.

Bathing waters are legally required to achieve "sufficient" microbiological status by 2015 (Article 5.3), although the numerical values will be reviewed in 2008 (Article 14).

However, bathing waters classified as "poor" in Table 2 may still remain in compliance with this Directive provided that Member States shall ensure that the following conditions are satisfied (Article 5.4a (i-iv)):

> adequate management measures, including a bathing prohibition or advice against bathing, with a view to preventing bathers' exposure to pollution and identification of the causes and reasons for the failure to achieve "sufficient" quality status is undertaken by Member States; and adequate measures to prevent, reduce or eliminate the causes of pollution; and in accordance with Article 12, alerting the public by a clear and simple warning sign and informing them of the causes of the pollution and measures taken, on the basis of the bathing water profile.

Member States must establish their bathing water profiles by March 24, 2011, which will be reviewed as specified in Annex III of the Revised Bathing Water Directive.

Article 12 further describes information which must be made available to the public at the bathing water and communicated promptly by means of a sign, which includes:

- the current bathing water classification and any bathing prohibition or advice against bathing;

- a general description of the bathing water, in non-technical language, based on the bathing water profile established in accordance with Annex III;
- in the case of bathing waters subject to short-term pollution: notification that the bathing water is subject to short-term pollution;
- an indication of the number of days on which bathing was prohibited or advised against during the preceding bathing season because of such pollution, and a warning whenever such pollution is predicted or present;
- information on the nature and expected duration of abnormal situations during such events;
- whenever bathing is prohibited or advised against, a notice advising the public and giving reasons;
- whenever a permanent bathing prohibition or permanent advice against bathing is introduced, the fact that the area concerned is no longer a bathing water and the reasons for its declassification; and
- an indication of sources of more complete information in accordance with paragraph 2.

In addition, "Member States shall use appropriate media and technologies, including the Internet, to disseminate actively and promptly the information concerning bathing waters referred to in paragraph 1 and also the following information in several languages, when appropriate" (Article 12.1 and 12.2).

Where a bathing water is subject to short-term pollution the public should also be informed on the following (Article 12.4d):

- conditions likely to lead to short-term pollution;
- the likelihood of such pollution and its likely duration; and
- the causes of the pollution and measures taken with a view to preventing bathers' exposure to pollution and to tackle its causes.

Member States are required to disseminate this knowledge using geo-referenced information and signage at bathing waters beginning March 24, 2008.

Member States are free to simply use the numerical standards in Table 2. However, they may take advantage of the opportunity to discount samples collected during short-term pollution events provided they have produced a bathing water profile and have complied with the requirement to provide public information specified in Article 12, which requires real time water quality prediction. No more than 15% of planned samples that are predicted to be of poor quality (i.e., resulting in public advisories) can be discounted in this manner prior to the calculation of the compliance statistics.

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APPENDIX E: INDICATOR TERMINOLOGY

Table E-1. Definitions for Indicator and Index Microorganisms of Public Health Concern.*

Group	Definition
Indicator	A group of organisms that demonstrates the efficacy of a process, such as total heterotrophic bacteria or total coliforms for chlorine disinfection
Fecal indicator	A group of organisms that indicates the presence of fecal contamination, such as the bacterial groups fecal coliforms or *E. coli*; thus, they only infer that pathogens may be present
Index and model organisms	A group/or species indicative of pathogen presence and behavior respectively, such as *E. coli* as an index for *Salmonella* and F-RNA coliphages as models of human enteric virus behavior in the environment
Pathogen indicator	A specific pathogen belonging to a broader group of pathogens which would serve as a surrogate for the presence and/or health risks for that group (e.g., *Cryptosporidium* serving as a surrogate for all parasitic protozoa), or an indicator microorganisms whose presence is correlated to the presence of a broad group of pathogens (e.g., spores of *Clostridium perfringens* serving as a surrogate for human or dog parasitic protozoa)

*See Text Box E-1 for definitions of microbial groups (adapted from Ashbolt et al., 2001).

Text Box E-1. Definitions of Key Fecal Indicator Microorganisms

Coliforms: Gram-negative, non spore-forming, oxidase-negative, rod-shaped facultative anaerobic bacteria that ferment lactose (with β-galactosidase) to acid and gas within 24 to 48 hours at 36±2°C. *Not* specific indicators of fecal pollution.

Fecal coliforms: coliforms that produce acid and gas from lactose at 44.5± 0.2°C within 24±2 hours, also known as thermotolerant coliforms due to their role as fecal indicators.

Escherichia coli (E. coli): thermotolerant coliforms that produce indole from tryptophan, but also defined now as coliforms able to produce β-glucuronidase (although taxonomically up to 10% of environmental *E. coli* may not). Most appropriate group of coliforms to indicate faecal pollution from warm-blooded animals.

Fecal streptococci (FS): Gram-positive, catalase-negative cocci from selective media (e.g., azide dextrose broth or m Enterococcus agar) that grow on bile aesculin agar and at 45°C, belonging to the genera *Enterococcus* and *Streptococcus* possessing the Lancefield group D antigen.

Enterococci: all fecal streptococci that grow at pH 9.6, between 10° and 45°C, and in 6.5% NaCl. Nearly all are members of the genus *Enterococcus*, and also fulfil the following criteria: resistance to 60°C for 30 minutes and ability to reduce 0.1% methylene blue. The enterococci are a subset of fecal streptococci that grow under the conditions outlined above. Alternatively, enterococci can be directly identified as micro-organisms capable of aerobic growth at 44±0.5°C and of hydrolysing 4-methlumbelliferyl-β-D-glucoside (MUD, detecting β-glucosidase activity by blue florescence at 366nm), in the presence of thallium acetate, nalidixic acid, and 2,3,5-triphenyltetrazolium chloride (TTC, which is reduced to the red formazan) in the specified medium (ISO/FDIS 7899-1 1998).

Sulphite-reducing clostridia (SRC): Gram-positive, spore-forming, non-motile, strictly anaerobic rods that reduce sulphite to H_2S.

Clostridium perfringens: as for SRC, but also ferment lactose, sucrose and inositol with the production of gas; produce a stormy clot fermentation with milk; reduce nitrate, hydrolyse gelatine, and produce lecithinase and acid phosphatase. Bonde (1963) suggested that all SRC in receiving waters are not indicators of fecal pollution; thus, *C. perfringens* is the appropriate indicator.

Bacteroidales: a family of strictly anaerobic bacteria present in the guts of warm-blooded animals. The family to which *Bacteroides* belongs.

Bacteriodes: Gram-negative, mainly straight *Bacteroides* species that are: (a) obligately anaerobic, chain saturated, anteiso-methyl, and iso-methyl branched acids, (b) saccharolytic, producing acetate and succinate as the major metabolic end products; (c) contain enzymes of the hexose monophosphate shunt-pentose phosphate pathway; (d) have a DNA-base composition in the range 40-48 mol% GC; (e) membranes contain sphingolipids, and contain a mixture of long-chain fatty acids; (f) possess menaquiones with MK-10 and MK-11 as the major components; and (g) contain *meso*-diaminopimelic acid in their peptidoglycan. This definition restricts the *Bacteroides* to the following ten species: *B. fragilis*, *B. thetaiotaomicron*, *B. vulgatus*, *B. ovatus*, *B. distasonis*, *B. uniformis*, *B. stercoris*, *B. eggerthii*, *B. merdae*, and *B. caccae*, with *B. fragilis* as the type strain. The *Bacteroides*, along with *Prevotella* and *Porphyromonas*, form one major subgroup in the bacterial phylum Cytophaga-Flavobacter-Bacteroides. This phylum diverged quite early in the evolutionary lineage of bacteria, and thus the *Bacteroides*, although Gram-negative organisms, are not closely related to the enteric Gram-negatives such as *Escherichia coli*.

Bacteriodes phages: Those viruses (bacteriophages) that use *Bacteroides* as a host for replication.

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APPENDIX F: SUMMARY OF MEASUREMENTS CURRENTLY PLANNED FOR THE DOHENY AND MALIBU BEACH (CALIFORNIA) EPIDEMIOLOGY STUDY

Table F-1. Summary of Measurements Currently Planned for the Doheny and Malibu Beach (California) Epidemiology Study.

Indicator	Method	Investigator
Traditional		
Enterococci	IDEXX	South Orange County Wastewater Authority (SOCWA)
Enterococci	Membrane-filtration (MF)	SOCWA
Fecal coliforms	MF	SOCWA
E. coli	MF or IDEXX	Southern California Coastal Water Research Project (SCCWRP)
Total coliforms	MF	SOCWA
Rapid Traditional		
Enterococci	Quantitative polymerase chain reaction (qPCR)	Noble
Enterococci	qPCR	Stewart
Enterococci	PCR-Luminex	Stewart
Enterococci	Transcription-mediated amplification/nucleic acid sequence-based amplification (TMA/NASBA)	Moore
Enterococci	Immunomagnetic separation (IMS)	Bushon
E. coli	qPCR	Noble
E. coli	IMS	Bushon
E. coli	IMS	Jay
Marker Genes		
Entero-Raptor, *Esp* gene	qPCR-Raptor	Harwood/Lim
Enterococci *Esp* gene	qPCR	Scott
E. coli virulence genes	qPCR	Sadowsky
Bacteroides human marker	qPCR	Field
Bacteroides human marker	qPCR	Wuertz
Phage		
Phage	Culture	Stewart
Phage	Culture	Sobsey
Rapid phage	Antibody	Sobsey
Human Virus		
Adenovirus	qPCR	Sobsey

Indicator	Method	Investigator
Enterovirus	qPCR	Stewart
Hepatitis A virus	qPCR	Fuhrman
Norovirus	qPCR	Stewart
Norovirus	qPCR	Sobsey
Polyomavirus	qPCR	Harwood
Community Profiling		
Bacteroides thetaiotamicron	Sequencing	Moorthy
Heliobacter pylori	Sequencing	Moorthy
Campylobacter jejuni	Sequencing	Moorthy
Clostridium perfringens	Sequencing	Moorthy
Salmonella enteritica serovar Typhimurium	Sequencing	Moorthy
Shigella dysenteriae	Sequencing	Moorthy
Shigella flexneri	Sequencing	Moorthy
Shigella boydii	Sequencing	Moorthy
Bacterial Markers		
Bacteroides thetaiotamicron	qPCR	Noble
Bacteroides thetaiotamicron	PCR	Leddy
Multiple methanogens	PCR	Ufnar
Methanobrevibacter smithii	PCR-Luminex	Stewart
Methanobrevibacter smithii	qPCR	Stewart
Legionella spp.	qPCR	Gast

APPENDIX G: DEVELOPMENT OF DETERMINISTIC MODELS

The discussion of the modeling workgroup members included the present and future use of statistically-based models. This relates to the fact that they are currently being used to supplement monitoring information and can be implemented in a resource-effective manner in existing beach advisory programs. In general, deterministic models have not been included in the main part of this discussion (see Chapter 6) because it was the common opinion of the workgroup members of the modeling workgroup that their application represents a longer-range measure that might be considered in the context of research and development beyond the 2 to 3 year (near-term) window envisioned by the current criteria development effort; however, there were differences of opinion on the importance of this relative to development of new or revised recreational water quality criteria.

Although not discussed in detail at this workshop, deterministic process-based models represent an entire range of additional modeling tools that could be used to inform water quality criteria development and implementation over the range of criteria framework options that have been discussed during the course of this conference. Applications of such models to beach environments are discussed in the EPA report *Review of Potential Modeling Tools and Approaches to Support the BEACH Program*, (US EPA, 1999). They range from those that are simply based on precipitation to newer models that consider other factors such as sediment resuspension, hydrodynamics, microbial growth and decay, and non-point source basin scale inputs. For example, a process-based deterministic model has been recently used to predict fecal indicator concentrations in coastal reaches of southern Lake Michigan (Liu et al., 2006) and Huntington Beach, California (Boehm et al., 2005; Grant et al., 2005). Deterministic models also are being used in the development of total maximum daily loads (TMDLs) for pathogens and in evaluations of non-point and sources of biological contaminants in watersheds.

In this appendix, deterministic models for evaluating pathogens in watersheds are briefly discussed. TMDLs often have to consider non-point sources from watersheds. This discussion is not intended to be comprehensive; rather, it is designed to illustrate the range of tools available to this area of consideration.

Commonly used TMDL models allow users to discretize the watershed spatially and bacteria loads spatially and temporally, although this capacity is limited. As discussed in ASABE (2006),

> the models are also limited in their ability to simulate bacterial life cycles and bacteria concentrations. Even with their limitations, these models are useful when developing TMDLs if for no other reason than their use provides educational opportunities for both stakeholders and modelers throughout the TMDL process. The load duration method of developing TMDLs provides a good representation of overall water quality and needed water quality improvement, but intra-watershed bacteria contributions must be determined through supplemental sampling or through subsequent hydrologic and water quality modeling. Identified research needs include improved bacteria source characterization procedures and supporting data, and specific modeling advances.

New models are now becoming available for evaluating non-point sources of pathogens derived from watersheds and catchments.

- The L-THIA model (http://www.ecn.purdue.edu/run-off/lthianew/) combined with GIS-referenced inputs from Digital Watershed are being used as tools to evaluate runoff of fecal coliform and fecal streptococcus (enterococci) in watersheds. Digital Watershed (http://www.iwr.msu.edu/dw/) allows the user to view the watershed tributary to any given point in the continental United States, on an 8-digit or (in parts of the Midwest) a 12-digit HUC code level of detail. L-THIA calculates the surface and groundwater impacts of current land use, land use changes and potential best management practices (BMPs) for quality and quantity for the bacteria. L-THIA will be directly linked to STORET water quality and SSURGO soils databases within a year. In the Midwest it is also available as a web-based GIS tool at the 12-digit HUC code level through the watershed delineation tool at http://pasture.ecn.purdue.edu/~watergen/.
- The SPARROW model (SPAtially Referenced Regression on Watershed attributes) (http://water.usgs.gov/nawqa/sparrow/index.html) is being used to investigate the sources and fate of fecal contamination in streams and to assess the effects of the spatial resolution of the stream network and landscape data on model parameters and predictions. SPARROW has been used to evaluate the following indicators: fecal coliforms, *E. coli*, *C. perfringens*, somatic coliphage, F+ RNA phage, and the bacterial pathogen *Campylobacter*. The explanatory data for the SPARROW models include land use and other data that describe the climatic, hydrologic, and physical conditions of the catchments. The models also reveal the effects of climate, soils, and instream processes on the transport of fecal contaminants.
- LSPC is the Loading Simulation Program in C++, a watershed modeling system that includes streamlined Hydrologic Simulation Program Fortran (HSPF) algorithms for simulating hydrology, sediment, and general water quality on land as well as a simplified stream transport model. LSPC has been used in Alabama for developing pathogen TMDLs (see http://www.epa.gov/athens/wwqtsc/Toolbox-overview.pdf and http://www.epa.gov/ATHENS/wwqtsc/html/lspc.html).

In addition to these models, a 1999 EPA report describes other potential models that can be used for evaluating non-point sources of biological contaminants from catchments. These include, for example, HSPF. HSPF is one of the models that is included in the BASINS3 watershed model system that is maintained by EPA (http://www.epa.gov/waterscience/BASINS/).

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Contents

List of acronyms and abbreviations

AFRI	acute febrile respiratory illness
AIDS	acquired immune deficiency syndrome
ASP	amnesic shellfish poisoning
BCC	basal cell carcinoma
CBO	community-based organization
CDC	Centers for Disease Control and Prevention (USA)
cfu	colony-forming unit
COGP	Code of Good Practice
CPR	cardiopulmonary resuscitation
DALY	disability adjusted life year
DSP	diarrhetic shellfish poisoning
EAP	emergency action plan or procedure
EC	European Commission
GAE	granulomatous amoebic encephalitis
GI	gastrointestinal
HACCP	hazard analysis and critical control point
HAV	hepatitis A virus
HEV	hepatitis E virus
HIA	health impact assessment
HIV	human immunodeficiency virus
IARC	International Agency for Research on Cancer
IBM	integrated basin management
ICAM	integrated coastal area management
ID_{50}	dose of microorganisms required to infect 50% of individuals exposed
ILS	International Life Saving Federation
i.p.	intraperitoneal
LOAEL	lowest-observed-adverse-effect level
MM	malignant melanoma
MOE	Ministry of Environment
MOH	Ministry of Health
MOT	Ministry of Tourism
NGO	nongovernmental organization
NMSC	non-melanoma skin cancer
NOAEL	no-observed-adverse-effect level
NSP	neurotoxic shellfish poisoning

PAM	primary amoebic meningoencephalitis
PDF	probability density function
PFD	personal flotation device
pfu	plaque-forming unit
PSP	paralytic shellfish poisoning
QA	quality assurance
QMRA	quantitative microbial risk assessment
SCC	squamous cell carcinoma
SLRA	screening-level risk assessment
SOP	standard operating procedure
SPF	sun protection factor
TCBS	thiosulfate–citrate–bile salts–sucrose
TDI	tolerable daily intake
USLA	United States Lifesaving Association
UV	ultraviolet
UVR	ultraviolet radiation
WHO	World Health Organization
WTO	World Tourism Organization

Preface

The World Health Organization (WHO) has been concerned with health aspects of the management of water resources for many years and publishes various documents concerning the safety of the water environment and its importance for health. These include a number of normative "guidelines" documents, such as the *Guidelines for Drinking-water Quality* and the *Guidelines for Safe Use of Wastewater and Excreta in Agriculture and Aquaculture*. Documents of this type are intended to provide a basis for standard setting. They represent a consensus view among experts on the risk to health represented by various media and activities and on the effectiveness of control measures in protecting health. They are based on critical review of the available evidence. Wherever possible and appropriate, such guidelines documents also describe the principal characteristics of the monitoring and assessment of the safety of the medium under consideration as well as the principal factors affecting decisions to be made in developing strategies for the control of the health hazards concerned.

The *Guidelines for Safe Recreational Water Environments* are published in two volumes:

- *Volume 1: Coastal and Fresh Waters* provides a review and assessment of the health hazards encountered during recreational use of coastal and freshwater environments. It includes the derivation of guideline values and explains the basis for the decision to derive or not to derive them. It addresses a wide range of types of hazard, including hazards leading to drowning and injury, water quality, exposure to heat, cold and sunlight, and dangerous aquatic organisms; and provides background information on the different types of recreational water activity (swimming, surfing, etc.) to enable informed readers to interpret the Guidelines in light of local and regional circumstances. With regard to water quality, separate chapters address faecal pollution, free-living microorganisms, freshwater algae, marine algae and chemical aspects. It describes prevention and management options for responding to identified hazards.

- *Volume 2: Swimming Pools, Spas and Similar Recreational Water Environments* provides a review and assessment of the health hazards associated with recreational waters of this type; their monitoring and assessment; and activities available for their control through education of users, good design and construction, and good operation and management. It includes the derivation of guidelines including guideline values and explains the basis for the decision to derive or not to derive them. It addresses a wide range of types of hazard, including water quality, hazards leading to drowning and injury, contamination of associated facilities and air quality.

In addition to the above volumes of the *Guidelines for Safe Recreational Water Environments*, a practical guide entitled *Monitoring Bathing Waters*,[1] has been produced. It describes the principal characteristics of and approaches to the monitoring and assessment of coastal and freshwater recreational water environments. It emphasizes the need to utilize information of diverse types and from diverse sources in order to develop a valid assessment; and the need to establish effective links between the information generated and interventions to control risk in both the short and long term. It includes comprehensive practical guidance for the design, planning and implementation of monitoring programmes and assessments; and a Code of Good Practice for the monitoring and assessment of recreational water environments, to assist countries in developing such codes for national use and to promote international harmonization. Material relating to toxic cyanobacteria, including that in chapters 7 and 8 is based upon *Toxic Cyanobacteria in Water*,[2] which was prepared by an international group of experts.

The development of WHO activity on 'recreational' or 'bathing' water can be traced back to two expert consultations in the 1970s.[3] These meetings highlighted the breadth of possible hazards associated with recreational water use and noted that prospective volunteer studies offered the "best hope of progress" in terms of establishing links between water quality and bather health. They also suggested the grading of beaches according to bands of indicator counts and the use of sanitary assessments for beaches. These initial meetings were followed by a series of expert consultations. The meeting in Valetta, Malta held during 1989, reviewed the status of microbial guidelines for bathing waters and examined the potential protocols for epidemiological investigations. The importance of protocol design was clear at the Valetta meeting, and two principal approaches were reviewed—namely, the prospective case–control and the randomized trial. Two years later in Athens, Greece the early results of epidemiological investigations that employed both protocols were reviewed. It was decided at this meeting that both approaches were appropriate and could yield useful data for Guidelines derivation. The results of a series of major epidemiological studies in the United Kingdom were presented and critically reviewed at a meeting held in Athens, Greece in 1993.

The preparation of the *Guidelines for Safe Recreational Water Environments* Volume 1 covered a period of almost a decade and involved the participation of numerous institutions, more than 130 experts from 33 countries worldwide, and further reviewers and meetings. The work of the individuals concerned (see Acknowledgements) was central to the completion of the work and is much appreciated.

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[1] Edited by J. Bartram and G. Rees, published in 2000 by E & FN Spon on behalf of WHO.

[2] Edited by I. Chorus and J. Bartram, published in 1999 by E & FN Spon on behalf of WHO

[3] Meetings: Ostend, 1972; Bilthoven, 1974; Valetta 1989; Athens 1991; Athens 1993; Bad Elster 1996; Jersey 1997; Farnham 1998; Annapolis 1999; Farnham 2001.

In 1994, following discussions between the WHO Regional Office for Europe and WHO Headquarters, it was agreed to initiate development of guidelines concerning recreational use of the water environment, examining all possible health outcomes from both natural waters and swimming pools, including those related to water quality. This was undertaken as a collaborative initiative between WHO Headquarters and the WHO European Centre for Environment and Health, Rome, Italy. A comprehensive review of the scientific literature on sewage pollution of recreational water and health, eventually published as Prüss (1998), provided the focus for an expert consultation in Bad Elster in 1996. This meeting concluded that the epidemiological basis had been laid for evidence-based normative guidelines on faecal pollution of recreational water. The consultation also received information on new research findings quantifying the impacts of non-sewage sources of faecal bacteria on recreational water compliance with microbial water quality criteria. The implications of these findings were that many bathing waters might fail current water quality norms because of the influence of diffuse source pollution, which would not be reduced by sewage treatment alone.

At a further expert consultation hosted and co-sponsored by the States of Jersey in 1997 drafts of all chapters of the two volumes of Guidelines were reviewed, these were revised and further reviewed at a meeting the following year in Farnham, UK 1998. The Draft Guidelines for coastal and fresh waters were then submitted for international expert appraisal and received intensive review.

In 1999, an expert consultation co-sponsored by the US EPA and held in Annapolis, USA, resulted in the "Annapolis Protocol" (WHO, 1999), which suggested a new approach towards evaluation and regulation of faecal pollution of bathing waters. The Annapolis Protocol outlines a combined sanitary inspection and microbial measurement approach that is used to classify recreational waters. In addition, the protocol suggests the use of relevant information to facilitate real-time public health protection. Thus, the principal focus of regulation is expanded from retrospective numerical compliance assessment to include real-time management and public health protection. A further expert consultation to take account of the Annapolis protocol and other newly available information in the draft guidelines was held in Farnham, UK, in 2001. The Guidelines were finalized through a series of chapter-by-chapter conference calls with selected experts, in November 2002.

During the development of the Guidelines, careful consideration was given to previous assessments, in particular the work of the Mediterranean Action Plan, the Black Sea Environmental Programme, the activities undertaken by and for the European Commission, the activities undertaken by the US Environmental Protection Agency, including its "BEACH" programme and others.

In light of the importance of the subject area for health and the degree of attention it receives from the political and scientific communities and the general public, it is envisaged that new information will become available rapidly during future years. WHO would be pleased to learn of major related developments and will endeavour to ensure the continuing validity of the Guidelines through issuing addenda or further editions as appropriate.

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Executive summary

This volume of the *Guidelines for Safe Recreational Water Environments* describes the present state of knowledge regarding the impact of recreational use of coastal and freshwater environments upon the health of users—specifically drowning and injury, exposure to cold, heat and sunlight, water quality (especially exposure to water contaminated by sewage, but also exposure to free-living pathogenic microorganisms in recreational water), contamination of beach sand, exposure to algae and their products, exposure to chemical and physical agents, and dangerous aquatic organisms. As well, control and monitoring of the hazards associated with these environments are discussed.

The primary aim of the Guidelines is the protection of public health. The purpose of the Guidelines is not to deter the use of recreational water environments but instead to ensure that they are operated as safely as possible in order that the largest possible population gets the maximum possible benefit. The adverse impacts of recreational use of coastal and freshwater environments upon the health of users must be weighed against the enormous benefits to health and well-being—rest, relaxation and exercise—associated with the use of these environments.

The Guidelines are intended to be used as the basis for the development of international and national approaches (including standards and regulations) to controlling the health risks from hazards that may be encountered in recreational water environments, as well as providing a framework for local decision-making. The Guidelines may also be used as reference material for industries and operators preparing development projects in recreational water areas, as a checklist for understanding and assessing potential health impacts of recreational projects, and in the conduct of environmental impact and environmental health impact assessments in particular.

The information provided is generally applicable to any coastal or freshwater area where recreational water use occurs. The preferred approaches adopted by national or local authorities towards implementation of the Guidelines, including guideline values, may vary depending on social, cultural, environmental and economic characteristics, as well as knowledge of routes of exposure, the nature and severity of hazards, and the effectiveness of control measures.

A guideline can be:

- a level of management;
- a concentration of a constituent that does not represent a significant risk to the health of members of significant user groups;

- a condition under which such exposures are unlikely to occur; or
- a combination of the last two.

When a guideline is not achieved, this should be a signal to investigate the cause of the failure and identify the likelihood of future failure, to liaise with the authority responsible for public health to determine whether immediate action should be taken to reduce exposure to the hazard, and to determine whether measures should be put in place to prevent or reduce exposure under similar conditions in the future.

Drowning and injury prevention

Drowning, which has been defined as death arising from impairment of respiratory function as a result of immersion in liquid, is a major cause of death worldwide, particularly for male children. Near drowning is also a serious problem as it may have life-long effects. The recovery rate from near drowning may be lower among young children than among teenagers and adults. Studies show that the prognosis for survival depends more on the effectiveness of the initial rescue and resuscitation than on the quality of subsequent hospital care.

Drowning may be associated with swimming as well as with recreational water uses involving minimal water contact, such as recreational use of watercraft (yachts, boats, canoes) and fishing. Alcohol consumption is one of the most frequently reported contributory factors associated with drownings for adults, whereas lapses in parental supervision are most frequently cited for children. In cold weather, immersion cooling may be a significant contributory factor.

Of sports-related spinal cord injuries, the majority appear to be associated with diving. Injuries in diving incidents are almost exclusively located in the cervical vertebrae, resulting in quadriplegia or paraplegia. Data suggest that body surfing and striking the bottom is the most common cause of spinal injury. Alcohol consumption may contribute significantly to the frequency of injury. Other injuries associated with recreational water use activities include brain and head injuries, fractures, dislocations and other minor impact injuries, and cuts, lesions and punctures.

Prevention is the best way to reduce the incidence of injury and death related to the aquatic environment, and the majority of injuries can be prevented by appropriate measures at a local level. Physical hazards should first be removed or reduced if possible, or measures should be taken to prevent or reduce human exposure. Physical hazards that cannot be completely dealt with in this way should be the subject of additional preventive or remedial measures. These include drowning prevention programmes, public information and warnings (such as signs, flags and general education and awareness raising), the provision of effective lifeguard supervision and rescue services, and the establishment of different recreation zones for different recreational activities using lines, buoys and markers.

Monitoring of a site for existing and new hazards should be undertaken on a regular basis. The frequency and timing of inspections will vary with the location.

Sun, heat and cold

The recreational use of water environments sometimes leads to exposure of individuals to extreme solar radiation and to extreme conditions of heat or cold.

Ultraviolet radiation (UVR) from sunlight can be divided into three bands: UVA, UVB and UVC. As the ozone layer becomes depleted, the protective filter provided by the atmosphere is progressively reduced, and human beings are exposed to higher UV levels, in particular higher UVB levels.

Overexposure to UVR may result in acute and chronic damage to the skin, the eyes and the immune system. The most noticeable acute effect of excessive UV exposure is erythema, the familiar inflammation of the skin commonly termed sunburn. Photokeratitis and photoconjunctivitis are other acute effects of UV exposure. Chronic effects include two major public health problems: skin cancers (both non-melanoma skin cancers and malignant melanoma) and cataracts. Chronic exposure to UVR also causes a number of degenerative changes in the skin (e.g., freckles) and accelerates skin aging. There is also increasing evidence for an immunosuppressive effect of both acute high-dose and chronic low-dose UV exposure on the human immune system.

Not all effects of UV radiation are adverse. The best known beneficial effect is the stimulation of the production of vitamin D in the skin. UVR from artificial sources is also used to treat several diseases and dermatological conditions, including rickets, psoriasis, eczema and jaundice.

Simple protective measures are available and should be adopted to avoid adverse health effects on the skin, eyes and immune system. These include minimizing the amount of time spent in the sun, including complete avoidance of midday sun exposure; seeking shade; and wearing loose-fitting and tightly woven clothing, a broad-brimmed hat and wrap-around sunglasses. Furthermore, a broad-spectrum sunscreen with sun protection factor of 15 or more should be applied liberally on all areas of the body not covered by clothing and should be reapplied often. Sun protection programmes to raise awareness and achieve changes in lifestyle are urgently needed to slow down and eventually reverse the trend towards more skin cancers. The global solar UV index is an important vehicle to raise public awareness of UVR and the risks of excessive UV exposure and to alert people to the need to adopt protective measures.

Exposure to cold water may cause considerable problems for users of recreational waters. The immediate effect of sudden immersion in cold water can be a debilitating reflex response called cold shock, which includes life-threatening respiratory and cardiovascular effects and may lead to drowning. Sudden immersion in cold water often results in impaired swimming ability, which is believed to be responsible for the majority of sudden cold-water immersion deaths. Safety precautions include wearing suitable protective garments when swimming in cold water and using a life-jacket when boating to keep airways clear of water even when unconscious.

In a hot environment, people can suffer serious physical ailments, such as heat cramps, heat exhaustion and heat stroke. The very young, the elderly, patients using drugs that interfere with temperature regulation, people suffering from pre-existing chronic diseases and frequent consumers of alcohol appear to be particularly susceptible. Avoidance measures include consumption of non-alcoholic, non-caffeinated beverages, replacement of salt lost through sweating and retreat to shaded areas. Disorders due to heat occur most frequently when there are rapid changes in thermal conditions, such as during heat waves.

Faecal pollution and water quality

The most frequent adverse health outcome associated with exposure to faecally contaminated recreational water is enteric illness. A cause–effect relationship between faecal or bather-derived pollution and acute febrile respiratory illness (AFRI), which is a more severe health outcome than gastroenteritis, has also been shown.

There is consistency in the overall body of evidence concerning health effects from faecally polluted recreational waters, and a series of randomized controlled trials performed in the United Kingdom form the key studies for derivation of guideline values for the microbiological quality of recreational waters. For marine waters, only intestinal enterococci (faecal streptococci) showed a dose–response relationship for both gastrointestinal illness and AFRI. The guideline 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.

There is inadequate evidence with which to directly derive a water quality guideline value for fresh water. Application of the guideline values derived for seawaters to fresh waters would be likely to result in a lower illness rate in freshwater swimmers, providing a conservative guideline in the absence of suitable epidemiological data for fresh waters. Studies under way may provide a more adequate basis on which to develop freshwater guideline values.

The guideline values 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 that are not associated with recreational water.

The initial classification of a recreational water environment is based upon the combination of evidence for the degree of influence of (human) faecal material (by sanitary inspection of beach and water catchment) alongside counts of suitable faecal index bacteria (a microbial quality assessment). Information to be collected during sanitary inspections should cover at least the three most important sources of human faecal contamination of recreational water environments for public health purposes: sewage; riverine discharges (where the river is a receiving water for sewage discharges and either is used directly for recreation or discharges near a coastal or lake area used for recreation); and bather contamination, including excreta. Where human inputs are minimal, investigation of animal faecal inputs should be explored.

In the microbial water quality assessment, the sampling programme should be representative of the range of conditions in the recreational water environment while it is being used. An important issue is that of collecting sufficient numbers of samples so as to make an appropriate estimation of the likely densities to which recreational water users are exposed. The precision of the estimate of the 95th percentile is higher when sample numbers are increased. The number of results available can be increased significantly by pooling data from multiple years, unless there is reason to believe that local (pollution) conditions have changed. For practical purposes, data on at least 100 samples from a 5-year period and a rolling 5-year data set can be used for microbial water quality assessment purposes.

The outputs from the sanitary inspection and the microbial water quality assessment can be combined to give a five-level classification for recreational water environments—very good, good, fair, poor and very poor. Following initial classification, it is proposed that all categories of recreational water environment would be subject to an annual sanitary inspection (to determine whether pollution sources have changed) and continued water quality monitoring.

Another component of the assessment of a recreational water environment is the possible "upgrading" of a recreational water environment if a significant change in management reduces human exposure to microbial risk.

Follow-up analyses are recommended when the intestinal enterococci counts are high but the sanitary inspection suggests low sanitary impact, or vice versa. A primary role of the follow-up is to help identify the source of the faecal pollution, thereby assisting in the assessment and management of faecal contamination in recreational water environments.

In certain circumstances, there may be a risk of transmission of pathogens associated with more severe health effects (such as infectious hepatitis or typhoid fever) through recreational water use. Public health authorities should be alert to such hazards where exposure may occur and should take appropriate action to protect public health.

Population groups that may be at higher risk of disease include the young, the elderly and the immunocompromised, as well as visiting populations susceptible to locally endemic disease. If such groups are significant water users, then this should be taken into account in risk assessment and management.

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

Free-living microorganisms

In addition to microorganisms introduced to recreational waters through human or animal faecal contamination, a number of pathogenic microorganisms are free-living in such areas or, once introduced, are capable of colonizing the environment.

Vibrio species are natural inhabitants of marine aquatic environments in both temperate and tropical regions. The occurrence of vibrios does not correlate with the occurrence of the traditionally used bacterial faecal index organisms, except perhaps in waters receiving human wastes from disease outbreaks (mainly cholera). Due to the ubiquitous nature of *Vibrio* species in the aquatic environment, their presence in bathing water cannot be controlled by water quality control measures such as wastewater treatment and disinfection. Human carriers and shedding appear to be of only limited importance in the epidemiology of *Vibrio* infections associated with recreational water use. For *V. cholerae*, 10^6 organisms or more are typically needed to cause cholera, so that it is unlikely that persons bathing or involved in other recreational water activities would ingest vibrios in numbers high enough to cause gastrointestinal disease. However, the risk of extraintestinal infections associated with human pathogenic *Vibrio* species, especially wound and ear infections, during recreational activities in water is of health importance, although the infectious doses for such infections are unknown.

Aeromonas spp. are considered autochthonous inhabitants of aquatic environments and are ubiquitous in surface fresh and marine waters, with high numbers occurring during the warmer months of the year. Clinical isolation of these microbes presents the same seasonal distribution. Numbers may be high in both polluted and unpolluted habitats with densities ranging from <1 to 1000 cells per ml. Sewage can also contain elevated numbers (10^6–10^8 cells per ml) of aeromonads. *Aeromonas* has been found to have a role in a number of human illnesses including gastroenteritis. Cases of wound infections in healthy people associated with recreational water have been described, as have cases of pneumonia following aspiration of contaminated recreational water.

Free-living amoebae are unicellular protozoa common to most soil and aquatic environments. Of the many hundreds of species of free-living amoebae, only members of the genus *Acanthamoeba*, *Naegleria fowleri* and *Balamuthia mandrillaris* are known to infect humans, often with fatal consequences. *Acanthamoeba* have been isolated from natural and artificial waters. Certain species are pathogenic to humans and cause two clinically distinct diseases affecting the central nervous system: granulomatous amoebic encephalitis (GAE) and inflammation of the cornea (keratitis). *Naegleria fowleri*, which is found in thermal freshwater habitats worldwide, causes primary amoebic meningoencephalitis (PAM) in humans. PAM is usually fatal, with death occurring in 3–10 days after exposure. Infection usually results from swimming in contaminated water, although the infectious dose for humans is not known. *B. mandrillaris* encephalitis is largely a disease of the immunocompromised host, and certain cases of GAE attributed to *Acanthamoeba* have in fact been shown to have been caused by *B. mandrillaris*.

Leptospires are excreted in the urine of infected animals, which can then contaminate soil, mud, groundwater, streams and rivers. Humans become infected either directly through contact with infected urine or indirectly via contaminated fresh water or soil. Virulent leptospires gain entry to the body through cuts and abrasions of the skin and through the mucosal surfaces of the mouth, nose and conjunctiva. In cases due to exposure to recreational water, the incubation period seems to vary between 2 and 30 days, but generally is between 7 and 14 days. The clinical manifestations of leptospirosis vary considerably in form and intensity, ranging from a mild flu-like illness to a severe and potentially fatal form of the disease, characterized by liver and kidney failure.

Evidence suggests that although infection with free-living microorganisms or pathogenic leptospires via recreational water use may be life-threatening, the incidence of such infection is very low and, in many cases, is limited to specific areas. As such, no specific guideline values have been recommended, although authorities should be aware of the potential hazards posed by these organisms and act accordingly. Assessment of the likely hazard (e.g., the likelihood of thermal warming of fresh waters) and education of water users and health professionals are important control measures.

Microbial aspects of beach sand quality

Bacteria, fungi, parasites and viruses have all been isolated from beach sand. A number of them are potential pathogens. Factors promoting the survival and dispersion of pathogens include the nature of the beach, tidal phenomena, the presence of sewage outlets, the season, the presence of animals and the number of swimmers. Transmission may occur through direct person-to-person contact or by other means, although no route of transmission has been positively demonstrated.

Concern has been expressed that beach sand or similar materials may act as reservoirs or vectors of infection. However, the capacity of microorganisms that have been isolated from beach sand to infect bathers and beach users remains undemonstrated, and the real extent of their threat to public health is unknown. There is therefore no evidence to support establishment of a guideline value for index organisms or pathogenic microorganisms on beach sand.

The principal microbial risk to human health encountered upon beaches and similar areas is that arising from contact with animal excreta, particularly from dogs. Regulations that restrict access seasonally on frequently used beaches or place an obligation upon the owner to remove animal excreta, increased public awareness and beach cleaning are preventive management actions.

Algae and cyanobacteria in coastal and estuarine waters

Several human diseases have been reported in association with many toxic species of dinoflagellates, diatoms, nanoflagellates and cyanobacteria (blue-green algae) that occur in the marine environment. The toxicity of these algae to humans is due to the presence of algal toxins. Marine algal toxins become a problem primarily because they concentrate in shellfish and fish that are subsequently eaten by humans, causing shellfish poisoning (not dealt with in this volume).

Marine cyanobacterial dermatitis ("swimmers' itch" or "seaweed dermatitis") is a severe contact dermatitis that may occur after swimming in seas containing blooms of certain species of marine cyanobacteria. The symptoms are itching and burning within a few minutes to a few hours after swimming in the sea where the cyanobacteria are suspended. Some toxic components, such as aplysiatoxin, debromoaplysiatoxin and lyngbyatoxin A, have been isolated from marine cyanobacteria. These toxins are highly inflammatory and are potent skin tumour promoting compounds.

Nodularia spumigena was the first cyanobacterium recognized to cause animal death. The toxin produced by *N. spumigena*, called nodularin, acts as a hepatotoxin, in that it induces massive haemorrhages in the liver of mammals and causes disruption of the liver structure. To date, there have been no reports of human poisoning by *N. spumigena*, but humans may be as susceptible to the toxins as other mammals. Therefore, it is possible that small children may accidentally ingest toxic material in an amount that may have serious consequences.

Inhalation of a sea spray aerosol containing fragments of marine dinoflagellate cells and/or toxins (brevetoxins) released into the surf by lysed algae can be harmful to humans. The signs and symptoms are severe irritation of conjunctivae and mucous membranes (particularly of the nose) followed by persistent coughing and sneezing and tingling of the lips.

Available data indicate that the risk for human health associated with the occurrence of marine toxic algae or cyanobacteria during recreational activities is limited to a few species and geographical areas. As a result, it is inappropriate to recommend specific guideline values.

Within areas subject to the occurrence of marine toxic algae or cyanobacteria, it is important to carry out adequate monitoring activities and surveillance programmes. In affected areas, it is appropriate to provide health information to general practitioners and the general public, in particular recreational water users. Precautionary measures include avoiding areas with visible algal concentrations and/or algal scums in the sea as well as on the shore, avoiding sitting downwind of any algal material drying on the shore and showering to remove any algal material.

Algae and cyanobacteria in fresh water

Many species of freshwater algae may proliferate quite intensively in eutrophic (i.e., nutrient-rich) waters. However, they do not form dense surface scums or "blooms," as do some cyanobacteria. Toxins they may contain therefore are not accumulated to potentially hazardous concentrations. For this reason, most adverse health impacts from recreational use of fresh waters have been associated with cyanobacteria rather than with freshwater algae.

Progress in analytical chemistry has enabled the isolation and structural identification from toxic cyanobacteria of three neurotoxins (anatoxin-a, anatoxin-a(s) and saxitoxins), one general cytotoxin, which inhibits protein synthesis (cylindrospermopsin), and a group of toxins termed microcystins (or nodularins, found in brackish waters), which inhibit protein phosphatases. Most of them have been found in a wide array of genera, and some species contain more than one toxin.

Allergic or irritative dermal reactions of varying severity have been reported from a number of freshwater cyanobacterial genera (*Anabaena*, *Aphanizomenon*, *Nodularia*, *Oscillatoria*, *Gloeotrichia*) after recreational exposure. Bathing suits and particularly wet suits tend to aggravate such effects by accumulating cyanobacterial material and enhancing disruption of cells and liberation of cell content. It is probable that these symptoms are not due to recognized cyanotoxins but rather to currently largely unidentified substances.

In contrast to dermal contact, uptake of cyanobacteria though ingestion or aspiration involves a risk of intoxication by cyanotoxins. Most documented cases of human injury through cyanotoxins involved exposure through drinking-water, and they demonstrate that humans have become ill—in some cases seriously—through ingestion or aspiration of toxic cyanobacteria. Symptoms reported include abdominal pain, nausea, vomiting, diarrhoea, sore throat, dry cough, headache, blistering of the mouth, atypical pneumonia and elevated liver enzymes in the serum, as well as hay fever symptoms, dizziness, fatigue, and skin and eye irritations.

Health impairments from cyanobacteria in recreational waters must be differentiated between the chiefly irritative symptoms caused by unknown cyanobacterial substances and the potentially more severe hazard of exposure to high concentrations of known cyanotoxins, particularly microcystins. A single guideline value therefore is not appropriate. Rather, a series of guideline values associated with incremental severity and probability of health effects is defined at three levels.

For protection from health outcomes not due to cyanotoxin toxicity, but rather to the irritative or allergenic effects of other cyanobacterial compounds, a guideline level of 20 000 cyanobacterial cells/ml (corresponding to 10 μg chlorophyll-a/litre under conditions of cyanobacterial dominance) can be derived. A level of 100 000 cyanobacterial cells/ml (equivalent to approximately 50 μg chlorophyll-a/litre if cyanobacteria dominate) represents a guideline value for a moderate health alert in recreational waters. The presence of cyanobacterial scum in swimming areas represents the highest risk of adverse health effects, due to abundant evidence for potentially severe health outcomes associated with these scums.

Because adequate surveillance is difficult and few immediate management options are available (other than precluding or discouraging use or cancelling water sports activities such as competitions), provision of adequate public information is a key short-term measure. Medium- to long-term measures are identification of the sources of nutrient (in many ecosystems phosphorus, sometimes nitrogen) pollution and significant reduction of nutrient input in order to effectively reduce proliferation not only of cyanobacteria, but of potentially harmful algae as well.

Aesthetic issues

The aesthetic value of recreational waters implies freedom from visible materials that will settle to form objectionable deposits, floating debris, oil, scum and other matter, substances producing objectionable colour, odour, taste or turbidity, and substances and conditions that produce undesirable aquatic life. Clean beaches are one of the prime parameters that are desired by recreational users. Local economies may depend on the aesthetic quality of recreational water areas, and the environmental degradation of beaches is known to lead to loss of income from tourism.

Water at swimming areas should ideally be clear enough for users to estimate depth, to see subsurface hazards easily and to detect the submerged bodies of swimmers or divers who may be in difficulty. Aside from the safety factor, clear water fosters enjoyment of the aquatic environment. The principal factors affecting the depth of light penetration in natural waters include suspended microscopic plants and animals, suspended mineral particles, stains that impart a colour, detergent foams and dense mats of floating and suspended debris.

Visitor enjoyment of any beach is generally marred by litter. The variety of litter found in recreational water or washed up on the beach is considerable and includes, for example, discarded food/wrapping, bottles/cans, cigarette butts, dead fish, discarded condoms, discarded sanitary towels, and syringes, needles and other medical wastes. Unlike most litter, medical waste and broken glass also represent hazards to health.

Objectionable smells associated with untreated sewage effluent, decaying organic matter such as vegetation, dead animals or fish, and discharged diesel oil or petrol can deter recreational water and bathing beach users. Odour thresholds and their association with the concentrations of different pollutants of the recreational water environment have not, however, been determined.

Marine debris monitoring can be used to provide information on the types, quantities and distribution of marine debris, to identify sources of marine debris, to explore public health issues relating to marine debris and to increase public awareness of the condition of the coastline. Management options include manual or mechanical beach cleaning.

Chemical and physical agents

Chemical contaminants can enter surface waters or be deposited on beaches from both natural and anthropogenic sources. Exposure is one of the key issues in determining the risk of toxic effects from chemicals in recreational waters. The form of recreational activity will therefore play a significant role. Routes of exposure will be direct surface contact, including skin, eyes and mucous membranes, inhalation and ingestion. In assessing the risk from a particular contaminant, the frequency, extent and likelihood of exposure are crucial parts of the evaluation.

pH has a direct impact on the recreational uses of water only at very low or very high pH values. Under these circumstances, it may contribute to irritation of the skin and eyes.

The potential risks from chemical contamination of coastal and freshwater recreational waters, apart from toxins produced by marine and freshwater cyanobacteria and algae, marine animals or other exceptional circumstances, will be very much smaller than the potential risks from microbial contaminants. It is extremely unlikely that water users will come into contact with sufficiently high concentrations of most contaminants to cause ill effects following a single exposure. Even repeated (chronic) exposure is unlikely to result in ill effects at the concentrations of contaminants found in water and with the exposure patterns of recreational users. However, it remains important to ensure that chemical hazards and any potential human health risks associated with them are controlled and that users can be reassured as to their personal safety.

In most cases, the concentration of chemical contaminants will be below drinking-water guidelines. As long as care is taken in their application, the WHO *Guide-*

lines for Drinking-water Quality can provide a starting point for deriving values that could be used to make a preliminary risk assessment under specific circumstances. These guideline values relate, in most cases, to lifetime exposure following consumption of 2 litres of drinking-water per day. For recreational water contact, an intake of 200 ml per day—100 ml per recreational session with two sessions per day—may often be reasonably assumed.

An assessment of the chemical hazards in recreational water may involve inspecting the immediate area to determine if there are any immediate sources of chemical contamination, such as outfalls; considering the pattern and type of recreational use of the water to determine whether there will be extensive contact with the water and/or a significant risk of ingestion; and chemically analysing the water to support a quantitative risk assessment.

It is important that the basis of any guidelines or standards that are considered to be necessary for chemical constituents of recreational waters be made clear. Without this, there is a danger that even occasional, trivial exceedances of guidelines could unnecessarily undermine users' confidence. It is also important in evaluating chemical hazards that the risks are not overestimated. The risks should be related to risks from other hazards such as drowning or microbial contamination, which will almost invariably be much greater.

Dangerous aquatic organisms

Dangerous aquatic organisms may be encountered during recreational use of freshwater and coastal recreational environments. Such organisms vary widely and are generally of local or regional importance. The likelihood and nature of human exposure often depend significantly on the type of recreational activity concerned.

Two types of risks can be distinguished in relation to dangerous aquatic species: injury or intoxication resulting from direct encounters with predators or venomous species, and infectious diseases transmitted by species that have life cycles which are linked to the aquatic environment.

Injuries from encounters with dangerous aquatic organisms are generally sustained by accidentally brushing past a venomous sessile or floating organism when bathing, inadvertently treading on a stingray, weeverfish or sea urchin, unnecessary handling of venomous organisms during seashore exploration, invading the territory of large animals when swimming or at the waterside, swimming in waters used as hunting grounds by large predators or intentionally interfering with, or provoking, dangerous aquatic organisms.

Disease vectors include mosquitoes, which transmit malaria parasites and the viruses responsible for dengue fever, yellow fever and various types of encephalitis; and certain species of freshwater snails, which host the larval development of trematode parasites of the genus *Schistosoma*, which can cause a chronic, debilitating and potentially lethal tropical disease known as bilharzia or schistosomiasis in humans. Preventive measures include asking local health authorities for guidance on the local vector-borne disease situation and risk prevention, wearing protective clothing, using repellents and avoiding skin contact with water in schistosomiasis endemic areas.

"In-water" hazardous organisms include piranhas, snakes, electric fish, sharks, barracudas, needlefish, groupers, and moray and conger eels. Many have been known to attack and wound humans. Preventive measures include avoiding swimming in areas where large sharks are endemic; avoiding wearing shiny jewellery in the water where large sharks and barracudas are common; avoiding attaching speared fish to the body where sharks, barracudas or groupers live; avoiding wearing a headlight when fishing or diving at night in needlefish waters; and looking out for groupers and moray or conger eels before swimming into caves or putting hands into holes and cracks of rocks.

"Water's-edge" hazardous organisms include hippopotami, crocodiles and alligators. Preventive measures include keeping the animals at a distance whenever possible, avoiding swimming in areas inhabited by crocodiles or alligators, and embarking on safaris in hippopotamus- and crocodile-infested waters with a knowledgeable guide who can assess risks properly and judge the territorial behaviour of hippopotami in water.

The effects of invertebrate venoms on humans range from mild irritation to sudden death. The invertebrates that possess some kind of venomous apparatus belong to one of five large phyla: Porifera (sponges), Cnidarians (sea anemones, hydroids, corals and jellyfish), Mollusca (marine snails and octopi), Annelida (bristleworms) and Echinodermata (sea urchins and sea stars). Preventive measures include wearing suitable footwear when exploring the intertidal area or wading in shallow water, avoiding handling sponges, cnidarians, cone shells, blue-ringed octopus, bristleworms or the flower sea urchin, avoiding brushing against hydroids, true corals and anemones, and avoiding bathing in waters where Portuguese man-of-war are concentrated.

Venomous vertebrates deliver their venom either via spines, as with many fish species (e.g., catfish, stingray, scorpionfish, weeverfish, surgeonfish), or through fangs, as in sea snakes. Injuries caused by venomous marine vertebrates are common, especially among people who frequently come into contact with these marine animals. Potent vertebrate toxins generally cause great pain in the victims, who may also experience extensive tissue damage. Preventive measures include shuffling feet when walking along sandy lagoons or shallower waters where stingrays frequent, exercising caution when handling and sorting a fishing catch, wearing suitable footwear in shallow water and snake-infested areas, and carrying anti-venom in snake-infested areas.

Monitoring and assessment

WHO has developed a book based upon a framework "Code of Good Practice for Recreational Water Monitoring". This Code comprises a series of statements of principle or objectives that, if adhered to, would lead to the design and implementation of a monitoring programme of scientific credibility. It applies in principle to the monitoring of all waters used for recreational activities that involve repeated or continuous direct contact with a water body.

The Code is published in *Monitoring Bathing Waters*. It provides a linkage to the various health effects associated with recreational waters and incrementally builds up the component parts of a successful programme—key health issues, monitoring and assessment strategies, and principal management considerations. It also provides sufficient detail to allow a manager to undertake such a programme, integrating all the component parts in a consolidated whole. Cross-referencing between the Code and the various chapters of these Guidelines should ensure that a valid and replicable monitoring and assessment programme is established.

The Code and *Monitoring Bathing Waters* provide guidance on the design and implementation of a monitoring programme, including the design of a monitoring programme that includes appropriate quality assurance, data collection, data handling, data interpretation and reporting. In addition to this general guidance, guidance is given in relation to specific hazards that may be encountered in recreational water use areas.

Application of guidelines and management options for healthy recreational water use

The possible negative health outcomes associated with the use of recreational water environments result in the need for guidelines that can be converted into locally (i.e., nationally or regionally) appropriate and applicable standards and associated management of sites to ensure a safe, healthy and aesthetically pleasing environment.

A number of points need to be considered in converting guidelines into regulations adapted to local circumstances. Using the recreational water quality classification system for faecal pollution as an example, the principal requirements that would need to be incorporated into provisions would normally include:

- the establishment of a water quality classification system;
- the obligation upon the national or appropriate regulatory authorities to maintain a listing of all recognized recreational water areas in a publicly accessible location;
- the definition of responsibility for establishing a plan for recreational water safety management and its implementation;
- independent surveillance and provision of information to the public;
- the obligation to act, including the requirement to immediately consult with the public health body and inform the public as appropriate on detection of conditions potentially hazardous to health; and
- a general requirement to strive to ensure the safest achievable recreational water use conditions;

Several management interventions can be identified:

- Regulatory compliance, which includes risk management, is the making of decisions on whether or not risks to well-being are acceptable or ought to be controlled or reduced and for which responsibility lies in the hands of society regulators and participants in the activities; regulatory action at both the local level (i.e., improvements to facilities to eliminate hazards and thereby to reduce

risks) and the policy level (usually taking the form of creating standards or guidelines to control risk); enforcement of regulatory compliance; and monitoring and standards, whose aim is to promote improvement.
- Control and abatement technology (e.g., the control and abatement of pollution discharges with respect to the various levels of sewage treatment). When planning for the development of new recreational water projects or for the upgrading of existing ones, a health impact assessment (HIA), which considers changes in environmental and social determinants of health resulting from development, should be incorporated. HIA results in a package of recommended measures to safeguard health or mitigate health risks, as well as health promotional activities.
- Public awareness raising and enhancing the capacity for informed personal choice are increasingly seen as important factors in ensuring the safe use of recreational water environments and an important management intervention. One important tool used by associations and governments to enhance the public's capacity for informed personal choice is beach grading or award schemes.
- The provision of public health advice, is a key input to public awareness and informed personal choice, since it is vital that the public receive the correct information. One aspect of this management intervention is response to short-term incidents and breaches of standards. Prevention and rescue services can also be considered to fall within this intervention.

Multiple stakeholders are involved in the process of adapting and applying guidelines and standards. One way in which all the relevant stakeholders can be brought together is through the establishment of an integrated management system for marine and freshwater recreational areas based on the concept of integrated coastal area management (ICAM). This involves comprehensive assessment, the setting of objectives, and the planning and management of coastal systems and resources. It also takes into account traditional, cultural and historical perspectives and conflicting interests and uses. In an ICAM programme, the exact package of management options to reduce or eliminate health hazards and risks related to recreational water uses will be driven by the nature (including frequency and severity) of the health impacts. Upon assessing the combined level of risk, three levels of response may be considered (basic, expanded and full), each geared for a certain level of intervention.

CHAPTER 1
Introduction

This volume of the *Guidelines for Safe Recreational Water Environments* describes the present state of knowledge regarding the possible adverse impact of the recreational use of coastal and freshwater environments upon the health of users. It also outlines monitoring, control and prevention strategies relating to the hazards associated with these environments. Any possible adverse impacts must be weighed against the enormous benefits to health and well-being associated with the use of recreational water environments.

Recreational uses of inland and marine waters are increasing in many countries worldwide. These uses range from whole-body water contact sports, such as swimming, surfing and slalom canoeing, to non-contact sports, such as fishing, walking, birdwatching and picnicking.

The hazards that are encountered in recreational water environments vary from site to site, as do the nature and extent of exposure. Most available information relates to health outcomes arising from exposure through swimming and ingestion of water. In the development of these Guidelines, all available information was taken into consideration, accounting for the different routes of exposure as much as possible.

In order to properly interpret and apply the Guidelines in a manner appropriate to local conditions, it will be necessary to take into account social, cultural, environmental and economic characteristics of the site, alongside knowledge of activities undertaken, routes of exposure and the nature and severity of hazards. In doing so, local, national and international standard-setting bodies may develop standards that differ between regions and within regions according to differences in these factors.

National and local agencies working in the area of recreational water use have a responsibility to promote and ensure a safe environment. Recreational water areas may be under some form of ownership or associated with a provider of facilities or services. Owners or service providers and their personnel are key players in the control of hazards to human health and in some jurisdictions may have a legal obligation to execute continued "due diligence" relative to the safety of water or beaches. Rural or undeveloped recreational water areas often have different management arrangements and priorities. In all cases, considerable capacity to limit health risks is in the hands of the user, who should assume a degree of responsibility when engaged in recreational activities. Nongovernmental organizations and special interest groups also have an important role to play.

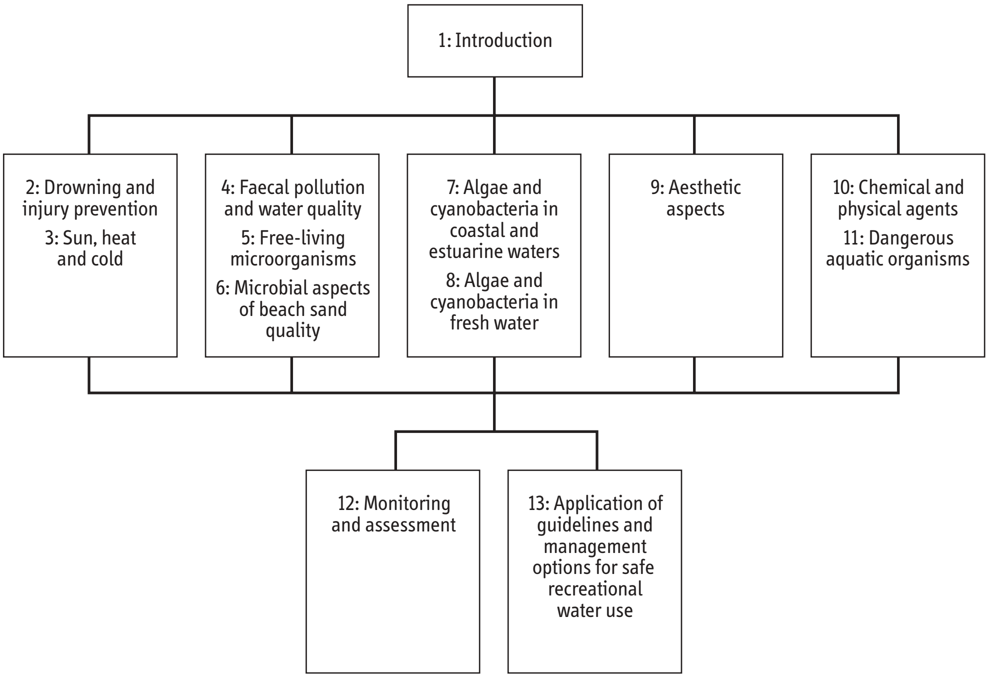

FIGURE 1.1. STRUCTURE OF *GUIDELINES FOR SAFE RECREATIONAL WATER ENVIRONMENTS. VOL. 1: COASTAL AND FRESH WATERS*

In seeking to control the health hazards associated with the recreational use of the water environment, responsible and concerned bodies have at their disposal a diverse range of interventions, including:

- monitoring and enforcing quality standards;
- general awareness-raising activities;
- adopting technical solutions to remediate problems; and
- preventing exposure to hazardous areas or conditions.

Ideally, these interventions should be adopted through proper planning and development of recreational water areas using a framework such as that provided by Integrated Coastal Area Management (see section 1.7.2)

In light of the diversity in exposure, hazard and nature of interventions, this Guidelines document is structured as shown in Figure 1.1.

1.1 General considerations

The primary aim of the *Guidelines for Safe Recreational Water Environments* is the protection of public health. The use of coastal and freshwater recreational water environments—and the resulting rest, relaxation and exercise—is associated with significant benefits to health and well-being. The purpose of the Guidelines is not to deter recreational water use but, instead, to ensure that recreational water areas are operated as safely as possible in order that the largest possible population gets the maximum possible benefit.

The Guidelines are intended to be used as the basis for the development of international and national approaches to controlling the health risk from hazards that may be encountered in recreational water environments, as well as providing a framework for local decision-making. The Guidelines may also be used as reference material for industries and operators preparing development projects in recreational water areas, as a checklist for understanding and assessing potential health impacts of recreational projects, and in the conduct of environmental impact and environmental health impact assessments in particular.

Where guideline values are presented, these are not mandatory limits, but measures of the safety of a recreational water environment. The main reason for not promoting the adoption of international standards for recreational water environments is the advantage provided by adoption of a risk–benefit approach. In the specific case of recreational water use, development of such approaches not only concerns health risks and benefits, but interrelates with other risks and benefits, especially those concerning environmental pollution/conservation, local and national economic development, and the health benefits and well-being derived from recreational use of the water environment.

This approach can often lead to the adoption of standards that are measurable and can be implemented and enforced. These would deal with, for example, water quality and dissemination of information. Other standards may relate to the education of children and adults or to the obligation to prepare and disseminate comparative studies of the safety of alternative locations for recreational water use. In developing strategies for the protection of public health, competent government authorities would take into account the general education of both adults and children and also the efforts and initiatives of nongovernmental organizations and industry operators in this area.

Clearly, a broad-based policy approach will be required that will include legislation as well as positive and negative incentives to alter behaviour and monitor situations. Such a broad base will require significant efforts in intersectoral coordination and cooperation at national and local levels, and successful implementation will require development of suitable skills and expertise as well as the elaboration of a coherent policy and legislative framework.

1.2 Types of recreational water environment

Coastal and freshwater recreational water environments are defined, for the purposes of these Guidelines, as any coastal, estuarine or freshwater area where any type of recreational usage of the water is made by a significant number of users. While uses may be diverse and the Guidelines are intended to be applicable to all types of use (see section 1.3), most concern relates to uses entailing water contact and, in the case of water quality, significant risk of water ingestion.

1.3 Types of use

There are many different types of recreational usage of water environments. These include, for example, sunbathing, wading, swimming, diving, boating, fishing and sailboarding.

Competition for suitable waters and the popularity of recreation often create conflicts between activities, as indicated in Table 1.1. These conflicts can be resolved by supervision, regulation, codes of good practice and voluntary agreements. High-activity sports often present an internal conflict between enjoyment of the excitement and hazard, resolvable by proper attention to safety, training and supervision.

Within the socioeconomic context of recreational water use, the importance of tourism is considerable—in terms of its size, impacts on socioeconomic and environmental spheres and the responsibility and means to intervene that it has at its disposal. Each year, millions of tourists flock to coastal areas. Tourism is the world's third largest industry and the prime economic sector in some states and regions, such as the Caribbean. This is creating increased competition for use of coastal waters and beach areas, increasing the need for clear regulations and codes of conduct.

TABLE 1.1. EXAMPLES OF CONFLICTING INTERACTIONS BETWEEN AND WITHIN DIFFERENT WATER RECREATIONAL ACTIVITIES, AND POSSIBLE CONTROL MEASURES

Recreational Activities	**Conflicting interactions**	**Possible control measures**
Whitewater rafting and canoeing, canoe slalom	Challenge and excitement enhance enjoyment but also present hazard of injury and drowning to participants and other water users	Wearing of buoyancy aids, safety helmets; organized training in life-saving; local and national codes of practice; classification of courses by difficulty; supervision and rescue cover at organized events; separation of conflicting uses
Waterskiing, jetskiing, windsurfing	Hazard of injury to swimmers; conflict with movements of commercial shipping, fishing and yachting; powered craft create noise and oil pollution, affecting enjoyment of other users	Creating local restriction zones to avoid conflict; engine designs and oil formulations to avoid visible emission of oil
Use of inland waterways for boating under power, canoe touring and angling	Injury to swimmers	Prohibit swimmers where water quality and conditions are unsuitable; otherwise create or designate swimming areas
Recreational use of drinking-water reservoirs	Contamination of drinking-water sources by faeces, litter, oil and fuel	Restrict uses to angling from shore or rowboat, dinghy sailing, birdwatching and walking, with local codes of practice, supervised by wardens and clubs; no dogs; provision of litter collection and toilets
Dog-walking and horse-riding on beaches	Fouling of beaches; potential transmission of toxocariasis from dog faeces, particularly to children; horses colliding with people on the beach	Banning dogs and horses from recognized swimming beaches during the swimming season

The recognition that all legitimate activities can be accommodated is the essence of integrated coastal area management (ICAM) or integrated river basin management (IBM). The process of ICAM or IBM (see section 1.7.2) introduces mechanisms to facilitate the resolution of conflicts between such competing sectors of the coastal zone or river basin and to help reach agreeable solutions, with respect to the carrying capacity of the environment, while satisfying the general needs of the area. In coming to agreement, management will usually have to adopt pragmatic solutions.

1.4 Types of user

Users of coastal and freshwater recreational water environments may include:

- the general public;
- children/babies;
- hotel guests;
- tourists;
- competitive swimmers;
- clients of camping parks; and
- specialist sporting users, including anglers, canoeists, boat users, scuba divers and so on.

Certain groups of users may be more predisposed to hazards than others. Children, for example, particularly when unattended, may cause an elevated risk of accidents for themselves and others because of their desire for attention and their general reluctance to observe formal rules of safety and hygiene. In addition, they generally play for longer periods of time in recreational waters and are more likely to intentionally or accidentally swallow water.

The elderly and disabled may have strength, agility and stamina problems that limit their ability to recover from problems encountered in recreational water environments. The elderly and immunocompromised individuals may also be at higher risk of health damage from microbial deterioration of water quality, as they are more susceptible to the pathogenic organisms that may occur in this environment.

1.5 Hazard and risk

Popularly, the terms hazard and risk are used interchangeably. Correctly, a *hazard* is a set of circumstances that could lead to harm—harm being loss of life, injury or illness. The *risk* of such an event is defined (Lacey & Pike, 1989) as the probability that it will occur as a result of exposure to a defined quantum of hazard. The *rate of incidence* or *attack rate* is the expected number of events that occur for this defined quantum of hazard. Strictly speaking, probabilities and rates obey different laws, but if the probabilities are small and events are independent, the two values will be approximately equal. Risks can vary from negligible—an adverse event occurring at a frequency of below one per million—to high—fairly regular events that would occur at a rate of greater than one in a hundred (Calman, 1996).

1.5.1 *Types of hazard encountered*

The hazards associated with the use of coastal and freshwater recreational water environments fall into a number of groups:

- physical hazards (leading, for example, to drowning or injury);
- cold, heat and sunlight;
- water quality (especially exposure to water contaminated by sewage, but also exposure to pathogenic microorganisms free-living in recreational water);
- contamination of beach sand;
- algae and their toxic products;
- chemical and physical agents; and
- dangerous aquatic organisms.

The existence of a diverse range of hazards in the recreational water environment indicates the need for an understanding of their relative importance for health. Examples of adverse health outcomes associated with these hazards are given in Table 1.2.

Drowning and spinal injury are severe health outcomes of great concern to public health. Other injuries, such as cuts from glass and other wastes, while less severe, cause distress and decrease the benefits to well-being arising from recreation. Human behaviour—especially alcohol consumption—is a prime factor that increases the likelihood of injury (see chapter 2), for example, up to 50% of drowning deaths are associated with alcohol in some countries.

Notwithstanding the above, much attention has focused in recent years upon microbial hazards. In particular, health risks associated with contamination of water by sewage and excreta and associated gastroenteric outcomes have been the topics of both scientific and general public interest (see chapter 4). The hazards concerned are not restricted to gastroenteric outcomes and potentially include acute febrile respiratory illness and ear infections arising from pollution of water by excreta and swimmers and other naturally occurring or non-faecally derived infectious agents, such as leptospires (see chapter 5). However, in general terms, it appears that contamination of recreational water with excreta and sewage is widespread and common and affects large numbers of recreational water users, the majority of whom exhibit mild gastroenteric symptoms.

Hazards to human health exist even in unpolluted environments. For example, eye irritation in bathers may occur as a result of a reduction in the eye's natural defences through limited contact with water and does not necessarily relate to water quality or pollution *per se*.

1.5.2 *Assessment of hazard and risk*

Assessments of hazard and risk inform the development of policies for controlling and managing risks to health and well-being in water recreation. Both draw upon experience and the application of common sense, as well as the interpretation of data. Isolated measurements of risk are not very helpful when decisions have to be made for managing risks or developing policies for controlling them.

TABLE 1.2. EXAMPLES OF ADVERSE HEALTH OUTCOMES ASSOCIATED WITH HAZARDS ENCOUNTERED IN RECREATIONAL WATER ENVIRONMENTS

Type of adverse health outcome	**Examples of associated hazards (with chapter references in parentheses)**
Drowning	• Caught in tidal or rip currents, cut off by rising tides, falling overboard, caught by submerged obstacles, falling asleep on inflatables and drifting into deep water far from shore, slipping off rocks or washed off rocks by wave, misjudging swimming ability (2).
Impact injury	• Impact against hard surfaces or sharp objects (2), driven by the participant (diving, collision, treading on broken glass or jagged metal) or by the force of wind and water. • "Needle stick" injuries from used needles that have washed up or have been discarded on the beach (2). • Coral cuts, oyster cuts and abrasions from slipping on rocks (2). • Attack by aquatic animals (shark, conger and moray eels, piranhas, seals) (11).
Physiological	• Chilling, leading to coma and death (3). • Acute exposure to heat and ultraviolet radiation in sunlight—heat exhaustion, sunburn, sunstroke (3). • Cumulative exposure to sun—skin cancers (basal and squamous cell carcinoma, melanoma) (3).
Infection	• Ingestion of, inhalation of, or contact with pathogenic bacteria, viruses, fungi and parasites, which may be present in water as a result of faecal contamination, carried by participants or animals using the water, or naturally present (4–6). • Bites by mosquitoes and other insect vectors of parasitic diseases (11).
Poisoning and toxicoses	• Ingestion of, inhalation of, or contact with chemically contaminated water (10). • Stings of poisonous and venomous animals (jellyfish, snakes, sting rays) (11). • Ingestion of, inhalation of, or contact with blooms of toxigenic cyanobacteria in fresh (8) or marine water (7) and/or of dinoflagellates in marine water (7).

The assessment of a beach or water should take into account several key considerations, including:

- the presence and nature of natural or artificial hazards;
- the severity of the hazard as related to health outcomes;
- the availability and applicability of remedial actions;
- the frequency and density of use; and
- the level of development.

Health risks that might be tolerated for an infrequently used and undeveloped recreational area, for example, may justify immediate remedial measures at other areas that are more widely used or highly developed.

Figure 1.2 provides a schematic approach to comparing health hazards encountered during recreational water use. A severe health outcome such as permanent paralysis or death, as a result of diving into shallow water, may affect only a small number

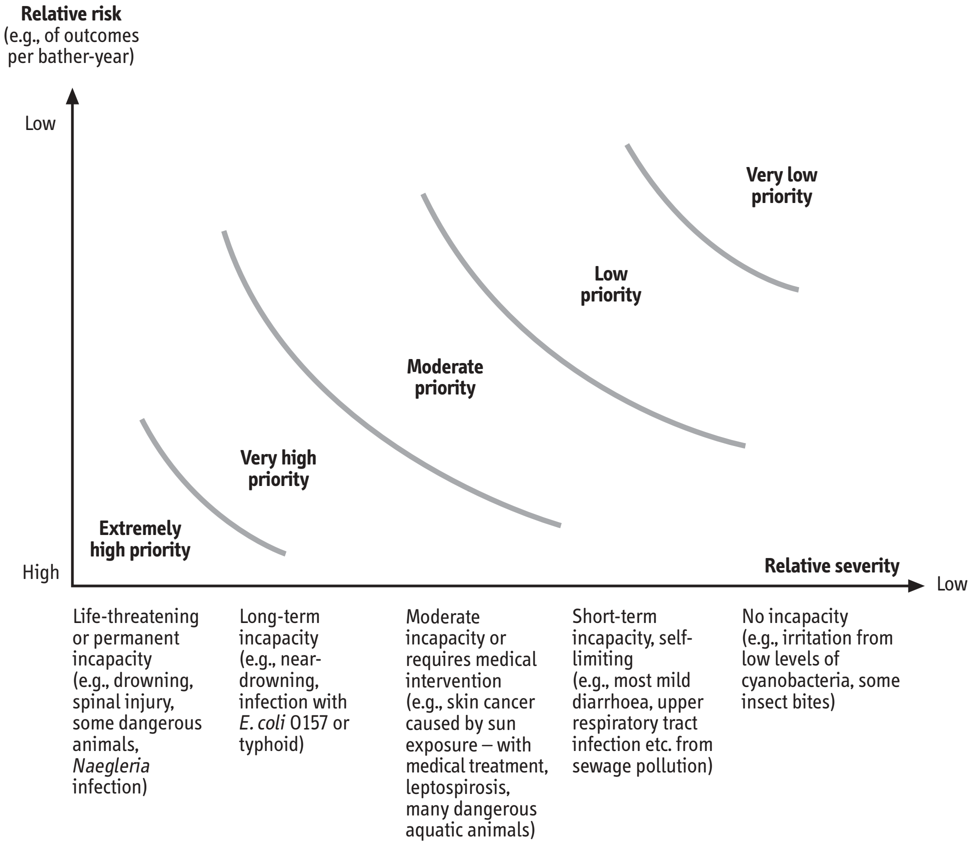

FIGURE 1.2. SCHEMATIC APPROACH TO COMPARING HEALTH HAZARDS ENCOUNTERED DURING RECREATIONAL WATER USE

of swimmers annually but may warrant a high management priority. Minor skin irritations, encountered at the other end of the scale, may affect a higher number of swimmers per year but do not result in any incapacity and thus require lower management priority.

Figure 1.2 can be applied throughout the Guidelines. For each hazard discussed, the "severity" of the hazard can be related to the relative risk in this figure and can serve as a tool to initiate further research or investigation into the reduction of risk as well as to highlight or emphasize priority protective or remedial management measures.

Data related to risk take four main forms:

- national and regional statistics of illness and deaths;
- clinical surveillance of incidence of illness and outbreaks;
- epidemiological studies and surveys; and

- accident and injury records held by recreational water area owners/managers and local authorities.

Although "incident records" held by local authoritative bodies are often comprehensive, published statistics are seldom sufficiently detailed for risk assessment. Processes of surveillance for drinking-water supplies (definined as the continuous and vigilant public health assessment of the safety and acceptability of supplies) have been recommended by WHO (1976, 1997) and involve a dual responsibility of a national, governmental regulator and the supplier or provider of the service. Systems for surveillance of public health operate in most countries. They serve the broad purpose of alerting either regulator or supplier to changes in incidence of disease and to the need for initiating immediate investigation of the causes and remedial action. Such investigation will involve epidemiology (the study of the occurrence and causes of disease in populations). Galbraith & Palmer (1990) give details of the use of epidemiology in surveillance. Epidemiology may also be used as a research tool to investigate hypotheses concerning the causes of illness (section 1.5.3).

There are other reasons why it is difficult to estimate risk directly, such as the following:

- In most active water sports, enjoyment arises from the use of skill to avoid and overcome perceived hazards. The degree of competence of participants and the use of properly designed equipment and protective clothing, accompanied by supervision and training, will considerably modify the risk.
- Risks of acquiring infectious disease will be influenced by innate and acquired immunity (for examples, see Gerba et al., 1996). The former comprises a wide range of biological and environmental factors (age, sex, nutrition, socioeconomic and geographic), as well as body defences (impregnability of the skin, lysozyme secretion in tears, mucus and sweat, the digestive tract and phagocytosis). Previous challenge by pathogens often results in transient or long-lasting immunity.
- Assessment of harm itself and the degree of harm suffered depend upon judgement at the time. Medical certification of injury and of physiological illness and infection, accompanied by clinical diagnosis, is the most reliable information. Information obtained by survey or questionnaire will contain a variable degree of uncertainty caused by the subjects' understanding of the questions, their memory of the events and any personal bias of the subject and interviewer. Survey information is only as good as the care that has gone into the design and conduct of the survey. Data for aesthetic insult are subjective by nature, but frequencies of occurrence of particular types of waste objects on stretches of beaches can be quantified.
- The causes of harm must be ascertained as much as possible at the time. There are considerable difficulties in the cases of low-level exposures to chemical and physical agents that have a cumulative or threshold effect and of infectious diseases caused by those pathogens that have more than one route of infection or have a long period of incubation. For example, gastroenteric infections at

resorts may result from person-to-person contact or faulty food hygiene in catering, as well as from ingesting sewage-contaminated water.

- Where data are in the form of published regional or national statistics giving attack rates, the exact basis on which the data are collected and classified must be ascertained. For example, national statistics on deaths by drowning will usually include suicides and occupational accidents (fishermen, mariners, construction workers) as well as misadventure in recreation.
- It cannot be assumed that risk is directly proportional to exposure or that risks from multiple exposures or a combination of different factors will combine additively.

1.5.3 The use of epidemiology in risk assessment

There is a considerable body of epidemiological information concerning the effects of faecal contamination of swimming waters on the incidence of gastroenteritis and other transmissible diseases in swimmers and other participants in water recreation. This has been critically reviewed (Pike, 1989, 1994; Cartwright, 1992; Fewtrell & Jones, 1992; Prüss, 1998) and is examined later in this volume (chapter 4). The level of epidemiological research concerning some other types of recreational water hazard is considerably less than that for faecal contamination. This may relate to a number of factors, including infrequent outcomes and ethical concerns.

Epidemiological information is more reliable than published statistics for assessing risks, since its rigorous disciplines are designed to eliminate sources of bias and errors in interpretation. On the other hand, this rigour limits epidemiological studies to single or a few closely related hazards and carefully defined populations. Hence, epidemiological approaches do not always measure the full range of variation in population responses (Grassman, 1996).

1.5.4 Degree of water contact

The overall basis for development of a risk reduction strategy depends on broad classifications of recreational activities. For hazards where contact with and/or ingestion of water are important, an understanding of the different degrees of contact associated with different recreational water uses is essential. The degree of water contact directly influences the degree of contact with infectious and toxic agents and physical hazards found in water and therefore the likelihood of being injured or contracting illness.

The degrees of water contact encountered in coastal and freshwater recreational water environments may be classified as follows:

- *No contact*—recreational activity in which there is normally no contact with water (e.g., angling from shore), or where water is incidental to enjoyment of the activity (such as sunbathing on a beach).
- *Incidental contact*—recreational activity in which only the limbs are regularly wetted and in which greater contact (including swallowing water) is unusual—for example, boating, fishing, wading.

- *Whole-body contact*—recreational activity in which the whole body or the face and trunk are frequently immersed or the face is frequently wetted by spray, and where it is likely that some water will be swallowed—e.g., swimming, diving or whitewater canoeing. Inadvertent immersion, through being swept into the water by a wave or slipping, would also result in whole-body contact.

Routes of exposure to infectious and toxic agents in water will vary depending on the degree of water contact. Generally, exposure of skin and mucous membranes during recreational water activities is most frequent. For whole-body contact activities, the probability that some water will be ingested will be greater, although actual data on the quantities of water ingested while indulging in water sports are difficult to obtain. Inhalation can be important in circumstances where there is a significant amount of spray, such as in waterskiing. The skill of the participant in water recreation will also be important in determining the extent of involuntary exposure, particularly water ingestion.

1.6 Measures to reduce risks in water recreation

Because hazards may give rise to health effects after short-term exposures, it is important that standards, monitoring and implementation enable preventive and remedial actions within real time frames. For this reason, emphasis in the Guidelines is placed upon identifying circumstances and procedures that are likely to lead to a continuously safe environment for recreation. This approach emphasizes monitoring of both conditions and practices and the use of threshold values as key indicators, assessed through programmes of monitoring and assessment.

Table 1.2 in section 1.5.1 lists and classifies the main adverse health outcomes associated with exposure to hazards encountered in water recreation. Study of the examples given indicates that reduction of most, if not all, of their associated risks can be obtained by avoiding the circumstances giving rise to the hazard or mitigating their effect. Table 1.2 also suggests particular types of recreation that may be prone to certain hazards and actions that may be taken to reduce the risk. For example, glass left on a beach will cause the hazard of cuts to walkers with bare feet, which may be mitigated by regular cleaning of the beach, provision of litter bins, prohibiting the use of glass on the beach and educational awareness campaigns. This suggests that the types of recreational activity undertaken in a given location should be subject to a hazard assessment and the type of control measures that will be most effective determined.

Examples of potential control measures and bases for developing guidelines and for reducing risks in non-contact, incidental contact and whole-body contact water recreation are presented in Tables 1.3 (page 12), 1.4 (page 13) and 1.5 (page 14), respectively. For each recreational use, more than one hazard will be encountered and the list of hazards for each use will differ depending on circumstances. Measures for risk reduction will therefore be specific to each form of recreation and to particular circumstances. Detailed examples of hazards and their associations with particular forms of recreation will be considered in later chapters.

TABLE 1.3. HAZARDS AND MEASURES FOR REDUCING RISKS IN NON-CONTACT RECREATION

Examples of non-contact recreational activities[a]	Principal hazards	Potential risk reduction measures
Angling from shore (1–6) Boating under power (1–4) Picnics (1–4, 6) Walking (1–4, 6) Sunbathing (2–4, 6) Birdwatching (1–4, 6)	1. Falling in, drowning	1. Where appropriate: safety rails, lifebelts/lifejackets, warning notices, broadcast gale warnings, education, legislation regarding use of lifejackets while boating. Personal care.
	2. Sunburn, sunstroke, skin cancer	2. General and local publicity. Use of sunscreen or sunblock, limit exposure. Wearing protective clothing.
	3. Aesthetic revulsion from fish deaths, anaerobic conditions, oil and other visible pollution	3. Control and licensing of discharges from sewage works, industry, storm sewer outfalls, agriculture, landfills and watercraft.
	4. Bites from mosquitoes and other insect vectors of disease	4. Health warnings to travellers, anti-malarial therapy, avoidance of infested regions, application of appropriate insect repellants.
	5. Infection following skin injury and exposure to water	5. Exercising care; covering all injuries with waterproof dressings.
	6. Injury; treading on broken glass or jagged metal waste	6. Litter control, cleansing recreational area. Putting rubbish in bins or taking it away. Prohibiting use of glass on beach.

[a] Numbers in parentheses refer to principal hazards (column 2) and potential risk reduction measures (column 3).

Participants in the whole-body contact sports of sub-aqua diving, surfing, water-skiing, whitewater canoeing, rafting and windsurfing normally wear wet suits or other protective clothing, which limit skin exposure to the agents of leptospirosis and schistosomiasis and to venomous animal stings, as well as to chilling and ultraviolet radiation (UVR), but which may aggravate symptoms caused by contact with toxic cyanobacteria under some circumstances or enhance the absorption of chemicals through the skin. The wearing of helmets and buoyancy jackets in sailing and canoeing activities (Table 1.5) protects against head injuries and drowning, respectively.

1.7 Managing recreational waters

1.7.1 Stakeholders

Mutually supportive actions should take place, coherently, at the local, national and international level in order to reduce risks encountered during recreational water use. Multiple stakeholders intervene in the assessment, use and protection of recreational waters. Their roles and responsibilities should be defined and their efforts harnessed through an integrated planning framework. Figure 1.3 (page 16) illustrates the variety of stakeholders and their roles in the process of assessing and using recreational waters and taking remedial actions to limit health hazards.

TABLE 1.4. HAZARDS AND MEASURES FOR REDUCING RISKS IN INCIDENTAL CONTACT RECREATION

Examples of incidental contact recreational activities[a]	**Principal hazards**	**Potential risk reduction measures**
Rowing, sailing, canoe touring (1, 2, 3, 5, 6) Wading (1–8) Fishing (1–8) Paddling, adults (1–8) (for paddling by young children,) see Table 1.5	1. Falling in, drowning	1. Where appropriate: safety rails, lifebelts/lifejackets, warning notices, broadcast gale warnings, education, legislation regarding use of lifejackets while boating, supervision and availability of rescue services. Personal care.
	2. Leptospirosis (fresh water)	2. Bankside management to control rodents, litter collection. Treating and covering cuts and abrasions prior to exposure. Seeking medical advice if influenza-like symptoms are noticed a few days after recreation.
	3. Cyanobacterial toxicoses (fresh water)	3. Control of eutrophication, monitoring and reporting of cyanobacterial populations, curtailing recreation during blooms. Local publicity. Personal awareness: reporting blooms, avoiding contact, washing down body and equipment after recreation.
	4. Injury; treading on broken glass or jagged metal waste	4. Litter control, cleansing recreational area. Putting rubbish in bins or taking it away. Prohibiting use of glass on beach.
	5. Sunburn, sunstroke, skin cancer	5. General and local publicity. Use of sunscreen or sunblock, limit exposure. Wearing protective clothing.
	6. Bites from mosquitoes and other insect vectors of disease	6. Health warnings to travellers, anti-malarial therapy, avoidance of infested regions, application of appropriate insect repellants.
	7 Fish stings	7 Local awareness raising where problem occurs.
	8. Swimmers' itch and schistosomiasis (freshwater)	8. Control weeds and aquatic snails. Avoiding warm, snail-infested ponds. Personal awareness raising. Information on occurrence of schistosomiasis.

[a] Numbers in parentheses refer to principal hazards (column 2) and potential risk reduction measures (column 3).

1.7.2 Integrated coastal area or river basin management

Integrated coastal area management (ICAM) and integrated river basin management (IBM) are usually initiated in response to issues relating to one or more of the following: fisheries, recreation/tourism, hazards and mangrove depletion. Therefore, recreational water hazards are just one of a wide range of issues, interests and constraints that affect the planning and management of coastal areas or river basins. Decisions relating to management of hazards should be made with reference to all relevant government policies and other factors that affect coastal/river basin amenity and use. Social, economic, aesthetic, recreational and ecological factors all need to be considered. Successful ICAM or IBM also requires "integration over time, with immediate

TABLE 1.5. HAZARDS AND MEASURES FOR REDUCING RISKS IN WHOLE-BODY CONTACT RECREATION

Examples of whole-body contact recreational activities[a]	Principal hazards	Potential risk reduction measures
Sub-aqua diving (1–12) Swimming (1–12) Surfing (1, 2, 5–9, 11, 12) Waterskiing (1–12) Whitewater canoeing, rafting (1–3, 5–7, 11, 12) Windsurfing (sailboarding) (1–12) Children's exploratory activities and paddling (1–12)	1. Drowning	1. Where appropriate: lifebelts/lifejackets, warning notices, broadcast gale warnings, education, legislation regarding use of lifejackets while boating, supervision and availability of rescue services. Personal care.
	2. Waterborne infections[b]	2. Microbial standards. Licensing, control and treatment of discharges of sewage, effluents, storm overflows. Improvements where indicated by unsatisfactory microbial quality. Personal awareness of local conditions.
	3. Leptospirosis (fresh water)	3. Bankside management to control rodents, litter collection. Treating and covering cuts and abrasions prior to exposure. Seeking medical advice if influenza-like symptoms are noticed a few days after recreation.
	4. Cyanobacterial toxicoses (fresh water)	4. Control of eutrophication, monitoring of cyanobacterial populations, curtailing recreation during blooms. Local publicity. Personal awareness raising: reporting blooms, avoiding contact, washing down body and equipment after exercise.
	5. Impact injury	5. Notices indicating hazards. Personal awareness raising and avoidance, wearing head and body protection, where appropriate. Supervision and presence of lifeguards and rescue services. Removal/mitigation of the hazard.
	6. Injury; treading on broken glass or jagged metal waste	6. Litter control, cleansing recreational area. Putting rubbish in bins or taking it away. Prohibiting use of glass on beaches.
	7. Collision with or entrapment by wrecks, piers, weirs, sluices and underwater obstructions	7. Notices to mariners, marker buoys, posting warnings. Personal awareness. Legislation requiring boater training. Rescue services to respond to accidents and mitigate injuries. Appropriate oversight (e.g., harbour patrol).
	8. Fish stings	8. Local awareness raising where problem occurs.
	9. Attack by marine animals (sharks, conger and moray eels, seals)	9. Posting warnings. Personal awareness raising and avoidance.
	10. Swimmers' itch and schistosomiasis (fresh water)	10. Control weeds and aquatic snails. Avoiding warm, snail-infested ponds. Personal awareness raising. Information on the occurrence of schistosomiasis.

TABLE 1.5. *Continued*

Examples of whole-body contact recreational activities[a]	Principal hazards	Potential risk reduction measures
	11. Bites from mosquitoes and other insect vectors of disease	11. Health warnings to travellers, anti-malarial therapy, avoidance of infested regions, application of appropriate insect repellants.
	12. Sunburn, sunstroke, skin cancer	12. General and local publicity. Use of sunscreen or sunblock, limit exposure. Wearing protective clothing.

[a] Numbers in parentheses refer to principal hazards (column 2) and potential risk reduction measures (column 3).

[b] Infections caused by pathogens derived from faecal pollution (see chapter 4).

day-to-day management objectives being co-ordinated and consistent with long-term national and international policy goals" (OECD, 1993, p. 16). It focuses on the interaction between various activities/resource demands carried out within the coastal zone or river basin as distinct from other regions.

Management should be coordinated to reconcile different, sometimes conflicting, uses:

- management of land resources for urban, industrial, mining, tourism and conservation activities;
- management of waters for recreation, aquaculture, conservation, transport and mining;
- management of living freshwater or marine resources; and
- provision of coastal and flood defences.

ICAM and IBM provide umbrellas for coordination among these areas of intervention, covering the economic, abiotic/biotic and social systems.

Current ICAM thinking encapsulates both coastal and river catchments. In the rest of this volume, therefore, only the term ICAM will be used.

1.7.3 Types of management action

Figure 1.4 (page 17) provides a management framework with different levels of health risk and accordingly suggested relevant interventions (which will have differing time frames for implementation), ordered in four major fields:

- compliance and enforcement;
- control and abatement technology;
- public awareness and information; and
- public health advice and intervention.

Clearly, however, there are linkages between these, with, for example, public health advice having an important input into public awareness.

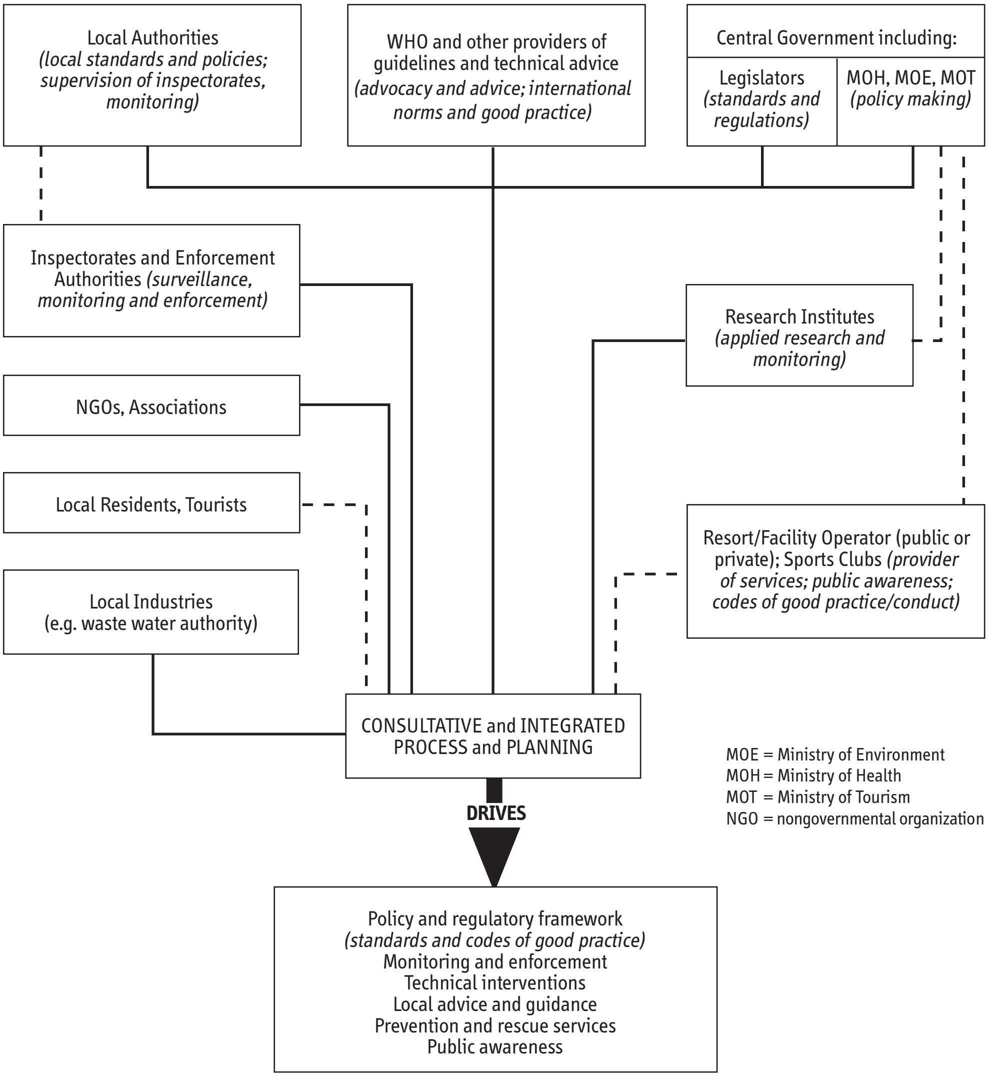

FIGURE 1.3. SOME STAKEHOLDERS IN RECREATIONAL WATER ENVIRONMENTS

The scheme shown in Figure 1.4 has general relevance and can be applied to all areas covered by the various chapters in this volume. The management interventions outlined in Figure 1.4 are discussed in more detail in chapter 13.

1.8 Nature of the guidelines

A guideline can be a level of management, a concentration of a constituent that does not represent a significant risk to the health of individual members of significant user groups, a condition under which such concentrations are unlikely to occur, or a combination of the last two. In deriving guidelines including guideline values, account

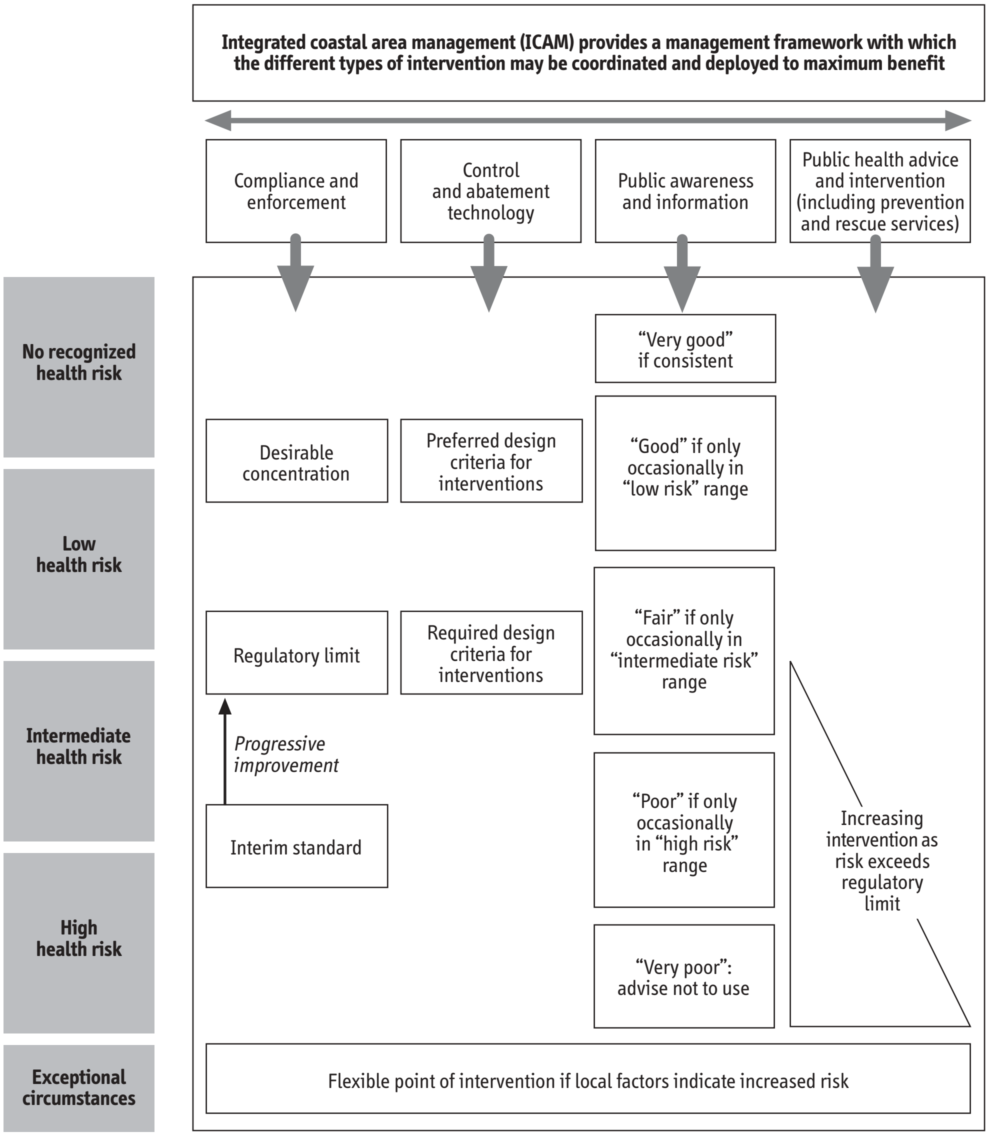

FIGURE 1.4. MANAGEMENT FRAMEWORK AND TYPES OF INTERVENTION IN RELATION TO DIFFERENT LEVELS OF RISK

is taken of both the severity and frequency of associated health outcomes. Water conforming to the guidelines may, however, present a health risk to especially susceptible individuals or to certain user groups.

When a guideline is exceeded, this should be a signal to investigate the cause of the failure and identify the likelihood of future failure, to liaise with the authority responsible for public health to determine whether immediate action should be taken to reduce exposure to the hazard, and to determine whether measures should be put in place to prevent or reduce exposure under similar conditions in the future.

For most parameters, there is no clear cut-off value at which health effects are excluded, and the derivation of guideline values and their conversion to standards therefore include an element of valuation addressing the frequency, nature and severity of associated health effects. This valuation process is one in which societal values play an important role, and the conversion of guidelines into national policy, legislation and standards should therefore take account of environmental, social, cultural and economic factors.

The existence of a guideline value or national standard does not imply that environmental quality should be degraded to this level. Indeed, a continuous effort should be made to ensure that recreational water environments are of the highest attainable quality.

Many of the hazards associated with recreational use of the water environment are of an instantaneous nature: accidents and exposures to infectious doses of microorganisms may occur in very short periods of time. Short-term deviations above guideline values or conditions are therefore of importance to health, and measures should be in place to ensure and demonstrate that recreational water environments are continuously safe during periods of actual or potential use.

This volume of the *Guidelines for Safe Recreational Water Environments* does not address:

- exposures associated with foodstuffs, in particular water products such as shellfish;
- protection of aquatic life or the environment;
- occupational exposures of individuals working in recreational water environments;
- especially susceptible individuals (or population groups);
- waters afforded special significance for religious purposes and which are therefore subject to special cultural factors;
- risks associated with ancillary facilities that are not part of the recreational water environment; for example, beach sand is addressed, while toilet facilities in adjacent areas are not considered beyond assertion of the need for them in order to minimize soiling of the recreational environment;
- guideline values for aesthetic aspects, since their valuation is one of societal and cultural values, which cannot be expressed solely in quantitative terms, and their control will not reduce adverse health effects; on the other hand, the importance of aesthetic factors in ensuring maximum benefit for well-being from recreational use of the water environment is discussed;
- seasickness;
- the "bends" decompression sickness and other phenomena restricted to sub-aqua and deep-sea diving;
- guidance on rescue, resuscitation or treatment; and
- therapeutic uses of waters (thalassotherapy, spas).

Swimming pools, spas and similar recreational water environments are addressed in the companion volume, *Guidelines for Safe Recreational Water Environments Volume 2: Swimming Pools, Spas and Similar Recreational Water Environments.*

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CHAPTER 2
Drowning and injury prevention

A number of injury-related health outcomes may arise through the recreational use of water and adjacent areas. Prominent among them are:

- drowning and near-drowning;
- major impact injuries (including spinal injuries resulting in various degrees of paraplegia and quadriplegia; and head injuries resulting in concussion, brain injury and loss of memory and motor skills);
- slip, trip and fall injuries (including bone fractures/breaks/dislocations resulting in temporary or permanent disability; facial injuries resulting in nose and jaw dislocations and scarring; and abrasions); and
- cuts, lesions and punctures

This chapter discusses these adverse health outcomes and their contributory factors, along with possible preventive measures. Bites, stings and so on from aquatic organisms are addressed in chapter 11.

2.1 Drowning

Drowning, which can be defined as death arising from impairment of respiratory function as a result of immersion in liquid, is a major cause of death worldwide. It has been estimated that, in 2000, 449 000 people drowned worldwide, with 97% of drownings occurring in low- and middle-income countries (Peden & McGee, 2003). It is the third leading cause of death in children aged 1–5 and the leading cause of mortality due to injury, with the mortality rates in male children being almost twice as high as those in female children (Peden & McGee, 2003). Not all drownings are related to recreational water use and the percentage that is attributable to recreational water is likely to vary from country to country. A study in the USA found that 50–75% of all drownings there occurred in natural waters (oceans, lakes, rivers, etc.), with both children and adults being victims (Dietz & Baker, 1974). Brenner et al. (2001) examined the location of drownings in children in the USA. They reported that for children aged between 1 and 4, 56% of drownings were in artificial pools and 26% were in other bodies of freshwater, while among older children 63% of drownings were in natural bodies of freshwater. In Australia, between 1992 and 1997, 17% of drownings occurred in non-tidal lagoons and lakes and 10% occurred at surf beaches (Mackie, 1999). In Uganda, drowning has been shown to be responsible for 27% of all injury fatality. Most of the drowning victims were young males who drowned in lakes and rivers during transportation or on fishing trips (Kobusingye, 2003). Data on drowning in many countries is inadequate, especially in terms of the location of the incident, and this can hamper the evaluation of interventions and prevention and rescue techniques.

Death by drowning is not the sole outcome of distress in the water. Near-drowning is also a serious problem. One study (Wintemute et al., 1988) found that for every 10 children who die by drowning, 140 are treated in emergency rooms and 36 are admitted to hospitals for further treatment (see also Spyker, 1985; Liller et al., 1993), although some never recover. In the Netherlands, it has been reported that on average there are about 300 drowning fatalities a year and an additional 450 cases who survive the drowning incident, of these 390 are admitted to hospital for further treatment (Bierens, 1996; Branche & Beeck, 2003).

It is possible to survive prolonged submersion in cold water (e.g., less than 21 °C). In rare cases, people have been submerged for significant periods (e.g., up to 40 min) with normal neurological recovery (Spyker, 1985; Winegard, 1997; Chochinov et al., 1998; Hughes et al., 2002; Perk et al., 2002).

The recovery rate from near-drowning may be lower among young children than among teenagers and adults. Some survivors suffer subsequent anoxic encephalopathy (Pearn et al., 1976; Pearn, 1977; Patrick et al., 1979), leading to long-term neurological deficits (Quan et al., 1989). Studies show that the prognosis depends more on the effectiveness of the initial rescue and resuscitation than on the quality of subsequent hospital care (Fenner et al., 1995; Cummings & Quan, 1999). Development of effective rescue resources, with on-scene resuscitation capabilities, may be important in reducing the frequency of drowning and consequences of near-drowning.

2.1.1 Contributory factors

Both drowning and near-drowning have been associated with many contributory factors (see, for example, Poyner, 1979). Data suggest, for example, that males are more likely to drown than females (Peden & McGee, 2003). This is generally associated with higher exposure to the aquatic environment (through both occupational and recreational uses), greater consumption of alcohol (leading to decreased ability to cope and impaired judgement) and their inclination towards higher risk-taking activity (Dietz & Baker, 1974; Mackie, 1978; Plueckhahn, 1979, 1984; Nichter & Everett, 1989; Quan et al., 1989; Howland et al., 1996).

Alcohol consumption is one of the most frequently reported contributory factors associated with the greatest proportion of adolescent and adult drownings in many countries (Howland & Hingson, 1988; Levin et al., 1993; Petridou, 2003). For children, lapses in parental supervision are the most frequently cited contributory factor in drownings (Quan et al., 1989), although alcohol consumption by the parent or guardian may also play a role in the lapse of supervision (Petridou, 2003).

Drowning and near-drowning may be associated with recreational water uses involving minimal water contact. Recreational use of watercraft (yachts, boats, canoes) and fishing (from watercraft, water's edge, rocks or solid structures) have been associated with drownings (Plueckhahn, 1972; Nichter & Everett, 1989; Steensberg, 1998). Such recreational water uses may occur during cold weather, and immersion cooling may be a significant contributory factor (see section 3.2; Bierens et al., 1990, 1995; Beyda, 1998; Lindholm & Steensberg, 2000). Non-use of lifejackets, even when readily available, is frequently cited as a significant contributory factor in these cases (Plueckhahn, 1979; Patetta & Biddinger, 1988; Steensberg 1998; Quan et al., 1998). In one study in North Carolina, USA, the activities most frequently associated with drownings were (in descending order) swimming, wading and fishing (Patetta & Biddinger, 1988).

Attempted rescue represents a significant risk to the rescuer. For example, a study in North Carolina reported the death by drowning of the would-be rescuer in a significant number of cases (Patetta & Biddinger, 1988). In Australia, Mackie (1999) reported that between 1992 and 1997 there were 1551 non boating-related drownings, of which over 2% were sustained while attempting a rescue.

Hyperventilation before breath-hold swimming and diving has been associated with a number of drownings among individuals, almost exclusively males, with excellent swimming skills. Although hyperventilation makes it possible for a person to extend their time under water, it may result in a loss of consciousness by lowering the carbon dioxide level in the blood (Craig, 1976; Spyker, 1985).

At beaches with surf, rip currents can be a major cause of distress. These currents, which pull swimmers away from shore, have been found to be a factor in as many as 80% of rescues by surf lifeguards (USLA, 2002). In Australia, 35% of rescues and 18.5% of resuscitation cases, over a ten year period, from surf beaches were due to rip currents (Fenner, 1999).

The presence of pre-existing disease is a risk factor for drowning and near-drowning, and higher rates of drowning are reported among those with seizure disorders (Greensher, 1984; CDC, 1986; Patetta & Biddinger, 1988; Quan et al., 1989). Further documented contributory factors include water depth and poor water clarity (Quan et al., 1989).

2.1.2 Preventive and management actions

It has been suggested that over 80% of all drownings can be prevented and prevention is the key management intervention (World Congress on Drowning, 2002; Mackie, 2003). Surprisingly, there is no clear evidence that drowning rates are greater in poor swimmers (Brenner, 2003) and the value of swimming lessons and water safety instruction as drowning preventive measures has not been demonstrated (Patetta & Biddinger, 1988; Mackie, 2003). There is also a significant debate regarding the age at which swimming skills may be safely acquired. Although the need for adult supervision is not decreased when young children acquire increased skills, the possibility that training decreases parental vigilance has not been assessed (Asher et al., 1995).

The availability of cardiopulmonary resuscitation (CPR) (including infant and child CPR) skills (Patetta & Biddinger, 1988; Orlowski, 1989; Liller et al., 1993; Kyriacou et al., 1994; Pepe and Bierens, 2003a) and of rescue skills among witnesses (Patetta & Biddinger, 1988) have been reported to be important in determining the outcome of unintentional immersions. It has been recommended that resuscitation skills should be learned by all professionals who frequent aquatic areas (Pepe & Bierens, 2003b) as early first aid and resuscitation are important factors in survival after a drowning incident.

The Centers for Disease Control and Prevention, USA (CDC) have suggested that legal limits for blood alcohol levels during water recreation activities should be mandated and enforced, and that the availability of alcohol at water recreation facilities should be restricted (CDC, 1998). Cummings & Quan (1999) report data that supports the theory that decreasing alcohol use around water is an effective safety intervention.

Education, aimed at making both locals and tourists knowledgeable about water-based hazards (such as rip currents), can play an important role in reducing drowning. Whittaker (2003) noted that an education package, started in 1998, apparently reduced the drowning rate on beaches in Victoria (Australia) by 31% over a 4 year period.

The principal contributory factors and preventive and management actions for drowning and near-drowning are similar and are summarized in Table 2.1.

2.2 Spinal injury

Data concerning the number of spinal injuries sustained as a result of swimming or water recreation incidents are not widely available or systematically collected. In the USA, it has been found that some 10% of all spinal cord injuries (an incidence of approximately 1000 per year) are related to diving into water (Think First Foundation, 2002).

Blanksby et al. (1997) tabulated data from a series of studies concerning diving incidents as the cause of acute spinal injury in various regions of the world. In one study (Steinbruck & Paeslack, 1980), 212 of 2587 spinal cord injuries were caused by sports or diving incidents, of which 139 were associated with water sports, the majority (62%) with diving. Diving incidents were found to be responsible for 3.8–14% of traumatic spinal cord injuries in a comparison of French, Australian, English and US studies (Minaire et al., 1979), for 2.3% of spinal injuries in a South African study and for 21% in a Polish study (Blanksby et al., 1997).

In diving incidents of all types, injuries are almost exclusively located in the cervical vertebrae (Minaire et al., 1979; Blanksby et al., 1997; Watson et al., 2001). Statistics such as those cited above therefore underestimate the importance of these injuries, which typically cause quadriplegia (paralysis affecting all four limbs) or, less commonly, paraplegia (paralysis of both legs). In Australia, for example, diving incidents account for approximately 20% of all cases of quadriplegia (Hill, 1984). The financial cost of these injuries to society is high, because those affected are frequently healthy younger persons—principally males under 25 years (Blanksby et al., 1997)—and treatment of persons with spinal injuries can be very expensive.

2.2.1 Contributory factors

Data from the USA suggest that body surfing at a beach and striking the bottom was the most common cause of aquatic spinal injury. Ten per cent of spinal injuries occurred when people dived into water, particularly from high platforms, including

TABLE 2.1. DROWNING AND NEAR-DROWNING: PRINCIPAL CONTRIBUTORY FACTORS AND PREVENTIVE AND MANAGEMENT ACTIONS

Contributory factors

- Alcohol consumption
- Cold
- Current (including rip currents, river currents, and tidal currents)
- Offshore winds (especially with flotation devices)
- Ice cover
- Pre-existing disease
- Underwater entanglement
- Bottom surface gradient and stability
- Waves (coastal, boat, chop)
- Water transparency
- Impeded visibility (including coastal configuration, structures and overcrowding)
- Lack of parental supervision (infants)
- Poor or inadequate equipment (e.g. boats or lifejackets)
- Overloading of boats
- Overestimation of skills
- Lack of local knowledge

Preventive and management actions

- Public education regarding hazards and safe behaviours
- Regulations that discourage unsafe behaviours (e.g., exceeding recommended boat loadings)
- Continual adult supervision (infants)
- Restriction of alcohol provision
- Provision of properly trained and equipped lifeguards
- Provision of rescue services
- Access to emergency response (e.g., telephones with emergency numbers)
- Local hazard warning notices
- Availability of resuscitation skills/facilities
- Development of rescue and resuscitation skills among general public and user groups
- Coordination with user group associations concerning hazard awareness and safe behaviours
- Wearing of adequate lifejackets when boating

trees, balconies and other structures. Special dives such as the swan or swallow dive are particularly dangerous, because the arms are not outstretched above the head but to the side (Steinbruck & Paeslack, 1980). There is no evidence to suggest that impact upon the water surface gives rise to serious (spinal) injury (Steinbruck & Paeslack, 1980). Alcohol consumption may contribute significantly to the frequency of injury through diminished awareness and information processing (Blanksby et al., 1997).

Minimum depths for safe diving are greater than frequently perceived, but the role played by water depth has not been conclusively ascertained. Inexperienced or unskilled swimmers require greater depths for safe diving. The velocities reached from ordinary dives are such that sight of the bottom even in clear water may provide an inadequate time for deceleration response (Yanai & Hay, 1995). Most diving injuries occur in relatively shallow water (1.5 m or less) and few in very shallow water (e.g., less than 0.6 m), where the hazard may be more obvious (Gabrielsen, 1988; Branche et al., 1991). Familiarity with the water body is not necessarily protective. In a study from South Africa (Mennen, 1981), it was noted that the typical injurious dive is into a water body known to the individual.

Data from the Czech Republic suggest that spinal injuries are more frequently sustained in open freshwater recreational water areas than in supervised swimming areas, although the number of injuries sustained in freshwater areas in this country appears to be declining (EEA/WHO, 1999).

A proportion of spinal injuries will lead to death by drowning. While data on this are scarce, it does not appear to be a common occurrence (see, for example, EEA/WHO 1999 regarding Portugal). In other cases, the act of rescue from drowning may give rise to spinal cord trauma after the initial impact (Mennen, 1981; Blanksby et al., 1997).

2.2.2 Preventive and management actions

Technique and education appear to be important in injury prevention (Perrine et al., 1994; Blanksby et al., 1997), as are preventive programmes. In Ontario, Canada, for example, preventive programmes established by Sportsmart Canada and widespread education decreased the incidence of water-related injuries substantially between 1989 and 1992 (Tator et al., 1993).

Because of the young age of many injured persons, awareness raising and education regarding safe behaviours are required early in life. Many countries have school-age swimming instruction that may inadequately stress safe diving, but which may also provide a forum for increasing public safety (Damjan & Turk, 1995). Education and awareness raising appear to offer the best potential for diving injury prevention, in part because people have been found to take little notice of signs and regulations (Hill, 1984). This is not to suggest that signs should not be utilized, but that both education and signage may provide significant benefits.

The principal contributory factors and preventive and management actions for spinal cord injury are summarized in Table 2.2.

TABLE 2.2. SPINAL CORD INJURY: PRINCIPAL CONTRIBUTORY FACTORS AND PREVENTIVE AND MANAGEMENT ACTIONS

Contributory factors
• Alcohol consumption • Diving into water of unknown depth • Bottom surface type • Water depth • Lack of adult supervision • Conflicting uses in one area • Diving into water from trees/balconies/structures • Poor underwater visibility
Preventive and management actions
• Local hazard warnings and public education • General public (user) awareness of hazards and safe behaviours, including use of signs • Early education in diving hazards and safe behaviours • Restriction of alcohol provision • Use separation • Lifeguard supervision • Emergency services, access

2.3 Brain and head injuries

Concussions, brain injury and skull/scalp abrasions have occurred through beach and aquatic recreational activities such as diving into shallow water. The contributory factors and preventive and management actions are similar to those for spinal injuries and for limb and minor impact injuries and are summarized in Table 2.2 and Table 2.3.

TABLE 2.3. FRACTURES, DISLOCATIONS AND OTHER IMPACT INJURIES: CONTRIBUTORY FACTORS AND PRINCIPAL MANAGEMENT ACTIONS

Contributory factors
• Diving into shallow water • Underwater objects (walls, piers) • Poor underwater visibility • Adjacent surface type (e.g., of water fronts and jetties) • Conflicting uses in one area
Preventive and management actions
• General user awareness of hazards and safe behaviours • Appropriate surface type selection • Adjacent fencing (e.g., of docks and piers) • Use separation • Lifeguard supervision • Warning signs

2.4 Fractures, dislocations and other impact injuries

Recreational water users have experienced injuries to the nose and jaw areas when swimming underwater, shallow diving or hitting underwater objects such as walls and piers or even other water users (depending upon the nature of the activity). These and other injuries have also been reported as a result of slipping, tripping or falling while entering or leaving the water. Injuries involving limb fractures or breaks of different types have many causes and may occur in a variety of settings in or around water. Broken bones (along with scarring, significant blood loss and amputation) have been reported as a result of injuries sustained from boat propellers (CDC, 2002), although it is not clear whether these were sustained from the boat or while in the water. The principal contributory factors and preventive and management actions associated with fractures, dislocations and other impact injuries are summarized in Table 2.3.

2.5 Cuts, lesions and punctures

There are many reports of injuries sustained as a result of stepping on glass, broken bottles and cans. Discarded syringes and hypodermic needles may present more serious risks (Philipp et al., 1995). Cuts and related injuries can also result from contact with shells, corals and so on. In the case of injury from such objects, wound infection from, for example, *Vibrio* spp. or *Aeromonas* spp. may be an additional problem (see chapter 5). The use of footwear on beaches should be encouraged. Adequate litter bins and beach cleaning operations contribute to prevention. In some areas, syringe/sharp objects disposal bins may be appropriate. Education policies to encourage users to take their litter home are a key remedial measure (see Table 2.4). Banning the possession of glass containers (bottles, jars, etc.) in some beach areas has been found to reduce the likelihood of injuries from broken glass.

2.6 Interventions and control measures

The majority of injuries can be prevented by appropriate measures especially at a local level. A relatively low cost way of promoting aquatic safety is through public education before the visitor even sets foot on the beach. Once the visitor arrives at the beach, additional public education efforts can further enhance public safety. A variety of measures to increase public awareness can be employed and these are reviewed in Chapter 13 (see 13.5).

At the beach, physical hazards should be removed or mitigated if possible, or measures should be taken to prevent or reduce human exposure. Physical hazards that cannot be completely dealt with in this way should be the subject of additional preventive or remedial measures—for example, open or rough water, rough waves, rip currents and bottom debris could all be the subject of general education, general warning notices or special warnings, especially at times of increased risk. It may be possible to rate recreational water areas according to certain characteristics, in order to provide objective, easily understandable information to the public. For example, a beach with a small tidal range, no sudden changes in water depth and so on might be rated as 'family friendly'. A river that is used for white water canoeing might be rated as 'suitable for beginners' under certain conditions or for 'experienced canoeists'

TABLE 2.4. CUTS, LESIONS AND PUNCTURES: PRINCIPAL CONTRIBUTORY FACTORS AND PREVENTIVE AND MANAGEMENT ACTIONS

Contributory factors
• Presence of broken glass, bottles, cans, medical wastes • Walking and entering water barefoot
Preventive and management actions
• Beach cleaning • Solid waste management • Provision of litter bins • Regulation (and enforcement) prohibiting glass containers • General public awareness regarding safe behaviours (including use of footwear) • General public awareness regarding litter control • Local first aid availability

in spate (flood) conditions. Such as system could complement the hazard ranking system outlined in section 2.7.1.

The term hazard is generally used in relation to the capacity of a substance or event to adversely affect human health (see 1.5). In this context, the absence of appropriate control measures may be treated as a component in the chain of causation. For example, the lack of lifeguards, rescue equipment, signs and other remedial actions can contribute to a variety of negative health outcomes.

2.6.1 Lifeguarding

At many coastal and fresh water beaches, people known as lifeguards or lifesavers protect recreational water users from injury and drowning. Depending upon local practice they may be volunteers or paid, or both. Here, the term "lifeguard" is used to refer to people trained and positioned at recreational water sites to protect the water user. Lifeguards, when adequately staffed, qualified, trained and equipped, seem to be an effective measure to prevent drowning. The report of a working group convened by the Centers for Disease Control and Prevention, USA states that "One effective drowning prevention intervention is to provide trained, professional lifeguards to conduct patron surveillance and supervision at aquatic facilities and beach areas" (Branche & Stewart, 2001).

Lifeguards can also assist in injury prevention (e.g., advising users not to enter dangerous areas, such as where a rip current is forming) and by playing a more general educational role (concerning water quality hazards and exposure to heat, cold or sunlight, for example). It has been estimated that lifeguards take 49 preventive actions for every rescue from drowning that they effect (USLA, 2002). According to Branche & Stewart (2001), "the presence of lifeguards may deter behaviours that could put swimmers at risk for drowning, such as horseplay or venturing into rough or deep water, much like increased police presence can deter crime". Further details on lifeguarding can be found in Appendix A.

2.6.2 Use separation

The waterfront may be used for diverse purposes, such as transit (pedestrian, vehicular), sunbathing, swimming, surfing, paddling, watercraft (yachts, powerboats, canoes, personal watercraft) and as a route of access, and the water itself may be used by both swimmers and non-swimmers. As a result of multiple and often dense use, conflicts may emerge, and in many cases zoning or other restrictions on certain uses may become necessary.

Use separation is a measure for minimizing risk where different user groups use the water in different ways within a confined area. Different zones are established for incompatible activities, including for example swimming, diving, sailboarding or powerboating, as well as for conservation and naturalist activities. Specific regulations may be identified for the use of surfboards or similar apparatus. For example, these may be banned within a distance of 70 m of any fishing pier or within 50 m of any swimmer, although this may be difficult to enforce.

At flat water beaches, lines, buoys and markers may be useful in limiting the water recreation area and separating different activities. Lines can also be used to prevent swimmers from entering dangerous areas, to warn of changing conditions or to indicate separation of shallow and deep areas, underwater obstructions, radical changes in slope, etc. The anchoring rope for buoys and markers should not create any risk of entanglement. The buoys are not intended as rest areas.

At coastal beaches where tide, current and wave action typically prevent the use of perimeter devices such as these, lifeguards may patrol and issue warnings or visual reference points onshore may help to keep activities in their proper areas.

It is of particular importance to separate boats from other water users, especially motorboats. If boat launching is to be permitted, special areas should be established that effectively separate it from zones for other uses. At the beachside warning signs and/or buoys should be provided. Boat lanes are generally perpendicular to the shoreline and delimited by floating lines. Boats should launch through this lane at a specified low speed—for example, not more than 3 knots. If boating areas are not delimited for all kind of boats (sailboats, powercraft and jet skis included), an exclusion zone may be defined—for example, in the 200-m zone.

2.6.3 Infrastructure and planning

Waterfront areas are accessed for a variety of purposes, some of which affect safety. Routes used for emergency access—for instance, during launching of rescue craft or to provide access to ambulances—should be suitably maintained, and continuous accessibility should be assured.

Ready access to telephones or other means of communication with emergency services may contribute to speed of rescue or resuscitation. Telephones should ideally be readily accessible and clearly visible, marked on local maps and posted with numbers of key emergency services.

In many recreational water use areas, certain locations or subareas may present significant continuous hazards to human health—for example, due to currents, weirs or rocks. Access to such areas may be discouraged or prevented by a combination of one or more interventions, such as signing, fencing and lifeguard supervision. In some instances, caution lines are used to discourage access, intentional or otherwise, by water users.

In areas with or without lifeguards, rescue equipment may be provided that is accessible for public use. All such safety equipment should be clearly visible from a distance and kept in good repair. Location intervals should be determined according to the response required for a given water recreation area. Public rescue equipment should normally be kept in place year-round.

2.6.4 Beach capacity

It has been suggested that recreational water areas should have an estimated load (number of bathers/visitors) that they may carry safely. While overcrowding may impede effective lifeguarding and therefore contribute to drowning, in practice this is difficult to enforce, and user needs and perceptions vary considerably between areas. Of more importance is the adequate management of the recreational water use area in order to minimize risk.

2.7 Monitoring and assessment

2.7.1 Assessing hazards

Physical characteristics that may present hazards to recreational water users consist of five interrelated phenomena, four of which are common to most coastal beaches:

- water depth, particularly when greater than chest deep;
- variable beach and surf zone topography, such as tides, bars, channels and troughs;
- breaking waves;
- surf zone currents, particularly rip currents; and
- localized hazards, such as reefs, rocks, offshore platforms, inlets, offshore winds, tidal currents, cold water, kelp beds, weirs and locks. The construction of jetties, piers, wharfs and other artificial structures can also contribute to the hazard.

The assessment of hazards in a beach or water environment is critical to ensuring safety. The assessment should take into account several key considerations, including:

- the presence and nature of natural or artificial hazards;
- the severity of the hazard characteristic as related to health outcomes;
- the ease of access to the recreational water area;
- the availability and applicability of remedial actions;
- the frequency and density of use; and
- the level of development for recreational use.

As outlined in chapters 1 and 4, health risks that might be tolerated for an infrequently used and undeveloped recreational area may result in immediate remedial measures at other areas that are widely used or highly developed.

Potential health outcomes associated with various hazards are summarized in Tables 2.1–2.4. The severity of the outcomes associated with a hazard can be related to the relative risk in Figure 1.2 and can serve as a tool to highlight or emphasize priority protective or remedial management measures and to initiate further research or investigation into the reduction of risk.

The hazard assessment could lead to a 'hazard rating'. Short (2003) outlines a beach hazard rating based on the physical characteristics of a beach (i.e., whether they are wave dominated, tide-modified or tide dominated). The resulting classification consists of a general beach hazard rating and a prevailing beach hazard rating, which depends on prevailing wave, tide and wind conditions. Such as rating could be expanded to include other hazards. This could form the basis for developing a safety plan, detailing the level of resources required to reduce the level of risk.

2.7.2 Inspection programmes and protocols

Inspection of a site for existing and new hazards should be undertaken on a regular basis in order to promote remedial action if required.

The inspection protocol for a recreational area, in terms of injury hazards may comprise the following:

1. Determining what is to be inspected and how frequently.
2. Monitoring changing hazards and use patterns periodically.
3. Establishing a regular pattern of inspection of conditions and controls.
4. Developing a series of checklists suitable for easy application. Checklists should reflect national and local standards where they exist.
5. Establishing a method for reporting faulty equipment and maintenance problems.
6. Developing a reporting system that will allow easy access to statistics regarding "when", "where", "why" and "how" questions needing answers.
7. Motivating and informing participants in the inspection process through in-service training.
8. Use of outside experts to critically review the scope, adequacy and methods of the inspection programme.

The frequency of inspection will vary according to the size of the recreational water area, the number of features, the density of use, the speed of change in both the hazards encountered and the remedial actions in place at a specific location, and the extent of past incidents or injuries. Timing of inspections should take account of periods of maximum use (e.g., inspection in time to take remedial action before major use periods) and periods of increased risk.

The criteria for inspections and investigations may vary from country to country. In some countries, there might be legal requirements and/or voluntary standard-setting organizations.

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CHAPTER 3

Sun, heat and cold

The recreational use of water environments may be associated with extreme temperature conditions. People may be unintentionally exposed to cold water (<15°C) which can result in a debilitating shock response. Or, at the other extreme, high air temperatures may result in heat stroke. As people engage in outdoor activities and recreation by the side of a lake or at a beach, they are often exposed to high levels of ultraviolet radiation (UVR) from the sun for prolonged periods of time. UVR can cause both acute and long-term damage to health. UVR and temperature deserve particular attention, as global climate change and ozone depletion are likely to aggravate existing health risks.

3.1 Exposure to ultraviolet radiation

3.1.1 UVR and ozone depletion

Everybody is exposed to UVR from the sun, and an increasing number of people are exposed to artificial sources used in industry, commerce and recreation. Emissions from the sun include visible light, heat and UVR. The UVR region covers the wavelength range 100–400 nm and consists of three bands—UVA (315–400 nm), UVB (280–315 nm) and UVC (100–280 nm). As sunlight passes through the atmosphere, all UVC and approximately 90% of UVB is absorbed by ozone, water vapour, oxygen and carbon dioxide. UVA is less affected by the atmosphere, and almost 70% of the solar emission in this band reaches the Earth's surface. Therefore, the UVR reaching the Earth's surface is largely UVA, with a small UVB component (usually less than 3%).

The amount of UVR at the Earth's surface is influenced by:

- *sun height*: The higher the sun in the sky, the higher the UVR level. Thus, UVR levels vary with time of day and time of year, with maximum levels occurring when the sun is directly overhead. Outside the tropics, the highest levels occur when the sun is at its maximum elevation, at around midday (solar noon) during the summer months.
- *latitude*: The closer the equator, the higher the UVR levels.
- *cloud cover*: UVR levels are highest under cloudless skies. Even with some cloud cover, UVR levels can be high as long as the solar disc is unobscured.
- *altitude*: At higher altitudes, a thinner atmosphere scatters and absorbs less UVR. With every 1000-m increase in altitude, UVR levels increase by approximately 14% (Blumthaler et al., 1994).

- *ground reflection*: UVR is reflected or scattered to varying extents by different surfaces. For example, fresh snow can reflect as much as 80% of UVR, sea foam about 25%, water up to about 15% (depending on the elevation of the sun) and dry beach sand about 15%.

As the ozone layer becomes depleted, the protective filter provided by the atmosphere is progressively reduced. Consequently, human beings and the environment are exposed to higher UVR levels, in particular higher UVB levels. Within the UV region, UVB has the greatest impact on human health, animals, marine organisms and plant life. International treaties to protect the ozone layer, such as the Vienna Convention (1985) and the Montreal Protocol on Substances That Deplete the Ozone Layer (1987), have gradually phased out the production of ozone-depleting substances; their ozone-depleting potential is expected to reach its maximum between 2000 and 2010. Due to the time delays in atmospheric processes, stratospheric ozone depletion will persist for many years (EEA, 1998), and the corresponding increases in UVR reaching the Earth's surface will exacerbate adverse health effects in all populations of the world. Computational models predict that a 10% decrease in stratospheric ozone could cause an additional 300 000 non-melanoma skin cancers (NMSCs), 4500 melanomas and between 1.6 and 1.75 million cataracts worldwide every year (UNEP, 1991).

3.1.2 Health outcomes

Overexposure to solar UVR may result in acute and chronic health effects on the skin, eye and immune system. Chronic effects include two major public health problems: skin cancers and cataracts. Furthermore, a growing body of evidence suggests that current environmental levels of UVR may increase the risk of infectious diseases and limit the efficacy of vaccinations (Halliday & Norval, 1997; Duthie et al., 1999). A comprehensive review and summary of UVR-related health effects can be found in the WHO Environmental Health Criteria monograph *Ultraviolet Radiation* (WHO, 1994) and in the United Nations Environment Programme report *Environmental Effects of Ozone Depletion* (UNEP, 1998).

1. Beneficial effects of UVR

Exposure to UVB stimulates the production of vitamin D in the skin. It has been estimated that more than 90% of vitamin D requirement is satisfied by this exposure and less than 10% from diet. Vitamin D has an important function in increasing calcium and phosphorus absorption from food and plays a crucial role in skeletal development, immune function and blood cell formation. Vitamin D deficiency is unlikely for most people, as, for example, a 10- to 15-min daily exposure of face, forearms and hands to normal northern European summer sun is sufficient to maintain vitamin D levels (McKie, 2000). An exception to this would be for people residing at high latitudes, where UVB levels in winter would be very low.

UVR from artificial sources is used to treat several diseases and dermatological conditions, including rickets, psoriasis, eczema and jaundice. While sunbed use for cosmetic purposes is not recommended (EUROSKIN, 2000), therapeutic treatment takes place under medical supervision, and the beneficial effects of therapeutic UVR exposure usually outweigh the harmful side-effects.

2. *Adverse effects of UVR on the skin*

The widespread perception that a tan is healthy and beautiful has led many people to actively seek a tan and expose their skin to excessive levels of UVR. This attitude, changed clothing habits, the popularity of outdoor activities and frequent holidays in sunny locations seem to be the major causes for the dramatic rise in skin cancer rates in all fair-skinned populations. Between 2 and 3 million NMSCs (WHO, Department of Evidence and Information for Policy, unpublished data) and 132 000 melanoma skin cancers (Ferlay et al., 2001) occur globally each year, according to WHO estimates. Since the early 1960s, the incidence of skin cancers has increased by between 3% and 7% in most fair-skinned populations (Armstrong & Kricker, 1994).

The sensitivity of skin to UVR is usually defined by six phototypes (Fitzpatrick et al., 1974). A more recent classification scheme relates skin type to short-term and long-term effects of UVR, based on the finding that 85–90% of registered skin cancers are found in the melano-compromised skin types I and II, while most of the remaining skin cancers are found in the melano-competent skin types III and IV (Fitzpatrick & Bolognia, 1995). These categories relate to the presence of melanin pigment in the epidermis, which determines human skin colour. Melanin absorbs UVR and in this way provides protection against exposure to UVR. Both classification schemes are depicted in Table 3.1.

TABLE 3.1. CLASSIFICATION OF SKIN TYPES[a]

Skin type classification		Burns in the sun	Tans after having been in the sun
I	Melano-compromised	Always	Seldom
II		Usually	Sometimes
III	Melano-competent	Sometimes	Usually
IV		Seldom	Always
V	Melano-protected	Naturally brown skin	
VI		Naturally black skin	

[a] Adapted from Fitzpatrick & Bolognia, 1995.

Many believe that only fair-skinned people need to be concerned about overexposure to the sun. Darker skin has more protective melanin pigment, and the incidence of skin cancer is lower in dark-skinned people. Nevertheless, skin cancers do occur and are often detected at a later, more dangerous stage. The risk of UVR-related health effects on the eye and immune system (see below) is independent of skin type (Vermeer et al., 1991).

Children are at a higher risk of suffering damage from exposure to UVR than adults, in particular because of the following:

- A child's skin is thinner and more sensitive, and even a short time outdoors in the midday sun can result in serious burns.
- Epidemiological studies demonstrate that frequent sun exposure and sunburn in childhood set the stage for high rates of melanoma in later life (IARC, 1992).
- Children have more time to develop diseases with long latency, more years of life to be lost and more suffering to be endured as a result of impaired health. Increased life expectancy further adds to people's risk of developing skin cancers.
- Children are more exposed to the sun. Estimates suggest that up to 80% of a person's lifetime exposure to UVR is received before the age of 18 (Marks et al., 1990; Wakefield & Bonett, 1990).
- Children love playing outdoors but usually are not aware of the harmful effects of UVR.

The most noticeable acute effect of excessive UVR exposure is erythema, the familiar inflammation of the skin commonly termed sunburn. The symptoms of a mild sunburn are reddening of the skin caused by vascular dilatation and some swelling, while in severe cases the skin will blister. In addition, most people will tan from darkening of existing melanin or through the UVR stimulation of melanin production, which occurs within a few days following exposure. A further, less obvious adaptive effect is the thickening of the outermost layers of the skin that attenuates UVR and decreases the penetration to the deeper layers in the skin. Current estimates suggest that a suntan can offer a sun protection factor (SPF) of between 2 and 3 (Young & Sheehan, 2001). Depending on their skin type, individuals vary greatly in their skin's initial threshold for erythema and their ability to adapt to UVR exposure (see Table 3.1).

Chronic exposure to UVR also causes a number of degenerative changes in the cells, fibrous tissue and blood vessels of the skin. These include freckles, naevi (moles) and lentigines, which are pigmented areas on the skin, and diffuse brown pigmentation. UVR accelerates skin aging, and the gradual loss of the skin's elasticity results in wrinkles and dry, coarse skin.

NMSCs comprise basal cell carcinoma (BCC) and squamous cell carcinoma (SCC). BCC is the commonest but rarely fatal, while SCC can metastasize and be fatal if left untreated. Surgical treatment for NMSC can be painful and is often disfiguring. The temporal trends of NMSC incidence are difficult to determine, because registration of these cancers has not been achieved. However, specific studies carried out in the USA, Australia and Canada indicate that between the 1960s and the 1980s, the prevalence of NMSC has increased by a factor of more than two.

Malignant melanoma (MM), although far less prevalent than NMSC, is the major cause of death from skin cancer and is more likely to be reported and accurately diagnosed than NMSC. Since the early 1970s, MM incidence has increased significantly—for example, an average 4% every year in the USA (American Cancer Society, 2000). A large number of studies indicate that the risk of MM correlates with genetic and personal characteristics and a person's UVR exposure behaviour. The following is a summary of the main human risk factors (WHO, 1994):

- A large number of atypical naevi (moles) is the strongest risk factor for MM in fair-skinned populations.
- MM is more common among people with a pale complexion, blue eyes and red or fair hair. Experimental studies have demonstrated a lower minimum erythema dose and more prolonged erythema in melanoma patients than in controls.
- High, intermittent exposure to solar UVR appears to be a significant risk factor for the development of MM.
- The incidence of MM in white populations generally increases with decreasing latitude, with the highest recorded incidence occurring in Australia, where the annual rates are 10 and over 20 times the rates in Europe for women and men, respectively.
- Several epidemiological studies support a positive association with history of sunburn, particularly sunburn at an early age.
- The role of cumulative sun exposure in the development of MM is equivocal. However, MM risk is higher in people with a history of NMSC and of solar keratoses (areas of skin marked by overgrowth of horny tissue), both of which are indicators of cumulative UVR exposure.

3. *UVR effects on the eye*

The eye is recessed within the anatomy of the head and shielded by the brow ridge, the eyebrows and the eyelashes. Bright visible light activates the constriction of the pupil and the squinting reflex to minimize the penetration of the sun's rays into the eye. However, the effectiveness of these natural defences in protecting against UVR exposure is limited under certain conditions, such as sunbed use or strong ground reflection from fresh snow and sometimes sand and water.

Acute effects of UVR exposure on the eye include photokeratitis and photoconjunctivitis. These inflammatory reactions are comparable to a sunburn of the very sensitive skin-like tissues of the eyeball and eyelids and usually appear within a few hours of exposure. Both can be very painful but are reversible and do not result in any long-term damage to the eye or vision. An extreme form of photokeratitis is snow blindness.

Sun exposure, in particular exposure to UVB, also appears to be a major risk factor for cataract development, although cataracts appear to different degrees in most individuals as they age. Cataracts occur when proteins in the eye's lens unravel, tangle and accumulate pigments that cloud the lens and eventually lead to blindness. They are the leading cause of blindness in the world, affecting some 12–15 million people. According to WHO (1994) estimates, up to 20% of cases of cataract-related blindness may be caused or enhanced by sun exposure, especially in India, Pakistan and other countries of the "cataract belt" close to the equator. As the world's population ages, cataract-induced visual dysfunction and blindness are on the increase; reducing ocular exposure to UVR and smoking prevention are the only interventions that can reduce risk of developing cataracts (Brian & Taylor, 2001).

4. *UVR effects on the immune system*

The immune system is the body's defence mechanism against infections and cancers and is normally very effective at recognizing and responding to an invading microorganism or the onset of a tumour. Although the data remain preliminary, there is increasing evidence for an immunosuppressive effect of both acute high-dose and chronic low-dose UVR exposure on the human immune system (Duthie et al., 1999).

Animal experiments have demonstrated that UVR can modify the course and severity of skin tumours (Fisher & Kripke, 1977). Also, people treated with immunosuppressive drugs have a greater incidence of SCC than the normal population. Consequently, beyond its role in the initiation of skin cancer, sun exposure may reduce the body's defences, which normally limit the progressive development of skin tumours.

Several studies have demonstrated that exposure to current environmental levels of UVR alters the activity and distribution of some of the cells responsible for triggering immune responses in humans. Consequently, sun exposure may enhance the risk of disease resulting from viral, bacterial, parasitic or fungal infections and may modify the course of disease progression in both animals and humans (Halliday & Norval, 1997; Yamamoto et al., 1999, 2000). Furthermore, especially in countries of the developing world, high UVR levels may reduce the effectiveness of vaccines.

3.1.3 Interventions and control measures

Damage from UVR to the skin, eyes and immune system is mostly preventable. Reducing both the occurrence of sunburn and cumulative UVR exposure can decrease harmful health effects and significantly reduce health care costs.

Spending a sunny day at the beach, sunbathing by the side of a lake or engaging in different kinds of water sports frequently lead to prolonged exposure to UVR, often including the time of day when UVR levels are highest. This can be exacerbated by the lack of shade and reflection of the sun's rays by water and sand—both can reflect up to about 15% of incident UVR. Sun protection consideration should take these particular environmental conditions into account.

1. *Personal protection against UVR*

It is the individual's choice as to whether to adopt sun protection or not. Simple protective measures are available and should be adopted to avoid adverse health effects on the skin, eyes and immune system caused by sun exposure (Box 3.1).

Children require special protection, as they are at a higher risk of suffering damage from exposure to UVR than adults. Encouraging children to take simple precautions will prevent both short-term and long-term damage while still allowing them to enjoy the time they spend outdoors. Shade, clothing and hats provide the best protection

BOX 3.1 EXAMPLE OF ACTION STEPS FOR SUN PROTECTION (US EPA, 2000)

LIMIT TIME IN THE MIDDAY SUN

The sun's UV rays are the strongest between 10 a.m. and 4 p.m. To the extent possible, limit exposure to the sun during these hours.

WATCH FOR THE UV INDEX

This important resource helps you plan your outdoor activities in ways that prevent overexposure to the sun's rays. While you should always take precautions against overexposure, take special care to adopt sun safety practices when the UV index predicts exposure levels of moderate or above.

USE SHADE WISELY

Seek shade when UV rays are the most intense, but keep in mind that shade structures such as trees, umbrellas or canopies do not offer complete sun protection. Remember the shadow rule: "Watch your shadow—No shadow, seek shade!"

WEAR PROTECTIVE CLOTHING

A hat with a wide brim offers good sun protection for your eyes, ears, face and the back of your neck. Sunglasses that provide 99–100% UVA and UVB protection will greatly reduce eye damage from sun exposure. Tightly woven, loose-fitting clothes will provide additional protection from the sun.

USE SUNSCREEN

Apply a sunscreen of SPF 15 or more liberally and reapply every 2 h, or after working, swimming, playing or exercising outdoors.

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for children; applying sunscreen becomes necessary on those parts of the body that remain exposed, like the face and hands. Infants of less than 12 months should always be kept in the shade.

2. *Information and education*

Information should be provided to the public on UVR and variation in UVR levels with time of day, time of year and geographical location; health effects of sun exposure on the skin, eyes and immune system; and available protective measures (see Box 3.1).

The global solar UV index (International Commission on Non-Ionizing Radiation Protection, 1995; WHO, 2002) is an important vehicle to raise public awareness of UVR and the risks of excessive UVR exposure and to alert people about the need to adopt protective measures (see Box 3.2). The UV index describes the level of solar UVR at the Earth's surface. The values of the index range from zero upward. The higher the index value, the greater the potential for skin and eye damage following exposure to UVR, and the less time it takes for harm to occur.

A standard graphic representation of UV index has been proposed (WHO, 2002) in order to promote consistency in reporting and improve understanding of the

BOX 3.2 GLOBAL SOLAR UV INDEX (WHO, 2002)

UV index values are grouped into exposure categories:

UV index values	Exposure category	Level of sun protection required	'Sound bite' messages
≤2	Low	None required	You can safely stay outside
3–5	Moderate	Protection required	Seek shade during midday hours. Slip on a shirt, slop on sunscreen and slap on a hat.
6–7	High		
8–10	Very high	Extra protection required	Avoid being outside during midday hours. Make sure you seek shade. Shirt, sunscreen and hat are a must
11+	Extreme		

Even for very sensitive, fair-skinned people, the risk of short-term and long-term UVR damage below a UV index of 3 is limited, and under normal circumstances no protective measures are needed. Above the threshold value of 3, protection is necessary and should include all protective means available. At the very high or extreme exposures of UV index values of 8 and above, this message must be reinforced, and people should be encouraged to use more sun protection and avoid being outdoors during midday hours.

concept. Ready made materials, such as those shown in Figure 3.1, are available from WHO.

The levels of UVR and therefore the values of the index vary throughout the day. In reporting the UV index, most emphasis is placed on the maximum UVR level on a given day. This generally occurs during the 4-h period around solar noon. Depending on geographical location and whether daylight saving time is applied, solar noon takes place between noon and 2 p.m. The media usually present a forecast of the maximum UVR level for the following day. In many countries, the UV index is reported along with the weather forecast in newspapers, on TV and on the radio. However, this reporting usually occurs during the summer months only, unless the location lies within or close to the tropics.

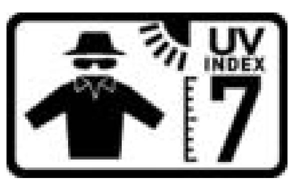

FIGURE 3.1. UVI GRAPHIC REPRESENTATION (WHO, 2002)

Information relating to the UV index should be especially targeted at vulnerable groups within the population, such as children and tourists, and should inform people about the range of UVR-induced health effects, including sunburn, skin cancer and skin aging and effects on the eye and immune system. Daily UV index levels can easily be presented on signboards and signposts on beaches or by the side of lakes and should be accompanied by a simple message encouraging people to adopt sensible sun behaviour. Furthermore, simple flyers could be made available free of charge at the entrance, at cashiers or at kiosks. Lifeguards, first aid providers and other employees should be educated about UVR and sun protection and trained to act as role models for the users of recreational water environments, in particular children. They may help to disseminate information about the dangers of UVR and could be important partners for organizations that are planning to hold educational events or skin cancer screening initiatives. Further information can be found on the website of WHO's INTERSUN Programme (http://www.who.int/peh/uv/).

Public education aims to improve people's knowledge about the health risks of excessive sun exposure and to achieve a change in attitudes and behaviour. Educational activities in the context of recreational water environments should mainly address children, adolescents and their parents. The best way of generating interest is through activities and games. The main message should be that the enjoyment of outdoor sport and recreation activities should not be compromised, but can even be enhanced by sun-protective behaviour. Sensible behaviour is relatively simple to incorporate and can eliminate sunburn and heat stroke, which are often associated with being in the sun for prolonged periods of time.

Reducing the occurrence of sunburn and cumulative UVR exposure during childhood and over a lifetime will eventually cause skin cancer rates to decline. It has been estimated that four out of five cases of skin cancer could be prevented by sensible behaviour (Stern et al., 1986). An effective campaign can have an enormous impact on public health: it has been estimated that the regular use of sunscreen with SPF 15+ up to the age of 18 could decrease the frequency of skin cancer in Australia by more than 70% (Stern et al., 1986).

The SunSmart Campaign of the Anti-Cancer Council of Victoria, Australia (Anti-Cancer Council of Victoria, 1999b), has made significant achievements in raising awareness of the issues of sun protection and skin cancer as well as encouraging changes in sun-related behaviour. Evaluations of the programme show that fewer people see tanning as desirable or attractive and more people wear hats, use sunscreen and cover up to avoid the sun. Most significantly, research during the 1990s revealed an 11% decrease in the incidence of common skin cancers in 14- to 50-year-old people (Staples et al., 1998).

Low-cost interventions can significantly decrease costs to the healthcare system. Skin cancer is the most costly of all cancers to the Australian health system. The direct costs of treatment have been estimated at US$5.70 per head per annum, while the cost of prevention campaigns has been calculated at US$0.08 per head per annum. Assuming that a 20-year prevention campaign costing US$0.17 per head per annum reduced UVR exposure by 20%, that melanoma rates began to fall after a 5-year lag and that NMSCs and solar keratoses began to fall after a 15-year lag, the predicted annual saving would be US$0.17 per person (Carter et al., 1999). This would mean that every dollar spent would save a dollar.

In order to change people's sun exposure habits and the societal view that associates a tan with good health and beauty, long-term strategies are required. To create a supportive environment for the integration of sun protection considerations in the use of recreational water environments, it is important to establish working relationships with authorities and organizations such as community health services, sporting clubs, skin cancer associations, and the service and tourism sector, including public transport services and restaurants. Efforts should be made to learn from and integrate existing community initiatives to promote sun protection in the planning and implementation process.

3. *Infrastructure and planning*
While it is the decision of the individual as to whether to adopt sensible sun behaviour or not, the management of recreational water environments has the responsibility to facilitate a positive choice through adequate structural and policy measures. One important infrastructural consideration is the provision of shade structures and the integration of shaded areas in the vicinity of recreational water bodies. This is especially important in areas where a lot of people congregate—e.g., at a snack stand where people may queue for prolonged periods of time. Shade can be either permanent or portable and can come from natural sources such as trees or hedges or from artificial structures such as gazebos, canopies or shelter sheds. An inexpensive means of shade provision is to give users of recreational water environments the opportunity to hire portable parasols at low cost.
An example of a guide for local government is shown in Table 3.2.

3.2 Exposure to cold

Cold water removes heat from the body 25 times faster than cold air. The immediate effects of sudden immersion in cold water (<15 °C) can be a debilitating, short duration (approximately 2–3 minutes), reflex response called cold shock. This response includes life-threatening respiratory and cardiovascular effects. The respiratory effect involves quick onset (less than 30 seconds) uncontrollable rapid breathing, which impairs breath-holding and facilitates aspiration of water (which can lead to drowning). The cardiovascular response involves an immediate constriction (closure) of the blood vessels near the surface of the body, an increase in heart rate and a surge in blood pressure. These factors may lead to incapacitation from a cardiovascular incident, such as heart attack or stroke and/or death from drowning following aspiration (Golden & Tipton, 2002; International Life Saving Federation, 2003).

If sudden immersion in cold water does not cause death immediately, the related effects will impair swimming ability. Research has shown that even strong swimmers can experience difficulty and drown within minutes of cold-water immersion unless they are habituated to cold (Golden & Hardcastle, 1982). These initial responses occur long before body temperature begins to fall and are believed to be responsible for the majority of sudden cold-water immersion deaths.

TABLE 3.2. AN EXAMPLE OF A RECREATIONAL FACILITY SUN PROTECTION GUIDE FOR LOCAL GOVERNMENT[a]

SunSmart components	Desirable actions/outcomes
Education	• Erect signage about the importance of sun protection • Ensure that employees are role models for users of facilities • Conduct sun protection information sessions for employees • Ensure that sun protection information is available to patrons and clients
Clothing	• Ensure that employees wear broad-brimmed hats, sunglasses and long-sleeved shirts on patrol • Sell broad-brimmed hats in kiosks
Sunscreen	• Sell low-priced (or subsidized) SPF 30+[b] broad-spectrum, waterproof sunscreen • Provide employees with SPF 30+ broad-spectrum sunscreen
Shade	• Review available shade at local government recreational facilities • Ensure that sufficient shade, either natural or built, is available or planned for when developing new recreational facilities or centres • Investigate the opportunities to make available portable shade structures to schools and organizations using local government-controlled facilities
Schedules	• Allow users to leave in the middle of the day and then return without extra cost
Policy guidelines	• Change any rules (e.g., clothing restrictions for employees or patrons) that prevent people from being adequately protected • Adopt the SunSmart Sport or Pool Policy available from the Anti-Cancer Council of Victoria, Australia • Promote and encourage schools or sporting clubs using local government facilities to introduce a sun protection policy of their own • Ensure that no facilities that increase the risk of skin cancer operate within local government recreation facilities (e.g., solariums)

[a] Source: Anti-Cancer Council of Victoria (1999a).

[b] IARC (2001) recommends the use of sunscreen with an SPF of 15 or higher. In geographical locations where UVR levels are always high, such as Australia, a sunscreen with SPF 30+ may be necessary.

After about three minutes, the initial effects of sudden cold-water immersion decline. Thereafter, progressive whole-body cooling occurs, leading to a gradual fall in deep body temperature—hypothermia. Before a significant level of hypothermia develops, however, there is a progressive cooling of the muscles and joints in the exposed limbs resulting in shivering and stiffening. This impairs locomotion and thus swimming performance (Tipton et al., 1999), which will likely lead to drowning before a life-threatening level of hypothermia develops—unless the victim is wearing a lifejacket or personal flotation device (PFD) capable of keeping the airway clear of the water. This impairment of locomotion also impedes the person's ability to assist in his or her own rescue effort.

For those wearing a proper lifejacket, drowning may be prevented; however, without timely rescue, hypothermia will eventually lead to loss of consciousness and death from cardiac arrest (Golden, 1973). Time to death in such victims will be influenced by body insulation (thickness of clothing worn and the amount of body fat, with men generally having less than women), age (young and elderly fair less well), water state (breaking waves increase the chances of water aspiration) and time to rescue. A person who has consumed alcohol will succumb to the effects of hypothermia more rapidly (Haight & Keatinge, 1973).

One very rare complication of contact with cold water is cold urticaria. This condition is an allergy-like reaction to contact with cold water, as well as other sources of cold (Bentley, 1993). Within minutes, the skin may become itchy, red and swollen. Fainting, very low blood pressure and shock-like symptoms can present.

Prevention is the best cure. Attempts should not be made to swim in cold water unless habituated to it or wearing suitable protective garments (such as a wet suit or survival suit). If at risk of immersion, precautions should be taken against becoming immersed (such as by use of a safety line). On boats, suitable clothing and a proper lifejacket (with sufficient buoyancy to keep the airway clear of the water even when unconscious) should be worn.

3.3 Exposure to heat

Human body temperature is maintained within a narrow range, despite extremes in environmental conditions and physical activity. In healthy individuals, an efficient heat regulatory system will normally enable the human body to cope effectively with a moderate rise in ambient temperature. Within certain limits of mild heat stress and physical activity, thermal comfort can be maintained. In extreme temperatures, the human body is able to react with a series of adaptation mechanisms. The most significant are sweating, dilatation of the peripheral blood vessels, an increase in some hormones (antidiuretic hormone and aldosterone) and an increase in respiratory rate and pulse. In the meantime, the body tries to lose as few salts as possible and decreases the blood flow to the kidney.

Heat acclimatization usually takes from 7 to 14 days, but complete acclimatization to an unfamiliar thermal environment may take several years (Babayev, 1986; Frisancho, 1991). Acclimatization lowers the threshold for sweating, which is the most effective natural means of combating heat stress and can occur with little or no change in the body core temperature. As long as sweating is continuous, people can withstand remarkably high temperatures, provided water and sodium chloride (the most important physiological constituents of sweat) are replaced.

Disorders due to heat most frequently occur with rapid changes in thermal conditions, especially in low latitudes and in densely populated urban areas (Weiner, 1984; WHO, 1990). This was well illustrated by the 1980, 1983, 1988 and 1995 heat waves in the USA (CDC, 1995) and the 1987 heat wave in Athens, Greece (Katsouyanni et al., 1988, 1993). The following population groups seem to be disproportionally affected by such weather extremes, probably because they have a lesser physiological coping ability (CDC, 1995):

- the elderly;
- the very young (0–4 years);
- persons with impaired mobility;

- persons suffering from pre-existing chronic diseases (such as arteriosclerosis, previous heart failures, diabetes and congenital absence of sweat glands); and
- frequent consumers of alcohol (Schuman, 1964; Kilbourne, 1982).

A comfortable temperature for most people is around 20–28 °C. Factors influencing thermal comfort include air temperature, humidity, wind speed and fluxes in shortwave and longwave radiation. Under normal conditions, recreational water bodies may influence people's perception of ambient temperature conditions, such that a middle-aged person walking on the beach at midday copes better with heat exposure than the same person walking on an urban road at midday (Jendritzky et al., 1997).

In recreational water areas, steps that can be taken to reduce body temperature are similar to those for reducing exposure to sun and include wearing lightweight clothing and broad-brimmed hats, seeking shady areas and swimming in cool water. Other initiatives to help cope with exposure to heat include ensuring an adequate supply of safe drinking-water and replenishing any salt loss. Educating people with increased susceptibility to heat exposure (e.g., the elderly) would also be useful.

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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.

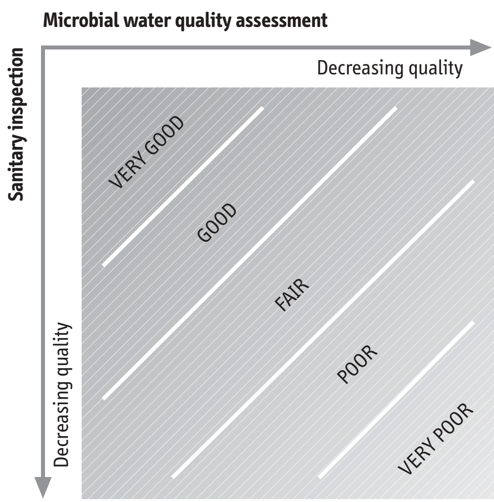


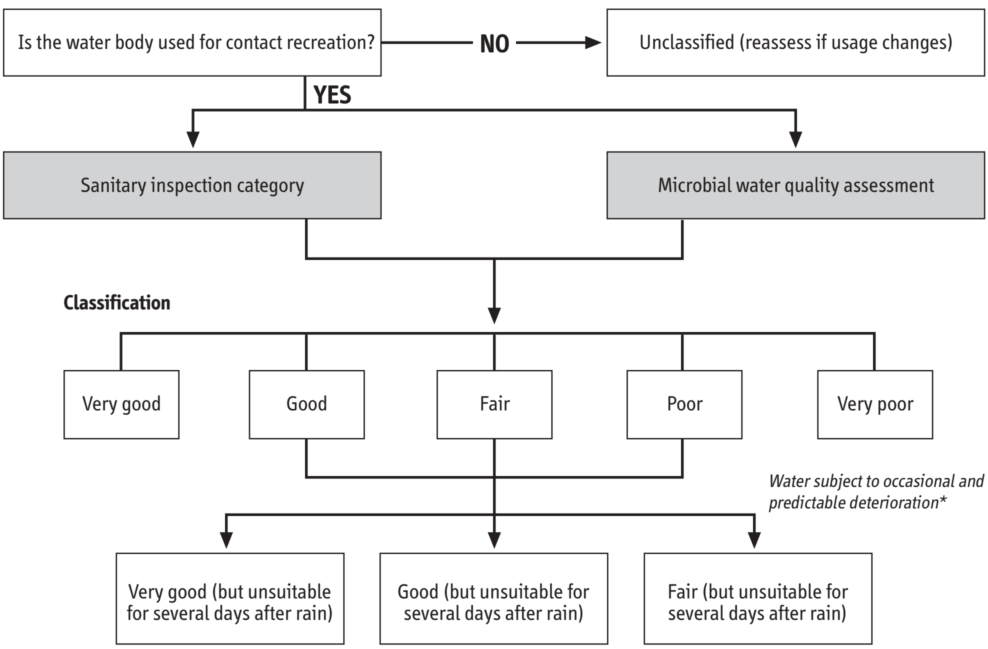
FIGURE 4.1. SIMPLIFIED CLASSIFICATION MATRIX

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

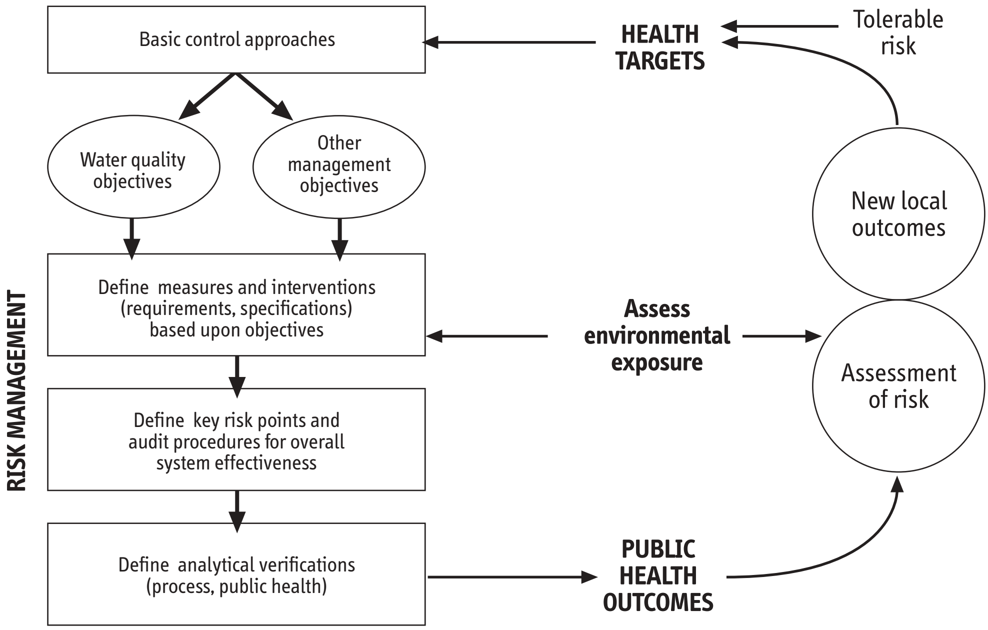


FIGURE 4.3. HARMONIZED APPROACH TO ASSESSMENT OF RISK AND RISK MANAGEMENT FOR WATER-RELATED EXPOSURE TO PATHOGENS (ADAPTED FROM BARTRAM ET AL., 2001)

- 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:

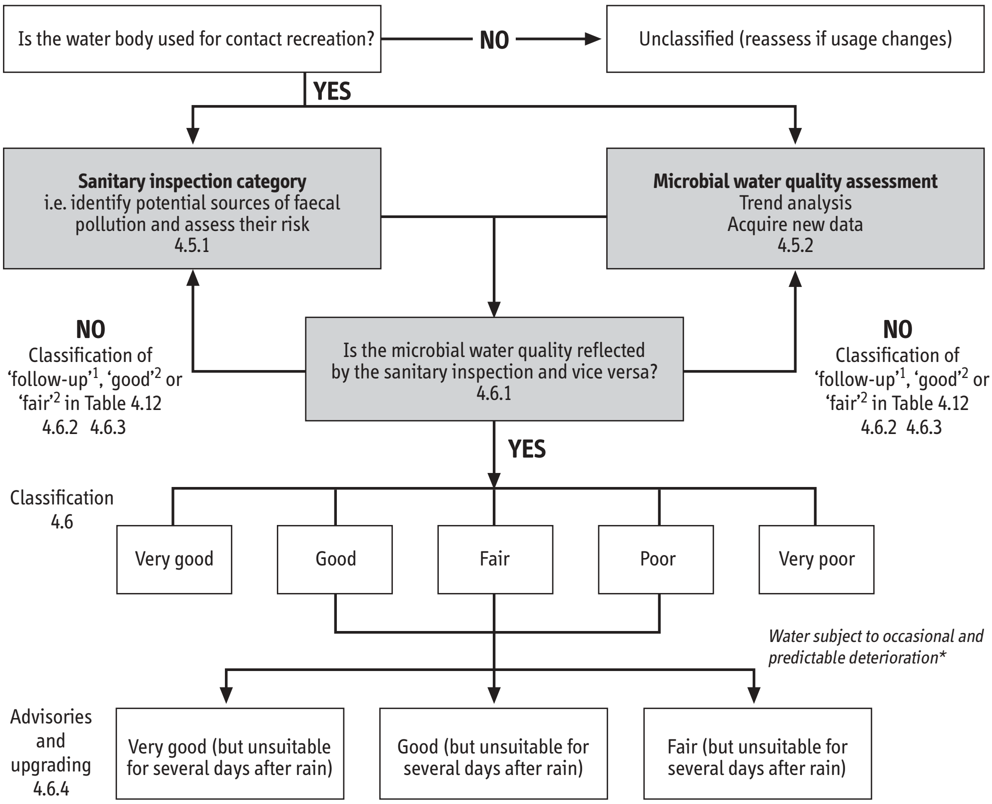
- assessment of evidence for the degree of influence of faecal material (i.e., derivation of a sanitary inspection category); and
- counts of suitable faecal index bacteria (a microbial water quality assessment).

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	**Very low**	Very good	Very good	Follow up[1]	Follow up[1]	Action
Inspection	**Low**	Very good	Good	Fair	Follow up[1]	
Category	**Moderate**	Good[2]	Good	Fair	Poor	
(susceptibility to	**High**	Good[2]	Fair[2]	Poor	Very poor	
faecal influence)	**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]	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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CHAPTER 5
Free-living microorganisms

In addition to microorganisms introduced to recreational waters through human or animal faecal contamination (see chapter 4), a number of pathogenic micro-organisms are indigenous to such areas or, (like the leptospires) once introduced are capable of colonizing the environment. This chapter describes the principal free-living microorganisms of concern; the diseases that they may cause associated with recreational water use; and potential control measures.

5.1 Human pathogenic Vibrio species

Vibrio species are motile, non-spore-forming, slightly curved Gram-negative rods with a single polar flagellum. They are both aerobic and facultatively anaerobic. Vibrios require, or their growth is stimulated by, sodium chloride, and they are capable of respiratory and fermentative metabolism, with only a single species (*V. furnissii*) producing gas. All human pathogenic *Vibrio* species except *V. metschnikovii* reduce nitrate and are oxidase positive.

Substantial evidence has been provided showing that *Vibrio* species are natural inhabitants of marine aquatic environments in both temperate and tropical regions, with most infections acquired by exposure to such environments or to foods derived from them (Kelly et al., 1991; Oliver & Kaper, 1997). *Vibrio* species have been isolated from a variety of environmental samples, including water, sediment, plankton, shellfish and finfish. Several studies suggest that the occurrence of vibrios does not correlate with the occurrence of the traditionally used faecal index organisms, although a positive correlation may be found in waters receiving human wastes from disease outbreaks (mainly cholera). There appears to be a positive correlation between water temperature and the numbers of human pathogenic vibrios isolated, as well as the number of reported infections. Seasonality is especially noted for *V. vulnificus* and *V. parahaemolyticus* (Oliver & Kaper, 1997). *Vibrio* species have been isolated in waters showing a broad range of salinities and varying pH values. *V. cholerae* and *V. mimicus* are the only species found in fresh water. Thus, due to the ubiquitous nature of *Vibrio* species in the aquatic environment, the presence of *Vibrio* species in bathing water cannot be controlled by water quality control measures such as wastewater treatment and disinfection. Human carriers and shedding appear to be of only limited importance in the epidemiology of *Vibrio* infections associated with recreational water use.

While there exists considerable variation in the severity of the various *Vibrio*-associated diseases, the most severely ill patients generally suffer from pre-existing illnesses, with chronic liver disease being one of the most common. An exception to this generalization is *V. cholerae* serogroups O1 and O139, the causes of cholera, which can readily cause disease in non-compromised individuals.

The *Vibrio* species of medical importance grow well on common media, including blood, chocolate and Mueller-Hinton agars. A more thorough search for *Vibrio* species can be done by culture on a selective-differential plating medium, such as thiosulfate–citrate–bile salts–sucrose (TCBS) medium. However, not all *Vibrio* species of medical importance grow well on TCBS agar (Kelly et al., 1991). In fact, different brands of TCBS can select different species of vibrios and be less selective for the target microorganisms. Increased yields of vibrios from environmental samples may be obtained by enrichment in alkaline peptone water before subculture on plating media. Since *Vibrio* species vary considerably in pathogenicity and epidemiological significance, isolates should be identified to species level. Such identification should be performed by reference laboratories. Although commercial systems for the identification of *Vibrio* species have improved, misidentifications remain a problem.

Currently, 12 *Vibrio* species are known to cause or to be associated with human infections: *V. alginolyticus*, *V. carchariae*, *V. cholerae*, *V. cincinnatiensis*, *V. damsela*, *V. fluvialis*, *V. furnissii*, *V. hollisae*, *V. metschnikovii*, *V. mimicus*, *V. parahaemolyticus* and *V. vulnificus* (Kelly et al., 1991). The infections can be classified as intestinal or extraintestinal, although this division is not absolute (Table 5.1).

Studies of the infectious dose for vibrios able to cause gastrointestinal disease have been carried out mainly for *V. cholerae*, where the infectious dose appears high. For *V. cholerae*, 10^6 organisms or more are needed to cause cholera. In hypochlorhydric persons, the infectious dose is reduced to 10^4–10^5 organisms (Oliver & Kaper, 1997). Thus, it is unlikely that persons bathing or involved in other recreational water

TABLE 5.1. *VIBRIO* SPECIES THAT MAY BE FOUND IN HUMAN CLINICAL SPECIMENS[a]

	Occurrence in human clinical specimens[b]	
Species	**Intestinal**	**Extraintestinal**
V. alginolyticus	–	++
V. carchariae	–	+
V. cholerae 01	++++	+
Non-01	++	+
V. cincinnatiensis	–	++
V. damsela	–	+
V. fluvialis	++	–
V. furnissii	++	–
V. hollisae	++	–
V. metschnikovii	+	+
V. mimicus	++	+
V. parahaemolyticus	++++	+
V. vulnificus	+	+++

[a] Modified from Kelly et al. (1991)

[b] The symbols +, ++, +++ and ++++ give the relative frequency of each organism in specimens from implicated infections; – = not found.

activities would ingest vibrios in numbers high enough to cause gastrointestinal disease in the absence of extreme contamination. However, the risk of extraintestinal infections associated with human pathogenic *Vibrio* species, especially wound and ear infections, during recreational activities in water is of health importance, although the infectious doses for such infections are unknown.

5.1.1 V. alginolyticus

V. alginolyticus is very common in the marine environment. The organism does not cause diarrhoea but may cause soft tissue infections following exposure to seawater (Kelly et al., 1991). Ear, wound and eye infections have been reported most often. It appears that a majority of patients with otitis associated with *V. alginolyticus* have predisposing conditions, including chronic otitis media and rupture or tubulation of the tympanic membrane. Infections are usually self-limiting and of moderate severity and short duration, and antibiotic treatment is only occasionally necessary.

5.1.2 V. cholerae

Among the vibrios, special attention has focused on the identity of those causing cholera. *V. cholerae* has been divided into more than 150 serological types on the basis of the O or somatic antigens.

Numerous outbreaks of cholera involving drinking-water and foods have been documented. Although *V. cholerae* O1 and O139 are occasionally isolated from the aquatic environment, especially in areas with cholera outbreaks, no confirmed cholera cases caused by activities in recreational waters, including bathing, seem to have been reported. Thus, probably because of the high infectious dose required to cause cholera, it appears that the isolation of *V. cholerae* O1 and O139 from marine waters represents a very low health risk to persons bathing or participating in other recreational activities in such waters. Other non-O1 serotypes cause gastroenteritis, with the range of symptoms found to vary, but common features include diarrhoea and occasional vomiting with abdominal cramps.

Although isolates have been recovered from wounds, ears and a variety of other sites, the clinical significance of *V. cholerae* as a cause of extraintestinal infections is uncertain, as other potential pathogens are often also isolated.

5.1.3 V. parahaemolyticus

V. parahaemolyticus is an agent of food poisoning, associated with the consumption of raw or insufficiently cooked seafood. It has also been associated with pneumonia (Yu & Uy-Yu, 1984), resulting from inhalation of contaminated aerosol, and wound infection. *V. parahaemolyticus* has been associated with severe life-threatening infections these, however, were subsequently found to be due to *V. vulnificus* (Kelly et al., 1991).

5.1.4 V. vulnificus

V. vulnificus is a foodborne pathogen, causing a rapidly fatal infection in persons with underlying liver diseases. Generally implicated is the consumption of raw or under-cooked oysters, although raw clams, octopus and other marine fish and shellfish have also been associated with the disease. In addition to being foodborne, *V. vulnificus* causes wound infections following entry into a skin lesion. Such infections are almost always associated with seawater and/or shellfish. Puncture wounds generally result from utensils used to clean shellfish or from the hard shell or fins of shellfish and fish. Symptoms, which develop after about 16 h, include intense pain, redness, swelling and rapidly developing tissue destruction. Although the pathogenesis of these infections has yet to be elucidated, it is likely that one or more of the several exoenzymes (e.g., collagenase, protease, elastase, phospholipase, cytotoxic haemolysin) produced by this species are essential for its ability to invade and cause tissue destruction. Surgical removal of tissue, skin grafting and even amputation are generally required. Mortality rates average 20–25%. Wound infections with *V. vulnificus* typically occur in healthy persons and remain localized, although the bacterium may become systemic in people with chronic liver disease, and this carries a high fatality rate. In the past, several reports of severe, life-threatening infections associated with *V. parahaemolyticus* were described; however, subsequent examination of the isolates from these cases has indicated that they were actually *V. vulnificus* (Kelly et al., 1991).

5.2 Aeromonas species

Aeromonas species are Gram-negative rod-shaped or coccoid cells that are facultative anaerobes and generally motile by a single polar flagellum (although non-motile species exist), which are currently assigned to the family Aeromonadaceae (Altwegg, 1999). They utilise carbohydrates with production of acid and gas, and the metabolism of glucose is both fermentative and respiratory. They are oxidase and catalase positive and reduce nitrates to nitrites.

Aeromonas spp. are considered autochthonous inhabitants of aquatic environments and are ubiquitous in surface fresh and marine waters, with high numbers occurring during the warmer months of the year (Ashbolt et al., 1995; Holmes et al., 1996). Clinical isolation of these microbes presents the same seasonal distribution (Joseph, 1996). Numbers may be high in both polluted and unpolluted habitats with densities ranging from <1 to 1000 cells per ml (Holmes et al., 1996; Borrell et al., 1998; Altwegg, 1999). A significant correlation has been reported between aeromonads and the trophic state of freshwater (Rippey & Cabelli, 1989; Ashbolt et al., 1995). Other authors, however, were unable to predict the trophic status of several lakes in relation to these microorganisms (Rhodes & Kator, 1994). Sewage can also contain elevated numbers (10^6–10^8 cells per ml) of aeromonads (Ashbolt et al. 1995; Holmes et al., 1996). In marine bathing waters they can be abundant and their presence is supported by organic matter coming from the land (Alucino et al., 2001). Study of *Aeromonas* spp. virulence factors (Chopra & Houston, 1999; Janda, 2001) in isolates recovered from bathing waters indicates the presence of potentially virulent strains (Ashbolt et al., 1995; Kingombe et al., 1999; Sechi et al., 2002; Soler et al., 2002).

The taxonomy of the genus is considered complex and has evolved rapidly since 1987 with the addition of nine new species. Presently, it includes fifteen accepted species (Altwegg, 1999). As a consequence of this difficult taxonomy, studies of environmental and clinical aeromonads have generally been limited to three species; A*eromonas hydrophila*, *Aeromonas sobria* and *Aeromonas caviae*. According to present taxonomy, however, strains under these names may belong to other species. The term *A. hydrophila* has, for example, frequently been used indistinctly to include all motile, mesophilic aeromonads, which comprise several named species (Carnahan & Altwegg 1996). The reason for such simplification is that most commercial identification systems tend to identify most of the strains as belonging to this species. This has led to an overestimation of the clinical and environmental relevance of this species and has hampered the establishment of the true incidence of the other species. When reliable identification methods are applied the most prevalent clinical species are *A. caviae, A. veronii* (correct terminology for the clinical strains referred to as *A. sobria*) and *A. hydrophila* accounting for more than 85% of all clinical isolates (Kühn et al., 1997; Janda & Abbott, 1998; Figueras et al., 2000a,b). These species have also been found to be prevalent in recreational waters (Borrell et al., 1998).

Aeromonas has been found to have a role in a number of human illnesses (Janda & Abbott, 1998). Its association with gastroenteritis has been seen both in industrialised countries and developing countries worldwide (Joseph, 1996; Janda & Abbott, 1998; Sixl et al., 1999). *Aeromonas*-associated diarrhoea is normally self-limiting and in many cases does not lead to a microbiological study of the faeces. This lack of investigation could explain why relatively few outbreaks have been identified (Montiel & Harf-Montiel, 1997). It affects mostly children under five years of age and immunocompromised adults. These are also the population groups most frequently associated with aeromonad-related septicaemia (Janda & Abbott, 1996, 1998). Underlying diseases (cancer, hepatobiliary disease and diabetes) play a major role in the acquisition and outcome of the diseases produced by *Aeromonas* (Ko & Chuang, 1995; Janda & Abbott, 1998). Although there are exceptions, *Aeromonas* sepsis normally arises secondarily to gastroenteritis or wound infections and is associated with high (30–85%) mortality rates (Janda & Abbott, 1996, 1998; Altwegg, 1999; Ko et al., 2000). The main risk of acquiring *Aeromonas*-associated infections is by water contact through open wounds. The consumption of contaminated water or food may be important (Bloom & Bottone, 1990; Joseph et al., 1991; Altwegg et al., 1991; Voss et al., 1992; Kelly et al., 1993; Krovacek et al., 1995). Cases of wound infections in healthy people associated with recreational-water have been described (Joseph et al., 1991; Voss et al., 1992; Altwegg, 1999) as have cases of pneumonia following aspiration of contaminated water (Goçalves et al., 1992; Janda & Abbott, 1998).

5.3 Free-living amoebae

Free-living amoebae are unicellular protozoa common to most soil and aquatic environments (Page, 1988). Of the many hundreds of species of free-living amoebae, only members of the genus *Acanthamoeba*, *Naegleria fowleri* and *Balamuthia mandrillaris* are known to infect humans, often with fatal consequences.

5.3.1 Acanthamoeba

Acanthamoeba is a genus of environmental free-living amoebae found in most soil and water habitats (Page, 1988). The organism can infect a variety of mammals, including humans, producing severe and often fatal consequences. The genus contains numerous species, of which *A. polyphaga*, *A. castellanii* and *A. culbertsoni* have been identified most frequently as causing human disease (Martinez, 1985; Ma et al., 1990; Kilvington & White, 1994).

Acanthamoeba is characterized by a feeding and replicating trophozoite that, under adverse conditions, can form a dormant cyst stage (Page, 1988). Trophozoites are 25–40 μm in length, depending on the species, and show numerous needle-like projections from the trophozoite body, termed acanthapodia. A central contractile vacuole is present in the trophozoite cytoplasm and is required for osmotic regulation. Cysts range in length from approximately 15 to 28 μm, depending on the species, and are double walled.

The resistance of *Acanthamoeba* cysts to extremes of temperature, disinfection and desiccation accounts for the almost ubiquitous presence of the organism in the environment (Martinez, 1985; Page, 1988; Kilvington & White, 1994). *Acanthamoeba* have been isolated from natural and artificial waters, chlorinated swimming pools and the atmosphere.

Acanthamoeba is an aerobic organism and as such cannot exist as the trophozoite stage in environments with low oxygen content. However, *Acanthamoeba* cysts have been isolated from anaerobic material such as faeces and sewage (Daggett et al., 1982; Martinez, 1985). The trophozoites are killed by saline concentrations of >1%, although the more environmentally robust cysts have been isolated from marine environments (Sawyer et al., 1982).

Acanthamoeba numbers in freshwater habitats vary according to the temperature of the water (either from seasonal variation or through thermal enrichment from industrial processes), availability of a bacterial food source and, possibly, the extent of human activity associated with the water (Daggett et al., 1982; Martinez, 1985; Kilvington & White, 1994). However, detailed ecological surveys have not been conducted. *Acanthamoeba* cysts have been isolated in significant numbers from marine sites, particularly those associated with sewage and waste effluent outlets.

Certain species of *Acanthamoeba* are pathogenic to humans and cause two clinically distinct diseases: granulomatous amoebic encephalitis (GAE) and inflammation of the cornea (keratitis) (Ma et al., 1990; Martinez, 1991; Kilvington & White, 1994). The taxonomic classification of *Acanthamoeba* is derived from microscopic observations of the trophozoite and cyst forms (Page, 1988). This is a subjective approach, and molecular typing methods have demonstrated that *Acanthamoeba* is a genetically complex group that correlates poorly with species identification based on morphological criteria (Kilvington et al., 1991a; Gast & Byers, 1995). As a consequence, the precise identity of the *Acanthamoeba* species/strains causing human infection and their possible environmental sources are unknown. However, *A. polyphaga* and *A. castellanii* are most frequently reported as causing keratitis, and *A. culbertsoni* is most frequently reported as causing GAE (Kilvington et al., 1991a; Gast & Byers, 1995).

GAE is a chronic disease of the immunosuppressed (as a result of chemotherapy or drug or alcoholic abuse) host (Martinez, 1985, 1991; Ma et al., 1990). Cases of GAE in patients with acquired immunodeficiency syndrome (AIDS) have also been reported. GAE is subacute or chronic and invariably fatal. Symptoms include fever, headache, seizures, meningitis and visual abnormalities. GAE is extremely rare, with only 60 cases reported worldwide. The route of infection in GAE is unclear, although invasion of the brain may result from the blood following a primary infection elsewhere in the body, possibly the skin or lungs (Martinez, 1985, 1991). The precise source of such infections is unknown because of the almost ubiquitous presence of *Acanthamoeba* in the environment.

Acanthamoeba keratitis affects previously healthy persons and is a severe and potentially blinding infection of the cornea (Ma et al., 1990; Kilvington & White, 1994). In the untreated state, acanthamoeba keratitis can lead to permanent blindness. Although only one eye is usually affected, cases of bilateral infection have been reported. The disease is characterized by intense pain and ring-shaped infiltrates in the corneal stroma. Contact lens wearers are most at risk from the infection and account for approximately 90% of reported cases (Kilvington & White, 1994). Poor contact lens hygiene practices (notably ignoring recommended cleaning and disinfection procedures and rinsing or storing of lenses in tap water or non-sterile saline solutions) are recognized risk factors, although the wearing of contact lenses while swimming or participating in other water sports may also be a risk factor. In non-contact lens related keratitis, infection arises from trauma to the eye and contamination with environmental matter such as soil and water (Sharma et al., 1990).

5.3.2 Naegleria fowleri

Naegleria fowleri is a free-living amoeba found in thermal freshwater habitats worldwide. The organism causes fatal primary amoebic meningoencephalitis (PAM) in humans. Infection usually results from swimming in contaminated water (John, 1982; Martinez, 1985; Warhurst, 1985).

N. fowleri is found in thermal aquatic environments and can tolerate temperatures up to 46 °C. Although *N. fowleri* is most likely to be isolated from sites where the temperature is above 30 °C, the cysts can survive at 4 °C for at least 12 months, with retention of virulence by the excysted trophozoites (Warhurst, 1985). *N. fowleri* has been isolated from both natural and artificial thermally enriched habitats, such as natural hot springs, freshwater lakes, domestic water supplies, chlorinated swimming pools, water cooling towers and effluent from industrial processes (Martinez, 1985). *N. fowleri* has also been isolated from water cooling circuits of electricity power stations and thermal effluents from industrial processes in Belgium, Czechoslovakia, France, the USA and the United Kingdom (De Jonckheere, 1987; Kilvington & Beeching, 1997). In the latter study, *N. fowleri* was also isolated upstream and downstream of the river supplying the power station (Kilvington & Beeching, 1997). Also, in Belgium, *N. fowleri* has been isolated from a fish farm exploiting thermal water from a nuclear power plant (De Jonckheere, 1987). Such ecological surveys have indicated that *N. fowleri* is more likely to predominate in artificial thermal habitats compared with natural environments such as hot springs, where other, non-pathogenic, thermophilic *Naegleria* species predominate (Kilvington et al., 1991b; Kilvington & Beeching, 1997).

Primary amoebic meningoencephalitis results from the instillation of *N. fowleri* into the nasal passages, usually while swimming. Young males are most at risk from infection, probably because of their more vigorous swimming habits. From the nostrils, the organism invades the nasal epithelium and migrates along the olfactory lobes, via the cribriform plate, to infect the brain and meninges. The infectious dose of *N. fowleri* for humans is not known. PAM is usually fatal, with death occurring in 3–10 days after exposure.

Since PAM was first recognized in 1965, several hundred cases have been reported worldwide. Clustering of cases can occur when a single site is the source of infection. In Usti, Czechoslovakia, 16 cases were associated with a public swimming pool (Cerva & Novak, 1968). The source of the contamination was eventually traced to a cavity behind a false wall used to shorten the pool length. The pool took water from a local river, which was the likely source of the organism.

Cases of PAM have been reported from Belgium and Czechoslovakia in people swimming in warm effluent water from industrial processes (De Jonckheere, 1987). In south-western Australia, infections have been associated with the reticulated mains supply water. In this region, water is supplied to remote localities via over-ground steel pipes. Solar heating of the water in the system enabled *N. fowleri* to proliferate and resulted in approximately 20 cases of PAM. The installation of chlorifiers at regular intervals along the pipelines and regular monitoring of the supply eliminated the problem (Robinson et al., 1996).

One confirmed case of PAM occurred in Bath Spa, England, in 1978. The victim was a young girl who swam in a public bathing pool fed with water from the historic thermal springs that rise naturally in the City (Cain et al., 1981). Subsequent analysis confirmed the thermal springs to be the source of the infection (Kilvington et al., 1991b).

5.3.3 Balamuthia mandrillaris

In 1990, Visvesvara and colleagues described cases of fatal encephalitis in humans and other primates due to a previously undescribed free-living amoeba. By morphological appearance, the amoebae resembled members of the genus *Leptomyxa*; on closer examination, however, they were found to be sufficiently distinct to be described as a new genus and species, *Balamuthia mandrillaris* (Visvesvara et al., 1993). Using antiserum to the organism, the investigators were able to demonstrate that certain cases of GAE attributed to *Acanthamoeba* were in fact caused by *B. mandrillaris*.

Unlike *N. fowleri* and *Acanthamoeba*, *B. mandrillaris* does not grow on the standard medium for isolating free-living amoebae, plain agar seeded with the bacterium *Escherichia coli* (Page, 1988). *B. mandrillaris* has been cultured from only a few cases of infection using mammalian tissue culture cell lines (Visvesvara et al., 1990, 1993). As a consequence of the difficulties in growing the organism, there have been no reports of the isolation of *B. mandrillaris* from water or other environmental samples.

Like *Acanthamoeba* GAE, *B. mandrillaris* encephalitis is largely a disease of the immunocompromised host and infects either sex and any age (Visvesvara et al., 1990, 1993). However, cases are being recognized in persons with no underlying immunosuppression and with no history of contact or swimming in water (Martinez & Visvesvara, 2001). The clinical course of the disease in humans ranges from 14 days to 6 months, with a mean of 75 days. Infection is invariably fatal. Clinical symptoms and histopathological findings are similar to those seen in GAE, and cysts are also found in the tissues. Approximately 85 cases of *B. mandrillaris* encephalitis have been described worldwide, with some 50% coming from the USA. At least 10 cases have occurred in patients with human immunodeficiency virus (HIV). Other cases have been identified from Argentina, Australia, Canada, Czechoslovakia, Japan, Mexico and Peru (Visvesvara et al., 1996; Martinez & Visvesvara, 2001).

5.4 Leptospira species

Leptospires are motile spirochaete (helically coiled) bacteria. Traditionally, the genus *Leptospira* consists of two species, the pathogenic *L. interrogans* sensu lato and the saprophytic *L. biflexa* sensu lato. Serological tests within each species revealed many antigenic variations and, on this basis, leptospires are classified as serovars. In addition, a classification system based on DNA relatedness is used (Brenner et al., 1999). The current species determination is based on this principle. The serological and genetic taxonomies are two different systems with only little correlation (Brenner et al., 1999). Free-living strains are ubiquitous in the environment (Faine et al., 1999); the pathogenic strains, however, live in the kidneys of animal hosts.

5.4.1 L. interrogans *sensu lato*

Leptospires live in the proximal renal tubules of the kidneys of carrier animals (including rats, cows and pigs) and are excreted in the urine, which can then contaminate surface waters (ponds, lakes, streams, rivers), groundwater soil and mud. Humans and animals (humans are always incidental hosts) become infected either directly through contact with infected urine or indirectly via contaminated fresh water or soil. Virulent leptospires gain entry to the body through cuts and abrasions of the skin and through the mucosal surfaces of the mouth, nose and conjunctiva. In cases due to exposure to recreational water, the incubation period seems to vary between 2 and 30 days but generally is between 7 and 14 days (Christie, 1974).

Diseases caused by *Leptospira interrogans* sensu lato have been given a variety of names, including swineherd's disease, Stuttgart disease and Weil's syndrome, but collectively all of these infections are termed leptospirosis. The clinical manifestations of leptospirosis vary considerably in form and intensity, ranging from a mild flu-like illness to a severe and potentially fatal form of the disease, characterized by liver and kidney failure and haemorrhages (Weil's syndrome). Severity is related to the infecting serovar as well as host characteristics, such as age and underlying health and nutritional status. Specific serovars are often associated with certain hosts. For example, serovar hardjo is associated with cattle; serovar pomona is associated with pigs, cattle and rodents; and serovars icterohaemorrhagiae, copenhageni, bataviae, autumnalis, australis and javanica are associated with rats and small rodents.

Due to the non-specific presentation of leptospirosis and its resemblance to many other diseases, there is a worldwide underdiagnosis of the disease, with mild cases probably being dismissed as flu and severe cases often confused with other diseases. Additionally, serological surveys suggest that subclinical and inapparent infections are common (Faine et al., 1999). As conclusive diagnosis cannot be made without laboratory confirmation of clinical samples, case determination may be dependent upon local facilities and expertise (Faine et al., 1999).

Leptospirosis is often considered to be an occupational disease related to proximity to animals or contaminated water. In developed countries, however, cases related to occupation seem to be on the decrease, while those related to recreation are increasing (Sandford, 1986; Waitkins, 1986; CDC, 1998; Kirsche, 2001) possibly reflecting increases in leisure time and the increasing trend of adventure or wilderness activities in the tropics and subtropics.

Compared with many other pathogens, leptospires have a comparatively low resistance to adverse chemical and physical conditions. They are seldom found in water of below pH 6.8, and they cannot tolerate drying or exposure to direct sunlight. Their survival in polluted water and seawater is poor (Noguchi, 1918; Alston & Broom, 1958). However, in the right circumstances, around neutral pH and when moderate temperatures and oxygen supersaturated conditions exist, leptospires are still detectable for up to about six months (Alston & Broom, 1958). It has been suggested that pathogenic leptospires may grow and multiply under certain environmental conditions, and large numbers (far in excess of what would be expected from contamination) have been found in some fast-flowing rivers (Alexander et al., 1975; Baker, 1965; Baker & Baker, 1964).

5.5 Guideline values

Evidence suggests that although infection with free-living microorganisms or pathogenic leptospires via recreational water use may be life-threatening, the incidence of such infection is very low and, in many cases, is limited to specific areas. As such, no specific guideline values have been recommended. Authorities should be aware of the potential hazards posed by these organisms and act accordingly.

5.6 Risk assessment and control measures

Given the nature of the microorganisms outlined in this chapter, assessment of the likely local hazard (e.g., the likelihood of thermal warming of fresh waters) and education of water users and health professionals will be important control measures.

5.6.1 Vibrios

Although human pathogenic *Vibrio* species are ubiquitous in marine waters, their presence represents only a minor risk of gastrointestinal disease to people bathing or involved in other recreational activities in the water. However, potentially life-threatening extraintestinal infections with *Vibrio* species may occur, and physicians should consider this diagnosis when people who have had recent contact with seawater present a wound infection or an acute ear infection.

5.6.2 Aeromonads

Aeromonas are ubiquitous both in freshwater and seawater recreational areas and therefore there is a high risk of exposure. The infective dose that produced colonisation in a single study in volunteers and in a 28-year-old laboratory worker after accidental ingestion was, in both cases, 10^9 cells (Morgan et al., 1985; Carnahan et al., 1991). The health significance of *Aeromonas* in consumed water is discussed in WHO (2002). However, the rapid onset of cellulitis in the setting of soft-tissue trauma with a history of water exposure should alert clinicians of the possible infection by *Aeromonas* (Gold & Salit, 1993). The presence of Gram-negative bacilli (after Gram-staining of purulent exudates) in a post-traumatic wound exposed to water is a sufficient basis to recommend empirical therapy (Gold & Salit, 1993).

Questioning patients about bathing water contact in *Aeromonas*-diarrhoeal processes may help to establish the epidemiological risk of acquiring these microbes during recreational activities. The use of typing techniques such as the ones employed in several studies (Kühn et al., 1997; Davin-Regli et al., 1998; Demarta et al., 2000; Martinez-Murcia et al., 2000; Bonadonna et al., 2002; Soler et al., 2003) may help to determine the epidemiological links between environmental and clinical strains.

5.6.3 Free-living amoebae

Acanthamoeba are common in soil and water habitats and occur in coastal and fresh waters. However, the incidence of human infections from this opportunistic pathogen is extremely low. Detailed risk assessments related to *Acanthamoeba* in coastal and freshwater bathing sites have not been undertaken and the possibility of acquiring an infection from such sites cannot be discounted. As such, immunocompromised individuals should be advised of the possible risk of acquiring GAE so that they can make an informed choice about swimming or immersion in natural waters.

Cases of PAM have been associated with freshwater lakes in the USA that receive solar warming in the summer months (Martinez, 1985), although Wellings et al. (1977) have estimated that only one case of PAM occurs for every 2.6 million exposures to water containing *N. fowleri* in Florida, USA. In some European countries, it is possible that solar heating could provide sufficient warming in the summer months to favour the proliferation of *N. fowleri*. However, no studies on the presence of *N. fowleri* at such sites and the potential risk to human health from recreational use of such water bodies have been conducted.

Studies have demonstrated that thermal discharges from electricity power stations in England and France can result in *N. fowleri* being present in rivers (Kilvington & Beeching, 1997). Cases of PAM have been reported from Belgium and Czechoslovakia in people swimming in warm effluent water from industrial processes (De Jonckheere, 1987). Accordingly, the direct use of such water for recreational purposes should be carefully evaluated. Surveys should also be undertaken to determine the presence and number of *N. fowleri* in rivers and lakes receiving thermally polluted water from industrial processes to enable risk assessments to be made from such sites.

B. mandrillaris encephalitis is a newly recognized disease, the incidence of which is still unknown. Infections would appear to be rare and, therefore, the risk to human health might be considered as minimal. Because the organism cannot be grown on standard media used for isolating other pathogenic free-living amoebae, the environmental habitat and possible sources of infection are undefined; as such, it is not possible to assess the risk from recreational water use at this point.

5.6.4 Leptospires

It is impractical to completely eradicate potential sources of leptospiral infection. Since the incidence of leptospirosis (certainly of the severe form) is relatively low, sensible precautionary measures for high-risk groups using freshwater environments are the best means of achieving the greatest protection. Public information highlighting sensible precautions, such as covering cuts and scratches with waterproof plasters or bandages prior to immersion, showering after water immersion and the provision of litter control and other measures to minimize the rodent population can be effective.

5.7 References

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CHAPTER 6

Microbial aspects of beach sand quality

Beaches represent the unconsolidated sediment that lies at the junction between water (oceans, lakes and rivers) and land and are usually composed of sand, mud or pebbles. From a recreational viewpoint, sand beaches are sought after. Especially in higher latitudes, a significant percentage of time is spent on the beach itself rather than in the water.

Microorganisms are a significant component of beach sand. Bacteria, fungi, parasites and viruses have all been isolated from beach sand. A number of genera and species that may be encountered through contact with sand are potential pathogens. Accordingly, concern has been expressed that beach sand or similar materials may act as reservoirs or vectors of infection (Nestor et al., 1984; Roses Codinachs et al., 1988; Mendes et al., 1997), although transmission by this route has not been demonstrated in epidemiological studies.

In this chapter, the incidence, dispersion and fate of microorganisms in beach sand are reviewed, as are potential management actions.

6.1 Microorganisms in beach sand

6.1.1 Faecal index microorganisms

Faecal index organisms are non-pathogenic microorganisms used to indicate the degree of faecal contamination. They are generally present in far greater numbers than pathogenic microorganisms and are easy to isolate, identify and enumerate. Faecal index organisms include coliforms (total coliforms, thermotolerant coliforms and *Escherichia coli*), intestinal enterococci (see Box 4.1), bacteriophages and clostridia.

The presence of total coliforms, thermotolerant coliforms, *E. coli* and intestinal enterococci in beach sand and the relationship between their counts in beach sand and their counts in adjacent waters have comprised a significant area of research, with apparently contradictory results. Total coliforms, thermotolerant coliforms and intestinal enterococci were isolated from surface sand samples in Marseilles and Agde, France. Counts of intestinal enterococci, probably originating from animals, were higher than counts of other indices (Conseil Supérieur d'Hygiène Publique de France, 1990). High numbers of thermotolerant coliforms and intestinal enterococci were isolated in beach sand along Taranto coastal waters in Italy (Signorile et al., 1992). Lower numbers of faecal index organisms were recorded in swimming areas in Tel Aviv, Israel, and in Barcelona, Spain (Figueras et al., 1992; Ghinsberg et al., 1994).

Low numbers of bacterial indices of faecal pollution were recovered in dry sand from a beach along the Thyrrenian coast (Italy). *E. coli* was recovered in 61% of the samples and enterococci outnumbered coliforms (Bonadonna et al., 2002).

In an Italian study, a significant correlation was found between contamination of beaches and contamination of adjacent seawaters, although the sand generally had higher bacterial counts than the water (Aulicino et al., 1985). A similar tendency was found at Barcelona beaches; in contrast to the Italian study, however, the level of contamination was not significantly different between sand and seawater (Roses Codinachs et al., 1988).

Papadakis et al. (1997) found no correlation between the indices of faecal pollution counted on the wet part of the beach and *Staphylococcus aureus* counts or the presence of fungi. A statistically significant correlation was detected between yeasts and molds, *E. coli* and enterococci, enterococci and spores of sulfite-reducing *Clostridium* and between clostridial spores and staphylococci in an investigation on wet and dry sands in Italy (Bonadonna et al., 2002). In an epidemiological study carried out on two beaches in Malaga, Spain, faecal index microorganisms, especially coliphages, were highly significantly correlated with dermatophyte fungi (microscopic fungi that grow on skin and mucous membranes) on one of the beaches. Only *E. coli* showed a significant correlation with *Candida albicans* (a pathogenic fungus). At the other beach, intestinal enterococci showed the best correlation with dermatophyte fungi. Again, coliphages were the indices that best correlated with *C. albicans* (Borrego et al., 1991).

6.1.2 Staphylococcus

According to some studies, *Staphylococcus* spp. predominate over other flora in the sand (Dowidart & Abdel-Monem, 1990). Of a total of 85 strains of Gram-positive cocci isolated from beach water and sand located at two popular beaches in Chile, 31% were classified as *S. epidermidis*, 9% as *S. haemolyticus*, 24% as *S. aureus* and 36% as *Staphylococcus* spp. (Prado et al., 1994).

The origin of *Staphylococcus* in beach sand is attributed to human activity. Its occurrence has been found to correlate with the number of swimmers on the beach, and the counts of *S. aureus* were found to correlate with the presence of yeasts of human origin in sand samples (Papadakis et al., 1997). Higher counts of *S. aureus* were recovered from the sand and water in summer, when there was a higher density of swimmers on the beach, than in winter. Also, higher counts of *S. aureus* were recovered from sand than from water samples (Ghinsberg et al., 1994; Papadakis et al., 1997).

Investigations carried out along the Tyrrhenian coast (Italy) showed higher densities of *Staphylococcus* spp. in sand of areas characterized by breakwaters than in sands found in open areas. *S. epidermidis* was the predominant species (Bonadonna et al., 1993a).

6.1.3 Pseudomonas aeruginosa

In a study in Israel, both seawater and sand on a number of beaches were found to contain various levels of *Pseudomonas aeruginosa*. The isolation of *P. aeruginosa* and of other *Pseudomonas* spp. was proportionally higher in sand than in seawater samples (Ghinsberg et al., 1994). *P. aeruginosa* was isolated from sandy beaches in Portugal under various tidal conditions, all beaches containing similar counts (Mendes et al., 1993).

6.1.4 Vibrio *spp.*

Vibrio parahaemolyticus isolates have been found in marine or brackish water and sand specimens collected from sand banks in Africa (Aldova, 1989). *Vibrio harvey* has been isolated from seashore water and sand samples collected on coarse sand or pebble beaches (Aldova, 1989; see also chapter 5).

6.1.5 *Enteric bacteria*

Species of bacteria that can cause gastroenteritis have been isolated from sand samples. However, their presence constitutes no apparent health threat to sunbathers. Sand beaches in Portugal contained similar counts of *Clostridium perfringens* under various tidal conditions (Mendes et al., 1993). Bonadonna et al. (1993b) suggested that *C. perfringens* could be a good index of faecal contamination in sand sediment. Low levels of *Campylobacter jejuni* were recorded in both coastal waters and sand on a number of Israeli beaches, with the beach sand containing higher counts than adjacent shore waters (Ghinsberg et al., 1994). In the United Kingdom, intertidal zone sediments appeared to serve as a substantial reservoir for thermophilic campylobacters, which could contribute significantly to bacterial numbers in surface waters, especially in rough weather (Obiri-Danso & Jones, 1997). Dabrowski (1982) isolated *Shigella* spp. from beach sand and water in the bay of Gdansk (Poland).

6.1.6 *Fungi*

Fungi that are often found in the environment as saprophytes may act as opportunistic pathogens, especially in immunocompromised patients (Hoog et al., 2000). Studies by Soussa (1990) in the Portuguese central coastal area showed dermatophytes in 42% of the sand beaches analysed. The most common were *Trichophyton mentagrophytes*, *T. rubrum* and *Microsporum nanum*, all isolated from sandy, non-flooded areas with organic residues. These species are all associated with skin infections, with *T. mentagrophytes* being the most common agent of dermatomycosis in Europe and *T. rubrum* the most common agent worldwide (Hoog et al., 2000). Saprophytic fungi (*Aspergillus candidus*, *A. ochraceus* and *A. fumigatus*) were isolated in the flooded and intermediate areas in high tidal conditions (Izquierdo et al., 1986).

Candida albicans and other *Candida* spp. have been isolated from sand beaches in the south of France (Bernard et al., 1988). In the same study, 8 keratinophilic fungi (i.e., those able to grow on keratin, a characteristic common to dermatophytes) and 11 non-keratinophilic species, all potential pathogens, were isolated. Izquierdo et al.

(1986) isolated 16 species of fungi from beach sand along the northeastern Mediterranean coast of Spain, among them some potentially pathogenic strains. Most of the species belonged to the genera *Penicillium*, *Aspergillus* and *Cladosporium*.

In Israel, Ghinsberg et al. (1994) isolated fungi in all beach sand samples, but not in seawater samples. In a study in Guadeloupe, Boiron et al. (1983) investigated fungal species in seawater and seashore sand, concluding that the similarity of bacterial species in sand and seawater, in conjunction with the fact that no *Candida albicans* was isolated, corroborated their hypothesis that the isolated yeasts were of marine origin. The isolated fungi belonged to the species *C. tropicalis*, *C. parapsilosis*, *C. langeronii*, *C. guilliermondii*, *Trichosporon cutaneum* and *Torulopsis* sp. The most frequently isolated genera from beach sand samples in a Spanish study were *Penicillium*, *Aspergillus*, *Cladosporium*, *Altenaria*, *Mucor*, *Monilia*, *Cephalosporium*, *Verticillium* and *Chrysosporium* (Roses Codinachs et al., 1988). Absence or low incidence of *C. albicans* has also been recorded by other researchers (Roses Codinachs et al., 1988; Figueras et al., 1992).

The fungal density of 180 samples of sand collected from 42 Spanish Mediterranean beaches was found to reach several hundred thousand colony-forming units per gram of sample. The most commonly isolated genera were *Penicillium*, *Cladosporium*, *Aspergillus, Acremonium, Altenaria* and *Fusarium* (Larrondo & Calvo, 1989). In a study carried out in the Attica area of Greece, fungal isolates included *Candida albicans*, *C. krusei*, *C. tropicalis*, *C. puilliermondi*, *C. rugosa*, *Pitirosporum orbiculare*, *Fusarium*, *Penicillium*, *Mucor*, *Helminthosporium* and *Aspergillus niger* (Papadakis et al., 1997), a number of which are pathogenic (Hoog et al., 2000).

6.1.7 Viruses and parasites

Very little information exists concerning the presence of viruses and parasites in beach sand. In a three-year study in Romania by Nestor et al. (1984), the incidence of enteroviruses was found to depend on season, with no viruses being present in water and beach sand during non-vacation seasons. In a study of two sand beaches in Marseilles, France, *Toxocara canis* was found to be the most common parasite, being present on average in 150 g of sand (Conseil Supérieur d'Hygiène Publique de France, 1990). However, in a study carried out on "dog beaches" in Perth, Australia, a total of 266 samples showed no traces of *Toxocara canis* eggs or other eggs/larvae of parasitic nematodes (Dunsmore et al., 1984). It was emphasized in this study that the major risk to humans was from an environment in which puppies, not older dogs, were found. The presence of other parasites transmitted by water (Marshall et al., 1997) that have not been investigated in recreational sand areas may be potentially significant.

6.2 Dispersion and fate of microorganisms in beach sand

The growth of microorganisms in beach sand is limited by nutrient input. Laboratory studies have shown that nutrients pass through the bacterial community into the protozoan and metazoan community (Khiyama & Makemson, 1973). Further studies have shown that microbial contamination is higher in sand than in adjacent waters, as the sand behaves as a passive harbour for cumulative pollution (Oliveira & Mendes, 1991, 1992; Oshiro & Fujioka, 1995). Higher levels of coliforms, *E. coli* and enterococci in sand from Hanauma Bay (Hawaii) were thought to originate from run off from the cliffs surrounding the bay (Oshiro & Fujioka, 1995). Faeces from pigeons and mongoose were also thought to be a source of beach sand contamination. This study concluded that the contaminated sand could be the major source of the periodically high levels of bacteria in the water. Sand contamination is highly variable over short distances, making interpretation of results difficult (Aubert et al., 1987; Figueras et al., 1992; Oshiro & Fujioka, 1995).

The survival of enteric bacteria on the surface of dry sand may essentially be of short duration, the bacteria being destroyed mostly by environmental pressure. Wet sand, the area where young children typically spend most of their time on the beach, is the most relevant. Wet sand, enriched with organic substances, provides a favourable environment for enteric bacteria, which enables them to survive longer than in seawater (Papadakis et al., 1997).

Various factors have been proposed as encouraging the survival and dispersion of faecal index microorganisms and pathogens on beach sand. These include the nature of the beach, tidal phenomena, sewage outlets, the season, the presence of animals and the number of bathers. Water movement, for example, causes erosion, transportation and deposition of beach sediment and redistribution of associated microorganisms. Obiri-Danso & Jones (1997) analysed sediment samples in the United Kingdom for thermophilic campylobacters and faecal index microorganisms before and after tidal cover over a 12-month period. Fifty-three per cent of the samples were positive for campylobacters before tidal cover; this figure was significantly lower than the 64% recovered after tidal disposition. However, there was no significant difference in index organism numbers with respect to samples taken before or after tidal cover. In the same study, a seasonal variation was observed in campylobacters, with the highest isolation rate in winter (100%), followed by secondary peaks in spring (33–67%) and autumn (67–78%). The lowest counts were found in summer, which correlated with the incidence of campylobacters in surface waters. In contrast, Mendes et al. (1993) studied the influence of tides on counts of faecal index microorganisms and pathogens in sand without finding any clear differences. Nestor et al. (1984) found that the incidence of some pathogens depended on the season, with no viruses present in seawater and sand of beaches outside the holiday season. Borrego et al. (1991) reported higher bacterial counts and longer survival time in beaches close to sewage outlets.

As outlined in the previous section, fungi are often encountered in sand, and their survival is longer than that of enteric bacteria due to their capacity to form resistant spores. It has been suggested that the presence and the level of fungi is related to direct or indirect contamination originating from the residues/detritus from beach users and/or tidal influence (Mendes et al., 1998). In an *in vitro* study, Anderson (1979) found that four pathogenic fungi (*Trichosporon cutaneum*, *Candida albicans*, *Microsporum gypseum* and *Trichophyton mentagrophytes*) survived for at least 1 month in non-sterile sand inoculated with propagules of such fungi. In a similar study, five species of dermatophytes (*Epidermophyton floccosum*, *Microsporum canis*, *M. gypseum*, *Trichophyton mentagrophytes* and *T. rubrum*) and *Scopulariopsis brevicaulis* survived for between 25 and 360 days (Carillo-Muñoz et al., 1990).

Intensively used water recreation areas provide opportunities for person-to-person transmission of pathogens (e.g., dermatophytes). Transmission may occur because individuals shed pathogens onto sand, by direct contact or through other means, although, with the exception of transmission via contaminated water (as discussed in chapter 4), none of these has been positively demonstrated. Papadakis et al. (1997) collected water and sand samples from two beaches—one more popular than the other—in summer and winter, and the numbers of swimmers present on the beaches were counted. Coliforms, thermotolerant coliforms, enterococci, *S. aureus*, yeasts and moulds were also investigated. Water and sand samples were very low in index organisms of faecal pollution. Human species of yeasts were present in water and sand samples from both sites. *S. aureus* was isolated from water and sand samples only twice in winter, when swimmer presence was exceptional. A significant correlation appeared between swimmer numbers present on the beach and *S. aureus* counts in water samples, the correlation being more pronounced on the more popular beach. In sand samples, *S. aureus* counts correlated with the number of swimmers present on the beach only at the more popular beach. Yeasts of human origin correlated with the number of swimmers on the more popular beach, both in water and in sand samples.

6.3 Guideline values

Bacterial indices of faecal pollution and several pathogens have been isolated from beach sand. However, the capacity of pathogens in beach sand to infect beach users remains undemonstrated, and the real extent of their threat to public health is unknown. There is, therefore, no evidence to support the establishment of a guideline value for index organisms or pathogenic microorganisms in beach sand. However, preventative measures, such as education campaigns, and the management actions described in section 6.5 are important precautionary measures.

6.4 Research and monitoring

Epidemiological evidence for health risks from exposure to sandy beaches has not been found. Epidemiological studies aimed at investigating cause–effect or at examining a possible dose–response relationship linking the microbial quality of beach sand with skin, eye, ear and gastrointestinal symptoms would improve understanding in this area.

Experience with systematic beach surveillance as part of pollution control is relatively limited, and routine monitoring of beach sand for index organisms is generally not justified. However, it has often been recommended for research. WHO/UNEP (1992, 1994) indicated that wet beach sand and sediments should be part of epidemiological and microbiological studies correlating recreational water quality with health effects, but evidence to date indicates that beach sand does not appear to constitute an infectious hazard (Chabasse et al., 1986; Conseil Supérieur d'Hygiène Publique de France, 1990).

6.5 Management actions

The principal microbial risk to human health encountered on beaches and in similar areas is that arising from contact with animal excreta—notably that of dogs, where, for example, such areas are used for exercising pets. Regulations, often local in character, may restrict access on a seasonal basis to frequently used beaches or place an obligation upon the owner to remove animal excreta. Increased public awareness may help to reduce exposure, especially among young children. While beach cleaning may contribute to the removal of animal excreta, it is more often undertaken for aesthetic reasons or to attempt to remove litter or sharp materials, such as broken glass. The majority of beach management award schemes would not give an award to a resort beach that allowed dogs during the swimming season.

In some countries, particularly at resort areas, mechanical sand cleaning is a common practice that can eliminate visible rubbish mixed with sand, reducing the amount of organic matter and therefore reducing the further development of microorganisms (Bartram & Rees, 2000 chapter 12). However, mechanical cleaning may disturb sand ecology (Llewellyn & Shackley, 1996). Studies that have investigated the microbiological quality of sand have shown that a clear improvement was achieved as a result of raising the general levels of hygiene and cleanliness (Fernandez & Ferrer, 1982).

Chemical products such as disinfectants are sometimes applied to sand without regard to their effectiveness or possible ecotoxicological effects. The Conseil Supérieur d'Hygiène Publique de France (1990) has argued that there is not enough evidence to demonstrate the need for and efficiency of sand disinfection. When sand treatment is necessary, simple methods, such as sweeping and aeration, could be applied (Figueras et al., 1992), together with constant beach supervision in order to prevent access by animals. The use of clean towels for use on the beach, good personal hygiene, the prohibition of animals and regular mechanical cleaning are considered, by some authorities, to be important (e.g., Conseil Supérieur d'Hygiène Publique de France, 1990).

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CHAPTER 7

Algae and cyanobacteria in coastal and estuarine waters

In coastal and estuarine waters, algae range from single-celled forms to the seaweeds. Cyanobacteria are organisms with some characteristics of bacteria and some of algae. They are similar in size to the unicellular algae and, unlike other bacteria, contain blue-green or green pigments and are able to perform photosynthesis; thus, they are also termed blue-green algae.

Algal blooms in the sea have occurred throughout recorded history but have been increasing during recent decades (Anderson, 1989; Smayda, 1989a; Hallegraeff, 1993). In several areas (e.g., the Baltic and North seas, the Adriatic Sea, Japanese coastal waters and the Gulf of Mexico), algal blooms are a recurring phenomenon. The increased frequency of occurrence has accompanied nutrient enrichment of coastal waters on a global scale (Smayda, 1989b).

Blooms of non-toxic phytoplankton species and mass occurrences of macro-algae can affect the amenity value of recreational waters due to reduced transparency, discoloured water and scum formation. Furthermore, bloom degradation can be accompanied by unpleasant odours, resulting in aesthetic problems (see chapter 9).

Several human diseases have been reported to be associated with many toxic species of dinoflagellates, diatoms, nanoflagellates and cyanobacteria that occur in the marine environment (CDC, 1997). The effects of these algae on humans are due to some of their constituents, principally algal toxins. Marine algal toxins become a problem primarily because they may concentrate in shellfish and fish that are subsequently eaten by humans (CDR, 1991; Lehane, 2000), causing syndromes known as paralytic shellfish poisoning (PSP), diarrhetic shellfish poisoning (DSP), amnesic shellfish poisoning (ASP), neurotoxic shellfish poisoning (NSP) and ciguatera fish poisoning (CFP).

Notwithstanding the importance of dietary exposure for humans, this chapter deals only with the possible risks associated with recreational activities in (or near) coastal and estuarine waters. Exposures through dermal contact, inhalation of sea spray aerosols and ingestion of water or algal scums are briefly considered, as are precautionary measures that can be taken. Chapter 8 deals with algae and cyanobacteria in freshwater. More detailed coverage of cyanobacteria and human health is available in *Toxic Cyanobacteria in Water* (Chorus & Bartram, 1999).

7.1 Exposure through dermal contact

Marine cyanobacterial dermatitis ("swimmers' itch" or "seaweed dermatitis") is a severe contact dermatitis (inflammation of the skin) that may occur after swimming in water containing blooms of certain species of marine cyanobacteria. The symptoms are itching and burning within a few minutes to a few hours after swimming in an area where fragments of the cyanobacteria are suspended. Visible dermatitis and redness develop after 3–8 h, followed by blisters and deep desquamation. Some marine beaches, for example, report widespread problems due to a benthic marine cyanobacterium, *Lyngbya majuscula*, which grows on rocks in tropical seas and may cause severe blistering if trapped under the bathing suits of swimmers; this generally happens following storm conditions, which cause the dispersal of the cyanobacterium (Grauer & Arnold, 1961). To date, incidents have been reported only from Japan, Hawaii and Australia (Grauer & Arnold, 1961; WHO, 1984; Yasumoto & Murata, 1993).

Some toxic components, such as aplysiatoxin, debromoaplysiatoxin and lyngbyatoxin A, have been isolated from marine cyanobacteria (Mynderse et al., 1977; Fujiki et al., 1985; Shimizu, 1996). The cyanobacterium *Lyngbya majuscula* is known to produce debromoaplysiatoxin and lyngbyatoxin A, and the cyanobacteria *Oscillatoria nigroviridis* and *Schizothrix calcicola* are known to produce debromoaplysiatoxin (Mynderse et al., 1977). These toxins are highly inflammatory and are potent skin tumour promoting compounds, utilizing mechanisms similar to those of phorbol esters (i.e., through the activation of protein kinase C) (Gorham & Carmichael, 1988; Fujiki et al., 1990). More research is needed to establish the possible tumour promotion risks for human populations.

Occasionally, skin irritation problems have also been reported by swimmers exposed to certain strains of the marine cyanobacterium *Trichodesmium*, as well as dense raphidophyte blooms of *Heterosigma akashiwo* (Falconer, pers. com.).

There is little information on the adverse effects of dermal contact with marine waters containing algal species producing DSP, PSP, ASP and NSP toxins or those species of marine dinoflagellates and flagellates that have been associated with the death of fish and invertebrates. However, people with occupational exposure to waterways (Pocomoke estuary in Maryland, USA) in which toxin-producing *Pfiesteria* or *Pfiesteria*-like dinoflagellates were present were found to be at risk of developing a reversible clinical syndrome characterized by difficulties with learning and higher cognitive function. The risk of illness appeared to be directly related to the degree of exposure (both dermal and exposure to aerosolized spray from the water) (Grattan et al., 1998). CDC (1997) noted that clinical features from exposure to *Pfiesteria piscicida* and related organisms include memory loss, confusion and acute skin burning.

7.2 Exposure through ingestion (of water or scum)

Nodularia spumigena was the first cyanobacterium recognized to cause animal death (Francis, 1878). The toxin produced by *N. spumigena*, called nodularin, is a cyclic pentapeptide. Nodularin is a hepatotoxin, in that it induces massive haemorrhages in the liver of mammals and causes disruption of the liver structure; it also has some effects on the kidneys (Eriksson et al., 1988; Sandström et al., 1990). Nodularin acts by inhibiting serine–threonine protein phosphatases (Fujiki et al., 1996). In the 19th century, several toxic blooms and accumulations of *Nodularia spumigena* were registered. Published literature relates to blooms of *N. spumigena* associated with poisoning of ducks (Kalbe & Tiess, 1964), dogs (Edler et al., 1985; Nehring, 1993), young cattle (Gussmann et al., 1985) and sheep (Main et al., 1977). To date, there have been no reports of human poisoning by *N. spumigena*, but humans may be as susceptible to the toxins as other mammals. Therefore, it is possible that small children, in particular, may accidentally ingest toxic material in quantities with potentially serious consequences.

Other than the study referred to in section 7.1, there is no evidence for adverse effects of ingestion of marine waters containing algal species producing DSP, PSP, ASP and NSP toxins, etc.

Some species of cyanobacteria are capable of causing dense scums, which contain high concentrations of cells. Since most toxin is intracellular, scums caused by toxigenic strains may contain elevated concentrations of toxin. The existence of a cyanobacterial scum caused by a toxigenic species represents an increased human health hazard. Scums are less of a problem in marine water than in fresh water, as the frequency of occurrence of scums is higher in lakes than in coastal areas.

7.3 Exposure through inhalation

Inhalation of a sea spray aerosol containing fragments of marine dinoflagellate cells and/or toxins (e.g., brevetoxins) released into the surf by lysed algae can be harmful to humans (Baden et al., 1984; Scoging, 1991). Brevetoxins are produced by the unarmoured marine dinoflagellate *Gymnodinium breve* (now referred to as *Karenia brevis*). For many years, these blooms were reported only from the south-east USA and eastern Mexico (Steidinger, 1993), but similar problems have now been reported in New Zealand (Fernandez & Cembella, 1995), which were thought to have been caused by *Karenia mikimotoi*. From 1998 to 2001 summer blooms of *Otreopsis ovata* occurred in the Apuan (Tuscany, Italy) benthic seawaters (Sansoni et al., 2002), with major consequences to the benthic flora. In 1998, on the tract of land inland from the bloom-affected area, some 100 people reported symptoms including coughing, sneezing and, in some cases, fever, which were associated with the inhalation of sea spray aerosol.

The signs and symptoms of exposure to brevetoxins by inhalation are severe irritation of conjunctivae and mucus membranes (particularly of the nose) followed by persistent coughing and sneezing and tingling of the lips. The asthma-like effects are not usually observed more than a few kilometres inland (Pierce, 1986).

7.4 Identification of marine toxic algae and cyanobacteria

Detailed information on sampling, identification and cell counts are described in Hallegraeff et al. (2003) for marine toxic phytoplankton and in Chorus & Bartram (1999) for cyanobacteria. Immunoassays are currently the most sensitive and specific methods for rapid screening of samples for microcystins, which are toxins produced by certain cyanobacteria (Ueno et al., 1996), they can also be used for algal toxins. These methods have also been developed for PSP toxins (Cembella et al., 1995), DSP toxins (Levine et al., 1988; Usagawa et al., 1989) as well as ASP and NSP toxins, although in recreational water health effects are not thought to be due to the toxins, but are more likely to be caused by different (largely uncharacterised) compounds, such as lipopolysaccharides.

In most cases, the identification of an algal or cyanobacterial species is not sufficient to establish whether or not it is toxic, because a number of strains with different toxicity may belong to the same species. As a consequence, in order to ascertain whether the identified species includes toxic strains, there is a need to characterize the toxicity. The most commonly employed method is the mouse bioassay, which has been successfully applied in the cases of cyanotoxins (Falconer, 1993), PSP toxins (WHO, 1984), NSP toxins (McFarren et al., 1960) and DSP toxins (Yasumoto et al., 1984). Toxicity is tested by intraperitoneal injection followed by 24-hour observation. This method is not specific but within a few hours provides a measure of the total toxicity. The mouse assay is not sensitive enough for testing ASP toxins. Many analytical methods based on high-performance liquid chromatography are now available to determine the occurrence of cyanotoxins (Lawton et al., 1994; Chorus & Bartram, 1999) as well as specific ASP (Lawrence et al., 1989), DSP (Lee et al., 1987), NSP (Pierce et al., 1985) and PSP (Sullivan & Wekell, 1987) toxins.

7.5 Guideline values

Available data indicate that the risk for human health associated with the occurrence of marine toxic algae or cyanobacteria during recreational activities is limited to a few species and geographical areas. As a result, it is inappropriate to recommend specific guideline values, although authorities should be aware of the potential hazard and act accordingly.

7.6 Precautionary measures

7.6.1 Monitoring

Within areas subject to the occurrence of marine toxic algae or cyanobacteria, it is important to carry out adequate monitoring activities and give information to the human population potentially affected. Monitoring programmes should be planned with the aim of preventing human exposure in areas affected by blooms of toxic algae or cyanobacteria. In some cases, satellite imagery can be used as a part of a proactive monitoring programme. For example, movements of the Gulf Stream and subsequent elevated water temperatures play a key role in *Gymnodinium breve* blooms; Gulf Stream temperatures monitored by remote sensing of infrared radiation can provide information on the likelihood of a bloom and its subsequent movement (Hungerford & Wekell, 1993).

Long data records on phytoplankton populations, toxic and otherwise, may contribute to a more comprehensive understanding of phytoplankton dynamics and ecosystem function, which could lead to more efficient monitoring. If, for instance, long time series of data concerning phytoplankton populations exist, it would be possible to decide if a species that has suddenly appeared is new to the area or if endemic species have become toxic. Important supporting parameters include temperature, salinity, chlorophyll (phytoplankton biomass) and surface current circulation (transport of harmful algae). Knowledge of the temporal and geographic distribution of inorganic nutrients and their sources, as well as other phytoplankton growth factors, are also important when planning and operating a monitoring programme (Andersen, 1996).

When conditions favourable to algal or cyanobacterial blooms are recognized, monitoring activities should be intensified and should include taxonomic ranking of potentially toxic species and eventually analysis of the algal toxins (Hallegraeff et al., 2003).

7.6.2 Information

In affected areas, it is appropriate to provide general practitioners and medical clinics with information regarding the health problems potentially associated with algal blooms and toxic algae, the diagnosis and treatment of poisonings, the surveillance of groups of people who could be at risk and procedures for reporting to public health authorities. Health information should also be made available to the general public and to recreational water users in particular. Information may be disseminated through various means, including schools, on-site notices, mass media and specific brochures. These should contain information about algal blooms and toxic algae, the possible health effects, reporting procedures for any health problems thought to be possibly linked with water-based recreation and recommended protective measures.

As a precaution, the following guidance is recommended for potentially affected areas and should be included in public information:

- Avoid areas with visible algal concentrations and/or algal scums in the sea as well as on the shore. Direct contact and swallowing appreciable amounts are associated with the highest health risk.
- On the beach, avoid sitting downwind of any algal material drying on the shore, which could form an aerosol and be inhaled (particularly in areas with *Gymnodinium breve* blooms).
- If sailing, windsurfing or undertaking any other activity likely to involve water immersion in the presence of algal blooms, wear clothing that is close fitting in the openings. The use of wet suits for water sports may result in a greater risk of rashes, because algal material in the water trapped inside the wet suit will be in contact with the skin for long periods of time.
- After coming ashore, shower or wash yourself down to remove any algal material.
- Wash and dry all clothing and equipment after any contact with algal blooms and scum.
- If any health effects are subsequently experienced and whatever the nature of the exposure, seek medical advice.

In some areas, information on harmful algal blooms is distributed rapidly to users of the monitoring system by telephone, telephone answering machine, fax, E-mail and/or Internet (e.g., the Baltic Sea Alg@line, found at http://www2.fimr.fi/project/algaline/algaetu.htm) (Andersen, 1996).

7.6.3 Prevention of marine algal blooms

There have been several attempts to develop practical methods for controlling algal blooms. The use of clays, herbicides, metals, chelators, artificial turbulence, dinoflagellate parasites and zooplankton all have been the subject of research. Unfortunately, many of these methods are not practical and may have adverse ecological side-effects.

Algal blooms result from a complex interaction between hydrographic, meteorological, biological and chemical conditions, of which only a few can be controlled. Without essential nutrients, principally nitrates and phosphates, algae will usually not reach bloom proportions. Excessive nutrient input from land-based sources is one of the most influential promoting factors, and minimization of nutrient availability will often contribute to controlling algal growth.

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CHAPTER 8
Algae and cyanobacteria in fresh water

The term algae refers to microscopically small, unicellular organisms, some of which form colonies and thus reach sizes visible to the naked eye as minute green particles. These organisms are usually finely dispersed throughout the water and may cause considerable turbidity if they attain high densities. Cyanobacteria are organisms with some characteristics of bacteria and some of algae. They are similar to algae in size and, unlike other bacteria, they contain blue-green and green pigments and can perform photosynthesis. Therefore, they are also termed blue-green algae (although they usually appear more green than blue).

Human activities (e.g., agricultural runoff, inadequate sewage treatment, runoff from roads) have led to excessive fertilization (eutrophication) of many water bodies. This has led to the excessive proliferation of algae and cyanobacteria in fresh water and thus has had a considerable impact upon recreational water quality. In temperate climates, cyanobacterial dominance is most pronounced during the summer months, which coincides with the period when the demand for recreational water is highest.

Livestock poisonings led to the study of cyanobacterial toxicity, and the chemical structures of a number of cyanobacterial toxins (cyanotoxins) have been identified and their mechanisms of toxicity established. In contrast, toxic metabolites from freshwater algae have scarcely been investigated, but toxicity has been shown for freshwater species of Dinophyceae and also the brackish water Prymnesiophyceae and an ichthyotoxic species (*Peridinium polonicum*) has been detected in European lakes (Pazos et al., in press; Oshima et al., 1989). As marine species of these genera often contain toxins, it is reasonable to expect toxic species among these groups in fresh waters as well.

Although many species of freshwater algae proliferate quite intensively in eutrophic waters, they do not accumulate to form dense surface scums (often termed blooms) of extremely high cell density, as do some cyanobacteria. The toxins that freshwater algae may contain are therefore not accumulated to concentrations likely to become hazardous to human health or livestock. For these reasons, this chapter will focus primarily on the health impacts of cyanobacteria. More detailed coverage of cyanobacteria and human health is available in *Toxic Cyanobacteria in Water* (Chorus & Bartram, 1999).

8.1 Occurrence of toxic cyanobacteria

Toxic cyanobacteria are found worldwide in inland and coastal water environments. At least 46 species have been shown to cause toxic effects in vertebrates (Sivonen & Jones, 1999). The most common toxic cyanobacteria in fresh water are *Microcystis* spp., *Cylindrospermopsis raciborskii*, *Planktothrix* (syn. *Oscillatoria*) *rubescens*, *Synechococcus* spp., *Planktothrix* (syn. *Oscillatoria*) *agardhii*, *Gloeotrichia* spp., *Anabaena* spp., *Lyngbya* spp., *Aphanizomenon* spp., *Nostoc* spp., some *Oscillatoria* spp., *Schizothrix* spp. and *Synechocystis* spp. Toxicity cannot be excluded for further species and genera. As research broadens and covers more regions over the globe, additional toxic species are likely to be found. Therefore, it is prudent to presume a toxic potential in any cyanobacterial population.

The most widespread cyanobacterial toxins are microcystins and neurotoxins (see section 8.3). Some species contain neurotoxin and microcystin simultaneously. Field populations of the most common bloom-forming genus, *Microcystis*, are almost always toxic (Carmichael, 1995), but non-toxic strains do occur. Generally, toxicity is not a trait specific for certain species; rather, most species comprise toxic and non-toxic strains. For microcystins, it has been shown that toxicity of a strain depends on whether or not it contains the gene for microcystin production (Rouhiainen et al., 1995; Dittmann et al., 1996) and that field populations are a mixture of both genotypes with and without this gene (Kurmayer et al., 2002). Experience with cyanobacterial cultures also shows that microcystin production is a fairly constant trait of a given strain or genotype, only somewhat modified by environmental conditions (see various contributions in Chorus, 2001). While conditions leading to cyanobacterial proliferation are well understood (the physiological or biochemical function of toxins for the cyanobacteria is the subject of many hypotheses—Chorus & Bartram, 1999), the factors leading to the dominance of toxic strains over non-toxic ones are not.

Worldwide, about 60% of cyanobacterial samples investigated contain toxins (see section 8.4). The toxicity of a single bloom may, however, change in both time and space. Demonstrations of toxicity of the cyanobacterial population in a given lake do not necessarily imply an environmental or human hazard as long as the cells remain thinly dispersed. Mass developments and especially surface scums pose the risks.

8.2 Formation of cyanobacterial blooms

In contrast to true algae, many species of planktonic cyanobacteria possess specialized intracellular gas vesicles. Stacks of these minute (<300 nm) proteinaceous hollow cylinders maintain a gas-filled space in the cell, which enables the organism to regulate its buoyancy and thus to actively seek water depths with optimal growth conditions. However, regulation of buoyancy by changing the amount of gas in the vesicles is slow. Cells adapted to turbulent mixing by enlarged gas vesicles will take a few days to reduce their buoyancy in order to adapt to more quiescent conditions. Thus, especially when the weather changes from stormy to fine (i.e., mixing conditions in the water change from turbulent to strongly stratified), many excessively buoyant cells or colonies may accumulate at the surface. Light winds drive them to leeward shores and bays, where they form scums (Figure 8.1). In extreme cases, such agglomerations may become very dense and even acquire a gelatinous consistency. More frequently, they are seen as streaks or slimy scums that may even look like blue-green paint or jelly. Such situations may change rapidly, within hours, or may remain unchanged for weeks (Chorus & Bartram, 1999).

Scums can be quickly broken by wave action and redispersed by renewed wind mixing. However, especially in shallow bays, scum material may take a long time to disperse, as a result of either wave wash or, ultimately, disintegration of the cells. Dying and lysing cells release their contents into the water, where pigments may adopt a copper-blue colour. Bacterial decomposition leads to rapid putrefaction of the material. The in-shore deposits are often repulsive and potentially very toxic.

Whereas agglomerations of cyanobacteria are usually caused by planktonic species in eutrophic waters, benthic mats in oligotrophic waters (which are relatively poor in plant nutrients) occasionally also cause problems; these surface-covering mats can grow only in clear water, in which sunlight penetrates to the bottom. During sunny days, their photosynthesis may lead to high rates of oxygen production, forming bubbles that loosen parts of the mats and drive them to the surface. Mats of benthic cyanobacteria washed to the shore and scavenged by dogs have been lethal (Edwards et al., 1992), and cattle deaths on Swiss alpine meadows may also be caused by benthic cyanobacteria (Mez et al., 1997, 1998). Although relevant for pets and live-stock, the human health impact of these cyanobacteria on beaches will be consider-ably lower than that of scums in the water. Awareness of the potential toxicity of such beached mats is, however, important, because they accumulate along shores of clear waters usually not recognized as potentially producing harmful cyanobacteria or algae.

8.3 Cyanotoxins

Progress in analytical chemistry has enabled the isolation and structural identification of three neurotoxins with somewhat different modes of blocking neuronal signal transmission (anatoxin-a, anatoxin-a(s) and saxitoxins), one general cytotoxin, which inhibits protein synthesis (cylindrospermopsin), and a group of toxins termed micro-cystins (or nodularins, found in brackish waters), which inhibit protein phosphatases. Phosphatase inhibition is generally cytotoxic, but microcystins are primarily hepato-toxic, because they use the bile acid carrier to pass through cell membranes. These toxins were named after the organism from which they were first isolated, but most of them have been found in a wider array of genera, and some species contain more than one toxin or both microcystins and neurotoxins.

Although the toxins listed in Table 8.1 are assumed to be the substances most significant for human health, it is unlikely that all of the important cyanotoxins have been discovered. Yoo et al. (1995) pointed out that an increasing variety of in-dividual toxins is continually being discovered. Numerous pharmacological working groups are conducting research for pharmacologically active substances from cyanobacteria (e.g., Mundt & Teuscher, 1988; Falch et al., 1995). Fastner et al.

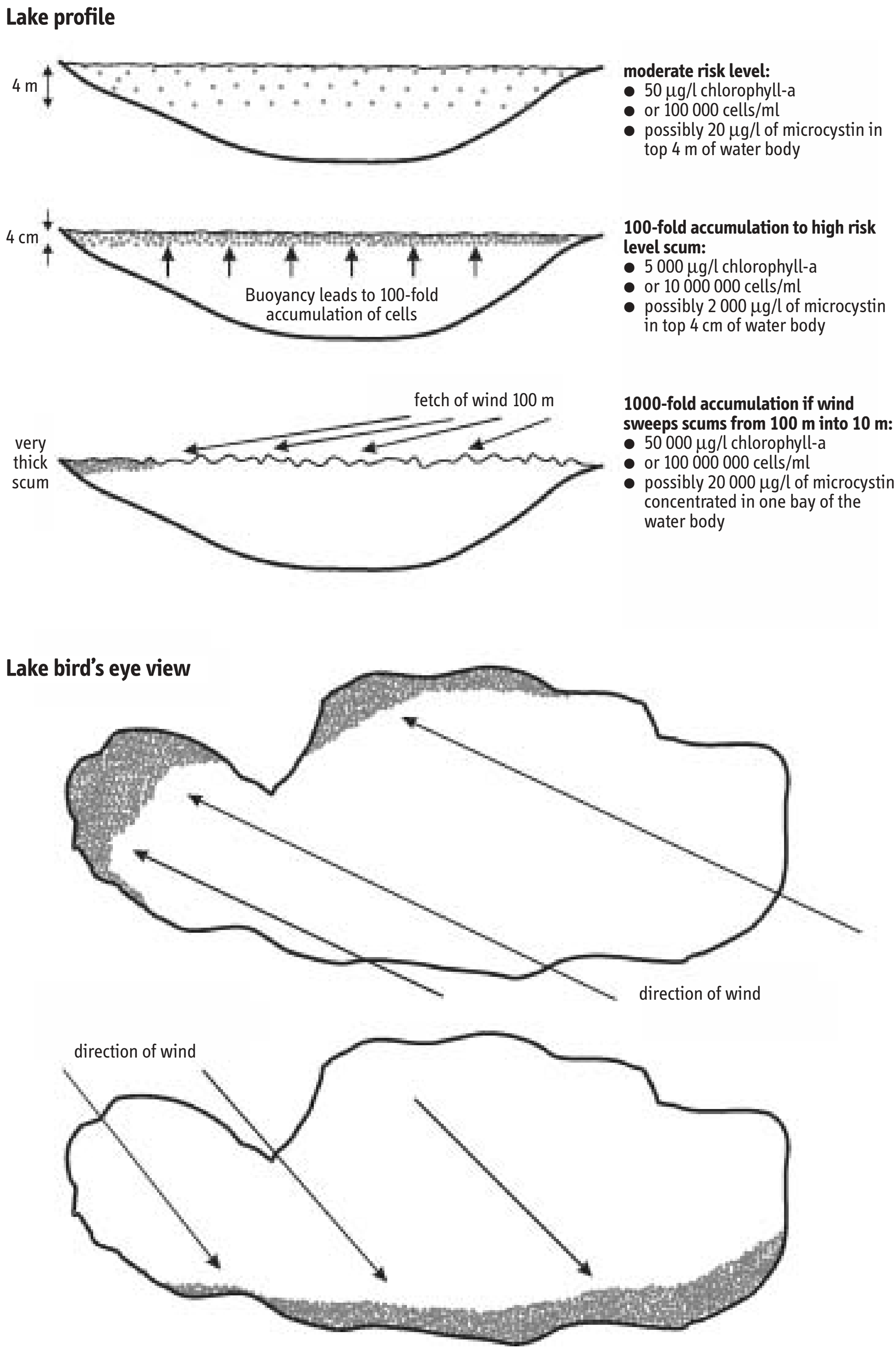

FIGURE 8.1. SCHEMATIC ILLUSTRATION OF SCUM FORMATION CHANGING THE CYANOTOXIN RISK FROM MODERATE TO HIGH (CHORUS & BARTRAM, 1999)

(2001) showed that primary rat hepatocytes reacted to microcystins in crude extracts of some strains of cyanobacteria in close correlation to their content of microcystins, but that this reaction was further enhanced by an unknown factor. Oberemm et al. (1997) demonstrated substantial toxicity of cyanobacterial crude extracts to fish eggs, the effects not being due to the content of any of the known cyanotoxins.

TABLE 8.1. CYANOBACTERIAL TOXINS AND THEIR ACUTE TOXICITY[a]

Cyanotoxins	LD_{50} (i.p. mouse)[b] of pure toxin (μg/kg)	Taxa known to produce the toxin(s)	Mechanism of toxicity
Protein phosphatase blockers (cyclic peptides with the amino acid ADDA)			
Microcystins in general (~60 known congeners)	45–>1000	*Microcystis*, *Planktothrix*, *Oscillatoria*, *Nostoc* *Anabaena*, *Anabaenopsis* *Hapalosiphon*	all block protein phosphatases by covalent binding and cause haemorrhaging of the liver; cumulative damage may occur
Microcystin-LR	60 (25–125)		
Microcystin-YR	70		
Microcystin-RR	300–600		
Nodularin	30–50	*Nodularia spumigena*	
Neurotoxins			
Anatoxin-a (alkaloid)	250	*Anabaena*, *Oscillatoria*, *Aphanizomenon*, *Cylindrospermum*	blocks post-synaptic depolarization
Anatoxin-a(s) (unique organophosphate)	40	known only from two species of *Anabaena*	blocks acetylcholinesterase
Saxitoxins (carbamate alkaloids)	10–30	*Aphanizomenon*, *Anabaena*, *Lyngbya*, *Cylindrospermopsis raciborskii*	block sodium channels
Cytotoxin			
Cylindrospermopsin (alkaloid)	2100 in 1 day 200 in 5–6 days	*Cylindrospermopsis raciborskii*	blocks protein synthesis; substantial cumulative toxicity

[a] derived from Turner et al., 1990; Kuiper-Goodman et al., 1999; Sivonen & Jones, 1999.

[b] LD_{50} = lethal dose$_{50}$ (the dose of a chemical that will, on average, kill 50% of a group of experimental animals); i.p. = intraperitoneal.

8.3.1 Microcystins

Microcystins are the most frequently occurring and widespread of the cyanotoxins. They are cyclic heptapeptides containing a specific amino acid (ADDA) side chain which, to date, has been found only in microcystins and nodularin (a cyclic pentapeptide toxin of cyanobacteria from brackish waters). About 70 structural analogues of microcystin have been identified (Rinehart et al., 1994; Sivonen & Jones, 1999). They vary with respect to methyl groups and two amino acids within the ring. This has consequences for the tertiary structure of the molecule and results in pronounced differences in toxicity as well as in hydrophobic/hydrophilic properties. Microcystins block protein phosphatases 1 and 2a (which are important molecular switches in all eukaryotic cells) with an irreversible covalent bond (MacKintosh et al., 1990).

The chief pathway for microcystins entry into cells is the bile acid carrier, which is found in liver cells and, to a lesser extent, in intestinal epithelia (Falconer, 1993). For vertebrates, a lethal dose of microcystin causes death by liver necrosis within hours up to a few days. Evidence for the permeability of other cell membranes to microcystins is controversial. It is possible that hydrophobic structural analogues can penetrate into some cell types even without the bile acid carrier (Codd, 1995). In addition, Fitzgeorge et al. (1994) published evidence for disruption of nasal tissues by the common hydrophilic analogue microcystin-LR. While toxicity by oral uptake is generally at least an order of magnitude lower than toxicity by intraperitoneal (i.p.) injection, intranasal application in these experiments was as toxic as i.p. injection, and membrane damage by microcystin enhanced the toxicity of anatoxin-a. This uptake route may be relevant for water sports activities that lead to inhalation of spray and droplets, such as waterskiing.

Microcystins are found in most populations of *Microcystis* spp. (which frequently form surface scums) and in strains of some species of *Anabaena* (which may also form scums). High microcystin content has also been observed in *Planktothrix* (syn. *Oscillatoria*) *agardhii* and *P. rubescens* (Fastner et al., 1999). *P. agardhii*, however, never forms scums, and where it occurs *P. rubescens* does not usually form scums during the recreational water use season, thus reducing the hazard to swimmers.

Fitzgeorge et al. (1994) demonstrated that microcystin toxicity is cumulative: a single oral dose resulted in no increase in liver weight (which is a measure of liver damage), whereas the same dose applied daily over seven days caused an increase in liver weight of 84% and thus had the same effect as a single oral dose 16 times as large. This may be explained by the irreversible covalent bond between microcystin and the protein phosphatases and subsequent substantial damage to cell structure (Falconer, 1993). Healing of the liver probably requires growth of new liver cells. Subacute liver injury is likely to go unnoticed for two reasons:

- liver injury results in externally noticeable symptoms only when it is severe;
- acute dose–response curves for microcystins are steep. Therefore, little acute damage may occur until levels close to severe acute toxicity are reached. As a result of the lack of apparent symptoms at moderate exposure, exposure is likely to be continued by people uninformed of the risk (e.g., for consecutive days of a holiday or a hot spell), which will increase the risk of cumulative liver damage.

There are two aspects of chronic microcystin damage to the liver—progressive active liver injury (Falconer et al., 1988) and the potential for promotion of tumour growth. Tumour-promoting activity of microcystins is well documented, although microcystins alone have not been demonstrated to be carcinogenic. Promotion of mouse skin tumours has been shown after initiation by topical exposure to a carcinogen (dimethylbenzanthracene) followed by ingestion of a *Microcystis aeruginosa* extract (Falconer & Buckley, 1989; Falconer & Humpage, 1996). In rat liver studies, the appearance of pre-neoplastic liver foci and nodules was promoted by pure microcystin-LR in a protocol involving one i.p. dose of diethylnitrosamine and i.p. doses of microcystin-LR over several weeks (Nishiwaki-Matsushima et al., 1992).

Studies on the mechanism of cell toxicity showed that microcystin interferes with cell structure and mitosis, and this may help to explain the tumour-promoting activity (Falconer & Yeung, 1992; Kaja, 1995). It has been suggested that, in China, cases of liver tumours in humans may be associated with the presence of cyanotoxins in drinking water (Ueno et al., 1996).

8.3.2 Neurotoxins

Irrespective of somewhat different modes of action, all three neurotoxins (Table 8.1) have the potential to be lethal by causing suffocation—anatoxin-a and a(s) through cramps, saxitoxins through paralysis. However, no human deaths from exposure to neurotoxins associated with recreational use of water are known.

Anatoxin-a(s) is the only known naturally occurring organophosphate cholinesterase inhibitor and causes strong salivation (the 's' in its name stands for salivation), cramps, tremor, diarrhoea, vomiting and an extremely rapid death (within minutes). Saxitoxins and anatoxin-a(s) are among the most neurotoxic substances known. However, evidence is accumulating that in lakes and rivers they do not occur as frequently as microcystins. This applies especially to anatoxin-a(s): to date, it has been found only in a small number of *Anabaena* blooms in North America. Furthermore, concentrations even of these highly toxic substances in scums will scarcely reach levels acutely neurotoxic to a human ingesting a mouthful. In contrast, neurotoxicity may be experienced by livestock that drink many litres of contaminated water and pets—especially dogs—that gather scum material in their fur and ingest it through grooming with the tongue.

After ingestion of a sublethal dose of these neurotoxins, recovery appears to be complete, and no chronic effects have been observed to date. For these reasons, the neurotoxins are a hazard to be aware of when using waters populated with cyanobacteria for recreation. On the basis of current knowledge, however, it is reasonable to consider them less dangerous than microcystins or cylindrospermopsin, which may cause ongoing injury.

8.3.3 Cylindrospermopsin

Cylindrospermopsin is an alkaloid isolated from *Cylindrospermopsis raciborskii* (Ohtani et al., 1992). It is a general cytotoxin that blocks protein synthesis, the first clinical symptoms being kidney and liver failure. In contrast to the pure toxin, crude extracts of the organism also cause injury to the lungs, adrenals and intestine, indicating further, unknown toxins in the organism. Clinical symptoms may become manifest only several days after exposure, so it will often be difficult to determine a cause-effect relationship. Patients intoxicated with cylindrospermopsin via drinking-water in an incident in Australia escaped death only through skilled and intensive hospital care (Falconer, 1996). *Cylindrospermopsis raciborskii* is considered to be a tropical and subtropical species, but has been reported to form blooms as far north as Vienna (Roschitz, 1996). Substantial populations have been reported from north-eastern Germany (C. Wiedner, personal communication), and generally *C. raciborskii* appears to be invading temperate regions (Padisák, 1997). Thus, cylindrospermopsin may become relevant in temperate zones in future.

8.3.4 Analysis

From the 1960s to the end of the 1980s, detection of cyanotoxin was primarily performed with the mouse bioassay (outlined in section 7.4), conducted to assess the safety of drinking-water supplies. Due to the high cost and lack of approved laboratories as well as ethical limitations of applicability, this method is not suitable for large screening or monitoring programmes. However, effective methods of chemical analysis are now available for the known cyanotoxins, and sensitive immunoassays as well as enzyme assays have become commercially available for the most important ones (e.g., microcystins and saxitoxins). This opens new possibilities for screening programmes targeted at assessment of the potential risk, as well as for regular surveillance (Chorus & Bartram, 1999).

8.4 Evidence for toxicity of cyanobacteria

Observations of lethal poisoning of animals drinking from water with mass developments of cyanobacteria are numerous. The first documented case of a lethal intoxication of livestock after drinking water from a lake heavily populated with cyanobacteria was published in the 1800s (Francis, 1878), and cases recorded since have included sheep, cattle, horses, pigs, dogs, fish, rodents, amphibians, waterfowl, bats, zebras and rhinoceroses (Codd et al., 1989). Dogs have died after grooming accumulations of cyanobacteria out of their fur or after ingesting beached mats of benthic cyanobacteria.

A number of human deaths have been reported through exposure to cyanobacterial toxins through renal dialysis (Carmichael, 1996; Jochimsen et al., 1998), and also implicated in drinking-water (Teixera et al., 1993). Health impairments are also seen from numerous anecdotal reports of irritations of the skin and/or mucous membranes and from documented cases of illness after exposure through drinking-water as well as accidental swallowing or aspiration of scum material. Other sources of information include toxicological data from animal experiments and data on concentrations of cyanobacterial toxins in waters used for drinking-water purposes and recreation.

Human health risk from exposure to cyanobacteria and their toxins during recreational water use arises through three routes of exposure:

- direct contact of exposed parts of the body, including sensitive areas such as the ears, eyes, mouth and throat, and the areas covered by a bathing suit (which may collect cell material);
- accidental uptake of water containing cells by swallowing; and
- uptake of water containing cells by aspiration (inhalation).

Different cyanobacterial metabolites are likely to be involved in evoking symptoms associated with these exposure routes.

8.4.1 *Exposure through dermal contact*

Allergic or irritative dermal reactions of varying severity have been reported from a number of freshwater cyanobacterial genera (*Anabaena*, *Aphanizomenon*, *Nodularia*, *Oscillatoria*, *Gloeotrichia*) after recreational exposure. Bathing suits and particularly wet suits tend to aggravate such effects by accumulating cyanobacterial material and enhancing disruption of cells and liberation of cell content. Reports from the USA have recorded allergic reactions from recreational exposure, and the cyanobacterial pigment phycocyanin has been shown to be responsible in one case (Cohen & Reif, 1953). In addition, cutaneous sensitization to cyanobacteria has been documented. Skin irritations were a frequent symptom found in an epidemiological study by Pilotto et al. (1997) on health effects after recreational exposure to cyanobacteria. This study showed correlation to cyanobacterial cell density and duration of exposure, but not to microcystin concentrations. It is probable that these symptoms are not due to the recognized cyanotoxins listed in Table 8.1, but rather to currently largely unidentified substances.

Allergic reactions to cyanobacteria are frequently reported at the level of "anecdotal evidence" from eutrophic recreational waters, and it has been claimed that "allergic reactions to cyanobacteria are relatively common" (Yoo et al., 1995, p. 77). However, these have been rarely investigated in scientific studies or published. Among the small number of publications available, Heise (1949) described ocular and nasal irritations in swimmers exposed to Oscillatoriaceae. McElhenny et al. (1962) applied extracts from four different algal species, including cyanobacteria and Chlorophyceae (as intracutaneous skin tests), to 20 non-allergic children, none of who responded, and to 120 children with respiratory allergies, 98 of who showed clear positive reactions to at least one of the test strains. Mittal et al. (1979) tested 4000 patients in India with respiratory allergies, 25% of who showed positive reactions to either cyanobacteria or Chlorophyceae, or to both.

Allergic reactions are not confined to cyanobacteria, but may also be evoked by planktonic algae. However, allergic reactions require elevated cell densities in water used for swimming, and mass developments in fresh waters are most frequently due to cyanobacteria. Furthermore, other groups of algae do not accumulate as surface scums, and therefore their metabolites will not occur in comparably high concentrations. Thus, cyanobacteria are likely to be the most frequently occurring cause of such reactions.

8.4.2 *Exposure through ingestion or aspiration*

Swallowing or aspiration was the exposure route in most of the documented cases of human illness that have been associated with cyanobacteria (Box 8.1). In contrast to dermal contact, uptake of cyanobacteria involves a risk of intoxication by the cyanotoxins listed in Table 8.1. This risk may be estimated from cell density, cellular toxin content and known mechanisms of toxicity. Acute mechanisms of toxicity are well known for the neurotoxins and microcystins, and some information is available to estimate risks due to repeated or chronic exposure.

BOX 8.1 HUMAN ILLNESS ASSOCIATED WITH EXPOSURE TO CYANOBACTERIA

ILLNESS ATTRIBUTED TO CYANOTOXINS IN RECREATIONAL WATER

1959: **Canada:** In spite of a kill of livestock and warnings against recreational use, people still swam in a lake infested with cyanobacteria. Thirteen persons became ill (headaches, nausea, muscular pains, painful diarrhoea). In the excreta of one patient—a medical doctor who had accidentally ingested water—numerous cells of *Microcystis* spp. and some trichomes of *Anabaena circinalis* could be identified (Dillenberg & Dehnel, 1960).

1989: **England:** Ten out of 20 soldiers became ill after swimming and canoe training in water with a heavy bloom of *Microcystis* spp.; two developed severe pneumonia attributed to the inhalation of a *Microcystis* toxin and needed hospitalization and intensive care (Turner et al., 1990). Swimming skills and the amount of water ingested appear to have been related to the degree of illness.

1995: **Australia:** Epidemiological evidence of adverse health effects after recreational water contact from a prospective study involving 852 participants showed elevated incidence of diarrhoea, vomiting, flu symptoms, skin rashes, mouth ulcers, fevers, and eye or ear irritations within 2–7 days after exposure (Pilotto et al., 1997). Symptoms increased significantly with duration of water contact and density of cyanobacterial cells, but were not related to the content of known cyanotoxins.

ILLNESS ATTRIBUTED TO CYANOTOXINS IN DRINKING-WATER

1931: **USA:** A massive *Microcystis* bloom in the Ohio and Potomac rivers caused illness of 5000–8000 people whose drinking-water was taken from these rivers. Drinking-water treatment by precipitation, filtration and chlorination was not sufficient to remove the toxins (Tisdale, 1931).

1968: **USA:** Numerous cases of gastrointestinal illness after exposure to mass developments of cyanobacteria were compiled by Schwimmer & Schwimmer (1968).

1979: **Australia:** Combating a bloom of *Cylindrospermopsis raciborskii* in a drinking-water reservoir on Palm Island with copper sulfate led to liberation of toxins from the cells into the water and resulted in serious illness (with hospitalization) of 141 people supplied from this reservoir (Falconer, 1993, 1994).

1981: **Australia:** In the city of Armidale, liver enzyme activities (a sign of exposure to toxic agents) were found to be elevated in the blood of the population supplied from surface water polluted by *Microcystis* spp. (Falconer et al., 1983).

1985: **USA:** Carmichael (1994) compiled case studies on nausea, vomiting, diarrhoea, fever and eye, ear and throat infections after exposure to mass developments of cyanobacteria.

1988: **Brazil:** Following the flooding of the Itaparica Dam in Bahia State, some 2000 cases of gastroenteritis were reported over a 42-day period, of which 88 resulted in death. Investigation of potential causes of this epidemic eliminated pathogens and identified a very high population of toxic cyanobacteria in the drinking-water supply in the affected areas (Teixera et al., 1993).

1993: **China:** The incidence of liver cancer was related to water sources and was significantly higher for populations using cyanobacteria-infested surface waters than for those drinking groundwater (Yu, 1995).

Continued

1994: **Sweden:** Illegal use of untreated river water in a sugar factory led to an accidental cross-connection with the drinking-water supply for an uncertain number of hours. The river water was densely populated by *Planktothrix agardhii* and samples taken a few days before and a few days after the incident showed these cyanobacteria to contain microcystins. In total, 121 of 304 inhabitants of the village (as well as some dogs and cats) became ill with vomiting, diarrhoea, muscular cramps and nausea (Anadotter et al., 2001).

ILLNESS ATTRIBUTED TO CYANOTOXINS IN WATER USED FOR HAEMODIALYSIS

1975: **USA:** Endotoxic shock of 23 dialysis patients in Washington, DC, was attributed to a cyanobacterial bloom in a drinking-water reservoir (Hindman et al., 1975).

1996: **Brazil:** In total, 131 dialysis patients were exposed to microcystins from the water used for dialysis; 56 died. At least 44 of these victims showed the typical symptoms associated with microcystin, now referred to as "Caruaru Syndrome", and liver microcystin content corresponded to that of laboratory animals having received a lethal dose of microcystin (Jochimsen et al., 1998).

Most documented cases of human injury through cyanotoxins involved exposure through drinking-water, and they demonstrate that humans have become ill—in some cases seriously—through ingestion or aspiration of toxic cyanobacteria. The low number of reported cases may be due to lack of knowledge about the toxicity of cyanobacteria; neither patients nor doctors associate symptoms with this cause. Symptoms reported include "abdominal pain, nausea, vomiting, diarrhoea, sore throat, dry cough, headache, blistering of the mouth, atypical pneumonia, and elevated liver enzymes in the serum, especially gamma-glutamyl transferase" (Carmichael, 1995, p. 9), as well as hay fever symptoms, dizziness, fatigue, and skin and eye irritations; these symptoms are likely to have diverse causes, with several classes of toxin and genera of cyanobacteria involved.

8.5 Evidence for toxicity of algae

Systematic investigation of the toxicity of freshwater algae is required, particularly for species related to toxic marine taxa (dinoflagellates, diatoms, haptophytes). However, as discussed above, freshwater algae are considerably less likely to pose recreational health hazards comparable to those of scum-forming cyanobacteria, because algae lack similarly effective mechanisms of accumulation.

Oshima et al. (1989) isolated and identified three ichthyotoxins (polonicumtoxins A, B and C) from a dinoflagellate, *Peridinium polonicum*. Toxicity in the mouse bioassay was 1.5–2 mg/kg, i.e., several orders of magnitude lower than the toxicity of microcystin-LR. The Ames test showed no mutagenicity, but the authors emphasized the need for studies on chronic toxicity to evaluate the potential health risk of these toxins.

Allergic reactions have been investigated as outlined in section 8.4.1. Skin reactions in response to a bloom of *Uroglena* spp. were observed in a small number of swimmers. These reactions were especially pronounced under bathing suits, where cells accumulated and were partially disrupted during swimming (Chorus, 1993). Divers frequently complain of dermal reactions to algal material accumulating under their wet suits, which tend to act as a strainer that lets out water but collects algae between skin and suit. Pressure and friction between fabric and skin lead to cell disruption, liberation of content and intensified dermal exposure, not only to algal cell wall material, but also to substances otherwise largely confined within the cells.

One of the few reports involved the raphidophyte algal species *Gonyostomum semen* (related to *Heterosigma* mentioned in chapter 7), which may develop high population densities in slightly acidic waters and emits a slimy substance causing skin irritation and allergic reactions. In Sweden, occurrence of this species led to closure of a number of freshwater recreational sites (Cronberg et al., 1988).

8.6 Health risk evaluation

Documented evidence of significant human health impairment exists only for cyanobacteria, not for freshwater algae. Data from surveys in a number of countries show that toxicity is to be expected in about 60% of all samples containing cyanobacteria (Table 8.2). Generally, the liver-toxic microcystins appear to be more common than neurotoxins, although the latter have caused severe animal poisonings in North America, Europe and Australia. Blooms containing cylindrospermopsin have been reported from Australia, Hungary, Japan, Israel and Germany.

While a general picture of the frequency of occurrence of cyanotoxins associated with certain cyanobacterial taxa is emerging, it is less clear what cyanotoxin levels may be expected in recreational waters containing cyanobacteria. Very few studies have addressed the variability of toxin content in the course of the development of cyanobacterial populations (Benndorf & Henning, 1989; Jungmann, 1995; Kotak et al., 1995; Fastner et al., 1999), although this knowledge would be important for risk assessment. This is because the cumulative toxicity of microcystins means that hazards are greatest for persons exposed regularly over a number of days or weeks. For management of recreational waters, a few years of regular investigation of the toxin content of prevalent cyanobacterial blooms may provide information on the variability of toxin content in both time and space. If the toxin content proves to show little variation during several weeks or even months of blooming for certain key species, a basis for future predictions of cellular toxin content from frequent cell counts and only occasional toxin analysis may be established.

Most studies have focused on the quantity of toxins contained in the cells of the dominant cyanobacteria. If the cell density is known in addition to the toxin content per cell, toxin concentrations per litre of water can be calculated. A few studies have directly addressed concentrations per litre, and sensitive detection methods now allow direct determination of toxin concentrations per litre rather than requiring enrichment of cell material.

Generally, the cyanotoxin content of cells can reach levels of several milligrams per gram dry weight. This has been established for microcystins, nodularin, cylindrospermopsin, anatoxin-a and saxitoxins, the maximum being found for nodularin:

18 mg/g dry weight (Sivonen & Jones, 1999). If both toxin content and cell density or biomass of cyanobacteria per litre is known for a given water body, maximum toxin concentrations to be expected can be estimated from such data. As the toxic concentrations depend upon cell density, scum formation is critical in determining cell density. In one study, microcystin concentrations ranged from 0.01 to 0.35 mg/litre while the cyanobacteria were evenly dispersed (Fastner et al., 1999). However, sampling of shoreline scums of the same water bodies showed microcystin concentrations of more than 1 mg/litre in 7 of 34 samples, and maxima reached 24 mg/litre (Chorus & Fastner, 2001). Some commonly occurring species, such as *Planktothrix agardhii*, never form scums. The maximum reported microcystin concentration per litre of water for *P. agardhii* is 0.35 mg/litre (Fastner et al., 1999).

TABLE 8.2. FREQUENCIES OF MASS OCCURRENCES OF TOXIC CYANOBACTERIA IN FRESH WATERS[a]

Country	No. of samples tested	% of toxic samples
Australia	231	42
Australia	31	84[b]
Brazil	16	75
Canada, Alberta	24	66
Canada, Alberta	39	95
Canada, Alberta (three lakes)	226	74[b]
Canada, Saskatchewan	50	10
China	26	73
Czech Republic and Slovakia	63	82
Finland	215	44
France, Brittany	22	73[b]
Germany	533	72[b]
Germany	393	22
Former German Democratic Republic	10	70
Greece	18	?
Hungary	50	66
Japan	23	39
Netherlands	10	90
Portugal	30	60
Scandinavia	81	60
Denmark	296	82
Norway	64	92
Sweden	331	47
United Kingdom	50	48
United Kingdom	50	28[b]
USA, Minnesota	92	53
USA, Wisconsin	102	25
Mean		**59**

[a] From Sivonen & Jones (1999).

[b] High-performance liquid chromatography was used to determine the toxin content of the samples.

For practical purposes, the present state of knowledge implies that health authorities should regard any mass development of cyanobacteria as a potential health hazard.

8.7 Guideline values

As discussed above, approaches to recreational water safety should address the occurrence of cyanobacteria as such, because it is as yet unclear whether all important cyanotoxins have been identified, and the health outcomes observed after recreational exposure—particularly irritation of the skin and mucous membranes—are probably related to cyanobacterial substances other than the well known toxins listed in Table 8.1. Additionally, the particular hazard of liver damage by microcystins should be considered. In face of the difficulty of representative quantitative sampling due to the heterogeneous distribution of cyanobacteria in time and space, particularly with respect to scum formation and scum location, approaches should further include addressing the capacity of a water body to sustain large cyanobacterial populations.

Health impairments from cyanobacteria in recreational waters must be differentiated between the chiefly irritative symptoms caused by unknown cyanobacterial substances and the potentially more severe hazard of exposure to high concentrations of known cyanotoxins, particularly microcystins. A single guideline value therefore is not appropriate. Rather, a series of guideline values associated with incremental severity and probability of health effects is defined at three levels (Table 8.3).

8.7.1 *Relatively low probability of adverse health effects*

For protection from health outcomes not due to cyanotoxin toxicity, but rather to the irritative or allergenic effects of other cyanobacterial compounds, a guideline level of 20 000 cyanobacterial cells/ml (corresponding to 10 μg chlorophyll-a/litre under conditions of cyanobacterial dominance) can be derived from the prospective epidemiological study by Pilotto et al. (1997). Whereas the health outcomes reported in this study were related to cyanobacterial density and duration of exposure, they affected less than 30% of the individuals exposed. At this cyanobacterial density, 2–4 μg microcystin/litre may be expected if microcystin-producing cyanobacteria are dominant, with 10 μg/litre being possible with highly toxic blooms. This level is close to the WHO provisional drinking-water guideline value of 1 μg/litre for microcystin-LR (WHO, 1998), which is intended to be safe for lifelong consumption. Thus, health outcomes due to microcystin are unlikely, and providing information for visitors to swimming areas with this low-level risk is considered to be sufficient. Additionally, it is recommended that the authorities be informed in order to initiate further surveillance of the site. The results of the epidemiological study (Pilotto et al., 1997) reported some mild irritative effects at 5000 cells but the level of health effect and the small number of people affected were not considered to be a basis to justify action.

8.7.2 *Moderate probability of adverse health effects*

At higher concentrations of cyanobacterial cells, the probability of irritative symptoms is elevated. Additionally, cyanotoxins (usually cell-bound) may reach concentrations with potential health impact. To assess risk under these circumstances, the data used for the drinking-water provisional guideline value for microcystin-LR

TABLE 8.3. GUIDELINES FOR SAFE PRACTICE IN MANAGING RECREATIONAL WATERS[a]

Guidance level or situation	How guidance level derived	Health risks	Typical actions[b]
Relatively low probability of adverse health effects			
20 000 cyanobacterial cells/ml *or* 10 μg chlorophyll-a/litre with dominance of cyanobacteria	• From human bathing epidemiological study	• Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness	• Post on-site risk advisory signs • Inform relevant authorities
Moderate probability of adverse health effects			
100 000 cyanobacterial cells/ml *or* 50 μg chlorophyll-a/litre with dominance of cyanobacteria	• From provisional drinking-water guideline value for microcystin-LR[c] and data concerning other cyanotoxins	• Potential for long-term illness with some cyanobacterial species • Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness	• Watch for scums or conditions conducive to scums • Discourage swimming and further investigate hazard • Post on-site risk advisory signs • Inform relevant authorities
High probability of adverse health effects			
Cyanobacterial scum formation in areas where whole-body contact and/or risk of ingestion/aspiration occur	• Inference from oral animal lethal poisonings • Actual human illness case histories	• Potential for acute poisoning • Potential for long-term illness with some cyanobacterial species • Short-term adverse health outcomes, e.g., skin irritations, gastrointestinal illness	• Immediate action to control contact with scums; possible prohibition of swimming and other water contact activities • Public health follow-up investigation • Inform public and relevant authorities

[a] Derived from Chorus & Bartram, 1999.

[b] Actual action taken should be determined in light of extent of use and public health assessment of hazard.

[c] The provisional drinking-water guideline value for microcystin-LR is 1 μg/litre (WHO, 1998).

(WHO, 1998) may be applied. Swimmers involuntarily swallow some water while swimming, and the harm from ingestion of recreational water will be comparable to the harm from ingestion of water from a drinking-water supply with the same toxin content. For recreational water users with whole-body contact (see chapter 1), a swimmer can expect to ingest 100–200 ml of water in one session, sailboard riders and waterskiers probably more.

A level of 100 000 cyanobacterial cells/ml (which is equivalent to approximately 50 μg chlorophyll-a/litre if cyanobacteria dominate) represents a guideline value for a moderate health alert in recreational waters. At this level, a concentration of 20 μg microcystin/litre is likely if the bloom consists of *Microcystis* and has an average toxin content of 0.2 pg/cell, or 0.4 μg microcystin/μg chlorophyll-a. Levels may be approximately double if *Planktothrix agardhii* dominates. With very high cellular microcystin content, 50–100 μg microcystin/litre would be possible.

The level of 20 μg microcystin/litre is equivalent to 20 times the WHO provisional guideline value concentration for microcystin-LR in drinking-water (WHO, 1998) and would result in consumption of an amount close to the tolerable daily intake (TDI) for a 60-kg adult consuming 100 ml of water while swimming (rather than 2 litres of drinking-water). However, a 15-kg child consuming 250 ml of water during extensive playing could be exposed to 10 times the TDI. The health risk will be increased if the person exposed is particularly susceptible because of, for example, chronic hepatitis B. Therefore, cyanobacterial levels likely to cause microcystin concentrations of 20 μg/litre should trigger further action.

Non-scum-forming species of cyanobacteria such as *Planktothrix agardhii* have been observed to reach cell densities corresponding to 250 μg chlorophyll-a/litre or even more in shallow water bodies. Transparency in such situations will be less than 0.5 m measured with a Secchi disc. *Planktothrix agardhii* has been shown to contain very high cell levels of microcystin (1–2 μg microcystin/μg chlorophyll-a), and therefore toxin concentrations of 200–400 μg/litre can occur without scum formation.

An additional reason for increased alert at 100 000 cells/ml is the potential for some frequently occurring cyanobacterial species (particularly *Microcystis* spp. and *Anabaena* spp.) to form scums. These scums may increase local cell density and thus toxin concentration by a factor of 1000 or more in a few hours (as illustrated in Figure 8.1), thus rapidly changing the risk from moderate to high for bathers and others involved in body-contact water sports. Cyanobacterial scum formation presents a unique problem for routine monitoring at the usual time intervals (e.g., 1 or 2 weeks) because such monitoring intervals are unlikely to pick up hazardous maximum levels. Because of the potential for rapid scum formation at a cyanobacterial density of 100 000 cells/ml or 50 μg chlorophyll-a/litre (from scum-forming cyanobacterial taxa), intensification of surveillance and protective measures are appropriate at these levels. Daily inspection for scum formation (if scum-forming taxa are present) and measures to prevent exposures in areas prone to scum formation are the two principal actions important in these situations.

Intervention is recommended to trigger effective public information campaigns to educate people on avoidance of scum contact. Furthermore, in some cases (e.g., areas with frequent scum formation), restriction of water contact activities may be judged to be appropriate. An intensified monitoring programme should be implemented, particularly looking for scum accumulations. Health authorities should be notified immediately.

8.7.3 High probability of adverse health effects

Abundant evidence exists for potentially severe health outcomes associated with scums caused by toxic cyanobacteria. No human fatalities have been unequivocally associated with cyanotoxin ingestion during recreational water activities, although numerous animals have been killed by consuming water with cyanobacterial scum material. This discrepancy can be explained by the fact that animals will drink greater volumes of scum-containing water in relation to their body weight, whereas accidental ingestion of scums by humans during swimming will typically result in a lower dose.

Cyanobacterial scums can represent thousand-fold to million-fold concentrations of cyanobacterial cell populations. Calculations suggest that a child playing in *Microcystis* scums for a protracted period and ingesting a significant volume could receive a lethal dose, although no reports indicate that this has occurred. Based on evidence that a lethal oral dose of microcystin-LR in mice is 5000–11 600 μg/kg body weight and sensitivity between individuals may vary approximately 10-fold, the ingestion of 5–50 mg of microcystin could be expected to cause acute liver injury in a 10-kg child. Concentrations of up to 24 mg microcystin/litre from scum material have been published (Chorus & Fastner, 2001). Substantially higher enrichment of scums—up to gelatinous consistency—is occasionally observed, of which accidental ingestion of smaller volumes could cause serious harm. Anecdotal evidence indicates that children, and even adults, may be attracted to play in scums. The presence of scums caused by cyanobacteria is thus a readily detected indicator of a risk of potentially severe adverse health effects for those who come into contact with the scums. Immediate action to control scum contact is recommended for such situations.

8.7.4 Conclusions

The approach outlined in this section does not cover all conceivable situations. Swimmers may be in contact with benthic cyanobacteria after a storm breaks off clumps of filaments or cyanobacterial mats naturally detach from the sediment and are accumulated on shorelines (Edwards et al., 1992). Measures of cyanobacterial cell density will not detect these hazards. Instead, this cyanotoxin hazard calls for critical and well informed observation of swimming areas, coupled with a flexible response.

It is difficult to define "safe" concentrations of cyanobacteria in recreational water for allergenic effects or skin reactions, as individual sensitivities vary greatly. Aggravation of dermal reactions due to accumulation of cyanobacterial material and enhanced disruption of cells under bathing suits and wet suits may be a problem even at densities below the guideline levels described above.

8.8 Management options

For purposes of management, it is important to understand that cyanotoxins are chiefly found within cyanobacterial cells. Liberation into the surrounding water is possible, particularly when cells die and lyse, and differences may occur between toxins and species regarding "leakage" from intact cells. However, toxin dissolved in water is rapidly diluted and probably also degraded, whereas hazardously high toxin concentrations usually result from the accumulation of cell material as scums.

Because adequate surveillance is difficult and few immediate management options are available (other than precluding or discouraging use or cancelling water sports activities such as competitions), provision of adequate public information is a key short-term measure. Medium- to long-term measures are identification of the sources of nutrient (in many ecosystems phosphorus, sometimes nitrogen) pollution and significant reduction of nutrient input in order to effectively reduce proliferation not only of cyanobacteria, but of potentially harmful algae as well.

8.8.1 Short-term measures

Providing adequate information to the public on the cyanobacterial risk associated with using a particular recreational water area is important not only for avoiding this hazard, but also for understanding symptoms potentially caused by exposure and identifying their cause. Communication of warnings to the public may occur through local news media, by posting warning notices and through other means. They may accompany information on other recreational water quality parameters regularly monitored by the authorities and/or some further information on cyanobacteria.

Differentiation between the degree of water contact in different types of water sports should be included in warning notices. Information on the frequently transient nature and very variable local distribution of scums is important to convey the message that recreational activities are restricted only temporarily and often only very locally, and that in such cases acceptable water quality may be found nearby, e.g., at another site of the same lake.

As a precaution, the following guidance is recommended for all freshwater-based recreation and should be included in public information:

- Avoid areas with visible cyanobacterial or algal concentrations and/or scums in the water as well as on the shore. Direct contact and swallowing appreciable amounts are associated with the greatest health risk.
- Where no scums are visible, but the water shows strong greenish discoloration and turbidity, test if you can still see your feet when standing knee-deep in the water (after wading in without stirring up sediment). If not, avoid bathing—or at least avoid ingestion of water, i.e., submersion of your head.
- In such situations, avoid water-skiing because of potentially substantial exposure to aerosol.
- If sailing, sailboarding or undertaking any other activity likely to involve accidental water immersion in the presence of cyanobacterial or algal blooms, wear clothing that is close fitting in the openings. The use of wet suits for water sports may result in a greater risk of rashes, because cyanobacterial or algal material in the water trapped inside the wet suit will be in contact with the skin for long periods of time.
- After coming ashore, shower or wash yourself down to remove cyanobacterial or algal material.

- Wash and dry all clothing and equipment after contact with cyanobacterial or algal blooms and scum.

8.8.2 Long-term measures

The aim of long-term measures to minimize health risks due to toxic algae and cyanobacteria is to prevent or reduce the formation of cyanobacterial blooms in water used for recreational water activities. This can be achieved by keeping total phosphorus concentrations below the "carrying capacity," which sustains substantial population densities. Experience from numerous water bodies shows that this can be achieved if total phosphorus concentrations are 0.01–0.03 μg/litre (depending somewhat on the size and mixing regime of the water body).

This threshold may be difficult to reach in water bodies with multiple sources of nutrient pollution. However, nutrient sources are locally very variable. Therefore, identifying the chief sources and developing strategies for preventing the formation of cyanobacterial blooms are recommended and may in many cases prove to be more feasible than initially assumed (Chorus & Mur, 1999). In particular, nutrient input from agricultural runoff may in many cases be reduced by decreasing the application of fertilizers to match the actual demand of the crop or by protecting the shoreline from erosion by planting shrubs along a buffer strip about 20 m wide along the shoreline, rather than ploughing and fertilizing to the very edge of the water.

8.9 References

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CHAPTER 9
Aesthetic issues

Aesthetic issues play an important role in the public's perception of a recreational water area, for example public opinion surveys about desirable seaside resort characteristics have found that some 10% of respondents cite the importance of a clean beach (Oldridge, 1992). The principal aesthetic concern is revulsion associated with obvious pollution of the water body, turbidity, scums or odour (which may relate to inadequate levels of dissolved oxygen, see chapter 10). Pollution may cause nuisance for local residents and tourists as well as environmental problems and may lessen the psychological benefits of tourism (WHO, 1980). In this chapter, the aesthetic parameters that affect the acceptability of a recreational water area are described.

9.1 Aesthetic parameters

The general aesthetic acceptability of recreational water can be expressed in terms of criteria for transparency, odour and colour. It has been suggested that values for light penetration, colour and turbidity should not be significantly increased over natural background. The aesthetic value of recreational water areas implies freedom from visible materials that will settle to form objectionable deposits, floating debris, oil, scum and other matter, substances producing objectionable colour, odour, taste or turbidity, and substances and conditions that produce undesirable aquatic life (Department of National Health and Welfare, Canada, 1992).

9.1.1 Transparency and colour

Safety hazards associated with turbid or unclear water depend on the intrinsic nature of the water itself. Ideally water at swimming areas should be clear enough for users to estimate depth, to see subsurface hazards easily and to detect the submerged bodies of swimmers or divers who may be in difficulty (see chapter 2). Aside from the safety factor, clear water fosters enjoyment of the aquatic environment. The clearer the water, the more desirable the swimming area (National Academy of Sciences, 1973). The principal factors affecting the depth of light penetration in natural waters include suspended microscopic algae and animals, suspended mineral particles, stains that impart a colour (iron, for example, may impart a reddish colour to water), detergent foams and dense mats of floating and suspended debris, or a combination of these factors.

There are two measures of colour in water—true and apparent. The true colour of natural water is the colour of water from which turbidity has been removed (i.e., filtered water). Natural minerals give true colour to water; for example, calcium carbonate in limestone regions gives a greenish colour, ferric hydroxide, red. Organic substances, tannin, lignin and humic acids from decaying vegetation also give true colour to water (Reid & Wood, 1976). Apparent colour is an aesthetic quality and cannot be quantified. It is usually the result of the presence of coloured particulates, the interplay of light on suspended particles and such factors as reflection of the bottom or sky. An abundance of (living) blue-green algae (cyanobacteria) may impart a dark green hue; diatoms give a yellow or yellow-brown colour; some algae impart a red colour. Zooplankton, particularly microcrustaceans, may occasionally tint the water red (Reid & Wood, 1976). The causes of colour in marine waters are not thoroughly understood, but dissolved substances are one of the contributory factors. The blue of the sea is a result of the scattering of light by water molecules, as in inland waters. Suspended detritus and living organisms give colours ranging from brown through red and green. Estuarine waters have a different colour to the open sea; the darker colours result from the high turbidity usually found in such situations (Reid & Wood, 1976). This characteristic colour can also impact on coastal recreational waters receiving estuarine input, where public perception may be that the colour change represents some form of pollution as illustrated in Box 9.1.

BOX 9.1 AESTHETIC REVULSION RELATING TO WATER COLOUR PRODUCED BY A NON TOXIC ALGAL BLOOM

Within the monitoring programme for bathing waters of Catalunya (NE Spain), which is the responsibility of *l'Agència Catalana de l'Aigua—Departament de Medi Ambient-Generalitat de Catalunya*, a persistent problem was detected at La Fosca beach (Costa Brava) characterised by the discoloration of water. Water that appeared to be clean in the early morning became green-brown by late morning and remained so into the evening. This generated numerous complaints from the public who assumed the problem to be related to wastewater and sewage inputs. An intensive monitoring programme was conducted, this included:

- sanitary inspection of the beach and sewage system to search for unauthorised outlets;
- inspection of possible inland water influence;
- study of the temporal and spatial variations of the microbial water quality;
- analysis of physico-chemical parameters;
- study of sediments and flora, and finally;
- an investigation of phytoplankton.

The programme (which cost US$35,000 at 1994–1996 prices) unequivocally ruled out wastewater or sewage inputs. The discoloration was eventually attributed to a non-toxic dinoflagellate *Alexandrium taylori* (Delgado et al., 1997). Once the origin of the problem was identified a series of press conferences and a local publicity campaign was undertaken to inform the public. *A. taylori* had not previously been identified in the Mediterranean. Since its identification at La Fosca, however, it has been reported at other Mediterranean locations (Garcés et al., 2000).

This incident illustrates that not all water discoloration should be assumed to be due to sewage pollution. In this instance a preliminary investigation to identify dinoflagellate species would have saved both time and money.

Some regulatory authorities have recommended absolute values for transparency/colour and turbidity in recreational waters. This approach can be difficult to apply at local level because many waters may have naturally high levels of turbidity/colour. It is, therefore, more common that reference to changes from the normal situation be used to indicate potential water pollution.

9.1.2 Oil, grease and detergents

Even very small quantities of oily substances make water aesthetically unattractive (Environment Canada, 1981). Oils can form films on the surface, and some oil-derived substances, such as xylenes and ethylbenzene, which are volatile, may also give rise to odours or tastes, even though they are of low toxicity. In some countries (e.g., Canada), it has been reasoned that oil or petrochemicals should not be present in concentrations that can be detected as a visible film, sheen or discoloration on the surface, be detected by odour or form deposits on shorelines and bottom sediments that are detectable by sight or odour (International Joint Commission, 1977; Department of National Health and Welfare, Canada, 1992) (see also chapter 10). It is difficult to establish criteria for oil and grease, as the mixtures falling under this category are very complex. Tar may also present a problem on the shore; this can be removed by mechanical cleaning of the sand (see chapter 6).

Detergents can give rise to aesthetic problems if foaming occurs, particularly since this can be confused with foam caused by the by-products of algal growth (see chapters 7 and 8; Bartram & Rees, 2000 chapter 10).

9.1.3 Litter

Beach litter is derived from three main sources: marine, riverine (including torrents) and beach user discards. Visitor enjoyment of any beach is likely to be marred by litter, although litter perception varies with respect to many parameters, such as age, socioeconomic status and gender. Although not litter, as such, large accumulations of seaweed and algae are likely to be an aesthetic problem (both in terms of visual impact and odour) and also, if associated with flying and/or biting insects, a nuisance (see chapter 11).

The variety of litter found in recreational water or washed up on the beach is considerable. Some examples of unwanted recreational water flotsam and jetsam include wooden crates and palettes, cardboard cartons, newspaper, steel drums, plastic containers and foam products, rubber goods such as vehicle tyres, bottles and cans, dead animals or animal bones, human hair, discarded clothing, hypodermic syringes, needles and other medical wastes, bottle tops, cigarette butts and packets, matchsticks, fish netting and rope ends.

Litter counts have been considered as possible proxy indicators for the likelihood of gastrointestinal effects associated with swimming. For example, high incidence rates of self-reported gastrointestinal illness after bathing in sewage-polluted water have been associated with public perceptions of different items affecting the aesthetic appearance of recreational water and beaches (University of Surrey, 1987). The presence of the following items was positively correlated with the likelihood of self-reported gastrointestinal symptoms: discarded food/wrapping, bottles/cans, broken bottles, paper litter, dead fish, dead birds, chemicals, oil slicks, human/animal excrement (particularly from dogs, cats, cattle or birds), discarded condoms and discarded sanitary towels.

9.1.4 Odour

Objectionable smells associated with sewage effluent, decaying organic matter such as vegetation, dead animals or fish, and discharged diesel oil or petrol can deter recreational water and beach users. Odour thresholds and their association with the concentrations of different pollutants of the recreational water environment have not been determined. The presence of dissolved oxygen in the water body will be of great importance in preventing the formation of undesirable amounts of odorous hydrogen sulfide (see chapter 10).

9.1.5 Noise

Traffic on nearby roads, trade hawkers and indiscriminate use of beach buggies, motorbikes, portable radios and hi-fi equipment, motorboats and jet skis can all impact on tranquillity for the beach and water user; at the same time, some people thrill to noisy activities (Velimirovic, 1990). Mindful of the need for mutual respect (WHO, 1989), zoning of areas for different activities is often undertaken.

9.2 Economic consequences

The public often perceives the quality of recreational water to be very different from its actual microbial and/or chemical quality (Philipp, 1994). Some studies have shown that rivers of good microbial or chemical quality have been perceived as poor by the public because of aesthetic pollution (Dinius, 1981; House, 1993). Poor aesthetic recreational water and beach quality may, however, also imply poor microbial/chemical water quality.

The economic aspects associated with cleaning the coastline have previously been reviewed (Bartram & Rees, 2000). Local economies may depend on the aesthetic quality of recreational water areas, and many fear that environmental degradation of beaches could lead to loss of income from tourism (WHO, 1990; Godlee & Walker, 1991; Philipp, 1992). At resort beaches, litter may have an economic effect on the region. During 1987 and 1988, beach closures in New York and New Jersey, USA, due to litter accumulation, together with the public's perception of degraded beach and water quality, cost the local economy several billion dollars (Valle-Levinson & Swanson, 1991).

The upper Adriatic coast of the Mediterranean Sea was hit during the 1989 summer season by a very severe episode of eutrophication, which, together with mucilage caused by the production of viscous substances from benthic micro-algae, generated considerable concern among tourists about their health. The unpleasant sight of large tracts of this viscous amorphous substance along the shoreline resulted in a large number of beaches along the Italian coastline becoming temporarily unsuitable for bathing (WHO, 1990). There was a 40% reduction in local tourism as a consequence of this (Philipp, 1992), and aesthetic considerations alone were sufficient to prevent would-be bathers from entering the water (WHO, 1990). The economic effects attributed to the loss of use of the environment for tourists and other economic purposes were:

- loss of tourist days;
- damage to the local tourist infrastructure (loss of income for hotels, restaurants, bathing resorts, other amenities, etc.);
- damage to tourist-dependent activities (loss of income for clothing manufacture, food industry, general commerce, etc.);
- damage to fisheries activities (reduction in fish catch, depreciation of the price of seafood);
- damage to fisheries-dependent activities (fishing equipment production and sales, fisheries products, etc.); and
- damage to the image of the Adriatic coast as a recreational resort at both national and international levels (WHO, 1990; Philipp, 1992).

A further economic factor that should be taken into consideration is the health care cost associated with beach litter, in particular hospital waste washed up on beaches (Philipp, 1991; Walker, 1991; Anon., 1994). The direct health care costs arising from discarded hypodermic syringe needles have been studied and found to be considerable (Philipp, 1993).

9.3 Marine debris monitoring

Methods to undertake marine debris surveys have been presented and discussed elsewhere (Bartram & Rees, 2000 chapter 12). The purposes of marine debris monitoring may include one or more of the following:

- to provide information on the types, quantities and distribution of marine debris (Williams & Simmons, 1997);
- to provide insight into problems and threats associated with an area (Rees & Pond, 1995);
- to assess the effectiveness of legislation and coastal management policies (Earll et al., 1997);
- to identify sources of marine debris (Earll et al., 1997);
- to explore public health issues relating to marine debris (Philipp et al., 1993, 1997); and
- to increase public awareness of the condition of the coastline (Rees & Pond, 1995).

In the United Kingdom, for example, one series of studies identified a 4-fold deterioration in coastal environmental quality during three consecutive years (Philipp et al., 1994). The results helped to justify national legislation for tighter controls on discharges from seawater sewage outfall pipes and the removal of screenings for disposal elsewhere, better provision and emptying of litter bins and improved advice for the public (Philipp et al., 1994, 1997). In Catalunya (Spain), a programme of aesthetic monitoring was undertaken to supplement microbial water quality data (Box 9.2).

BOX 9.2 VISUAL INSPECTION AND MICROBIAL WATER QUALITY

The monitoring programme conducted in the Catalunya region of NE Spain has been implemented to provide the public with information on the aesthetic aspects of water and sand in combination with data on microbial water quality. Microbial water quality monitoring is conducted once a week, while aesthetic aspects are assessed more frequently (up to five times a week). Data is collected on the presence and amount of:

- plastics;
- sanitary residues;
- algae;
- tar;
- oil;
- litter;
- abnormal water colour; and
- anything else that may cause aesthetic revulsion.

In addition, information on how thoroughly a beach is machine cleaned and how frequently litter containers are emptied is recorded.

The aesthetic data are processed alongside the microbial water quality data and result in a combined grading for the beach. Aesthetic aspects are considered to be so important that an excellent microbial grading may be reduced to good or even poor if the beach looks bad.

Municipalities, tourist information offices, NGOs, local newspapers, TV and radio are informed weekly of the results. In addition, municipalities receive a report outlining raw microbial data for each of the evaluated parameters and the results of the visual inspection along with suggestions for improvements. This system gives confidence to the public that their concerns are being taken seriously and has also encouraged many municipalities to improve the aesthetic aspects of their bathing areas.

The reliability and validity of litter counts as measures of health protection need to be tested among different populations and in different exposure situations (Philipp et al., 1997). Beach surveys for the extent of littering are, however, useful as indicators of the need for behavioural change (WHO, 1994). To be worthwhile in the research context, litter counts, as measures of aesthetic quality and as potential indicators of the likelihood of illness associated with the use of the recreational water area, must be able to:

- classify different levels of beach and water quality and the density of different litter and waste items before and after any environmental improvements or cleansing operations;

- be useful when compared with conventional microbial and chemical indicators of recreational water and beach quality;
- differentiate the density of different pollutants deposited by the public on beaches from pollutants that originated elsewhere and were then washed ashore;
- show consistent findings when used in studies of similar population groups exposed to the same pollutant patterns; and
- show a correlation with variations in the human population density of recreational water and beaches (Philipp, 1992; IEHO, 1993; Philipp et al., 1997).

Large-scale monitoring programmes for marine debris often rely on volunteers to survey the beaches and collect data (Marine Conservation Society, 2002). It is, however, not usually possible, with staffing constraints, to verify the findings in a sample of locations before the next high tide. Tide changes can, too, be accompanied by changes in water currents and wind direction. Nevertheless, reliable data can be collected if comprehensive guidance is given to ensure comparable approaches by different groups of volunteers and if validated questionnaire methods are used in consistent and uniform ways. Internal cross-checks of such methods have been undertaken, and they have confirmed consistency of the data collected (Philipp et al., 1993).

9.4 Guideline values and management

As guidelines are aimed at protecting public health, no guideline values have been established for aesthetic aspects. Aesthetic aspects, however, are important in terms of maximizing the benefit of recreational water use.

In terms of aesthetic factors, questions frequently raised for local managerial consideration include the following (Philipp, 1993):

- Are wastes there?
- If present, where are the wastes coming from?
- Are they causing aesthetic problems?
- Could the aesthetic problems be responsible for economic losses in the local community?
- Can the effects (if any) be stopped?
- Who should control the problems?
- What will it cost, and can any loss of environmental opportunity be measured?

Mechanical beach cleaning (see also chapter 6 and Bartram & Rees, 2000 chapter 12) usually involves motorized equipment utilizing a sieve that is dragged through the top layer of the sand. The sieve retains the litter, but usually cigarettes and other small items pass through. Resort beaches use such equipment because it is fast and provides an aesthetically clean recreational areas for visitors. In areas with, for example, medical waste, sewage-related debris or other potentially harmful items, it reduces health risks for those cleaning the beach, because no manual picking up of material is involved. The utilization of mechanical cleaning at rural beaches has been questioned, as such cleaning affects local ecology (Llewellyn & Shackley, 1996).

Other strategies for keeping beaches free of litter include providing waste bins on beaches and emptying them frequently, suggesting that recreational water users take their litter home with them and using people to manually pick up litter.

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CHAPTER 10
Chemical and physical agents

Chemical contaminants can enter surface waters or be deposited on beaches from both natural and anthropogenic sources. These may be either point sources, such as an industrial outfall or a natural spring, or non-point (diffuse) sources, such as runoff from land. In most cases, there will be significant dilution or attenuation of contaminants, depending on circumstances. In all cases, chemical and physical contamination must be assessed on a local basis.

The potential risks from contamination of recreational water environments by chemical and physical agents are described in this chapter. Chemical and physical agents may also lead to degradation of the aesthetic quality of recreational water environments, which is addressed in chapter 9. Toxins from cyanobacteria and algae, while chemical in nature, are addressed in chapters 7 and 8.

10.1 Exposure assessment

Exposure is one of the key issues in determining the risk of toxic effects from chemicals in recreational waters. The form of recreational activity will therefore play a significant role. Routes of exposure will be direct surface contact, including skin, eyes and mucous membranes, inhalation and ingestion. In assessing the risk from a particular contaminant, the frequency, extent and likelihood of exposure are important parts of the evaluation.

Generally, exposure of skin and mucous membranes is most frequent. For activities involving whole-body contact, the probability that some water will be ingested increases. The skill of the participant in their water recreation activity will be important in determining the extent of involuntary exposure, particularly by ingestion.

Inhalation can be important in circumstances where there is a significant amount of spray, such as in waterskiing or white water canoeing. Generally, however, inhalation is of greater significance in swimming pools and related environments where disinfection is practised (see Volume 2 of *Guidelines for Safe Recreational Water Environments*).

The use of wet suits implies that long periods may be spent in the water. In addition, by trapping water against the skin, the wet suit will create a micro-environment that will enhance the absorption of chemicals through the skin and potentially the development of skin irritation or allergy (see also chapters 7 and 8).

Many substances of potential concern are of low water solubility and will tend to migrate to sediments, where they may accumulate. Where the sediments remain undisturbed, this is of low concern. However, where the sediment is disturbed and resuspended or where recreational water users are in intimate contact with sediment, then this may contribute to exposure. This can result in increased skin exposure, but little is known of the quantitative movement of chemicals adsorbed on sediment through skin. In general, it is probable that this will make only a minor contribution to overall exposure.

10.2 Hydrogen ion concentration (pH)

pH has a direct impact on the recreational users of water only at very low or very high values. Under these circumstances, pH may have effects on the skin and eyes.

Primary irritation of the skin appears to be linked to high pH, although the mechanism is unclear. It is unlikely that irritation or dermatitis would be caused directly by high or low pH, although these conditions may be exacerbated, particularly in sensitive subjects.

High or low pH may also to and exacerbate irritation of the eye by chemicals. However, no adverse effects on the eye were noted in a study by Basu et al. (1984), who examined the capacity of water from two inland lakes in Ontario, Canada (Clearwater Lake: pH ~4.5, acid neutralizing capacity 40 μeq/litre; Red Chalk Lake: pH ~6.5, acid neutralizing capacity 70 μeq/litre), to cause eye irritation in rabbits and human volunteers.

Water of high pH could have an adverse effect on hair condition by causing the hair fibres to swell and by cleaving the cystine bridges between adjacent polypeptide chains of hair protein. However, the impact will also be dictated by the buffering capacity of the water.

In very soft and poorly buffered waters with an alkalinity of less than about 40 mg of calcium carbonate per litre, pH will be more susceptible to wide fluctuations. In well buffered waters, pH is much less likely to reach extreme values, but the significance of high or low pH for skin reactions and eye irritation will be greater.

10.3 Dissolved oxygen

Dissolved oxygen will not have a direct effect on users, but it will influence microbial activity and the chemical oxidation state of various metals, such as iron. It will be of great importance in preventing the formation of undesirable amounts of hydrogen sulfide. These factors are not a human health concern, but may give rise to aesthetic issues (see chapter 9). These problems will not occur in waters with sufficient dissolved oxygen.

10.4 Chemical contaminants

In general, the potential risks from chemical contamination of recreational waters, apart from toxins produced by marine and freshwater cyanobacteria and algae (chapters 7 and 8), marine animals (chapter 11) or other exceptional circumstances, will be very much smaller than the potential risks from other hazards outlined in chapters 2–5). It is unlikely that water users will come into contact with sufficiently high concentrations of most contaminants to cause adverse effects following a single exposure. Even repeated (chronic) exposure is unlikely to result in adverse effects at the concentrations of contaminants typically found in water and with the exposure patterns of most recreational water users. However, it remains important to ensure that chemical hazards and any potential human health risks associated with them are recognized and controlled and that users can be reassured as to their personal safety.

For recreational water area users, the dangers of chemical contamination will depend on the particular circumstances of the area under consideration. For example, a fast-flowing upland river, remote lakes or drinking-water reservoir used for recreation will be unlikely to suffer from significant chemical contamination. However, slow-flowing lowland rivers, lowland lakes and coastal waters may be subject to continuous or intermittent discharges and may have suffered from past pollution, which could result in contaminated sediments. Where motorboats are used extensively, chemical contamination of the water by gasoline additives may cause concern. Where a water body used for recreational purposes receives significant wastewater discharges, its chemical constitution and how recreational areas will be influenced should be considered, taking into account both the dilution and dispersion of the discharge.

In general, significant contamination by naturally occurring contaminants is less likely than contamination by industrial, agricultural and municipal pollution, but there may be circumstances where small recreational water bodies containing water from mineral-rich strata could contain high concentrations of some substances. Such waters, however, are more likely to contain metals, such as iron, that may give rise to aesthetic degradation of the water (see chapter 9).

There is a great deal of anecdotal evidence regarding skin rashes and related effects in individuals coming into contact with chemically contaminated water. Except in circumstances of extreme contamination or the presence of algal blooms (covered in chapters 7 and 8), evidence amenable to critical scientific evaluation is not available.

10.5 Guideline values

The chemical quality of recreational waters does not seem to represent a serious health risk for recreational water users, and in most cases the concentration of chemical contaminants will be below drinking-water guideline values. There are no specific rules that can easily be applied to calculate guideline values for chemical contaminants in recreational waters. However, as long as care is taken in their application, the WHO *Guidelines for Drinking-water Quality* (WHO, 1993, 1998) can provide a starting point for deriving values that could be used to make a screening level risk assessment under specific circumstances.

WHO drinking-water guideline values relate to water ingestion and, in most cases, to lifetime exposure. However, drinking-water guidelines may be related to recreational exposure. Mance et al. (1984) suggested that environmental quality standards for chemicals in recreational waters should be based on the assumption that recreational water makes only a relatively minor contribution to intake. They assumed a contribution for swimming of an equivalent of 10% of drinking-water consumption. Since most authorities (including WHO) assume consumption of 2 litres of drinking-water per day, this would result in an intake of 200 ml per day from recreational contact with water.

A simple screening approach is therefore that a substance occuring in recreational water at a concentration ten times that stipulated in the drinking-water guidelines may merit further consideration.

10.5.1 Inorganic contaminants

Most recreational exposure to inorganic contaminants will be by ingestion, with dermal contact and inhalation contributing little to exposure. Based on the assumptions given above, screening values for the ingestion of inorganic contaminants in recreational waters can be calculated from the WHO *Guidelines for Drinking-water Quality* (WHO, 1993, 1998). However, if the corresponding value for a particular inorganic contaminant is exceeded, this does not necessarily imply that a problem exists. Rather, it suggests the need for a specific evaluation of the contaminant, taking into consideration local circumstances and conditions of the recreational water area (see section 10.6). These could include, for example, the characteristics of the typical recreational water user, the degree of water contact of the recreational water activities carried out, effects of winds/currents/tides on contaminant concentration and the chemical form of the inorganic contaminant. For example, the chemical form of metals may significantly affect their solubility and absorption, and this should be taken into account in assessing any potential risks from exposure.

10.5.2 Organic contaminants

There are many organic contaminants that can be present in surface waters as a consequence of industrial and agricultural activity. Many of these substances will primarily be associated with sediments and particulate matter. This is particularly true of substances that are highly lipophilic, such as chlorinated biphenyls.

Skin absorption from contact with sediment is a possibility that cannot be ruled out, however, for most recreational purposes the extent of contact is likely to be small. However, consideration should be given to the likelihood of sediment being disturbed and the possibility of ingestion by some groups, such as infants and small children.

Some small chlorinated molecules (e.g., chloroform or tri- and tetrachloroethene) and hydrocarbons (e.g., toluene) have been shown to be absorbed through skin from water. A study by the US EPA (1992) concluded that the contribution from skin absorption and inhalation could contribute as much again as water ingestion.

As with inorganic contaminants, the WHO *Guidelines for Drinking-water Quality* (WHO, 1993, 1998) can be used as a basis for screening the potential risk from specific organic chemicals. Again, if the screening value for a particular organic contaminant is exceeded, this does not necessarily imply that a problem exists (see section 10.6). Rather, it suggests the need for a specific evaluation of the contaminant, taking into consideration local circumstances and conditions.

10.6 Approach to assessing chemical hazards in recreational waters

1. An inspection of the recreational water area will show if there are any obvious sources of chemical contamination, such as outfalls. These are a problem if they

are easily accessible or if the effluent does not receive immediate and significant dilution. Intelligence on past industry in the recreational area and upstream will give an indication of whether contaminated sediments are likely to be present and also the identity of possible contaminants. Knowledge is required of upstream industry and whether direct or indirect discharges are made to the water.

2. The pattern and type of recreational use of the water need to be carefully considered to determine the degree of contact with the water and if there is a significant risk of ingestion.
3. If it is probable that contamination is occurring and there is significant exposure of users, then chemical analysis will be required to support a quantitative risk assessment. Care should be taken in designing the sampling programme to account for variation in time and water movement. If resources are limited and the situation complex, then samples should first be taken at the point considered to give rise to the worst case; only if this gives rise to concern is there a need for wider sampling.
4. The quantitative risk assessment should consider the anticipated exposure in terms of both dose (i.e., is there significant ingestion?) and frequency of exposure. The WHO *Guidelines for Drinking-water Quality*, which provide a point of reference for exposure through ingestion, with a few exceptions described in the guideline summaries, relate to lifetime exposure.
5. It is important that the basis of any guidelines or standards, which are considered to be necessary, be made transparent. Without this, there is a danger that even occasional or trivial exceedances could unnecessarily undermine users' confidence.

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CHAPTER 11
Dangerous aquatic organisms

Dangerous aquatic organisms may be encountered during recreational use of freshwater and coastal environments (Halstead, 1988; Williamson et al., 1996). Such organisms vary widely and are generally of local or regional importance. The likelihood and nature of human exposure often depend significantly on the type of recreational activity concerned.

Because of the wide variety of organisms that may be encountered, this chapter summarizes only those known to have caused significant ill-health, injury or death to recreational water users. These include both non-venomous organisms (disease vectors, "in-water" hazardous organisms and "water's-edge" hazardous organisms) and venomous vertebrates and invertebrates (see Table 11.1). Space prohibits full coverage of their geographic distribution, identification, management or first aid treatment. Readers are advised to turn to specialized texts for such information, such as the WHO publication *International Travel and Health*, which is updated annually and is available on the internet (http://www.who.int/ith). Rats, which may spread illnesses such as leptospirosis, are not included but are covered in chapter 5.

Two types of risks can be distinguished in relation to dangerous aquatic species. The first type of risk is infectious disease transmitted by species with life cycles that are linked to the aquatic environment. The second type is injury or intoxication (e.g., ciguatera, histamine poisoning, shellfish and so on) resulting from direct encounters with large animals or venomous species. Injuries from encounters with dangerous aquatic organisms are generally sustained in one of the following ways:

- accidentally brushing past a venomous sessile or floating organism when swimming;
- entering waters frequented by dangerous jellyfish (e.g., box jellyfish);
- inadvertently treading on a stingray, weeverfish or sea urchin;
- unnecessary handling of venomous organisms during seashore exploration;
- invading the territory of large animals when swimming or at the waterside;
- swimming in waters used as hunting grounds by large predators; or
- intentionally interfering with, or provoking, dangerous aquatic organisms.

Perceived risks involving dangerous aquatic organisms may have important economic repercussions in areas that depend to a large extent on recreational tourism as a source of income. An example is the decline in South African tourists visiting Lake Malawi because of news reports about schistosomiasis (bilharzia) cases. Similarly, news about malaria outbreaks in East Africa and dengue outbreaks in the Caribbean

TABLE 11.1. RELATIVE RISK TO HUMANS POSED BY SEVERAL GROUPS OF AQUATIC ORGANISMS

Organism	Discomfort	Requires further medical attention	May require emergency medical attention
Non-venomous organisms			
Sharks		✓	✓✓
Barracudas		✓	
Needlefish		✓	✓✓
Groupers		✓	
Piranhas		✓	
Conger eels		✓	
Moray eels		✓	
Electric fish		✓	✓✓
Seals and sea lions		✓	
Hippopotami		✓	✓✓
Crocodiles and alligators		✓	✓✓
Venomous invertebrates			
Sponges	✓	✓	
Hydroids	✓	✓	
Portuguese man-of-war	✓	✓	✓✓
Jellyfish	✓	✓	
Box jellyfish		✓	✓✓
Hard corals	✓	✓	
Sea anemones	✓	✓	
Blue-ringed octopus		✓	✓✓
Cone shells		✓	✓✓
Bristleworms	✓	✓	
Crown of thorns starfish	✓	✓	
Sea urchins (most)	✓		
Flower sea urchin		✓	✓(✓)
Venomous vertebrates			
Stingrays		✓	✓✓
Stonefish		✓	✓(✓)
Other spiny fish (e.g., catfish, weeverfish, etc.)	✓	✓	
Surgeonfish	✓	✓	
Sea snakes		✓	✓✓
Water moccasin		✓	✓

✓✓—associated with fatalities ✓(✓)—probably associated with fatalities.

have had a serious impact on local economies. Incidents that have less effect on general public health, such as repeated shark attacks, usually have a less intense, and shorter-lived, impact in this sense.

Many serious incidents can be avoided through an increase in public education and awareness. It is therefore important to identify and assess the hazards posed by various aquatic organisms in a given region and bring the results to public attention. Awareness raising should be targeted at groups at particular risk (such as those known to have suffered adverse health effects), which may include local and/or visiting populations. In addition, at locations where hazards involving dangerous aquatic organisms have been identified, procedures should be developed for treating any injuries sustained.

11.1 Disease vectors

Animals that carry diseases are typically small and in themselves relatively harmless, with only a few individuals of a population carrying the disease. If present in large numbers, however, they may represent a major nuisance and also be an aesthetic issue (see chapter 9).

11.1.1 Mosquitoes

Tropical freshwater or brackish water environments are havens for mosquitoes. Female mosquitoes require a blood meal (from humans or other animals) to develop their eggs. In the process of taking a blood meal, mosquitoes may ingest pathogens (e.g., the parasite causing malaria) from an infected person or animal. At the next blood meal (mosquitoes go through various cycles of egg production), they then inject the pathogen into the next person, and this will spread the disease. All mosquitoes go through an aquatic larval stage, but the exact ecological requirements vary for the different species in different regions.

Two groups of mosquito-borne diseases are of particular public health importance for those who visit areas where transmission takes place (so-called endemic areas): malaria and arboviral diseases. (The HIV virus, which causes AIDS, is not transmitted by mosquitoes.)

Malaria is caused by one of four species of parasite belonging to the genus *Plasmodium*. Malaria parasites are transmitted by *Anopheles* mosquitoes. These mosquitoes bite between dusk and dawn. Their breeding places are generally in clean fresh water, standing or slowly running, with some species breeding in brackish water coastal lagoons. They never breed in polluted water. Unlike *Culex* mosquitoes (see below), *Anopheles* mosquitoes do not produce the typical high-pitched buzz that is part of the nuisance experienced in mosquito-infested areas. The position of the mosquito body with respect to the wall (at a 45-degree angle) when the insect is resting is probably the easiest way to distinguish anopheline mosquitoes from culicine ones.

Arboviral diseases (arbo = arthropod-borne) are caused by infections that are exclusively transmitted by mosquitoes. They include yellow fever, dengue and various types of encephalitis, such as Japanese encephalitis, when it is associated with flooded rice fields in south, south-east and east Asia. Many of these infections, notably yellow fever and Japanese encephalitis, are preventable by vaccination. For dengue fever (also known as break-bone fever in some parts of the world) and its more severe variant, dengue haemorrhagic fever, there is, however, no vaccine available.

The *Aedes* mosquitoes, which transmit the dengue virus, breed in small water collections in a human-made environment—hence the urban/human settlement-associated distribution of the disease. While dengue haemorrhagic fever is an important cause of death among children during outbreaks of the disease, classic dengue is a much less severe but very debilitating disease lasting for 4–6 weeks. *Aedes* mosquito species have black and white banded legs, and they (sometimes ferociously) bite during daytime.

Culex mosquitoes, which breed in organically polluted water, are mainly known for the transmission of filariasis (which can eventually develop into elephantiasis). This disease is likely to develop only in people who have been exposed to infectious bites for many years.

11.1.2 Freshwater snails and Schistosoma

Certain species of small freshwater snails (*Bulinus* sp., *Biomphalaria* sp. and *Oncomelania* sp., the last one being amphibic) are the essential intermediate hosts for the larval development of trematode parasites of the genus *Schistosoma*. These snails live in tropical lakes (either natural or man-made), in slow-flowing rivers and in the irrigation and drainage channels of agricultural production systems. Contamination of these waters with human excreta from parasite carriers releases first stage larvae (miracidia) that invade the snails. Once the larvae have developed into their infectious stage inside the snail (cercariae), they are released into the water. They adhere to and penetrate the human skin. Following a complex trajectory through the human body (and an associated metamorphosis), they grow into adult trematode worms living in the veins of the liver or the bladder.

Humans infected by *Schistosoma* suffer from a slowly developing chronic, debilitating and potentially lethal tropical disease known as bilharzia or schistosomiasis. Typical symptoms include fever, anaemia and tissue damage. Upon diagnosis, complete cure is possible using the drug praziquantel.

Trichobilharzia ocellata is a schistosome parasite of ducks, which occurs in temperate areas and leads to a far less serious form of infection than outlined above. Cercarial dermatitis or "swimmers' itch" results when the infectious stage of the parasite, known in some cultures as "duck fleas" invade humans. Symptoms may include a prickling sensation shortly after leaving the water, which is followed by an itchy papular dermatitis. The rash is confined to immersed areas of the body. In severe cases the rash can be accompanied by fever, nausea and vomiting (Fewtrell et al., 1994).

11.1.3 Preventive measures

Preventive measures can be taken by the individual:

- Always try to obtain information from appropriate international agencies (e.g., WHO, 1997, 2002) or local health authorities about the local vector-borne disease situation and follow their guidance in risk prevention.
- In malaria endemic areas, take the recommended prophylactic medicine.
- Wear protective clothing (long-sleeved shirts, long trousers) at the indicated biting times.
- Protect exposed parts of your body with repellents (e.g., N,N-diethyl-meta-toluamide—DEET).
- Screened windows and air-conditioning help keep mosquitoes out of houses.
- On return from a malarial area, consult your physician about the possible risk of having contracted the disease, should you have symptoms such as fever, headaches, chills or nausea.

- Avoid swimming or wading in fresh water in countries in which schistosomiasis occurs. Wearing full–length boots, which prevent water contact if wading in the water, will decrease the chances of infection. Although vigorous towel drying after an accidental, very brief, water exposure may help to prevent the *Schistosoma* parasite from penetrating the skin, do not rely on vigorous towel drying to prevent schistosomiasis.

11.2 "In-water" hazardous organisms

Although attacks by "in-water" hazardous organisms, such as sharks, usually attract a lot of public and media attention, the organisms are endemic to certain regions only, and their real public health significance is variable.

11.2.1 Piranhas (freshwater)

Piranhas are restricted to the fresh waters of northern South America, in the Amazon Basin. The largest species is *Pygocentrus piraya*, which reaches a size of 60 cm. Piranhas have powerful jaws with very sharp teeth, which they use to communally attack and kill large prey animals. They can be dangerous to humans. Splashing of the surface water is sufficient to attract a school of piranhas.

11.2.2 Snakes (freshwater)

Some non-venomous but large freshwater snakes such as the semi-aquatic anaconda (*Eunectes murinus*) can present a danger. The anaconda, which reaches lengths of up to 7.6 m, lives in tropical South America. Anacondas generally constrict and suffocate large prey, often viciously (non-venomous) biting the victim before coiling. Attacks on humans have occurred, but the snake is not generally aggressive towards people and will usually endeavour to escape if approached (see section 11.5.6 for venomous snakes).

11.2.3 Electric fishes (freshwater and marine)

Approximately 250 species of fish have specialized organs for producing and discharging electricity and are capable of delivering powerful electric shocks. These specialized organs are used by the fish to locate and stun prey, as a means of defence and for navigation. The electric shock is delivered to a person when contact is made with the animal's skin surface. The majority of electric fishes continuously emit a low-voltage electric charge in a series of pulses, with only two groups of electric fishes posing a serious threat to humans. The most dangerous of these is the freshwater electric eel (*Electrophorus electricus*), capable of producing an electric field of more than 600 volts. It can grow up to 3.4 m and lives in shallow rivers in tropical and subtropical South America. The fish is probably the only electric fish capable of killing a full-grown human.

The most powerful marine electric fishes are the torpedo rays (*Narcine* sp. and *Torpedo* sp.), which are bottom dwellers in all shallow temperate and warm seas. Electric rays vary greatly in their electric potential, some generating an electric field of up to 220 volts. Although the shocks are strong enough to be dangerous, no fatalities are known. Fishermen in European waters have been known to receive a shock from their line before seeing what was caught (Dipper, 1987).

11.2.4 Sharks (mainly marine)

Sharks live in all the oceans (excluding the Southern Ocean around the Antarctic continent) but are most abundant in tropical and subtropical waters. The majority of shark species are marine, and representatives are found at all depths. Some shark species migrate regularly from salt to fresh water, and a few inhabit freshwater lakes and rivers. Not all shark species are dangerous to humans.

Sharks are attracted by brightly coloured and shiny metallic objects, by the scent of blood, e.g., radiating from speared fish, and also by low-frequency vibrations and explosions. Sharks are furthermore attracted to nearshore garbage dumping grounds. In tropical waters, most shark attacks on humans occur during their habitual feeding times during late afternoons and at night. Sharks rarely "attack" humans, and such incidents are usually cases of mistaken identity, with the shark confusing the swimmer for its prey. Many attacks are a "bite," simply as a taste of the possible prey (Last & Stevens, 1994).

Shark species include the following:

- The great white shark (*Carcharodon carcharias*) lives mainly in the open ocean, although some swim into shallow water. Most of the attacks on people have happened in estuaries. The great white shark is responsible for the largest number of reported attacks on humans. It is thought that humans might be mistaken for its normal seal prey.
- The tiger shark (*Galeocerdo cuvier*) is extremely widespread in the tropics and subtropics. Following the great white shark, the second most reported attacks on humans are attributed to tiger sharks.
- The mako shark (*Isurus oxyrinchus*) is mainly an open ocean shark and occurs in all temperate and tropical oceans. It is often aggressive and dangerous when close to shore.
- The smooth hammerhead shark (*Sphyrna zygaena*), with its very distinctive head shape, lives in all warm water oceans.
- The silvertip shark (*Carcharhinus albimarginatus*) is very abundant around reefs and islands in the Pacific and Indian oceans.
- The bull shark (*Carcharhinus leucas*) is mainly located in the warm oceans of the world, although it can at times be found up the Amazon and rivers in Australia, Central America and south-eastern Africa (Halstead et al., 1990).

11.2.5 Barracudas and needlefish (marine)

The great barracuda (*Sphyraena barracuda*) is widely distributed throughout the subtropical and tropical regions of the open oceans. It is 1.8–2.4 m long and very rarely attacks humans. Barracudas, however, may intimidate divers and snorkellers by closely shadowing them. Like sharks, barracudas are attracted to shiny metallic objects and dead fish.

The various species of needlefish pose a more significant threat to humans. Needlefish are slender, possess very long, strong and pointed jaws and reach an average length of 1.8 m. They are most often found swimming in surface waters. At night, they are strongly attracted by bright lights. Cases of fishermen or divers on night expeditions being severely wounded and even killed by jumping needlefish have been reported (Halstead et al., 1990). Needlefish occur in the Caribbean, around the equatorial western African coast and near Japan and are widespread throughout the western Indian Ocean.

11.2.6 Groupers (marine)

Groupers live in the shallow waters of the Indo-Pacific on coral reefs and in sandy areas. Their size (the giant grouper, *Promicrops lanceolatus*, can reach 3 m) means that these generally non-aggressive fish are potentially dangerous. They are territorial fishes, and divers should look out for groupers before entering underwater caves and ensure that an exit is always open should a grouper wish to escape.

11.2.7 Conger and moray eels (marine)

The majority of eels are harmless, although they may attack and inflict fairly deep puncture wounds when provoked. Moray eels (*Gymnothorax* spp.) live in tropical waters on coral reef platforms, where they hide in crevices and holes among the dead coral. Conger eels (*Conger conger*) live in temperate waters of the Atlantic in rocky areas that offer them hiding places inside caves, holes and cracks.

11.2.8 Preventive measures

Preventive measures can be taken by the individual:

- Treat all animals with respect, and keep at a distance whenever possible.
- Avoid swimming at night or in the late afternoon in areas where large sharks are endemic.
- Avoid swimming in shark waters where garbage is dumped.
- Avoid wearing shiny jewellery in the water where large sharks and barracudas are common.
- Avoid attaching speared fish to the body where sharks, barracudas or groupers live.
- Avoid wearing a headlight when fishing or diving at night in needlefish waters.
- Look out for groupers and moray or conger eels before swimming into caves or putting hands into holes and cracks between rocks.

11.3 "Water's-edge" hazardous organisms

As with "in-water" hazardous organisms, attacks by "water's-edge" hazardous organisms, such as alligators, attract a lot of attention; however, the organisms are endemic to certain regions only, and their real public health significance is variable.

11.3.1 Hippopotami (freshwater)

The hippopotamus (*Hippopotamus amphibius*) is an aquatic mammal chiefly inhabiting freshwater rivers and lakes from the Upper Nile down to South Africa. Despite being a herbivore, the hippopotamus is responsible for a significant number of human deaths in Africa. Due to their sudden and violent nature and ability to swim quickly, hippopotami pose a serious threat to humans in the water. They are generally peaceable creatures, and most often a herd will scatter, or at least submerge, at the approach of humans, but attacks are not uncommon. The majority of incidents are due to ignorance of their habits, in particular moving between a group of hippopotami on shore and water.

11.3.2 Crocodiles and alligators (freshwater and marine)

Crocodiles are found in tropical areas of Africa, Asia, the western Pacific islands and the Americas. The majority of species live in fresh water. The largest living crocodiles may exceed 7.5 m in length. Crocodiles normally hunt at night and bask during the day, but might also hunt during the day if food is in short supply.

All crocodiles are capable of inflicting severe harm or causing death to humans. The more dense their populations, the more dangerous are individual crocodiles. The saltwater crocodile (*Crocodylus porosus*) of south-eastern Asia is probably the most dangerous of all the marine animals. It lives mainly in mangrove swamps, river mouths and brackish water inlets, but has been seen swimming far offshore (Halstead et al., 1990). The Nile crocodile (*C. niloticus*) has been rated as second only to the saltwater crocodile in danger to humans (Caras, 1976).

There are only two species of alligator: the Chinese alligator (*Alligator sinensis*) and the American alligator (*A. mississipiensis*). The Chinese alligator, found in the Yangtze River basin of China, is quite small (<2.5 m) and timid and is not considered to be a significant threat to humans. The American alligator, which lives in freshwater swamps and lakes in the south-eastern USA, is larger (up to 6 m in length) and potentially dangerous to humans. Attacks occur infrequently.

11.3.3 Seals and sea lions (marine)

Seals and sea lions are not aggressive towards humans under normal circumstances. During the mating season, however, or when with pups, bulls might turn aggressive and attack intruders. Of particular concern are the Californian sea lion (*Zalophus californianus*), found along the west coast of North America and the Galapagos, and the bearded seal (*Erignathus barbatus*), found on the edge of the ice along the coasts and islands of North America and northern Eurasia (Halstead et al., 1990).

11.3.4 Preventive measures

Preventive measures can be taken by the individual:

- Treat all animals with respect, and keep at a distance whenever possible.
- Avoid swimming in murky brackish water inlets, river mouths and mangrove swamps inhabited by saltwater crocodiles.

- Always try to obtain information from local authorities about the risk from hazardous organisms and ask for their guidance in risk prevention. If so advised, use a knowledgeable guide who can assess risks properly.

11.4 Venomous invertebrates

The effects of invertebrate venoms on humans range from mild irritation to sudden death. The invertebrates that possess some kind of venomous apparatus belong to one of five large phyla: Porifera (sponges), Cnidarians (sea anemones, hydroids, corals and jellyfish), Mollusca (marine snails and octopi), Annelida (bristleworms) and Echinodermata (sea urchins and sea stars).

11.4.1 Porifera (freshwater and marine)

Sponges are simple multicellular animals, living mainly in shallow coastal and fresh waters around the world. They either attach to some form of substrate (be it rock, seaweed or a hard-shelled animal) or burrow into calcareous shells or rock. Although most sponges are harmless to humans, examples of toxic sponges are found worldwide. Painful skin irritations, sometimes persisting for many hours, are the most common syndrome. No fatalities are known.

11.4.2 Cnidarians (marine)

Cnidarians are relatively simple, with a radially symmetrical body structure. Their body cavity has a single opening surrounded commonly by tentacles equipped with special cells known as cnidocytes. These cnidocytes contain characteristic capsule-like structures called cnidae, which in turn contain a thread that is mechanically discharged upon touch.

Cnidarians are separated into four groups: the Hydrozoa (plume-like hydroids, "fire corals," medusae and Siphonophora), Scyphozoa (free-swimming jellyfish), Cubozoa (box-shaped medusae) and Anthozoa (hard corals, soft corals and anemones). Hydroids and jellyfish possess so-called nematocysts (stinging capsules), which, when the cnidae thread is discharged, penetrate the integument (tough outer protective layer) of their prey and inject a toxin. Sea anemones and true corals, on the other hand, have spirocysts or ptychocysts with adhesive cnidae threads.

1. Hydrozoa

Most of the 2700 species of hydrozoa are harmless, but some can inflict painful injuries on humans. Well known examples of these are the sea firs, fire corals and Portuguese man-of-war. Apart from severe stinging cases from the Portuguese man-of-war, hydrozoan stings are not generally life threatening, although the pain can last for several days.

Stinging or fire corals (e.g., *Millepora alcicornis*) have nematocysts that vary in stinging intensity according to species (Sagi et al., 1987). These hydroid corals can cause a painful skin rash. They are generally found together with true corals in warm waters of the Indo-Pacific, the Red Sea and the Caribbean.

The stinging hydroid or fire-weed (*Aglaophenia cupresina*) is a hydroid colony. It resembles seaweed and grows on rocks and seaweeds in the tropical Indo-Pacific. If touched, it causes a nettle-like rash lasting several days (Rifkin et al., 1993).

The Portuguese man-of-war (*Physalia* spp.) is a free-swimming colony of open-water hydrozoans that lives at the sea–air interface. *Physalia* is easily recognized by the prominent floating blue or purple gas-filled bubble that supports the stinging cells on the tentacles and zooids hanging below. The tentacles may reach a length of up to 10 m. Different species of *Physalia* are widespread throughout all oceanic regions, except the Arctic and Antarctic, and may be blown onto beaches in swarms after strong onshore winds. The nematocysts remain active even when beached. Stings by the various *Physalia* species are the most common marine stings known at present. The Atlantic species (*Physalia physalis*) is the most dangerous and has been responsible for some severe stings (Spelman et al., 1982; Burnett et al., 1994) and three deaths (Burnett & Gable, 1989; Stein et al., 1989).

2. Scyphozoa and Cubozoa

The number and variety of potentially harmful Scyphozoa and Cubozoa are too numerous to mention here, but the subject has been widely reviewed by Burnett (1991) and Williamson et al. (1996). Williamson et al. (1996) give detailed accounts of the dangerous jellyfish species and describe the harm they can inflict on humans and the recommended treatment for stings from each of the individual species. Although most stings result in only a short-lived burning sensation, some can be dangerous, especially if the swimmer has a severe allergic reaction (Togias et al., 1985) or if the jellyfish is one of those rare species where the stings can be fatal.

The Scyphozoa, or true jellyfish, are typically pelagic and exist for the greater part of their life as medusae. They move by gentle pulsations of the bell, but are frequently driven ashore and stranded by wind and currents. All jellyfish are capable of stinging, but only a few species, particularly *Stomolophus nomurai* and *Sanderia malayensis*, are considered a significant hazard to human health (Mingliang, 1988; Williamson et al., 1996). Species of some genera, such as *Cyanea*, *Catostylus* and *Pelagia*, may occur in large groups or swarms.

The Cubozoa are the most dangerous cnidarians (Fenner & Williamson, 1996; Williamson et al., 1996). They are characterized by a roughly cube-shaped body or bell, with tentacles arising from fleshy extensions in each lower corner of the bell. Several species of box jellyfish have been implicated in human deaths, with *Chiropsalmus quadrigatus*, which causes some 20–50 deaths each year in the Philippines (Fenner & Williamson, 1996), and the Chironex box jellyfish *Chironex fleckeri*, found in summer months in the northern tropical waters of Australia (Baxter & Marr, 1969), being among the most venomous of all marine creatures. A death has also occurred in the south-eastern USA from the box jellyfish *Chiropsalmus quadrumanus* (Bengston et al., 1991). Respiratory failure may occur within a few minutes of being stung by *Chironex fleckeri* (Lumley et al., 1988).

3. *Anthozoa*

Hard corals can cause abrasion injuries if a swimmer simply brushes against their hard branches. Certain coral colonies also possess stinging nematocysts (*Goniopora*, *Plerogyra*, *Physogyra*), which can leave a rash if touched.

The majority of sea anemones are harmless, except when their tentacles come into contact with delicate parts of the body, such as the face, lips and underarms, resulting in a painful sting. One example is the common intertidal beadlet anemone (*Actinia equina*), found in the eastern Atlantic. More hazardous sea anemones include the hell's fire sea anemone (*Actinodendron plumosum*), found on the shady side of rocks and under coral ledges in the tropical Pacific. A sting from this anemone can cause skin ulcerations lasting for several months. *Triactis producta*, found in the Red Sea, gives painful stings that may later ulcerate (Halstead et al., 1990). A death has occurred after complications following a sting by *Condylactis* species (Garcia et al., 1994).

11.4.3 Mollusca (marine)

Molluscs are found in marine, freshwater and terrestrial environments. They all possess a distinct and well developed head, a muscular foot and a soft, variable-shaped body. Of the aquatic representatives of this large group, only some cephalopods and the cone shells (*Conus*) produce venoms harmful to humans.

All octopi possess two powerful horny jaws, which they can use to bite humans. The bites from the non-venomous (the majority) octopi result in small puncture wounds causing moderate pain. Certain species of octopus, such as the blue-ringed octopus (*Hapalochlaena* (= *Octopus*) *maculosa*) or the spotted octopus (*Octopus lunulatis*), are equipped with venom that aids in the capture of prey. Bites from these species can be deadly (the poisons are neuromuscular, producing muscular weakness and eventually respiratory paralysis) and should be treated with urgency (Williamson, 1987). Both species inhabit shallow coastal waters of the tropical Indo-Pacific and normally show no aggression towards humans. The majority of reported bites have resulted from handling or interfering with the octopi (Flecker & Cotton, 1955; Sutherland & Lane, 1969; Sutherland, 1983).

There are between 400 and 500 species of cone shells, all of them possessing a highly developed venom apparatus. The tropical and subtropical cone shells, *Conus* sp., are usually found in shallow waters along reefs and on or in sandy bottoms. They use their harpoon-like darts carrying the venom supply to catch prey and to discourage predators (Hinegardner, 1958). They often cause intense, localized pain at the site of the injury, accompanied by nausea, vomiting, dizziness and weakness. In more severe cases, victims experience respiratory distress with chest pain, difficulties in swallowing, marked dizziness, blurring of vision and an inability to focus. Fatalities are caused by respiratory paralysis (Flecker, 1936; Kohn, 1958; Endean & Rudkin, 1963; Russell, 1965). Most reported cases are from those organisms being handled.

11.4.4 Annelids (marine)

Of the annelids (segmented worms), only some bristleworms, named after two bristle-like setae attached to all their segments, are venomous. Bristleworms live under rocks and boulders. In venomous species, the setae sting; in the Caribbean fire worm (*Hermodice carunculata*), the sting leads to intense pain and a burning sensation.

11.4.5 Echinoderms (marine)

Very few of the radially symmetrical adult echinoderms are hazardous to humans. Most common minor injuries are abrasions or punctures acquired from contact with the spines or skin of echinoderms. Examples of venomous species are found only within the starfish and sea urchins.

The crown of thorns starfish (*Acanthaster planci*) is the only venomous starfish and lives on coral reefs in the Indo-Pacific. Its upper surface is covered with many long, sharp and venomous spines, which can inflict painful wounds if handled (Heiskanen et al., 1973). No serious injuries from *Acanthaster* have been recorded.

Sea urchins are found in all oceans, normally located on rocky foreshores and reefs. Most sea urchins can be handled safely, but a few species possess venomous spines or jaw-like pedicellariae capable of delivering very painful injuries (Halstead, 1971). These venomous species tend to be confined to the tropical and subtropical marine regions. Fatal incidents are said to have occurred from handling the flower sea urchin (*Toxopneustes pileolus*) from the Indo-Pacific, the most venomous sea urchin known, but these are difficult to confirm (Hashimito, 1979; Smith, 1977).

11.4.6 Preventive measures

Preventive measures can be taken by the individual:

- Always wear suitable footwear when exploring the intertidal area or wading in shallow water.
- Avoid handling sponges, cnidarians, cone shells, blue-ringed octopus, bristle-worms or the flower sea urchin.
- Avoid brushing against hydroids, true corals and anemones.
- Avoid swimming in waters where Portuguese man-of-war are concentrated (often indicated by beached specimens).
- If swimming where jellyfish are prevalent, wear a wet suit or other form of protective clothing, such as the full-length stretch-fitting suits used by divers in tropical waters.

11.5 Venomous vertebrates

Venomous vertebrates deliver their venom either via spines, as with many fish species, or through fangs, as in sea snakes. Injuries caused by venomous marine vertebrates are common, especially among people who frequently come into contact with these marine animals. Potent vertebrate toxins generally cause great pain in the victims, who may also experience extensive tissue damage.

11.5.1 *Catfish (freshwater and marine)*

Catfish are bottom dwellers living in marine, freshwater or estuarine environments. They possess venomous dorsal spines, which can inflict painful wounds even when the fish is dead (Halstead, 1988). The majority of catfish stings result from handling catfish while sorting fish catches. Some species, such as *Heteropneustes fossilis* from India, have been known to actively attack humans, leaving a painful sting (Williamson et al., 1996).

11.5.2 *Stingray (freshwater and marine)*

Stingrays are found in the Atlantic, Indian and Pacific oceans. They are predominantly marine, but the South American river ray (Pontamotrygonidae) lives in fresh water. Stingrays tend to be partially buried on sandy or silty bottoms in shallow inshore waters. Up to six venomous spines in their tails can stab unwary swimmers who happen to tread on or unduly disturb them. All stingray wounds, no matter how minor, should receive medical attention to avoid the risk of secondary infection. Some injuries caused by venomous stingrays can be fatal for humans if the spine pierces the victim's trunk; deaths have been reported for both marine (Rathjen & Halstead, 1969; Fenner et al., 1989) and freshwater (Marinkelle, 1966) species.

11.5.3 *Scorpionfish (estuarine and marine)*

All species of scorpionfish possess a highly developed venom apparatus and should therefore be treated with respect. The estuarine stonefish (*Synanceia horrida*, syn. *S. trachynis*) is the most venomous scorpionfish known and occurs throughout the Indo-Pacific. The reef stonefish (*Synanceia verrucosa*) resembles coral rubble and lies motionless in coral crevices, under rocks, in holes or buried in sand or mud, where divers often mistake it for a rock. The pain associated with stings by a stonefish is immediate and excruciating and can last for days (Williamson et al., 1996). The lionfish and true scorpionfish are also venomous. Deaths have been attributed to stonefish but are very difficult to confirm (Smith, 1957; Cooper, 1991).

11.5.4 *Weeverfish (marine)*

Weeverfish are confined to the north-eastern Atlantic and Mediterranean coasts. All four species (*Trachinus* spp. and *Echiichthys* sp.) contain venomous dorsal and gill cover spines. They are small (less than 4.5 cm) and lie partly buried in sandy bays at extreme low water where swimmers and beach walkers frequently step on them. Weeverfish are regarded by some as the most venomous fish found in temperate European waters (Halstead & Modglin, 1958; Russell & Emery, 1960).

11.5.5 *Surgeonfish (marine)*

Surgeonfish are herbivorous reef dwellers equipped with a sharp, moveable spine on the side and base of the tail. When excited, the fish can direct the spine forward, making a right angle with the body, ready to attack. Large surgeonfish, such as the Achilles surgeonfish (*Acanthurus achilles*) and the blue tang (*Acanthurus coeruleus*) of the warm seas of the western Atlantic, use their spines in defence and cause deep and painful wounds with a quick lashing movement of the tail (Halstead et al., 1990).

11.5.6 *Snakes (freshwater and marine)*

Poisonous snakes are air-breathing, front-fanged venomous reptiles, and many are associated with both the marine and freshwater environments. Of the 50 species of sea snake, the majority live close inshore or around coral reefs. They appear similar to land snakes, but have a flattened tail to aid in swimming. They are curious, generally non-aggressive creatures, but can be easily provoked to attack. All sea snakes are venomous and can inflict considerable harm if disturbed. White (1995) estimated a worldwide sea snake fatality rate of at least 150 per year.

Of the freshwater aquatic snakes, possibly the water moccasin or cottonmouth (*Agkistrodon piscivorus*) is the most dangerous to humans, the venom attacking the nervous and blood circulatory systems of the victim. The water moccasin is a pit-viper found throughout the south-eastern part of the USA. The species is never far from water and swims with its head well above the surface. When threatened, the snake opens its mouth wide to reveal the almost white lining, which gives it its common name. The species can be aggressive and is densely populated in some areas. Its bite can result in gross tissue damage, with amputations of the affected limb not uncommon (Caras, 1976). Other species of the genus *Agkistrodon* are found throughout North America and south-eastern Europe and Asia.

11.5.7 *Preventive measures*

Preventive measures can be taken by the individual:

- Always "shuffle" feet when walking along sandy lagoons or shallower waters where stingrays frequent.
- In catfish waters, fishermen should be extremely careful when handling and sorting their catch.
- Suitable footwear should be worn to avoid accidentally treading on weeverfish or stonefish.
- Wear boots in snake-infested areas.
- If possible, carry anti-venom in snake-infested areas.

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CHAPTER 12

Monitoring and assessment

In order to support the attainment of safety in recreational water environments, the responsible management authorities should establish a programme for evaluating existing hazards and monitoring the area for any changes that may occur. Threats to human health may include natural hazards, such as surf, rip currents or aquatic organisms or may have a man-made aspect, such as discharges of wastewater. Comprehensive review of the recreational area and monitoring for any changes enables a responsive strategy to protect public health be implemented.

To provide practical guidance concerning the design and implementation of monitoring programmes for recreational water use areas, WHO developed a book *Bathing Water Monitoring* (Bartram & Rees, 2000). The structure is based upon a framework "Code of Good Practice (COGP) for Recreational Water Monitoring," which is presented in this chapter. The Code was developed through an extensive process of consultation and within the context of cooperation between WHO and the European Commission. The application of the Code under specific circumstances is described in greater detail in the book.

This framework COGP constitutes a series of statements of principle or objectives that, if adhered to, would lead to the design and implementation of a credible monitoring programme. It applies in principle to the monitoring of all waters used for recreational activities that involve repeated or continuous direct contact with a water body. In many circumstances, there are different approaches or methods that can be applied to achieve the objectives stated in the Code. While equally valid in isolation of one another, the adoption of diverse approaches within a single programme may mean that results are not comparable between different locations or enforcement programmes.

The framework COGP provides a linkage to the various health effects associated with recreational waters and incrementally builds up the component parts of a successful programme—key health issues, monitoring and assessment strategies, and principal management considerations.

12.1 Design and implementation of monitoring programmes

12.1.1 Design of monitoring programmes

1. The objective(s) of a monitoring programme or study should be identified formally before the design of the programme and stated prior to data gathering. Ideally objectives would be based on assessment of the frequency and

severity of different adverse health outcomes, with the subsequent monitoring programme designed around those with the greatest public health benefit.

2. Objectives should be described in a manner that can be related to the scientific validity of the results obtained. The required quality of any data should be derived from the statement of objectives and stated at the outset.
3. Where data (such as results from water quality analyses) are to be compared between laboratories or between sites, all available measures to ensure comparability of results should be implemented:
   - A quality assurance programme based on internal controls and external controls (interlaboratory comparisons) is essential.
   - Criteria should be developed for dealing with participating laboratories consistently failing to comply with minimum analytical quality. These should be stated prior to data collection.
4. In designing and implementing monitoring programmes, all interested parties (legislators, nongovernmental organizations, local communities, laboratories, etc.) should be consulted. Every attempt should be made to address all relevant disciplines and involve relevant expertise.
5. The scope of any monitoring programme or study should be defined. This would normally take the form of definition of criteria for inclusion/exclusion of recreational water use areas and preparation of an inventory of recreational water use areas.
6. The catalogue of basic characteristics of all recreational water use areas should be prepared and updated periodically (generally annually)—and also in response to specific incidents—in a standardized format. It should include as a minimum the extent and nature of recreational activities that take place at the recreational water use area and the types of hazards to human health that may be present or encountered. Unless specifically excluded, the list of potential hazards to human health would normally include drowning and injury-related hazards, known or anticipated dangerous aquatic organisms, microbial quality of water and cyanobacteria or harmful algae. Monitoring programmes frequently also address aesthetic aspects and amenity parameters because of their importance to health and well-being.
7. Programme or study design should take account of information derived from the inventory of recreational water use areas and catalogue of basic characteristics, which may require refinement of programme objectives.
8. The logistical planning of any monitoring programme or study should take account of socioeconomic, technical/scientific and institutional capacities, staffing, equipment availability, consumable demands, travel and safety requirements and sample numbers, without compromising achievement of the objectives or scientific validity of the programme or study.
9. The hierarchy of authority, responsibility and actions within a programme or study should be defined. All persons taking part in the programme or study should be aware of their roles and inter-relationships.

10. Staff should be adequately trained and qualified, including with regard to health and safety aspects.
11. Monitoring programmes should include appropriate quality assurance (QA), which does not infringe on health and safety and which covers the integrity of all observation, interview, field sampling and water quality analyses as well as data input, analysis and reporting.
12. A QA Officer should be appointed who reports directly to senior management. The QA Officer should regularly audit all aspects of the operation with special regard to procedures, traceability of the data and reporting.
13. Essential elements of QA programmes include:
    — The writing and implementation of a Quality Manual and Standard Operating Procedures (SOPs). All SOPs should be regularly overhauled and updated as necessary and any deficiencies reported and appropriate remedial action taken.
    — SOPs should include maintenance and updating of inventories and catalogues; methodologies for all major equipment; all sampling and analytical procedures; sample receipt, screening and storage; and reporting.
14. Where samples are taken for laboratory analysis they should be registered on arrival at the laboratory. The applied laboratory procedures should conform to the SOPs defined at the laboratory. Where possible, all analytical procedures should follow defined protocols (e.g., International Organization for Standardization or American Public Health Association protocols). All equipment should be calibrated regularly and the operational procedures submitted to quality control staff in order to guarantee traceability of the data.
15. Laboratory accreditation can form a valuable part of activities relating to analytical quality, e.g., through pursuit of requirements for ISO/IEC 17025.
16. The programme should be evaluated periodically and whenever the general situation or any particular influence is changed. Commitment to support such evaluations should be built into the monitoring programme's design and authorization.

12.1.2 Data collection

17. Collection of data and information should utilize the most effective combination of methods of investigation, including:
    — observation;
    — historical review of deaths, injuries and accidents (including details on lifeguard positioning, number of rescues effected, preventive actions and attendance figures);
    — water quality sampling and analysis;
    — interview of appropriate persons; and
    — review of published and unpublished literature.
18. Frequency and timing of analytical sampling and selection of sampling sites should reflect recreational water area types, use types and density of use, as

well as temporal and spatial variations in the recreational water use area, which may arise from seasonality, tidal cycles, rainfall and discharge and abstraction patterns.

19. Analytical sampling should provide a data set amenable to statistical analysis.

12.1.3 Data handling

20. Data handling and interpretation of results should be done objectively, without personal or political interference.
21. The need for transformation of raw data, before analysis, to meet the conditions for statistical analysis should be agreed upon with a statistical expert before commencing analysis. In addition, procedures should be defined for handling censored data (such as 'less than' and 'greater than' data).
22. Data handlers and collectors should agree on a common format for recording results of analyses and surveys and should be aware of the ultimate size of the data matrix. The preferred approach is to use a database or spreadsheet that allows automatic logical verifications (i.e., only allows entries for certain date and numeric ranges). Forms and survey instruments should be compatible with this format. Likewise, data handlers should agree on a format for the output of results with those responsible for interpreting and presenting the data. Data entry should be double checked to ensure accuracy.
23. Procedures for dealing with inconsistencies such as omissions in records, indeterminate results (e.g., indecipherable characters, results outside the limits of the analytical methods) and obvious errors should be agreed upon in advance of data collection. On receipt from the data collectors, record forms should be examined and the agreed procedure followed. Discrepancies should be referred immediately to the data collector for correction or amendment. Where correction is not possible, resampling is generally the preferred option (with due regard for prevailing conditions); estimates may be preferable to leaving gaps in the record. Such estimates, however, must be recorded as such and the methodology of the estimate outlined.
24. Ideally, arrangements should be made to store data in more than one location and format, to avoid the hazards of loss and obsolescence. At all locations, data should be backed up regularly. Data should be transcribed accurately, handled appropriately and analysed to prevent errors and bias in the reporting.
25. The statistical methodologies should be reviewed by a statistical expert and comments taken into account in finalization.
26. Data should be handled and stored in such a way to ensure that the results are available in the future for further study and for assessing temporal trends.

12.1.4 Data interpretation

27. Data should be interpreted and assessed by experts with relevant recommendations for management actions prior to submission to decision-makers. Interpretations should always refer to the objectives and should also

propose improvements, including simplifications, in the data gathering activities, identifying future research needs and guidelines for environmental planning.

28. Interpretation of results should take account of all available sources of information, including those derived from inventory, catalogue of basic characteristics, sanitary and hazard inspection, water quality sampling and analysis, and interview, including historical records of these.

12.1.5 Reporting

29. The findings should be discussed with the appropriate local, regional and/or national authorities and others involved in management (including integrated water resource management), such as the industrial development and/or national planning boards.
30. Results should be reported to all concerned parties, including the public, legislators and planners. Any information relating to quality of recreational water use areas should be clear, should be concise and should integrate safety, microbial and aesthetic aspects.
31. In issuing information to concerned parties (the public, regulators, non-governmental organizations, legislators, etc.), it is essential that their requirements are kept in mind.
32. Where specific or extreme events that may threaten public health occur, the competent public health authority should be informed and recommendations should be made to the water user population about the risks of dangerous water conditions or poor water quality (see chapter 13).
33. Reports addressing the quality of recreational water use areas should be accompanied by reference to local and visitor perceptions of the aesthetic quality and risks to human health and safety (see chapter 9).
34. The deleterious impacts of human health hazards and aesthetic pollution and control measures to avoid or reduce such impacts should be introduced into environmental health education programmes in both formal and informal educational establishments.
35. The usefulness of the information obtained from monitoring is limited unless a supportive administrative and legal framework (together with an institutional and financial commitment to appropriate follow-up action) exists at local, regional and international levels.

12.2 Aspects relevant to specific hazards

The following items apply in addition to the general guidance given above in relation to specific hazards. As noted in chapter 1 (Figure 1.2), in order to maximise public health gains management authorities should prioritize according to the hazards having the most serious outcomes. Thus, generally speaking, drowning prevention measures would be prioritized over general beach cleaning. The reader should also refer to the relevant chapters in this volume.

12.2.1 *Drowning and injury hazards*

1. The catalogue of basic characteristics should include, wherever relevant, hazards such as beach slopes, tides, flows and currents, actual user groups, nearby hazardous areas such as cliffs, shallow waters dangerous for diving, weirs and other such hazards as identified from local knowledge and records of health effects.
2. Information regarding measures to prevent or ameliorate hazard exposure or outcomes, including, for example, lifeguard provision, staff training, signs, emergency telephone numbers, access to first aid, medical facilities, fencing, warning systems for adverse conditions and emergency routes, should be included in the catalogue of basic characteristics.
3. Monitoring and assessment programmes should address those hazards and preventive measures, described in 1) and 2), that are subject to change.
4. When assessing the significance of hazards, account should be taken of the severity and likelihood of adverse health outcomes, together with the extent of exposure.

12.2.2 *Microbial water quality assessment and sanitary inspection*

5. Sanitary inspection should be undertaken as a necessary adjunct to microbial water quality analysis to identify all real and potential sources of microbial contamination. It should assess their impact on the quality of the recreational water use area and water user health. During inspection, the temporal and spatial influences of pollution on water quality should receive full consideration.
6. An exhaustive sanitary inspection should be carried out immediately prior to the principal bathing season. Inspections of specific conditions should be conducted in conjunction with routine sampling during the bathing season. Pertinent information should be recorded on standardized checklists and used to update the catalogue of basic characteristics. If a problem is identified, it may be necessary to collect supplementary samples or information to characterize the problem.
7. Visual faecal pollution or sewage odour should be considered a definite sign of elevated microbial pollution, and necessary steps should be taken to prevent health risks to bathers.
8. SOPs for sanitary inspections, water sampling (including depth) and analyses should be well described to ensure uniform assessments.
9. Sample point location and distance between each should reflect local conditions (overall water quality, recreational usage, predicted sources of faecal pollution, temporal and spatial variations due to tidal cycles, rainfall, currents, onshore winds and point or non-point discharges) and may vary widely between sites.
10. Sterile sample containers should be used for microbiological samples. Scrupulous care should be taken to avoid accidental contamination during handling

and sampling collection. Every sample should be clearly identified with time of collection, date and location.

11. A sampling depth relevant for the exposure of concern should be selected and adhered to consistently in order to allow comparison between locations.
12. Samples should be kept in the dark and maintained as cool as possible within a chilled insulated container and delivered to the laboratory promptly after collection. Samples should be analysed as soon as possible and preferably within 8 h of collection. Sample storage is recommended not to exceed 24 h at 5 °C.
13. Additional information should be collected at the time of sampling, including water temperature, weather conditions, water transparency, presence of faecal material, abnormal discoloration of the water, floating debris, cyanobacterial or algal blooms, flocks of seabirds and any other unusual factors. All information should be recorded on standardized checklists.
14. Local and national conditions should be taken into account when selecting appropriate microbial indicators.
15. The influence of specific events such as rain on the recreational water use areas, especially in relation to the duration of the peak contamination period, should be established and prior agreed procedures implemented.
16. Extreme events such as epidemics and engineering and natural disasters may require additional measures to ensure there is no additional risk associated with recreational water use areas.
17. The procedures to be used for transformation of raw data to meet the statistical requirements should be agreed upon with the statistical expert prior to analysis. The most usual need is to transform bacterial counts to logarithms and to convert their approximately lognormal frequency distribution to normality.
18. When unexpectedly high microbiological results are obtained, resampling should be undertaken to help determine whether this was due to sporadic events or persistent contamination. In the latter case, the source of pollution should be established and appropriate action taken.

12.2.3 Cyanobacteria and algae

19. Monitoring of recreational water use areas should be sufficient to identify risk of blooms, taking into account actual or potential accumulation of toxic cyanobacteria and algae.
20. Sampling points should be located to represent different water masses (stratified waters, waters coming from river mouths, etc.) in the investigation area and the sources of nutrients (discharges, upwellings, etc.). Possible transport mechanisms of toxic phytoplankton should be considered, wind induced accumulations of scum should be identified and sampling schemes should be arranged accordingly.
21. In areas of high risk, sampling for algae should be carried out at least weekly. During development of blooms, sampling should be intensified to daily.

22. Monitoring of toxicity (using bioassays, chemical or immunological procedures) is justified only where reason exists to suspect that hazards to human health may be significant. In such cases, long-term information on phytoplankton populations (toxic, harmful and others) should be collected where appropriate.
23. Analyses of toxins should be undertaken only where standard, replicable and reliable analyses can be performed.
24. Where conditions are such that monitoring is considered essential, temperature, salinity (in marine coastal areas), dissolved oxygen, transparency, presence of surface water stratification, phytoplankton biomass (chlorophyll), surface current circulation (transport of algae) and meteorological patterns such as seasonal rainfall, storms and special wind regimes should be considered.

12.2.4 Other biological, physical and chemical hazards

25. Monitoring for other locally important hazards is justified only where reason exists to suspect that hazards to human health may be significant. Such occurrence may be highly localized.
26. Only where standard, replicable and reliable analyses may be undertaken for known parameters should such analyses be undertaken.
27. Approaches to the assessment of the significance of locally important hazards will depend on the type of hazard and should take account of their magnitude and frequency, severity and occurrence of health effects, and other local factors.

12.2.5 Aesthetic aspects

28. Selection of aesthetic pollution parameters for monitoring should take into account local conditions and should consider parameters such as surface accumulation of tar, scums, odours, plastic, macroscopic algae or macrophytes (stranded on the beach and/or accumulated in the water) or cyanobacterial and algal scums, dead animals, sewage-related debris and medical waste.
29. Assessment of aesthetic pollution indicators should take into account the perception and requirements of the local and any visiting populations in reference to specific polluting items as well as the feasibility of their monitoring.

12.3 Progressive implementation of monitoring and assessment

To protect health it is necessary to develop monitoring orientated towards hazards to human health in response to public health priority. This will normally mean that several aspects (beach safety, pollution control, etc.) will be developed in parallel. There are different levels of monitoring (as there are with management, see chapter 13), although each level deals with each of the major hazard groups (as outlined in Table 12.1). Typically, monitoring proceeds through local activities in isolation

TABLE 12.1. LEVELS OF MONITORING IN RELATION TO RESOURCE REQUIREMENTS[a]

Level	Basic information /visit rate	Accident hazards	Microbiological parameters	Cyanobacteria and algae	Other
Local (no national organization)	Local action comparable to basic level, in some locations only.	Local action comparable to basic level, in some locations only.	Local action comparable to basic level, in some locations only.	Local action comparable to basic level, in some locations only.	Local action comparable to basic level, in some locations only.
Basic (no access to equipment or staff resources at national level; limited local resources)	At least one pre-season visit; creation of a catalogue of basic characteristics; all recreational waters registered, but more–used and higher–risk beaches inspected and monitored.	Annual inspection for identification of any hazards and interventions (e.g., signs, warning systems).	Inspection for faecal pollution or sewage odour; delimitation of high risk areas; initial screening of microbial indicator parameters for primary classification; internal quality control at laboratories; at least one sample a month once the recreational water is classified.	Inspection for scum, type and transparency.	Register of local special problems.
Intermediate (limited access to resources both local and national level)	Comprehensive cataloguing and timetabling of visits; additional visits during peak seasons (e.g., monthly); greater proportion of recreational waters monitored.	Periodic verification of interventions during bathing season; central capacity for incident investigation.	Identification and cataloguing of potential sources of contamination; all recreational waters at primary classification; monthly sampling; additional sampling and investigation of unexpected peak values; reclassification scheme initiated; investigation of rain effects and design of preventive measures; internal quality control at laboratories; occasional inter-laboratory comparison studies.	Phosphate analysis (freshwater) Chlorophyll a (freshwater) where bloom events probable.	Check on local information availability; active warning and management response.
Full (no significant resource limitations)	Additional visits during peak seasons (e.g., fortnightly or weekly); complete cataloguing, including updating for each recreational area; all beaches with significant use monitored.	Central register of recorded incidents; decentralised capacity and procedure for incident investigation.	Additional microbiological parameters if necessary; possible reclassification investigated where indicated; internal and external quality controls regularly operated; convergence among participating laboratories.	Toxicity detection and toxin analysis capacity if necessary (not routine); remote sensing methods where relevant.	Chemical monitoring (for appropriate parameters).

[a] adapted from Bartram & Rees, 2000.

of any national or regional framework through basic, intermediate to full scale monitoring.

Extensive guidance on the development of practical and effective monitoring programmes for the safety of recreational water environments is presented in Bartram & Rees (2000).

12.4 References

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

ISO/IEC 17025 (1999*) General requirements for the competence of testing and calibration laboratories*. International Organization for Standardization, Geneva, Switzerland.

CHAPTER 13
Application of guidelines and management options for safe recreational water use

Recreational use of inland and marine waters is increasing in many countries worldwide. These uses range from whole-body contact sports (where there is a significant risk of water ingestion), such as swimming, surfing and slalom canoeing, to non-contact activities, such as fishing, walking, birdwatching and picnicking. Although these activities can benefit health, they may also be associated with adverse health outcomes, as described in previous chapters. These possible adverse health outcomes result in the need for guidelines that can be converted into locally (i.e., nationally or regionally) appropriate standards and associated management of sites to ensure a safe, healthy and aesthetically pleasing environment. The management interventions that may be required to ensure a safe recreational water environment include compliance and enforcement measures, application of control and abatement technology, public awareness and information initiatives and public health advice, which are best brought together in an integrated management framework (summarised in Figure 1.4). This chapter brings together the conclusions of the management strategies and options discussed in previous chapters.

13.1 Application of guidelines

Recommended guidelines for a number of hazards and associated risks to public health have been outlined in preceding chapters. The guidelines and recommendations range from identifying the need for providing advice to the public (chapter 3) to numerical guidance levels (chapter 8) to a system of classification (chapter 4). Chapter 1 and several other chapters have emphasized the need to adapt these guidelines to suit local circumstances.

Guidelines are intended to be flexible and should be adapted to suit regional, national and/or local circumstances by taking into consideration socio-cultural, environmental and economic conditions. An initial assessment of issues and priorities can, for example, include an assessment of the number of drownings or serious injuries sustained (i.e., severe health effects) in comparison to, say, cases of mild illness as a result of bathing in microbially contaminated water (see Figure 1.2). Initial assessment would preferably be complemented by a risk-benefit approach (qualitative or quantitative) and in some circumstances a full cost-effectiveness or cost-benefit analysis may be undertaken. The outcome of such analyses should inform the process of standards development and the measures that are put in place to implement the standards.

The agency responsible for health will take a leading and coordinating role in the application of guidelines. However, the health authority should ensure the active participation of the other key stakeholders as outlined in chapter 1 (Figure 1.3). A wide variety of elements of legislation and regulation may contribute to ensuring and/or improving the safety of the recreational water environment. Not all are relevant or appropriate to all types of hazard and the balance among them will depend on the nature of the hazards of priority concern for human health. Experience suggests that overall health protection is most effective when a number of complementary mechanisms are employed. The potential "actors" and functions involved in improving safety are outlined in Table 13.1.

TABLE 13.1. EXAMPLES OF ACTORS AND FUNCTIONS THAT MAY BE EXERCISED IN MANAGING RECREATIONAL WATER ENVIRONMENTS FOR SAFETY

Example of authority or activity	**Comments**
Facility operator/service provider	Agencies developing facilities or providing services may be responsible for the safety of those locations, or this may be seen as an element of 'duty of care' or 'due diligence'. Recreational water-specific requirements may include the establishment and implementation of a 'safety plan' (in consultation with other stakeholders, including agencies responsible for safety and health—see section 13.2). This would normally include an assessment of hazards, including reference to user groups; a programme for monitoring and assessment; a water safety plan (which would include 'normal' and 'incident' circumstances and include a communication strategy to stakeholders).
National authority responsible for public health	Responsible to maintain and update national standards, e.g., recreational water quality standards, including sampling regimes and methods, analytical methods, analytical quality control and inter-laboratory comparisons, reporting. Maintenance of lists of national recreational water use locations. Surveillance of injury and illness in the community.
Local authority responsible for public health	Authority and responsibility to advise local facility developers/service providers and municipalities on public health aspects of the activities and resources under their supervision. Authority and responsibility to intervene when made aware of imminent or actual severe threat to public health at a recreational water location, including advising against use for a determined period or until safe conditions are re-established.
Authority responsible for safety	May be multiple and some may be non-governmental (e.g., lifesaving federations). Often responsible for development and implementation of voluntary codes of good practice (e.g., for lifeguard qualifications and activities). The fact that they are voluntary does not reduce their importance and they may be a major aspect of safety promotion.
Local tourism body	Provision of information to the public.
Certification agencies	The certification process is used to verify that devices (such as life belts) meet a given level of quality and safety based on agreed standards.
Recreational water/facility user	Exercise informed choice and take personal responsibility (e.g., use of sunscreen, avoiding excess alcohol).

In regulatory monitoring programmes factors, such as frequency of inspection and/or sampling, analytical methods, data analysis, interpretation and reporting, sample site selection and criteria for recreational water use areas will generally be defined by the regulatory agency and should take account of the principles outlined in chapter 12.

Box 13.1 uses the implementation of the recreational water quality classification system (outlined in chapter 4) to illustrate points to be considered in adaptation of guidelines to specific, locally-appropriate regulatory provisions, as it is potentially the most complex.

BOX 13.1 GUIDELINE ADAPTATION (USING THE RECREATIONAL WATER QUALITY CLASSIFICATION SYSTEM AS AN EXAMPLE)

The principal requirements that would need to be incorporated into provisions would normally include:

1) The definition of "water user" or "bather", "recreational water" and, if the use is seasonal for the majority of users, "bathing season".
2) The establishment of a water quality classification system based on:
   a) defined statistics from microbial water quality assessment;
   b) defined levels of the probability of the sewage pollution of the recreational water (with the assessment to be based on inspection of the conditions during the defined bathing season);
   c) defined means to combine a) and b) to provide a broad classification of risk to public health.
3) The obligation upon national/regulatory authorities to maintain a listing of all recognized recreational water areas in a publicly accessible location. This would, typically, be the same location as used to inform the public of the recreational water classification.
4) The establishment of procedures, responsibilities and authority for progressively updating 2a) and 2b) in light of new scientific information and developments.
5) The definition of responsibility for:
   a) establishing a water safety plan (including "posting" to warn of poor water quality, monitoring and sanitary inspection) and its implementation (e.g., local authority, private facility manager or service provider, lifeguard association, etc.);
   b) independent surveillance, including recreational water classification (e.g., local government, public health body, environment agency/authority);
   c) provision of information to the public (e.g., public health body, local authority, local tourism body);
   d) interpretation of the significance of "exceptional circumstances" (e.g., public health body).
6) The obligation to act. This would include:
   a) the requirement that on detection of conditions potentially hazardous to health, or uncharacteristic of the location, to immediately consult with the public health body and inform the public as appropriate;
   b) a general requirement to strive to ensure the safest achievable recreational water use conditions, including implementation of measures in order to improve classification of recognized recreational water areas by available means (including pollution control and abatement) and the discouragement of the use of locations that present an especially high risk (i.e., the worst classification category);
   c) encouragement of advisory action at times of high risk of disease at locations where water quality deterioration is sporadic and predictable. Where such action can be shown to be effective, this should be taken into account in the classification scheme outlined in 2).

Most of the factors outlined in Box 13.1 will apply to the derivation and adaptation of any recreational water standard. In addition to illustrating the application of guidelines, Box 13.1 also highlights the importance of the multiple stakeholders (see Figure 1.3) involved in the process of adapting and applying guidelines and standards.

13.2 Recreational water safety plan

One way in which all the potential hazards outlined in previous chapters can be brought together, on a location specific basis, is through a recreational water safety plan. As outlined in Table 13.1 this would include an assessment of locally relevant hazards (including reference to user groups), a programme for monitoring and assessment and a management plan, which would detail both normal and incident (or exceptional) circumstances. It is suggested that such a safety plan is adapted from a country or regionally specific generic plan which could include a hazard rating scheme and also an overall recreational water rating, as outlined in chapter 2. The advantage of adapting a generic plan is that all recreational water areas in a specific area would then be rated against the same scale, improving informed personal choice.

13.3 Compliance and enforcement

"Watchdog" institutions responsible for the programmed process of monitoring quality indicators—i.e., sampling, measurement and subsequent recording of various characteristics (e.g., governmental environmental agencies/local authorities, with analysis being carried out by hospitals, public health or university laboratories)—should assess the conformity of recreational water areas to local or national standards. In those countries where it is difficult to achieve guideline objectives, central and local governments may set interim standards to ensure a progressive improvement towards local regulatory limits and possibly to desirable conditions.

13.3.1 Responsibility for risk management

Risk management is the making of decisions on whether or not risks to well-being are acceptable or ought to be controlled or reduced, based on evaluation of risks together with the identification and application of preventative or control strategies. The making of these judgements involves value judgements of some kind, whether a formal evaluation of costs of detriment from the hazard and the benefits of improvements or a subconscious personal evaluation.

Responsibility for managing risks in water recreation takes place at two distinct levels:

- society regulators, through central and local government and providers of recreational facilities; and
- participants in the activities, whether personally or collectively (see section 13.5).

The regulatory functions in risk management are very much the same as in other systems where public health and well-being are involved, such as drinking-water supply and food hygiene. They involve a devolvement of responsibilities downward and of reporting upwards. Responsibilities for monitoring may be devolved to an environmental agency or to local authorities, with analysis being carried out by hospital, public health or university laboratories. Local authorities may own or control access to public beaches and recreational water areas and thus fall into the category of provider. This role should be independent of a local authority's responsibility for public health (e.g., closing beaches and other recreational facilities deemed hazardous to health and safety). This latter responsibility is a well-defined role of a local authority's department of environmental health, the local medical officer for environmental health or equivalent. Central government and local authorities have a responsibility for informing the public about health issues in water recreation (Table 13.1).

13.3.2 Regulatory compliance

A number of problems affect the application of regulatory compliance and restrict the usefulness of this approach. For example, a marginal failure in water quality may be due to one of a number of contributing pollution sources. In the case of microbial quality, it is frequently the case that a number of sources—which may include riverine discharge, sewage, storm outflows, solid waste and agriculture—may all contribute and may be the responsibility of different authorities (hence the usefulness of an "umbrella" type of management framework, such as that provided by ICAM). A further problem concerns the issue of temporal variation. While most regulatory regimes require compliance based on a proportion of time, periods of high risk may be brief and either undetected by such regimes, which exposes the public to increased risk, or overestimated, thereby condemning an otherwise safe location. Finally, it should be recalled that legislation generally applies to specifically designated areas, e.g., government-defined bathing beaches, rather than to all potential recreational water use areas. Special interest groups and users of less-frequented locations may not be properly protected under such regimes.

An alternative approach to assessing regulatory compliance as a failure of a recreational water to achieve a certain water quality is provided by the "obligation to act" stipulation outlined in Box 13.1 and Section 4.7.3. This requirement would mean that failure to respond to the detection of conditions potentially hazardous to health would lead to non-compliance rather than the measured water quality (for example) falling below a certain measure.

The role of regulatory compliance is not, however, restricted to pollution control and may successfully be extended to the implementation of policy regarding areas suitable for development and provision of minimum facilities and supervision by local operators—for instance, in terms of lifeguards (see Appendix A) and first aid facilities.

There are two kinds of regulatory action. *Local action* consists of improvements to facilities to eliminate hazards and thereby to reduce risks. Examples are the construction of sewage treatment works and long sea outfalls to reduce contamination of the sea with sewage or designating areas to be used for waterskiing, which do not conflict with bathing. *Policy implementation* (regional, national or international) usually takes the form of creating standards or guidelines to control risk. Inherently, standards provide a means of judging whether conditions are acceptable or not and, therefore, whether improvements are needed. They also provide a means of identifying whether intervention to reduce exposure is required, such as through provision of public advice, closing areas etc. Purpose-designed programmes of monitoring (see chapter 12) and analysis must accompany them, which provide information on quality.

13.3.3 Enforcement

Enforcement is an essential component of the regulatory system. Strong enforcement of a regulatory approach, however, may also focus attention on the high cost of, for example, pollution control intervention, and in some cases it has been argued that this is disproportionate to the public health benefit obtained. Again, an "obligation to act" regulation may minimise this problem. As point 6b) in Box 13.1 suggests, a general requirement to ensure the safest achievable recreational water use includes the discouragement of use of inappropriate or very polluted sites and not just pollution control measures.

Pollution control measures are most effectively deployed within a wider context of ICAM (see Box 13.2). In order to be effective, standards, guidelines and codes of practice must address the root causes of hazards. For example, medical waste found on a beach should be cleared, but sourcing it and preventing ongoing contamination is the prime consideration.

13.3.4 Monitoring and reporting

One purpose of guidelines, standards and regulations should be to promote improvement, and thus monitoring and enforcement should focus upon this. Proper information and positive incentives are often more effective ways to achieve improvement than the implementation of sanctions.

Results of monitoring programmes should be made readily available to participants in a timely manner, so that they can make informed decisions on using the facilities, and to regulators, so that they can take decisions with facility owners to carry out needed improvements. The public is also entitled to receive the results of monitoring so that individuals can choose whether or not to visit a particular beach or recreational water (see section 13.5 and Box 9.2).

13.4 Control and abatement technology

As health risks in certain environments become apparent (because of either changing risks or improved detection), responsible institutions (e.g., water companies, agricultural agencies, beach or recreational facility managers and so on) should identify the causes and put in place measures to combat the risks. Detection of health risks should be objective, e.g., based on systematic surveys. Implementation of remedial action should be in accord with an integrated management framework as outlined earlier and may include the control and abatement of pollution discharges with respect to the various levels of sewage treatment (chapter 4), control of agricultural runoff (chapter 8), fencing of dangerous areas (chapter 2), beach cleaning (chapters 2 and 6), provision of lifeguards (chapter 2 and appendix A), etc. Zoning and use separation can also be simple, but effective, control measures (chapter 2).

In terms of pollution control and abatement technology, the *required* design criteria for an intervention would result in a low to intermediate health risk, while the *preferred* design criteria would result in a minimal or low health risk (Figure 1.4).

Within an integrated planning process, tools (such as environmental health impact assessment, environmental audits and quality standards) can be designed and enforced. Stakeholders including industrial representatives should be involved in the discussion throughout the whole process to ensure that priority concerns are taken into consideration and that the proposed tools are generally acceptable, which would facilitate compliance. A development plan would also include land use plans, overall legislation and regulation and could advocate the use of such tools as economic instruments to manage the recreational waters.

13.4.1 Health impact assessment

Planning for the development of new recreational water projects or for the upgrading of existing ones offers ample and timely opportunities to incorporate human health considerations. A health impact assessment (HIA) provides the method and procedures to ensure such incorporation in a systematic, comprehensive and focused manner. The HIA approach considers changes in environmental and social determinants of health resulting from development. Both types of health determinants are relevant in the context of recreational water projects. HIA should be linked to the environmental assessment, but must maintain a distinct profile.

The rationale for HIA is firstly economic. It allows design options and management measures to be integrated into the project rather than relying on strengthening health services or the need for subsequent remedial action to be implemented at a generally higher cost. Such an after-the-fact remedial approach is undesirable, because it usually signifies a transfer of hidden costs to the health sector. HIA will also contribute to improving the health status in the project area. It aims to identify not only adverse health effects but also health opportunities. The measures recommended on the basis of the HIA should be designed to take into account inequities in health status and to overcome a disproportionate burden of exposure to health risks of vulnerable groups.

HIA starts by setting boundaries and priorities, a process known as scoping and screening. In recreational water projects, the physical boundaries for HIA will often coincide with the project boundaries, but they may stretch beyond, to include communities downstream from a project on a river system or further along the coast, depending on prevailing currents. HIA is a predictive exercise, and it should, therefore, not only include communities currently inhabiting the project area, but also groups of people that may enter and settle temporarily or permanently. In recreational water projects, these may include temporary labour employed during the construction phase, new staff that have come to work at the recreational facilities and the project's target group itself: people who come to use the recreational facilities. Within these groups, those with particular vulnerabilities should be identified.

HIA should cover the full range of health issues potentially affecting these different groups: accident and injury, communicable diseases, non-communicable diseases, malnutrition, and psycho-social disorders. In the case of recreational water projects, it is likely that the screening process will result in a health focus on two major groups:

- accidents and injury
  - — hazards and risks to construction workers related to increased traffic and transportation once the project becomes operational;
  - — accidents and injury due to tourists' engagement in high-risk activities (whitewater rafting or scuba diving, for example) or their increased exposure to natural risks (shark attack, snake bite, jellyfish sting).
- communicable diseases
  - — mainly waterborne (and foodborne) diseases associated with a deterioration of water quality;
  - — water-related vector-borne diseases, because of ecosystem changes resulting in the increased breeding of mosquitoes and other insect vectors. Such changes include hydrological changes, biodiversity loss and an increased air humidity.

    (There may also be an impact on respiratory infections (non water pollution related), sexually transmitted infections and HIV/AIDS. There are, however, no design, engineering or water management measures that can help prevent these.)

There may be other health issues associated with recreational water development such as, increased risks of excessive UV exposure due to sunbathing, psycho-social effects especially among indigenous communities in the project vicinity or malnutrition among groups primarily depending on fisheries that are adversely affected by the project. There is, however, little that improved environmental management can do about such problems, with the exception, perhaps, of the last example.

Screening and scoping should lead to a decision concerning the need for a full HIA. A number of countries have legislation in place that contains criteria with minimum values only, about which an impact assessment is required. Some authorities have also advocated sentinel health monitoring to identify health impacts of development projects.

HIA results in a package of recommended measures to safeguard health or mitigate health risks, as well as health promotional activities. In recreational water projects, the resulting environmental management plan will aim to tackle the risks resulting from changes in the environmental determinants of health; moreover, regulatory measures, including financial instruments such as taxes or subsidies, will deal with risks resulting from social change.

Environmental management may involve permanent and capital-intensive measures often of an infrastructural nature. In the context of recreational water projects, this may include:

- wastewater treatment plants;
- systems of dykes, sluice gates, pumps, weirs and other hydraulic structures to optimize hydrological features;
- construction of pipelines to drain or desalinate coastal lagoons where mosquitoes breed; and
- protection of water storage ponds and tanks.

Environmental manipulation aimed at eliminating health risks is a recurrent action. Cleaning aquatic weeds that may harbour vector insects or snails from water bodies is an example.

Some remedial measures may themselves need an assessment for their possible environmental and health impacts. This is particularly true for the chemical control of insect vectors of disease using residual insecticides. Indoor residual spraying poses risks to the members of the spray team. Larviciding introduces the insecticides into the environment at large, where it may disrupt ecosystems and enter the food chain.

Once an action plan based on the HIA recommendations has been initiated, monitoring is a critical component. It should ensure compliance with the agreed design, construction and management changes. It should also follow the health status of the various groups to identify any unexpected health issues arising.

13.5 Public awareness and information

Awareness raising and enhancing the capacity for informed personal choice are increasingly seen as important factors in ensuring the safe use of recreational water environments and an important management intervention. They act both directly (i.e., users are less likely to choose an area that is known to be less safe or to practise unsafe behaviours, so that overall exposure of the population, and hence adverse health outcomes, will be reduced) and indirectly (the exercise of preference for safer environments may induce competition between resorts/destinations based upon relative safety and encourage investment in improvements). In order that these contribute to improved safety, it is essential that the public is generally aware and that information is available, comprehensible, delivered in a timely manner and standardized to enable comparison between alternative locations.

The general public has to rely on information about safety, hazards to health and well-being and facilities as it is able to gain from the news media, local authority notice boards, environmental groups and tourist publicity, as well as its own perceptions. Local NGOs, the tourism industry and local authorities contribute to the distribution of information brochures, the training of consumers in safe conduct and practice, the posting of warning notices, the zoning of dangerous areas and provision of lifeguards. In so doing, they need to translate data gathered by scientists and technicians into understandable and user-friendly messages. The media is also a powerful tool in awareness raising and information dissemination.

Awareness raising is of particular importance among certain specialist user groups and should concern both the hazards that they may reasonably encounter together with the hazards that they may present to other users. With the increasing use of recreational water areas by multiple user types (e.g., beaches used for swimming, jetskiing and sailboarding), this is of particular importance. Clubs and other user group associations have a special role to play in this regard.

Participation in leisure activities is essentially a voluntary activity. Committed participants may choose to belong to clubs and, in turn, clubs may be affiliated to regional and national organizations, which promote development of the sport at the highest national and international levels and issue rules and codes of practice to clubs and the wider membership. Clubs may own facilities and stretches of water. In general, the level of organization shown in Table 13.2 will ensure that club members enjoy the advantages of well maintained facilities, training in proficiency and personal safety and knowledge and awareness of hazards. The degree of development of this structure is dependent upon economic factors and the degree of commitment of participants to the development of their sport.

Participants can control risks actively by acting on knowledge provided to them in the form of guidance, codes of good practice, rules, training and information on the existence of local hazards (such as poor water quality, strong tidal currents, the existence of wrecks underwater and so on).

TABLE 13.2. PUBLIC AWARENESS INFORMATION: ORGANIZATIONAL LEVELS AND RESPONSIBILITIES

Participant	Expert advice	Regulator
National sports organizations Issue codes of practice and newsletters for membership, regulate competitive sport, promote training. International liaison.	**Public health body** Provide public health information. Liaise with user groups and media to disseminate appropriate health messages.	**Central government** Legislate standards, publish results of national monitoring, conduct national health surveillance, involved in finance of capital improvements.
Affiliated clubs Informing members of codes of practice, setting rules of conduct for members, supervising organized events, promoting high standards of performance, providing training.	**Professional institutions, experts** Current awareness of health and safety issues, legislation, research. Liaison with, and expert representation on, government committees and national sports organizations.	**Local authorities and government agencies** Monitoring, reporting results to central government, displaying results to public. Giving information on health. Enforcing public health measures, closing facilities if conditions are hazardous to health.
Club members Responsible to club for conduct and act on club's advice, in addition to making their own value judgements.	**National and international lifesaving federations** Lobby group. Dissemination of safety information.	**Providers of facilities** May be local authorities (public facilities) owners or service providers, including clubs with their own facilities. Adopting and implementing local codes of operational practice, providing safety facilities, preparing a recreational water safety plan, carrying out improvements. Publicizing facilities and results of monitoring.
General public Make own value judgements from personal awareness and knowledge.		

Increased public awareness regarding recreational water use and health is likely to lead to a number of direct benefits where the principal factor leading to accident or disease is individual error of judgement. This may be the case regarding a number of accident hazards, including, for example, diving into shallow water or overestimating swimming abilities. Increased awareness may also lead to greater availability of rescue and lifesaving skills among the general and water user population. The objective of awareness-raising activities is not only to raise the individual's ability to correctly appraise the risk but also to raise the level of confidence of the public that the issue is being addressed and monitoring measures are being undertaken.

Personal perceptions of pollution are most influenced by sight and odour, while physical danger is often based on a visual assessment. Choice of venue is strongly influenced by the availability of appropriate water conditions and areas most suitable for the activity (Cutter et al., 1979). The general public is therefore largely reliant on effective risk management.

One important tool used by associations and governments to enhance the public's capacity for informed personal choice is beach grading or award schemes. For example, since 1987, the Foundation for Education and Environment in Europe has attributed a quality label (in the form of the "Blue Flag") to European beaches and also to marinas. The Blue Flag award takes into account water quality, as well as restrictions on dogs, toilet facilities and so on. It encourages coastal municipalities to improve the public awareness of both visitors and residents.

Government authorities are also developing effective incentive systems. The tourism industry is increasingly conscious of the need to promote safety and environmental concerns and now sponsors "green quality labels". In addition, users and sports participants may develop schemes, such as that used to assess the conditions of surfing areas initiated by a surfers' association.

Although such schemes can improve public awareness and act to inform public choice, a lack of coherence and compatibility among award schemes may undermine their effectiveness and credibility. Issues related to such schemes are discussed in greater detail in Box 13.2.

13.6 Public health advice and intervention (including prevention and rescue services)

Public health advice is a key input to public awareness and informed personal choice, be it with regard to avoiding excessive UV exposure (chapter 3), being aware of what precautions to take against leptospirosis (chapter 5) or malaria (chapter 11) or knowing that an area is unsafe for swimming (chapter 2).

Public health advice and intervention includes response to short-term incidents and breaches of standards. When a guideline or standard is exceeded, the authority responsible for public health should determine if immediate action is required to reduce exposure to the hazard and whether measures should be put in place to prevent or reduce exposures under similar conditions in the future.

BOX 13.2 GRADING AND AWARD SCHEMES

A number of international and national award/grading schemes for water use areas (most commonly beaches) that include safety-related information have been developed. International examples include the Blue Flag (which is the most popular in Europe) and Coastwatch programmes. In addition, many countries also have one or more national equivalents. In the United Kingdom, for example, there are a number of other rating schemes in use, including the Seaside Awards, Good Beach Guide and Beachwatch. These schemes are used at a variety of recreational-water environments, ranging from large-scale resorts to undeveloped rural beaches. Award schemes can have a large influence on tourism (e.g., the beach award schemes in the USA) (Leatherman, 1997) and, as a result, are generally seen as desirable by local authorities and agencies responsible for tourism.

These schemes were designed to inform the public about a recreational area's quality so that users and potential users can make an informed choice regarding the area. Nevertheless, it appears that confusion exists about the implications associated with these schemes (Williams & Morgan, 1995). They are used to:

- give consumers information about water quality so that they can make informed choices about holiday destinations and assess risks when bathing in coastal waters;
- advise businesses that operate nearby and that want to reduce the risks caused by adverse publicity about poor water quality; and
- help resort managers and local authorities that wish to ensure that there are common standards and a common system for measuring those standards (Nelson et al., 1999).

In some of these programmes, however, human health concerns comprise only a small component, or it is possible for areas that present a significant public health risk to receive a high grading if other facilities are good or extensive. Such approaches are likely to undermine the contribution of informed personal choice to the promotion of user safety. In general, health-related aspects in such schemes should assume a dominant character in classification if there is any likelihood that users will interpret them as indicating safety.

A specific problem that is commonly encountered in the development of award schemes is that information may not be comparable between locations. For example, it may be difficult to generate comparable information on microbial water quality because of problems with interlaboratory comparability; where such information is locally generated, it may be difficult to ensure the impartiality of laboratories and surveyors. At an international scale, differing legislation, practice and interpretation between countries compound such problems.

The success of award schemes in terms of informing the public depends upon active information dissemination as well as the required technical interventions. While comparison of different locations constitutes an important part of the information required for improved personal choice, active information dissemination at a local level and related to short-term changes is also necessary. For example, in some recreational water use areas, changes in local conditions may be extreme or rapid, such that areas are unsafe for physical or quality reasons. Such areas require "posting" and also information dissemination where a beach is unsafe at certain times—for instance, because of weather conditions or because of local water quality changes. Ideally, the requirement for such information dissemination should constitute an important part of award schemes.

Available evidence suggests that many hazards associated with the recreational use of the water environment are of an instantaneous or short-term nature. Drowning has been associated with offshore winds carrying inflated toys and buoyancy aids away from the coast. In the case of water quality, certain beaches or areas are known to register increased pollution under certain conditions, relating to tide, wind direction or rainfall, for example. In eutrophic fresh water, wind may be associated with the accumulation of cyanobacterial "scums" in some areas, which may present a special hazard to children who are tempted to play in the scum material. Whenever such conditions occur and constitute a risk to public health, short-term advisory notices may be considered necessary, and the decision to place such notices should be based upon public health considerations. This approach may, through low-cost measures, enable safe use of areas that might otherwise be considered inappropriate for recreational use. Examples of conditions that may result in a severe health outcome and thus merit a public health advisory and levels at which they may be implemented are summarised in Table 13.3. Specific conditions require definition on a location by location basis depending upon local circumstances and the user groups and activities typically undertaken (a white water canoeist, for example, may be actively seeking river flood conditions).

TABLE 13.3. CONDITIONS THAT MAY MERIT INTERVENTION BY SAFETY OR PUBLIC HEALTH AUTHORITIES

Hazard	Examples of conditions meriting immediate action
Drowning	High surf conditions Development of a strong rip current Dam release of water on an impounded river
Microbial	Presence of human sewage (e.g., due to a pipeline breakage) 95% percentile value of intestinal enterococci/100ml greater than 500 (or greater than 200 if source mainly human faecal pollution) in consecutive samples. Presence of a large outbreak of faecal-oral illness in the local community (especially if the agent is resistant to sewage treatment processes and has a small infectious dose)
Algal and cyanobacterial	Presence of scums or detection of 100,000 cells/ml
Chemical	Chemical spill or significant contamination
Dangerous aquatic organisms	Presence of organisms associated with human fatalities such as sharks, hippopotami, crocodiles, alligators or box jellyfish (for example) close to the recreational area.

Prevention and rescue services can also be considered to fall within this intervention. Provision of lifeguards (see Appendix A) is a highly visible measure that may contribute to safety in various ways: by directly assisting in prevention of drowning (rescue, resuscitation), by assisting in injury prevention (e.g., advising users not to enter dangerous areas) and by playing a more general educational role (concerning water quality hazards and exposure to heat, cold or sunlight, for example).

13.7 Operating within an integrated coastal area management framework

One way in which all the relevant stakeholders can be brought together is through the establishment of an integrated management system for marine and freshwater recreational areas based on the concept of integrated coastal area management (ICAM), as outlined in chapter 1 and Box 13.3.

BOX 13.3 INTEGRATED COASTAL AREA MANAGEMENT

An integrated coastal area management (ICAM) framework is a "continuous and dynamic process that unites government and community, science and management, sectoral and public interests in preparing and implementing plans for the protection and development of coastal systems and resources" (GESAMP, 1996).

The main premises of ICAM are:

- Natural resources are finite, and their use must be allocated prudently.
- The functional integrity of the resource systems must be protected.
- Resource management involves changing human perceptions and behaviour.
- Resolution of multiple-use conflicts needs a holistic approach through policy, management and technical innovations.
- Planning and management processes are dynamic and should respond to ecological and socioeconomic conditions and evolve with time.

Ideally, ICAM seeks to address all activities and resources within a defined area. Thus, the need for and requirements of such economic and social activities, including fisheries, non-renewable resource extraction, waste disposal, agriculture and aquaculture, tourism, recreation, transportation and development, should be considered.

ICAM involves comprehensive assessment, the setting of objectives and the planning and management of coastal systems and resources. It also takes into account traditional, cultural and historical perspectives and conflicting interests and uses. The individual elements and some of the main linkages of the ICAM framework are shown in Figure 13.1.

ICAM is an iterative and evolving process for achieving sustainable development (UNCED, 1992) and continuous management capability that can respond to changing conditions. As such, the framework permits integration of the various needs and requirements for the coastal area and coordination of the actions, whether preventive or remedial. Integration relates to both vertical (levels of government and NGOs) and horizontal (cross-sectoral) coordination among stakeholders whose actions influence the quality/quantity of water-based resources reflected in the planning and management strategies.

Continued

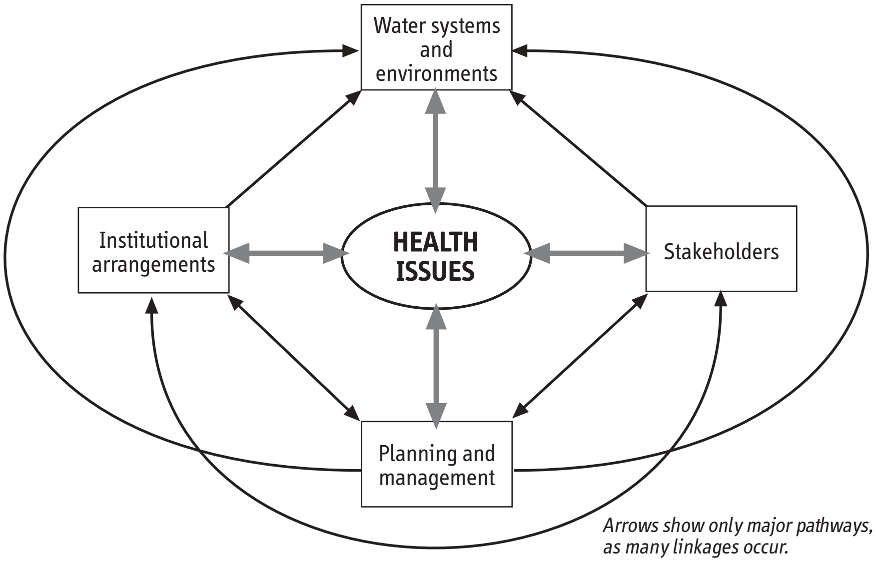

FIGURE 13.1. A SCHEMATIC VIEW OF THE INTEGRATED MANAGEMENT OF RECREATIONAL WATERS

An ICAM programme can be directed to one or more types of coastal areas, which can extend from coastal mountain watersheds to offshore coastal boundaries, and can also encompass river catchment areas.

Management options may vary, for example, from educational projects to construction work or from no-cost actions to heavily funded development. The exact package of management options to reduce or eliminate health hazards and risks related to recreational water uses will be driven by the nature and severity of the health impacts.

Based on assessment of risk, three levels of response may be considered:

- The basic response should guarantee that an ICAM management framework is established to prevent the occurrence of significant adverse health outcomes and facilitate the implementation of remedial actions. This could include the dissemination of minimum public awareness messages, the establishment of an integrated recreational water committee with participation of various stakeholders and the development of a streamlined monitoring programme.
- The expanded response would provide an enhanced institutional setting with more sophisticated legislation and increased participation of stakeholders in the development and implementation of solutions, targeted intervention to areas prone to health hazards, rapid response when problems are identified, and a greater public awareness activity together with the mobilization of local NGOs to support the effort.

- The full response would ensure a comprehensive package of management options with a clear strategic plan for implementation of the various interventions and establishment of an integrated coastal area/recreational waters management system, which would, in turn, develop appropriate tools (legislation, incentives, economic instruments, participation, etc.).

These three levels of response correspond to the assessed level of health risk in a recreational water area and should be complemented by the corresponding levels of monitoring outlined in chapter 12. Levels of response apply both at progressive national implementation and that appropriate to specific local circumstances.

A basic response may suffice in an area that is rarely frequented, with little or no record of health effects due to recreational activities and with no development plans to alter the nature and use of the recreational water zone in the medium term. The response should ensure that a potential danger situation can be dealt with effectively and immediately. The expanded and full levels of response would need to be adapted to local conditions, taking into consideration past occurrences and likely trends. Preventive actions are effective in areas with good general awareness levels, which have available resources and no imminent health danger and threats. Remedial actions would be required to minimize existing negative health effects. Usually a combination of the two would be selected, with respect to local conditions, availability of resources and valuation of the danger and impacts. The selection of level of response is clearly also linked to the availability of funding, technical support and advice.

Four major management interventions were identified in chapter 1. These comprise compliance and enforcement, control and abatement technology, public awareness and information (this includes support for informed choice, such as clear recreational water grading/award schemes) and public health advice and intervention (including prevention and rescue services). Some activity in each of these areas is possible and advisable at all three levels of response in terms of national implementation.

13.8 References

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APPENDIX A
Lifeguards

This annex draws upon the extensive experience of the International Life Saving Federation (ILS)[1] and comments received during the preparation of these Guidelines. It relates to people who are trained and positioned at recreational water sites to protect water users and who may be paid or voluntary. They may be referred to as lifesavers, lifeguards or given some other title. For simplicity, the term lifeguard has been used throughout this annex. The following sections outline points for consideration when setting up or running a lifeguarding scheme.

A.1 Lifeguard qualifications

Lifeguards are generally responsible for observation of a beach or recreational water area to anticipate problems and identify an emergency quickly, carry out rescues, give immediate first aid, communicate with swimmers and recreational water users, enforce regulations where appropriate, promote awareness of specific and general hazards and report incidents.

Lifeguards should have appropriate training and hold a suitable current qualification. This would normally be from an appropriate and recognized training and assessment agent. Lifeguards should, for example, be competent in lifesaving methods, swimming and the most current methods of resuscitation. Requalification should be undertaken at regular intervals, and practical rescue and resuscitation skills should be practised frequently. Both fitness and technical knowledge are required. Good practice would generally require that records be kept of all training and qualifications and be available for inspection.

Lifeguards should have locally-specific knowledge concerning the presence of natural and artificial features, the topography of the area, tides and currents, hazards posed by local animals, the distance to qualified medical assistance, hazards and risks, public relations, crowd management and local operating procedures.

Minimum standards for the training of lifeguards have been proposed (http://www.ilsf.org).

A.2 Lifeguard observation points

Lifeguard observation points must have a clear and unobstructed view of the area of supervision, including both the water and the beach. Lifeguard observation points

[1] The ILS is a non-profit confederation of major national lifesaving federations worldwide.

are ideally elevated (the higher the better, within reason) and provide the lifeguard with protection from the elements. These are often referred to as lifeguard "towers". They should provide adequate space to allow the lifeguard to stand and move while observing the water and a place for necessary rescue and first aid equipment. The design of a lifeguard observation point should include a way to respond on foot to a rescue without breaking observation of a swimmer in distress.

Lifeguard observation points should be placed to allow observation of the area under control. At coastal recreation areas, they should be placed as close to the water edge as practical at high tide and may be moved at intervals with the changing tide, so that they will be close to the water edge at all tidal stages. Where a beach is divided by a jetty or other obstacle to clear observation, each part should ideally be independently observed.

A.3 Lifeguarding equipment

Lifeguards on duty should be easily identifiable at a distance, in a manner that sets them apart from others at the beach, such as by a uniform. To properly perform their duties, lifeguards require appropriate rescue equipment. The most basic rescue device is a rescue float. The most common of these are tubes of flexible closed-cell foam rubber and buoys of hard plastic. Other examples of basic lifesaving equipment are the rescue board (a surfboard adapted for rescue), binoculars and swim fins. Lifeguards are frequently involved in first aid and need to be appropriately equipped for this work. Lifeguards are often be provided with a telephone or radio for communication. As record keeping is necessary, report forms should normally be provided.

More advanced rescue equipment can be useful. Rescue craft have proven effective in offshore rescue of swimmers, boaters and others. While costly, they receive a high degree of public support. They are most frequently deployed in areas of dense use or particular hazard. For effective use of rescue craft, good communication linked to rapid deployment is important. In some cases, the provision of a motor vehicle may also be appropriate.

All equipment should be inspected frequently and replaced or repaired as necessary.

A.4 Lifeguarding policies

Lifeguard organizations should develop written "standard operating procedures". These would contribute to the water safety plan (section 13.2) and should contain details on risk assessment, a plan of the recreational water (outlining hazards, access points, vantage points and blind spots, information points, zones, positioning of public rescue equipment and protective features), supervision requirements (e.g., lifeguard provision, rotation systems, qualification, surveillance levels and daily routines) and the duties of other recreational water staff.

An "emergency action plan" should be formulated to guide lifeguards in handling emergencies that can be reasonably anticipated. It should provide step-by-step procedures for each member of the team: rescue management, continuity of supervision during rescue, communication procedures during an incident (both within the team and with external agencies), aftercare and peer support.

Lifeguard levels of performance should be established and incorporated within the policies.

A.5 Lifeguard duty period

Lifeguard supervision should be maintained during times of significant use. Sufficient regular breaks should be incorporated into duty periods. When on duty, lifeguards should not perform other tasks that might detract from observation.

Warning signs should be posted if lifeguard service is interrupted, and the beginning and end of this period should be communicated with, for example, megaphones and signs.

A.6 Lifeguard staffing levels

Lifeguard staffing levels should be appropriate to the use of the area of responsibility and provide for public safety in a manner consistent to ensure safety. Responsibility should not be left with a single individual. Lifeguards work more effectively in teams. These teams should ideally be managed through a central administration capable of providing necessary relief, backup and resources.

Two primary factors influence the staffing level needs for lifeguards: attendance and risk. Attendance typically varies according to season, day of the week, weather and other factors. Risk can vary according to surf, rip current intensity (which is usually directly related to surf), wind (which may enhance surf size), water temperature and other factors. The number of lifeguards staffing a beach area should be adequate, regardless of fluctuations in attendance and risks. The provider of lifeguard protection must therefore either adopt a system to effectively vary staffing according to fluctuations or set a consistent staffing level aimed at the highest levels of attendance and risk. Most lifeguard providers address this via a mix of the two. That is, they set regular staffing levels somewhat below the level needed to address the highest levels of risk and attendance, but somewhat above the average levels. Then, they develop a system to enhance staffing levels when unexpected crowds and/or risks present themselves. Varying staffing levels by day of the week is also common in areas where attendance fluctuates predictably.

People in distress in the water rarely wave or call for help, being panicked and occupied in trying to keep themselves afloat, and even nearby swimmers are often unaware of the problem. Thus, lifeguard vigilance is of key importance. It is a tremendous challenge to maintain concentration in the face of the monotony of watching swimmers for extended periods of time. Training may help, but does not eliminate normal human reactions to boredom. Regular breaks are therefore important and may also be necessitated by the environment in which lifeguards operate, which may be hot and/or windy. Lifeguards must consume generous quantities of fluid to prevent dehydration and assist with concentration. Breaks allow for simple human needs along with relief from prolonged periods of scanning and physical inactivity.

A.7 References

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The World Health Organization's (WHO) new *Guidelines for Safe Recreational Water Environments* describes the present state of knowledge regarding the impact of recreational use of coastal and freshwater environments upon the health of users – specifically drowning and injury, exposure to cold, heat and sunlight, water quality (especially exposure to water contaminated by sewage, but also exposure to free-living pathogenic microorganisms in recreational water), contamination of beach sand, exposure to algae and their products, exposure to chemical and physical agents, and dangerous aquatic organisms. As well, control and monitoring of the hazards associated with these environments are discussed.

The primary aim of the Guidelines is the protection of public health. The Guidelines are intended to be used as the basis for the development of international and national approaches (including standards and regulations) to controlling the health risks from hazards that may be encountered in recreational water environments, as well as providing a framework for local decision-making. The Guidelines may also be used as reference material for industries and operators preparing development projects in recreational water areas, as a checklist for understanding and assessing potential health impacts of recreational projects, and in the conduct of environmental impact and environmental health impact assessments in particular.

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Monitoring Bathing Waters - A Practical Guide to the Design and Implementation of Assessments and Monitoring Programmes

Edited by Jamie Bartram and Gareth Rees

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Foreword

Coastal waters, rivers and lakes are used for a variety of recreational activities, including swimming, diving, fishing and sailing. If these activities are to be enjoyed safely, attention must be given to health hazards, as well as to the prevention of accidents.

Between 1993 and 1998, *Guidelines for Safe Recreational Water Environments* were developed by the World Health Organization (WHO) Headquarters in collaboration with the WHO European Centre for Environment and Health, Rome, Italy. These guidelines were released in the form of a draft for consultation in two volumes, *Coastal and Freshwaters* and *Swimming Pools, Spas and Similar Recreational Water Environments.* They comprise an assessment of the health risks associated with recreational use of water and outline linkages to monitoring and assessment and management practices. They are intended to provide guidance in identifying, characterising and minimising the risks to human health associated with recreational use of water and to promote the adoption of a risk-benefit approach to the management of such risks. The development of such an approach involves issues such as environmental pollution, conservation, and local and national economic development and may lead to the adoption of standards that can be implemented and enforced. To implement such an approach successfully requires considerable intersectoral co-operation and co-ordination at national and local levels as well as a coherent policy and legislative framework.

This book is a practical guide to the monitoring and assessment of freshwater and marine water used for recreation and builds upon the health risk assessment described in *Guidelines for Safe Recreational Water Environments.* It provides comprehensive guidance for the design, planning and implementation of assessments and monitoring programmes for water used for recreation. It addresses the wide range of hazards that may be encountered and emphasises the importance of linking monitoring programmes to effective and feasible management actions to protect human health. It also defines elements of good practice that together constitute the Code of Good Practice for the Monitoring and Assessment of Recreational Waters.

This book will be an invaluable source of information for anyone concerned with monitoring and assessing water used for recreation, including field staff. It will also be useful for national and regional government departments concerned with tourism and recreation, undergraduate and postgraduate students and special interest groups.

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This book is based on the Code of Good Practice for the Monitoring and Assessment of Recreational Waters, which was prepared in co-operation with the European Commission. The Code was developed through a review process in which comments were received from 55 persons in 28 countries and reviewed at a meeting of an international group of experts.

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Chapter 1*: INTRODUCTION

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* *This chapter was prepared by G. Rees, J. Bartram, K. Pond and S. Goyet*

From 1993 to 1998 the World Health Organization (WHO) worked on the progressive development of its *Guidelines for Safe Recreational-Water Environments* (WHO, 1998). The Guidelines comprise a health risk assessment of recreational water use to be published in two volumes *(Volume 1: Coastal and Fresh-Waters* and *Volume 2: Swimming Pools, Spas and Similar Recreational-Water Environments).* This present book is designed to complement the *Guidelines for Safe Recreational-Water Environments,* providing a practitioners' guide to the monitoring and assessment of coastal and freshwater recreational environments. It presents, in a methodological format, the information necessary to design and implement a monitoring and assessment programme for recreational water environments.

Surface and coastal waters are used for a variety of leisure and recreational activities, and for other purposes including transport, food production, hydroelectricity generation, as a transport medium and as a repository for sewage and industrial waste. Such activities are not always compatible with one another. Water and its recreational use have long been recognised as major influences on health and well being. The health benefits of bathing in saltwater were, and still are, promoted with enthusiasm. Sea water was once considered as an alternative medicinal treatment to spa water. Water-based recreation is an important component of leisure activities and tourism throughout the world. Tourists are responsible for the significant movement of economic resources both within and between countries. This may be typified by the annual influx of tourists from northern European countries to the countries surrounding the Mediterranean. A similar effect may occur within some countries where certain regions are favoured holiday destinations by those from other regions within the country.

Recreational use of the water environment may offer a significant financial benefit to the associated communities but it also has implications for health and for the environment. Visitors exert a variety of pressures on the very environment that attracts them. Water-based recreation and tourism can also expose individuals to a variety of health hazards, ranging from exposure to potentially contaminated foodstuffs and potable water supplies, through to exposure to sunshine and ultra violet (UV) light and to bathing in polluted waters. Water, however clean, is an alien environment to humans and thus it can pose hazards to human health even when it is of pristine quality.

The varied nature of the hazards to human health and well-being posed by recreational waters demands a full audit of the relative importance of the resultant health effects and the resources required to mitigate those effects. Undoubtedly, the public health outcomes of accidents (including drowning and trauma associated predominantly with diving incidents) and potential infections acquired from contaminated waters are those that demand most attention world-wide.

All the trends indicate that leisure activities, including water-based recreation, will continue to increase. Thus the effects of the health hazards that face recreational water users are likely to gain more prominence in the future. Those responsible for monitoring the likely health impacts of recreational water use are going to face increasingly complex challenges as recreational uses diversify and the number of users increases.

1.1 Health hazards in recreational water environments

Amongst the unequivocal adverse health outcomes resulting from recreational water exposure are drowning and near-drowning. Such injuries account for a significant annual death toll, often associated with reckless behaviour and/or alcohol consumption. Unsafe diving into water bodies can lead to a range of traumatic injuries, including spinal injury, which ultimately may result in quadriplegia. More common, but less severe, incidents include those arising from discarded materials, such as glass, cans and needles on beaches and on the bottom of the bathing zone. Of particular concern is the presence of medical waste, particularly hypodermic needles. Chapter 7 addresses the dangers due to accidents and injuries including those associated with drowning and spinal injury.

The pathogenic micro-organisms that can be found in water bodies have a wide range of sources. These include sewage pollution, organisms naturally found in the water environment, agriculture and animal husbandry and the recreational users themselves. Sewage of domestic origin comprises a particularly unhealthy mixture of micro-organisms. The microbiological hazards encountered in water-based recreation include viral, bacterial and protozoan pathogens. Primary concern has usually been directed towards gastro-intestinal illnesses acquired from recreational waters, although acute febrile respiratory illness and infections of the eye, ear, nose and throat have all been identified as acquired through bathing. The link between recreational water use and more serious infections such as meningitis, hepatitis A, typhoid fever and poliomyelitis is difficult to determine unequivocally.

A key environmental effect of sewage discharges is nutrient enrichment largely, but not exclusively, attributable to phosphate and nitrate in the sewage. This nutrient enrichment can lead to localised eutrophication, which in turn is associated with more frequent or severe algal blooms. Prolonged and excessive eutrophication has also been responsible for algal blooms on a regional basis, such as those in the Adriatic and Baltic Seas in recent years.

A review of human health effects arising from exposure to toxic cyanobacteria, as well as discussion of the detailed analysis of toxic cyanobacteria in water and of their monitoring and management, is available in a companion volume in this series, *Toxic Cyanobacteria in Water* (Chorus and Bartram, 1999). Chapter 10 in this book deals with the monitoring and assessment of toxic cyanobacteria and algae in recreational waters and also provides a framework for assessing under what circumstances such organisms may pose a priority hazard.

A number of other health hazards may be encountered during recreational water use but which are typically local or regional in distribution. These include: chemical contaminants, arising principally from direct waste or wastewater discharge; non-venomous disease-transmitting organisms (e.g. mosquitoes as malaria and arboviral disease vectors and freshwater snails as intermediate hosts of the schistosomes that cause bilharzia or schistosomiasis); hazardous animals encountered near water (such as crocodiles and seals) and venomous invertebrates (such as sponges, corals, jellyfish, bristleworms, sea urchins and sea stars); and venomous vertebrates (catfish, stingrays, scorpionfish, weaverfish, etc.). The health risks associated with these hazards are outlined in the *WHO Guidelines for Safe Recreational-Water Environments* (WHO, 1998). Approaches to monitoring and management of these health hazards are often strongly influenced by local factors and for this reason the issue is dealt with here in generic terms, allowing the reader to make informed responses to circumstances where such hazards may arise (Chapter 11).

Excessive exposure to UV, although not exclusive to water-based recreation, may pose a significant health risk if recreational water users do not take appropriate care. Acute effects, such as the discomfort and injury associated with sunburn, or delayed effects (which may include malignant melanoma) are direct adverse health outcomes. Cold water is an important contributory factor in many cases of drowning, and excessive exposure to heat and/or cold can also be associated with adverse health outcomes. These issues are fully addressed in the *WHO Guidelines for Safe Recreational-Water Environments* (WHO, 1998). Hazards attributable to physical components, such as exposure to extremes of temperature or to excess UV radiation, are not included in this book due to the limited contribution that monitoring and assessment can make to risk management in this context.

1.2 Factors affecting recreational water quality

The health risks posed by poor quality recreational waters generally relate to infections acquired whilst bathing. A range of pollutants enters recreational waters from a number of sources - coastal waters can be regarded as the ultimate sink for the by-products of human activities. In terms of the quality of coastal recreational waters and the resultant impact on human health, the key sources of pollutants are riverine inputs of domestic, agricultural and industrial effluents and direct sewage discharges from the local population. Apart from regular discharges through short and long sea outfalls, irregular discharges may occur through storm water and overflow outfalls, and through unregulated private discharges. Freshwater bathing sites are subject to the same polluting sources as coastal sites, although the scale and extent of these sources may be easier to predict in more clearly delimited freshwater sites.

In both coastal and freshwaters the point sources of pollution that cause most health concern are those due to domestic sewage discharges. Diffuse outputs and catchment aggregates of such pollution sources are more difficult to predict. Discharge of sewage to coastal and riverine waters exerts a variable polluting effect that is dependent on the quantity and composition of the effluent and on the capacity of the receiving waters to accept that effluent. Thus enclosed, low volume, slowly-flushed water systems will be affected by sewage discharges more readily than will water bodies that are subject to rapid change and recharge.

Water-based recreation and leisure activities contribute relatively little to pollution inputs and associated adverse health outcomes when compared with other sources of aquatic pollution, such as sewage outfalls. Pollution originating from water-based recreation and leisure craft includes sanitation discharges, fuel spillages, the environmentally toxic effects of antifouling compounds and general debris. Because water-based recreational activities often occur in estuaries or embayments, any polluting effects may be exaggerated due to the enclosed nature of the system and the subsequent accumulation of pollutants. This is particularly evident in the large number of boating marinas that have been developed over recent years, where there is often a high density of craft and the associated crew, adjacent to bathing waters. Appropriate controls on sanitation discharges from pleasure craft are dependent on suitable holding tanks and port reception facilities. The contribution that such vessels make to the total sewage inputs may be small, but may become more significant in the situations described above where vessels aggregate.

Environmental hazards attributable to pleasure craft may also arise from fuel spillages or discharges. Oil, petrol and diesel may be spilt at filling barges and bilge waters may be discharged, adding to pollution of the coastal environment. Two-stroke outboard engines are thought to exert an annual polluting effect several times greater than that attributable to high profile oil-tanker disasters such as that of the Exxon Valdez (Feder and Blanchard, 1998). Apart from the fuels themselves, the emissions from the engines may be harmful and the oil and petrol mix in two-stroke fuels has been implicated in the tainting of fish and shellfish products. This type of pollution, therefore, can pose an indirect threat to health.

The toxic nature of antifouling paints applied to prevent the attachment and subsequent growth of organisms on pleasure craft hulls and on coastal installations defines them as environmental pollutants. They may thus have an effect on water quality, particularly where vessels are concentrated or the area is enclosed. Such effects are usually considered to affect the marine biota rather than human health.

Pleasure craft and their users undoubtedly contribute significantly to the load of marine debris. Plastics, fishing gear, packaging, food and other wastes are discarded overboard, even when there are controls to prevent this. About 70-80 per cent of marine debris comes from land-based sources and the rest comes from vessels and installations. Such materials rarely affect water quality and human health but do have environmental effects and contribute enormously to aesthetic pollution.

1.3 Effective monitoring for management

Particular types of recreational activities may be associated with certain hazards and therefore discrete patterns of action may be taken to reduce the risk of these hazards. For example, untreated sewage discharges will pose one type of risk - that of infection to bathers; glass discarded on a beach will pose a different type of hazard - injury to walkers with bare feet. Effective sewage discharge procedures can address the former and regular cleaning of the beach coupled with provision of litter bins and educational awareness campaigns can reduce the latter hazard. Therefore, each type of recreational activity should be subject to assessment to determine the most effective control measures. This assessment should include factors that may have a moderating effect on the particular type of risk, such as local features, seasonal effects and the competence of the participants in the activity where the risk is encountered.

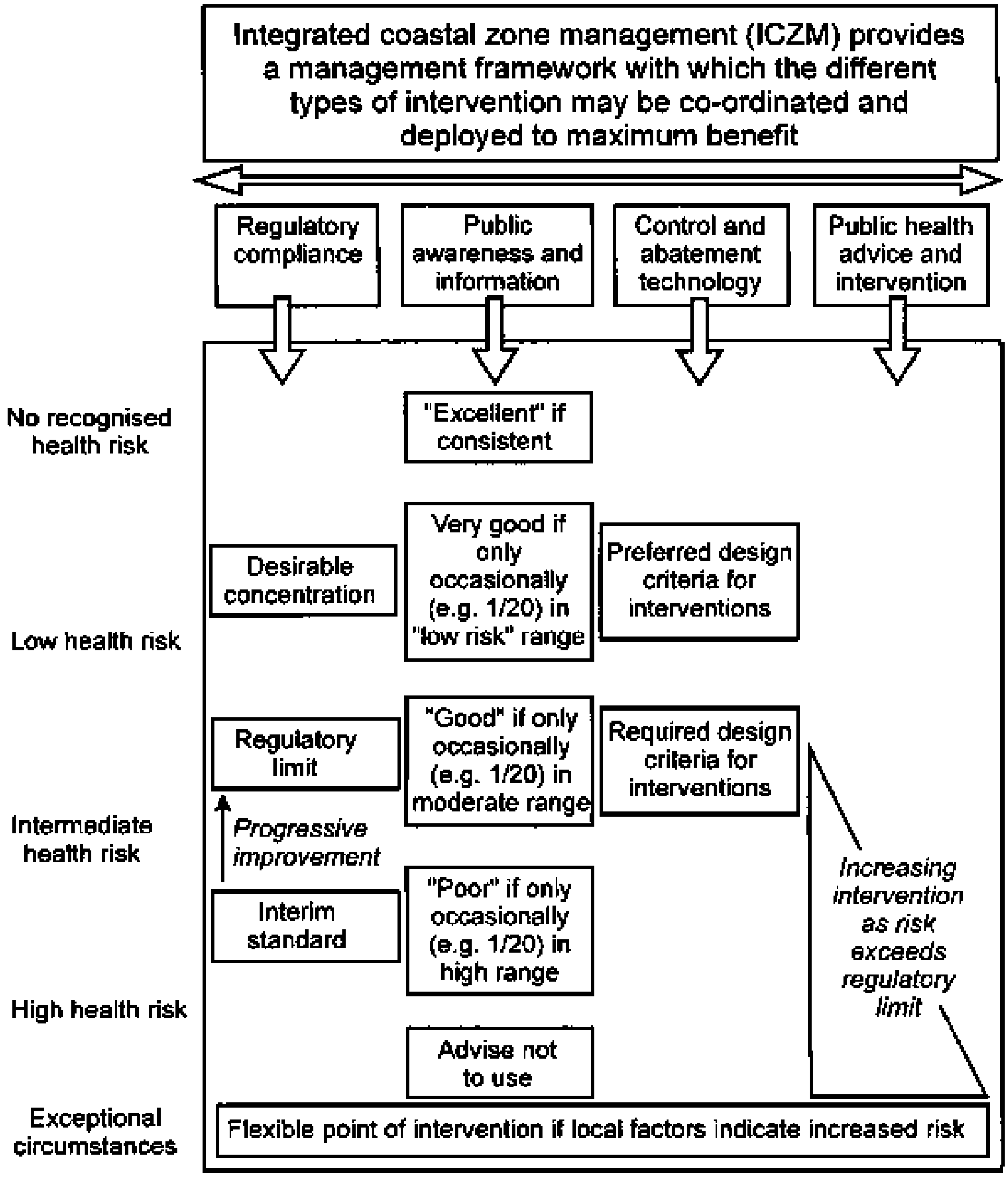
The importance of effective use of information from monitoring must be stressed. There is little point in generating monitoring data unless they are to be used. The eventual use of the information products resulting from monitoring should guide and determine all the stages of the monitoring process from the setting of objectives through to design and implementation, reporting and to co-ordination of follow-up. The principal components of management of recreational water use areas for the protection of public health are described in Chapter 5.

For any monitoring and assessment programme to be effective there must be clear management outcomes from the use of the data produced. Effective monitoring requires collection of adequate quantities of data of the appropriate quality, and an understanding of the link between monitoring and management (and therefore to whom, in what format and when information would be best provided). Subsequent actions may be remedial, may provide public information or may inform planning. For example, microbiological monitoring data can be used to justify improved treatment of coastal sewage discharges or, alternatively, to indicate that a small investment in injury and prevention measures will yield more substantial public health benefit. The information links between regulators, government and industry, and those that can provide the financial support for remedial initiatives, must be based on good quality data.

In order for individuals to be able to make knowledgeable decisions about their ultimate recreational destinations, based on the existing facilities and the environmental quality of the available options, they must be aware and informed. Ideally, such judgements should be based on good quality, readily understandable and easily accessible information. Individual choice of site of recreational activity may indirectly result in improved levels of recreational water and bathing beach management by local and national governments. Furthermore, individuals can take responsibility for some important actions to protect their own health and well-being whilst involved in water-based recreation.

The *WHO Guidelines for Safe Recreational-Water Environments* (WHO, 1998) describe management actions that support improved safety in recreational water use in four broad areas under the umbrella of integrated coastal or basin management (Figure 1.1). All four broad areas rely on the output of sound monitoring and assessment processes for effective implementation. Several activity levels can be defined at international, national, regional and local levels. Actions that may be taken at international and national level consist primarily of the setting of standards, and such actions are the province of government. There are many local actions that can be undertaken and which may have a significant impact on the well-being of recreational water users. These include basic beach management schemes, comprising lifeguard provision, appropriate sanitation facilities, potable water, parking, medical facilities, beach cleaning, emergency communication and zoning of activities to avoid conflict. These initiatives are largely the domain of local municipalities or of the owners of private beaches. Although all these management measures have attributable direct costs, they may have major indirect benefits in the form of increased recreation and leisure at the location. Beach award schemes (see Chapter 6) harness the willingness of municipalities to provide such facilities.

Figure 1.1 Management framework and types of intervention in relation to different types and degrees of hazard associated with recreational water use



Effective monitoring also requires the participation of all authorities, organisations, industries etc. with a vested interest. This implies that an agency involved in monitoring should ensure and maintain effective channels of communication with non-governmental organisations, industry (especially tourism), local and central government, trade associations, resort and tourism operators and elements of the media. Often monitoring and regulatory agencies are concerned with the quality and veracity of data, but are less aware of the need to package the information and display the results in ways that are easily understood by participating partners, i.e. the same partners with which they are trying to maintain links.

Many uses of the water environment have the capacity to conflict with each other. Such competing pressures must be monitored in a coherent fashion to minimise conflict and, where appropriate, risk to health. Conflict may arise between different groups of water users or between local users and those visiting an area. It is one of the primary roles of effective management to accommodate competing, and often conflicting, uses. When tourism and leisure-based activities are major revenue sources in a region, it is important to strike the correct balance between the demands of water-based recreation and the needs of the environment. Different recreational activities can also interact in a counter-productive fashion - angling, boating, surfing, bathing and water-skiing cannot all take place on the same area of water at the same time without some form of regulation. It is essential that effective planning and consultation processes exist to ensure that appropriate management practices are implemented and monitored. Such planning may enable what is generally a limited resource to be channelled into the most appropriate activities at a particular location. Visitor pressure can be managed more effectively by such means and the health and well-being of the individual and the environment is usually best served in this way.

1.4 Good practice in monitoring

The chapters of this book each include an element of good practice in the monitoring and assessment of recreational waters. Together these elements constitute a Code of Good Practice. This Code of Good Practice comprises a series of statements of principle or objectives which, if adhered to, would lead to the design and implementation of a monitoring programme of scientific credibility. The Code applies to the monitoring of all waters used for recreational activities that involve repeated or continuous direct contact by people with the water. In many circumstances there are different approaches or methods that can be applied to achieve the objective stated in the Code. Although each approach is equally valid in isolation, adoption of diverse approaches within a single programme would not lead to the comparability of results that may be required by an inter-location study or by an enforcement programme. Where data are to be compared between laboratories or between sites, all available measures should be implemented to ensure comparability of results. These include:

- A quality assurance programme based on internal and external (inter-laboratory comparisons) controls.

- The development of criteria for dealing with participating laboratories consistently failing to comply with minimum analytical quality. This should be stated prior to data collection and it should be adhered to.

- Procedures for dealing with data and sampling anomalies and omissions, which should be agreed prior to data collection and adhered to. In regulatory monitoring programmes, factors such as sampling frequency, analytical methods, data analysis, interpretation and reporting, sample site selection and criteria for recreational water use areas, are usually defined by a regulatory agency and should take account of the principles outlined in the Code of Good Practice.

1.5 Legislative context

Effective coastal or freshwater zone management requires an effective legislative framework to define the roles of different bodies and levels of government, as well as to provide environmental objectives. Management is not restricted to national issues; water quality, pollution control, international tourism and shipping are amongst the activities that affect the coastal zone and that also extend beyond national boundaries. No single government or agency can be responsible for the wide range of issues that need to be addressed in the coastal-freshwater zone and legislation should be considered at the international, national and local levels. In general, however, the fragmented and often duplicated responsibilities in the coastal zone are severe impediments to effective planning and management in many countries.

The structure and responsibility of local government differs throughout the world. In the UK, for example, County Councils are responsible for the strategic planning, structure plans and waste disposal, while the District Councils are responsible for housing, local planning, local plans, environmental health, coast protection, waste collection and noise control. In Australia, the Local Councils have general responsibilities for the production of coastline management plans, coastline hazard mitigation, hazard awareness and beach management, as well as specific responsibilities under the Environmental Planning and Assessment Act.

1.6 Socio-economic issues

The development of tourism may create conflict. Displacement effects, such as movement of the indigenous population out of coastal areas, banning fishing from tourist beaches, or inappropriate adaptation of cultural and historic resources, are pervasive and may lead to political and social reactions against tourism. These social pressures also tend to reinforce pressures for enclosed resorts and concentrated tourism enclaves that may increase the adverse environmental effects of tourism. Private beaches that charge entry fees provide a means of socio-economic selection. Furthermore, social problems may arise in a range of situations that include but are not limited to:

• *Conflicts between beach users.* Conflicts may develop particularly on intensively used beaches. They may occur between visitors, swimmers, surfers and boat users and may be resolved by delimiting zones of the beach and nearshore sea.

• *Complaints by people using a beach.* Noise from radios, vehicles or boats and other potentially insensitive behaviour may lead to conflict situations.

• *Camping on beaches.* Camping on beaches is tolerated in many countries, especially away from seaside resorts and other urbanised areas. However, this may be a cause of conflict between visitors and local people.

• *Visual intrusion.* Structures such as fishermen's huts and beachcombers' shacks may be considered unsightly.

• *Animals.* Fouling by dogs can be a nuisance and a health risk.

• *Alcohol.* Some authorities prohibit the drinking of alcohol on the beach, mainly because of unpleasant behaviour and the increased litter resulting from empty drinks cans and broken bottles.

Measures for coping with the adverse social and cultural impacts of tourism include studies of a social carrying capacity for tourism, public education programmes and improved security measures to address increased crime and drug problems. Observations of beach behaviour suggest that there is a tendency for beach users to segregate themselves on the basis of race and class (Bird, 1996). The benefits of tourism, including tourism in coastal areas, derive from direct and indirect income, new jobs, foreign investment, infrastructure development, increased local support for environmental amenities, and conversion to less stressful use. The main costs of beach tourism could be ascribed to the effects on water quality and availability, sewage and solid waste disposal, loss of non-renewable resources (e.g. sand mining from beaches is a major negative impact related to tourism development in the Caribbean), overharvesting of renewable resources, increased social tensions, stimulation of imports of foods and other consumables, costs of damage from natural hazards, increased densities of people and conversion to more stressful uses.

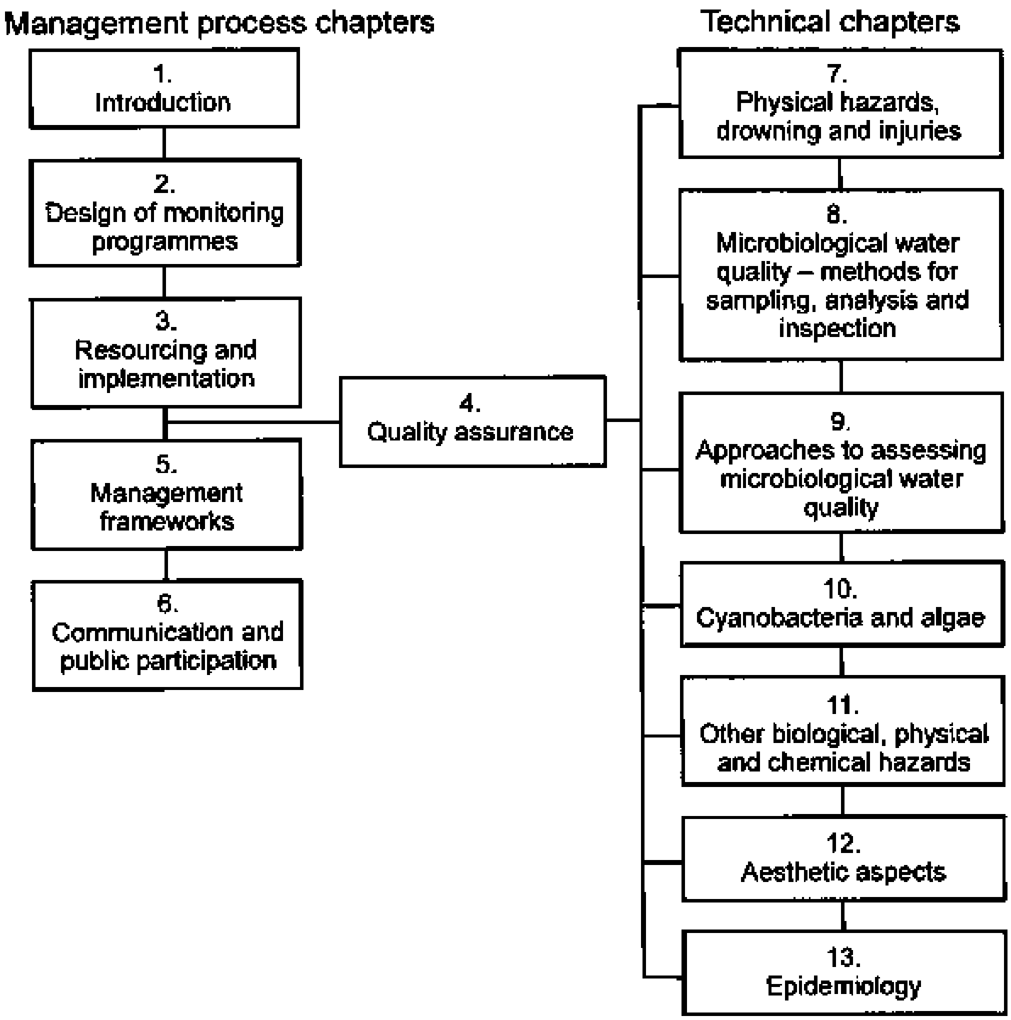
Benefits and costs can be measured in quantitative, financial terms if there have been sufficient econometric studies to determine shadow prices for known differences in environmental and social effects of tourism. National and regional development planners need studies that define the local rate of retained earnings or the local factors enhancing the effects of various types of tourism facilities. Given this basic information, local planners could relate these economic benefits to their assessment of the costs of environmental effects of alternative types of tourism development, or of specific project proposals. In the absence of these data it is not possible to make an assessment of the relative costs and benefits from the various mixtures of different tourism facilities. The valuation of beaches is a key element for developing a cost analysis of the environmental impacts of tourism (Houston, 1995).

1.7 Framework

This book concentrates on providing the practical information necessary to design and implement monitoring programmes and studies of recreational water and bathing beach quality and to create the link between the information generated and action to protect human health. The elements outlined throughout this book should be implemented flexibly according to the different objectives and priorities that exist in the area under consideration.

This book therefore comprises a series of 12 further chapters (Figure 1.2), each culminating in a section on elements of good practice as described above. The introductory comments in this section lead into Chapters 2 and 3 where guidelines on the selection of appropriate variables for the successful monitoring and assessment of recreational water and bathing beach quality are elaborated. This includes selection of suitable areas that can be designated for recreational use and the location of sample collecting stations and necessary variables at each sample station. Logistic issues in implementing the monitoring programme are also explored.

Figure 1.2 Management processes and technical aspects of monitoring bathing waters as discussed in the various chapters of this book



Chapter 4 provides the background for implementing a full and reliable analytical quality assurance system to ensure confidence in, and reliability of, the data gathered. Chapter 5 introduces the management of bathing beaches to maximise health protection of recreational users. A variety of methods of involving the public in the whole process are discussed in Chapter 6. Construction of a public information strategy is discussed together with involving those with a vested interest in ensuring that the most suitable strategy is adopted. Chapters 7 to 11 focus specifically on issues related to a specific type, or group of types, of hazard. These include hazards related to drowning and injury (Chapter 7).

The microbiological quality of recreational waters, its assessment in a consistent and coherent fashion and the interpretation of microbiological water quality data are the themes of Chapters 8 and 9. These chapters look at approaches to microbiological quality and sanitary assessment and the methods employed. The concept of pollution or sanitary inspection is introduced as a rapid and effective means of providing information for the monitoring process. Chapter 10 moves on to examine the potential health effects due to concentrations of cyanobacteria and microalgae in recreational waters; methods of assessing reliably the likely risks from such sources are addressed. Chapter 11 elaborates on monitoring associated with other biological, chemical and physical hazards and Chapter 12 extends the principles of monitoring into the context of aesthetic aspects of beach quality. Issues of perception of recreational water and bathing beach quality derived from aesthetic pollution indicators are explored further. The applied and effective nature of such monitoring and assessment schemes is seen as particularly suitable to coastal and estuarine environments. Chapter 13 introduces the complex practice of epidemiological surveys and their application to recreational waters.

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Monitoring Bathing Waters - A Practical Guide to the Design and Implementation of Assessments and Monitoring Programmes
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Chapter 2*: DESIGN OF MONITORING PROGRAMMES

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* *This chapter was prepared by K. Pond, W. Robertson, G. Rees and H. Salas*

Traditionally the primary reason for the assessment of the quality of an environment has been to verify suitability for intended uses. Monitoring has also evolved to determine trends in the quality of the environment and to determine how quality is affected by anthropogenic activities, including for example waste treatment operations (the latter is known as impact monitoring). Monitoring the background quality of recreational water environments is also now widely carried out to provide a means of assessing impacts and to check whether unexpected change is occurring. In regulatory monitoring programmes, factors such as sampling frequency, analytical methods, data analysis, interpretation and reporting, sample site selection and criteria for recreational water-use areas are generally defined by the regulatory agency.

General definitions for various types of environmental observation programmes have been proposed (e.g. Chapman, 1996) which may also be modified and interpreted in relation to recreational water use, as follows:

• *Monitoring.* Long-term, standardised measurement and observation of the environment in order to define status and trends.

• *Survey.* A finite duration, intensive programme to measure and observe the quality of the environment for a specific purpose.

• *Surveillance.* Continuous, specific measurement and observation for the purpose of management and operational activities.

Each of the above activities are often not clearly distinguished one from another and all may be referred to as “monitoring”, because they all involve collection of information at set locations and intervals. They do, nevertheless, differ in relation to their principal use in the recreational water quality assessment process.

2.1 Aims and objectives of monitoring

No assessment programme should be started without identifying the specific need(s) for information. Thus water quality assessment should take into account hydrological factors, water uses, economic development, policy and legislation, etc. The decisions that will result from the assessment programme determine whether emphasis should be put on concentrations or loads and on spatial or temporal distribution, as well as determining the most appropriate monitoring media. There are generally several competing beneficial uses of the recreational water-use area, and the monitoring activities should reflect the data needs of the various users involved.

The objectives of the assessments may focus activities on the spatial distribution of quality (a large number of sample stations), on trends (high sampling frequency), or on pollutants (in-depth inventories) (Box 2.1). Full coverage of all three requirements is virtually impossible and costly. Preliminary surveys are generally necessary in order to determine the appropriate focus of activities. Table 2.1 summarises the principal types of water quality operations in relation to their main objectives.

Box 2.1 Setting objectives for microbiological monitoring of bathing areas

The objectives of microbiological monitoring programmes can be diverse. However, microbiological monitoring of bathing areas is, in most cases, undertaken to comply with regulations and/or to establish the degree of microbiological pollution in order to protect public health and the environment. Those macro-objectives only answer the question: *Why?* More specific objectives need to be defined which will also tackle the question of *Where?* (location of bathing area, sampling points and frequency). Some aspects are fixed by the regulations. Others, such as the location of sampling points, are only generally defined and require preliminary screening. Such screening will establish the spatial and temporal variations of microbiological water quality to select the optimal sampling points and frequency to obtain data representative of those fluctuations. Questions of *What?* (variables or indicators to be determined) and *How?* (methodology of inspections and analysis) are sometimes only partially defined by the regulations. These variables must be those that are more representative of sewage pollution as a measure of health risk. Specific comparative studies of standardised indicators and procedures are advisable at each specific geographical area.

Public information and participation may be another objective included in the regulations. Microbiological results given to the public should include visual inspections for aesthetic factors that bathers will be confronted with and should be expressed in a clearly understandable ranking system.

Other specific objectives will assess the impact on the microbiological quality of river outlets at the sea bathing area and any discharges at inland reservoirs as well as the effects of rain. Spatial and temporal variations identified before will have to be taken into account and their impact assessed over a representative period so that remedial and/or preventative measures (such as indications of risk) can be encouraged.

Source: Based on the approach used by the Unit of Microbiology, Faculty of Medicine, University Rovira i Virgili, Spain

Table 2.1 Types and objectives of principal water quality assessment operations

Type of assessment	Major focus of water quality assessment
Multipurpose monitoring	Space and time distribution of water quality in general
Trend monitoring	Long-term evolution of pollution (concentrations and loads)
Basic survey	Identification and location of major problems and their spatial distribution
Operational surveillance	Water quality and related water quality descriptors (variables) for specific uses
Background monitoring	Background levels for studying natural processes; often used as reference point for pollution and impact assessments
Preliminary surveys	Inventory of pollutants and their space and time variability; usually prior to designing and establishing a routine monitoring programme
Emergency surveys	Rapid inventory and analysis of pollutants for rapid situation assessment following a catastrophic event
Impact surveys	Sampling limited in time and space, generally focusing on a few variables near pollution sources
Modelling surveys	Intensive water quality assessment limited in time, space and choice of variables to support, for example, eutrophication models or oxygen balance models
Early warning surveillance	At critical water use locations (continuous and sensitive measurements)

Source: Bartram and Ballance, 1996

It cannot be overemphasized that the benefits of careful preliminary planning and investigation far outweigh the efforts spent during this initial phase. Mistakes and oversights during this part of the programme may lead to costly deficiencies, or overspending, during many years of routine monitoring.

2.2 Elements of recreational water quality assessment

Once objectives have been set, the scope of the monitoring programme should be defined. This includes definition of criteria for inclusion or exclusion of recreational water-use areas and the preparation of an inventory of areas included or excluded as recreational water-use areas. A review of existing data and the compilation of a catalogue of the basic characteristics of the area, supported by preliminary surveys, determines the monitoring design. The completed review and catalogue should be followed by recommendations to relevant authorities for management, pollution control and, eventually, the adjustment or modification of monitoring activities (Box 2.2).

Box 2.2 Beach monitoring programme Lima, Peru

The city of Lima, capital of Peru, is located on the coast of the Pacific Ocean and has a current population of about 8 million (1998). In spite of its tropical latitude of about 12° south, the marine waters of the area are relatively cold due to the Humboldt Current emanating from the polar ice cap waters of the South. During the summer months (December to March), the beaches within and near Lima are used extensively by the local population for recreational activities such as swimming.

Lima generates about 16.5 $m^3 s^{-1}$ of wastewater, the major part of which is discharged untreated directly or via the Rimac River to the coastal marine waters. There are no existing submarine outfalls although plans call for the construction of long sea outfalls with treatment during the next decade. Due to the arid climate of the region, the reuse of sewage for crop irrigation is practised to some extent and will increase in the future.

The water quality standards of Peru classify marine waters as safe for primary contact recreation when 80 per cent of five samples taken over one month period show less than 1,000 MPN per 100 ml of faecal coliforms and 5,000 MPN per 100 ml for total coliforms. This standard drives the frequency of measurement of the monitoring programmes described below.

Because of its proximity to populated areas as well as its accessibility by public transportation, Miraflores Bay is a very popular beach area referred to as the “Costa Verde”. A major trunk sewer discharges about 6-7 $m^3 s^{-1}$ of raw sewage directly to Chira Beach (not used by the public) approximately 4 km east (upstream) of the closest popular beach of Costa Verde. Predominate currents in this area are parallel to the coast from east to west.

For years the media attributed the contamination of the “Costa Verde” area to the Chira outfall. In 1986, the Environmental Health Division of the Ministry of Health initiated a beach vigilance monitoring programme with 21 stations sampled on a weekly basis during the summer in the Costa Verde area. The data clearly demonstrated that there was gross pollution in the vicinity of the Chira discharge and that this contamination was beginning to encroach on the popular beach areas of Costa Verde. However, the pockets of contamination observed on some of the most popular beaches of Costa Verde could only be attributed to local direct discharges from sanitary sewer overflows, restaurants and other installations. Bather density could also have been a contributing factor.

The monitoring programme was expanded to 24 stations in Costa Verde that were sampled weekly during the summer bathing season and monthly during the winter from 1987 to 1989. These data confirmed that local discharges rather than the Chira outfall caused the contamination at some popular beaches in Costa Verde. Management action was taken in 1991 with the construction of a small trunk sewer and six pumping stations in Costa Verde to transfer sewage to the main sewer system with subsequent discharge via the Chira outfall.

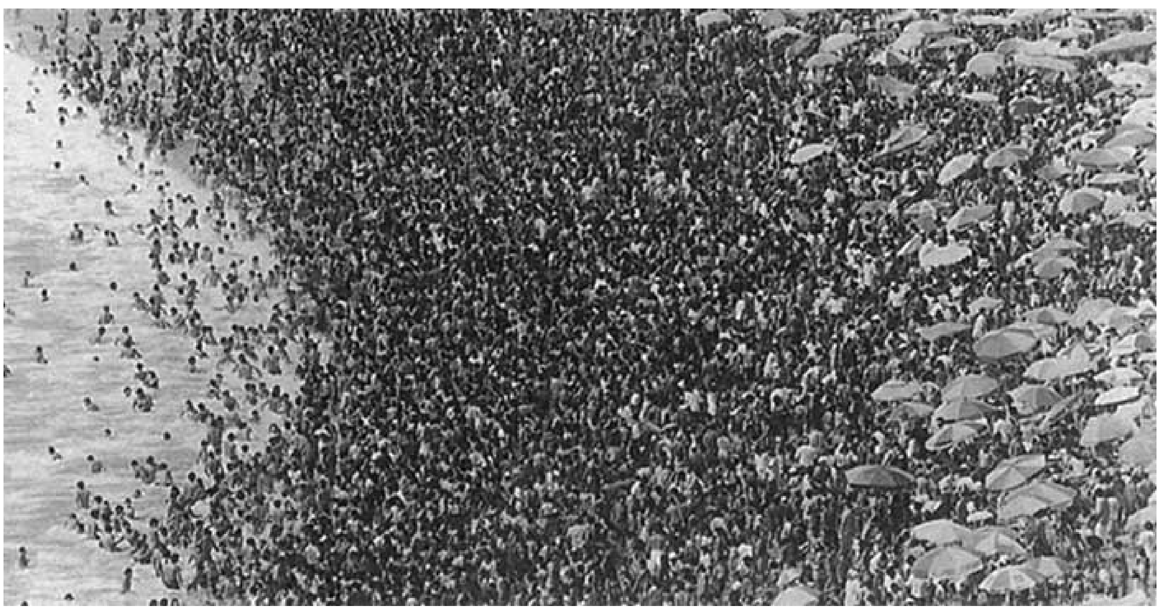
Budget cuts reduced the monitoring programme to as little as 10 stations in 1991 at the most contaminated beaches. In 1992 and 1993 weekly monitoring was resumed at the 24 Costa Verde stations during the summer. These data clearly demonstrated the water quality improvements at some beaches due to management action taken in 1991. The monitoring programme was even able to pick up the impact on beach water quality when the Costa Verde trunk sewer pumping stations were not operating during the energy blackouts caused by the drought of 1992.

In 1994 the monitoring programme was expanded to beaches to the north and south of Costa Verde which are also used by the Lima population. A total of 61 stations were sampled weekly in the summer and monthly in the winter.

The monitoring programme was expanded to national coverage in 1997. The programme presently takes samples at 148 stations: 77 stations at the beaches used by the Greater Lima Metropolitan Area population (these are sampled weekly in the summer and fortnightly in the winter) and 71 stations from Tumbes in the north to Tacna in the south (sampled weekly in the summer and monthly in the winter). In 1996, enterococci, *Escherichia coli* and *Vibrio cholerae* were added to the total and faecal coliform measurements at 46 stations. The programme conducts approximately 13,560 microbiological measurements per year.

The Lima monitoring programme was the key to ascertaining the real sources of the contamination of the popular Costa Verde beaches and instrumental in convincing the authorities, which in turn led to the implementation of sound management actions. The greater investment that would have been required to dispose of the sewage discharged properly via the Chira outfall would not have changed the situation at the popular Costa Verde beaches that were contaminated by small local sources. Nevertheless, if the monitoring approach described in this guidebook had been applied, action might have been taken sooner based on the information provided by the initial sanitary inspection. Furthermore, the sanitary inspection would have served to modify the monitoring programme to focus on those beaches where potential sources of pollution were identified, thus making the monitoring programme source-and use-driven as opposed to only use-driven.

Bathers on the beach at Costa Verde, Peru. Photograph courtesy of “El Comercio”



There are certain elements that are common to all water quality monitoring and assessment programmes. They are more, or less, extensively developed depending on the type of assessment required. These elements are:

• *Preliminary surveys.* Short-term, limited activities to determine the type of monitoring media and pollutants to be considered, and the technical and financial feasibility of a complete monitoring programme.

• *Monitoring design.* The selection of variables, station location, sampling frequency, sampling apparatus, etc. (Chapters 3, 8 and 12).

• *Field monitoring.* This includes *in situ* measurements, sampling of appropriate media, sample pre-treatment and conservation, identification, storage and shipment (Chapter 8).

• *Hydrological monitoring.* Measurements of water discharge, currents, tides, water levels, thermal profiles, etc. Hydrological data should always be related to the water quality assessment activities.

• *Laboratory activities.* These include concentration measurements, biological determinations, etc.

• *Data quality control.* This consists of analytical quality assurance within each laboratory and amongst all laboratories participating in the same programme (Chapter 4).

• *Data storage and treatment.* This is now widely computerised and involves the use of databases, for data storage reporting statistical analysis, trend determinations, multifactorial correlation, etc. together with presentation and dissemination of results in appropriate forms (graphs, tabulated data, data diskettes, etc.) (Chapters 3 and 8).

• *Data interpretation.* This involves the comparison of water quality data from different stations. For specific problems, and the evaluation of the environmental significance of observed changes, external expertise may be needed. Publication and dissemination of data and reports to relevant authorities, the public, and the scientific community is the necessary final stage of assessment activities (Chapter 6).

• *Water management.* Decisions will be taken at various levels involving local, national and international bodies, and by water authorities as well as by other environmental authorities. Important decisions concern the redesign of assessment operations in order to improve the monitoring programme and to make it more cost-effective (Chapter 5).

In recreational water quality investigations, the purpose of sampling is to obtain samples that are as representative as possible with respect to the microbiological, physicochemical and aesthetic properties of the area (Chapters 8, 9, 11 and 12). Sampling should be conducted during the bathing season, but is most appropriate when recreational waters are suspected of being contaminated or a source of waterborne disease. Historical data, combined with an annual environmental health assessment, may indicate that only occasional sampling is necessary. If deterioration in quality has occurred then monitoring of the area should be undertaken. Such an approach will allow health officials to concentrate their resources on beaches of questionable quality.

2.3 Data collection

2.3.1 Beach registration

In order to improve the quality of recreational water-use areas and to select beaches that can be developed as tourist areas, planners and managers may wish to keep a continuous record of selected information. The information necessary for selecting those parts of the coastline that will be used as bathing beaches now or in the near or distant future can be stored in a beach registration system. The aim of such a system is to establish a catalogue of all beaches, to use a checklist to collect the information needed to plan a monitoring programme and to decide if and how the beach will be developed in the future.

The checklist for the registration of beaches may be amended as required; for example, to assess the hazards present for swimmers in a particular area in order to develop the beach for tourism, and to prepare a beach management plan for the planning and co-ordination of all the resources related to providing a safe aquatic environment for the public. This approach could be especially useful when resources are not abundant and have to be employed as efficiently as possible (Chapter 3).

Beach registration is typically divided into four components:

• *Description of the surroundings.* The registration should include information on accessibility (roads, tracks, public transport, no access), hazard mitigating measures (information signs and information sources, lifeguards, showers, first aid posts, swimming and diving safety warnings) and facilities (restaurants, hotels, bars, toilets, drinking water, litter bins, car parks and camping grounds).

• *Description of the beach.* This should include an estimation of the area of the beach (length, width), beach material and visitors per day (estimate the peak numbers according to season, whole bathing season, main holiday period, public holidays and weekends). The number of visitors per day should be compared with the visitor capacity of the area.

• *Description of the water environment.* This includes details of the bathing zone (direction and speed of the current, slope, bottom material) and its use (fishing, jetskiing, intensive yachting, swimming, diving, etc.).

• *Counter indications.* Designated sensitive areas (resting place for water fowl, breeding place for rare birds, sanctuary, conservation area and other kinds of protected area such as military sites or other areas where public access is prohibited).

The information listed above should be collected by means of a desk survey of the existing information and during a subsequent field inspection on the beach. The baseline information should be revised annually. Ideally a map should accompany each registration and should show the extension of the beach, the accessibility, the surroundings, etc. The information may then be transferred to a computer system and amended as necessary.

In gathering data for inclusion in a beach registration system it is important to involve the local community. Often local people, local politicians, shopkeepers and, in particular, those charged with the operation of the beach (the local authority, beach operator, lifeguard) will have valuable information. Relevant non-governmental organisations (NGOs) such as nature conservation groups, angling clubs and yacht clubs, water skiing clubs and lifesaving associations, may provide useful information (Chapter 6).

The development of a registration system in the form of a database could aid coastal managers in their decision-making process through highlighting the suitability of beaches for particular uses. For example, it may become apparent that some beaches should not be promoted for recreational use because the area is ecologically sensitive or because bathing might be dangerous due to currents, bottom conditions or particular health hazards.

2.3.2 Environmental health assessment

In the past, sanitary inspections or surveys were directed primarily towards microbiological contamination of recreational waters but in recent times they have been broadened to include chemical contamination and other biological and physical hazards. The term environmental health assessment is now used to reflect this broadened scope. An environmental health assessment can be defined as a comprehensive search and evaluation of existing and potential microbiological and chemical pollution and biological and physical hazards that could affect the overall safety of a particular stretch of recreational water or bathing beach. Potential influences on water quality (such as river mouths, sewage outlets, harbour areas, other wastewater outlets) and physical hazards (rocks, open and rough water, rip-tides, shallow water, etc.) should also be considered. A comprehensive environmental health assessment consists of pre-inspection preparations, an on-site visit and the preparation of an assessment report. The environmental health assessment typically relies upon on-site inspection of hazards and mitigating factors for physical and microbiological hazards (Chapters 7 and 8), and on water quality testing, especially for microbiological quality (Chapter 8). When undertaking an assessment the sampling techniques employed are particularly important. Some guidance for obtaining statistically valid measurements is given below. Full details of methods and quality assurance procedures to be followed in sampling programmes are provided in Chapters 3, 4 and 8.

2.3.3 Quality monitoring

During routine visits, the site should be surveyed for signs of microbiological and chemical contamination. For example, visible sewage plumes, oil slicks, suspicious odours and fish or bird kills should be considered as immediate indications of unacceptable water quality. Beach monitoring for litter, tar balls, etc. should also be undertaken. The task of deciding the optimum number of samples to take and the most suitable locations in order to characterise quality in a meaningful way, and with the most economic use of resources, can be quite daunting. Statistically-based methods of sampling design can help this task and can also ensure that the data collected are appropriate for later statistical analysis and interpretation (Chapters 3 and 8). Basic sampling design naturally falls into seven aspects:

- Reasons to sample.
- What to sample.
- How to sample.
- When to sample.
- Where to sample.
- How many samples to take.
- Sampling evaluation.

The issues of what, when and how to sample are defined by the assessment programme objectives.

The results of any quality monitoring programme depend on where the pollution comes from, and therefore on where the samples have been taken. The physical factors characterising sampling stations may vary widely between stations, resulting in large differences in analytical values; for example, in bathing water monitoring stations water depth, current speed and direction, existence of haloclines and/or thermoclines, mixing processes, sampling depth and distance to sewage outlets and other pollution sources may all affect water quality. In reservoirs and lakes the phenomenon of thermal stratification is a source of complexity in sampling design because of its potential affect on vertical water quality differences. The most usual basis for design in such circumstances is stratified random sampling.

Sampling sites should be selected on the basis of information gathered during the beach registration and the first on-site inspection. Ideally, the sites chosen should be representative of the water quality or beach area throughout the whole area where users are exposed. The selection of sites should pay particular attention to site-specific conditions that may influence the concentrations and distribution of indicator organisms and pathogens.

Monitoring and surveillance programmes generally rely on observations made on discrete samples obtained within spatial and temporal constraints. An essential component of a monitoring programme is ensuring that the sample obtained is representative of the phenomenon under study. Errors introduced during sample collection and preparation are usually several orders of magnitude higher than errors due to analytical determinations.

The main aspects to be considered for obtaining a representative sample are: the adequate selection of the sampling points, sampling stations, frequency and timing of sampling; the strict adherence to proper sampling and quality assurance procedures; the complete identification of the sample; the adequate preservation of the sample; and the prompt transport of the sample to the laboratory.

The exact location of sampling points in any monitoring programme, including the distance between them, varies with each individual beach. Chapter 9 provides a sampling protocol including sampling and analysis criteria to be followed for microbiological water quality. This should be used in conjunction with the guidance provided in Chapter 3 to adapt the monitoring programme to the resources available. An environmental health assessment in the recreational area provides a good basis for establishing the location and number of sampling stations. The results of an intensive sampling programme, together with a detailed survey of water currents and water discharges, will identify any particular pattern of water quality deterioration that has to be considered when selecting sampling stations representative of the whole recreational area (Chapter 9). The intensive sampling programme should include the analysis of water samples taken at different water depths, at different hours of the day, during different tidal phases, and during any other known source of possible variation. The experience gained during the implementation of the monitoring programme should serve to modify and improve the initial sampling programme. Advanced statistical analysis may be used to identify spatial and temporal patterns amongst sampling results drawn from special surveys. These patterns may then be used as the basis for more general sampling site allocation. This is a particularly important aspect for dealing with the variability associated with reservoirs.

Details of relatively simple types of water sampling equipment are contained in Bartram and Ballance (1996) (see also Chapter 8). For surface and subsurface sampling, the containers used should be bottles of dark-coloured borosilicate glass of 200 to 300 ml capacity, with wide-mouths and ground glass stoppers. The same type of bottle may be used for subsurface sampling with the addition of an extension arm and clamp. Specific sampling procedures are contained in the recommended methods for determination of specific indicator organisms and pathogenic bacteria described in Chapters 8 and 9. For sediments, several types of bottom samplers are available commercially that can be used for collection of samples of sediments for microbiological analysis. The equipment required to monitor aesthetic aspects of recreational water-use areas is generally less than for water quality monitoring and varies depending on the method used (Chapter 12).

Sampling should be performed in a systematic manner to reduce variation between individual results. For this reason, it is necessary to keep constant as many factors as possible. These include the period of sampling (i.e. time of day) and the sampling method, as well as the location and depth (in the case of water sampling) of individual sampling points. Sampling can be considered as completed once the sample is transferred to the sterile container, whether on the beach or aboard a sampling vessel.

When sample transit does not allow the use of a central laboratory, other alternatives must be considered. These may include analysis of samples in an approved laboratory nearby, use of an approved laboratory field kit or use of a mobile laboratory. Such alternatives should undergo thorough testing and comparison before they are adopted.

2.4 Elements of good practice

• The objective(s) of a monitoring programme or study should be identified formally before designing the programme and they should be stated prior to data gathering.

• Objectives should be described in a manner that can be related to the scientific validity of the results obtained. The required quality of any data should be derived from the statement of objectives and should be stated at the outset.

• In designing and implementing monitoring programmes, all interested parties (legislators, NGOs, local communities, laboratories, etc.) should be consulted. Every attempt should be made to address all relevant disciplines and to involve relevant expertise.

• The scope of any monitoring programme or study should be defined. This would normally take the form of definition of criteria for inclusion and exclusion of recreational water-use areas and preparation of an inventory of recreational water-use areas.

• A catalogue of basic characteristics of all recreational water-use areas should be prepared and updated periodically (generally annually) (and also in response to specific incidents) in a standardised format. It should include as a minimum the extent and nature of recreational activities that take place at the recreational water-use area and the types of hazards to human health that may be present or encountered. Unless specifically excluded, the list of potential hazards to human health would normally include the microbiological quality of water, cyanobacteria or harmful algae, drowning and physical hazards. Monitoring programmes frequently also address aesthetic aspects and amenity parameters because of their importance to health and well being.

• Programme or study design should take account of information derived from the inventory of recreational water-use areas and catalogue of basic characteristics which, in turn, may require refinement of programme objectives.

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Chapter 3*: RESOURCING AND IMPLEMENTATION

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Monitoring programmes, by necessity, must be commensurate with the socio-economic and technical and scientific development of the country where they are implemented. For example the extent of development of national legislation and co-ordinating or oversight programmes will affect activities undertaken. Similarly, more complex analytical variables require highly trained technicians and costly laboratory facilities. In general terms, it is possible to distinguish three levels into which monitoring programmes can be classified (Table 3.1). All elements of assessment, from objective setting to data interpretation, are related to these three levels. The aim is to progressively develop monitoring operations from the basic level to the more comprehensive levels. Each level is associated with increasing demands on staff (expertise and numbers), inspection and fieldwork (complexity and frequency), laboratory facilities (range of analysis, throughput) and data management and reporting capacity.

Table 3.1 Levels of monitoring competence in relation to resource requirements

Level	**Basic information/visit rate**	**Accident hazards**[1]	**Microbiological parameters**[2]	**Cyanobacteria and algae**[3]	**Other**
Local (no national organisation yet)	Local action comparable to basic level in some locations only	Local action comparable to basic level in some locations only	Local action comparable to basic level in some locations only	Local action comparable to basic level in some locations only	Local action comparable to basic level in some locations only
Basic (no access to equipment or staff resources at national level; limited local resources)	At least one pre-season visit; creation of a catalogue of basic characteristics; all beaches registered, but more used and higher risk beaches inspected and monitored	Annual inspection for identification of any hazards and interventions (e.g. signs, warning systems)	Inspection for faecal pollution or sewage odour; delimitation of high risk areas; initial screening of faecal streptococci (marine or freshwaters), *E. coli* or faecal coliforms	Inspection for scum, type, and transparency	Register of local special problems

			(freshwater) for primary classification; internal quality control at laboratories; at least one sample a month in season once the beach is classified		
Intermediate (limited access to resources both local and national level)	Comprehensive cataloguing and timetabling of visits; additional visits during peak seasons (e.g. monthly); greater proportion of beaches monitored	Periodic verification of interventions during bathing season; central capacity for incident investigation	Identification and cataloguing of potential sources of contamination; all beaches at primary classification; monthly sampling; resampling and investigation of unexpected peak values; reclassification scheme initiated; investigation of rain effects and design of preventative measures; internal quality control at laboratories; occasional inter-laboratory comparison studies	Phosphate analysis (freshwater) Chlorophyll a (freshwater)	Check on local information availability; active warning and management response
Full (no significant resource limitations)	Additional visits during peak seasons (e.g. fortnightly or weekly); complete cataloguing, including updating for each recreational area; all beaches with significant use monitored	Central register of recorded incidents; decentralised capacity and procedure for incident investigation	Additional microbiological parameters if necessary; possible reclassification investigated where indicated; internal and external quality controls regularly operated; convergence amongst participating laboratories	Toxicity detection and toxin analysis capacity if necessary (not routine); remote sensing methods where relevant	Chemical monitoring (for necessary parameters)

Each level also demands inclusion of the requirements specified at lower levels

[1] See Chapter 7
[2] See Chapters 8 and 9
[3] See Chapter 10

3.1 Staffing and training

The personnel in charge of sample collection, field handling and field measurements must be trained for these activities (Table 3.2). The choice of personnel for sampling depends on a number of factors, including the geographic features of the region and the systems for transportation. For example, in a small country with good transport infrastructure, sampling may be carried out by laboratory personnel going to the field to take samples, conduct field analyses and transport samples back to the laboratory. In countries of a larger size that possess a more developed monitoring system, specially-trained field personnel often conduct the sampling and inspection. In large countries that have a poor transportation system, relatively more personnel are required. In this situation specialists from decentralised facilities, such as health centres or hydrometereological and hydrological stations, may be involved in sampling, inspection and testing. Such personnel may not always possess all appropriate training.

Table 3.2 Principal tasks undertaken by staff type as an indication of skills and training requirements for differing levels of monitoring competence

Level	**Field staff**[1]	**Laboratory staff**
Local	Basic inventory for beach registration Collection of water samples for microbiological and cyanobacteria analysis Transparency measurement (Secchi disc) Cyanobacterial scum recognition Basic sanitary survey	Data analysis and management Analysis of faecal indicator bacteria (according to indicator and method available)
Basic	More complex sanitary survey Selection of sampling sites Intensive microbiological sampling for primary classification Observational verification of effectiveness of interventions	Phosphate analysis Chlorophyll a analysis Organisation and implementation of necessary quality assurance
Intermediate	Local follow-up of unexpected peak microbiological values (including liaison with local authorities)	Participation in occasional, informal interlaboratory comparisons (probably “round-robin”)
Full	Participation in accident investigation	Cyanobacterial toxicity detection and

		toxin analysis, where relevant Participation in regular, formal inter-laboratory comparisons

For a description of the different levels of monitoring competence, see Table 3.1

[1] The balance between field and laboratory staff is determined by local factors

This guidebook provides a major part of the information needed for carrying out successful and adequate fieldwork and sampling. The quality of information produced by a monitoring programme depends on the quality of the work undertaken by field and laboratory staff. The importance of appropriate training cannot be over-stressed.

Separate training packages should be developed for field staff, laboratory staff and others. It should be emphasised that training is not a “once-only” activity, but should be continuous. Supervision, as a form of training, is especially relevant to laboratory and field staff. It is vital that the training function is flexible, responding to experience and feedback and taking account of specific needs. Training is especially important when programmes for monitoring are implemented in several countries or independently managed areas or regions, from which the results will be compared and used outside the country or region where the monitoring has taken place.

Assuming a good general education or relevant previous experience, the training period for staff responsible for fieldwork and sampling is about one week, with approximately one additional week of further training as monitoring develops through basic, intermediate to full levels. In order to maintain motivation, it is recommended, that short follow-up events are provided. If the staff are less experienced more training will be necessary.

Although not exhaustive, training for field staff should include the objectives of the water quality monitoring programme and its local, national and international significance. The training should stress the importance of samples being of good quality and representative of the water body from which they are taken and it should give guidance on how to ensure that the samples meet those requirements. The training should also include planning of field sampling and map reading, as well as how to make field notes describing the sampling site and station and how to undertake on-site inspections. Safety aspects of field sampling are an important component of the training programme.

Staffing requirements for servicing a monitoring programme may vary widely and it is not possible to make general statements about the number of staff needed for fieldwork. Estimating staffing requirements includes allowing for travel time or distance between beaches and laboratory, the choice of laboratory infrastructure (e.g. centralised or decentralised) and co-ordination between participating institutions.

The head of the laboratory and/or the programme manager are generally responsible for:

- Laboratory management.

- Determining and procuring the equipment and supplies that will be required.

- Ensuring that Standard Operating Procedures (SOPs) are being followed (Chapter 4).

- Ensuring that adequate quality control procedures are being followed.

- Enforcing safety procedures.

Laboratory technicians must have suitable training. They will generally be responsible for:

- Maintenance of the laboratory including, cleanliness and safe storage of all equipment, glassware and other reusables.

- Storage and preparation of reagents and media.

- Checking the accuracy of field equipment.

- Training of junior staff.

- Performing the tests and recording the results of field analyses.

3.2 Laboratory and analytical facilities

The choice of laboratories to be involved in the monitoring programme can have a major influence on the time required for collecting the samples. There may be reasons such as ensuring staff training, equipment repair and analytical quality control, that suggests using a central laboratory is more appropriate. Nevertheless, when total sample numbers are high and samples are transported over long distances, it may be preferable to use more than one laboratory. Under these circumstances it is normal practice for one of the laboratories to act as the central or co-ordinating laboratory. Some countries have developed mobile laboratories (typically in small vans or minibuses) as an alternative solution to the problems of sample transport. Where long distances are encountered this has often been found to be an effective approach. The need for co-ordination between institutions, for example when sampling is undertaken by one agency and analysis by another, may reduce the necessary workload but may lead to inefficiency if the co-ordination is not effective.

In many countries, monitoring laboratories are organised on two tiers: regional laboratories (lower level) to conduct basic determinations not requiring very complex equipment, and central laboratories (higher level) to conduct more complex analyses requiring elaborate equipment and well trained personnel. In addition, the central laboratories often provide the regional laboratories with methodologies and analytical data quality control.

During the initial stages of development of a recreational water monitoring system, it is reasonable to focus on the basic variables that, as a rule, do not require expensive and sophisticated equipment. Gradually, the number of variables measured can be increased in relation to the financial resources of the monitoring agency. Even in fully-developed monitoring programmes, the elaborate equipment and technical skills necessary for the measurement of complex variables are not needed in every laboratory.

Laboratories must be selected or set up to meet the objectives of each assessment programme. Attention should be paid to the choice of analytical methods. The range of concentrations measured by the chosen methods must correspond to the concentrations of the variable in a water body and to the concentrations set by any applicable water quality standards. Ideally, a laboratory should consist of four sections:

- Reception and registration of samples.
- Production of analytical media.
- Sample analysis.
- Washing up and autoclaving equipment.

However, it is possible for a laboratory to function properly in fewer sections, depending on the number of analyses to be undertaken.

3.3 Transport and scheduling

Resource elements in fieldwork include transportation, personnel requirements and equipment. Problems of transport are largely related to distance and accessibility of sampling sites. Where access to sites is known to be problematic, reliance has been placed traditionally on four-wheel drive vehicles, but these are often expensive to purchase and to operate. It may generally be assumed that the main part of the beaches included in the routine monitoring programme can be reached using ordinary vehicles because beaches difficult to access have most probably been excluded when selecting the participating beaches. Some beaches may be located such that using motorcycles or public transport is a possibility; however, most frequently, ordinary cars equipped for transportation of the samples will be required. Sampling transport equipment can consist of an insulated box with melting ice, or a refrigerator installed in the vehicles, to ensure that the samples are kept cool.

Travel time to and from sampling sites is a major constraint for staff undertaking fieldwork. Realistic estimates of travel time to each sampling site should be made as early in the programme as possible. An inventory of sampling stations should be developed, including actual travelling time, in order to facilitate programme planning. When planning the route for visiting the sites, the demand of having the samples analysed at the laboratory within the appropriate time frame must also be taken into account (see Chapter 8 for microbiological analyses and Chapter 10 for cyanobacteria).

3.4 Inspection forms and programmes

Public health authorities should have, at least, a basic inspection programme in place for all recreational-water sites within their jurisdiction. The primary purpose of the inspection programme is to minimise the risk of illness or accident to bathers. The programme should be based on an SOP (refer to Chapter 4). This will ensure that the on-site inspection, laboratory analyses and interpretation of data are carried out in an objective and uniform manner. An on-site survey form should be prepared as a guide for inspectors to make certain that all aspects of the site receive adequate review and evaluation.

In most programmes at least two forms will be used. The first “basic registration” form collects the minimum background information to construct a register. The beach register is a high priority during the basic level of monitoring. The principal uses of the beach register are:

- To determine which beaches should be considered by the programme, based upon criteria such as extent of use, degree of development (or plans to develop) and already-recognised hazards.

- To provide data (such as transport options and travel times) to assist in programme planning.

Details of the information to be included in a beach registration form are provided in Chapter 2.

Once a beach enters into a monitoring programme it will be subject to periodic, usually annual, inspections. These inspections are principally orientated towards the identification of hazards that might lead to physical injury or contribute to drowning; towards the adequacy of measures (signs, lifeguards, communications) in place to reduce these risks; and towards sources of microbiological pollution such as sewage outfalls, combined sewer overflows, rivers and storm drains. At freshwater sites, the inspection may also be concerned with the likelihood of cyanobacterial blooms (see Chapter 10). An example of a sanitary survey form is included in Chapter 8 (see Box 8.1) and the components of an on-site inspection are discussed further in Chapter 7 (physical hazards), Chapters 8 and 9 (microbiological aspects) and Chapter 12 (aesthetic aspects). Other inspections may be required sporadically depending on local circumstances, for example to assess bather load (Chapter 8), cyanobacterial hazards (Chapter 10) or to verify the effectiveness of interventions to control microbiological quality (Chapter 9).

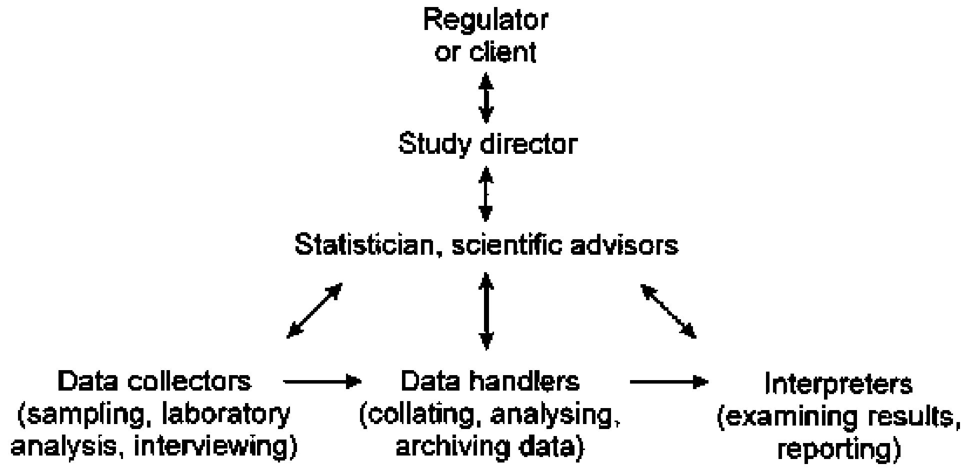
A key requirement of any inspection programme is the availability of qualified personnel. Ideally, inspectors should have a basic knowledge of public health microbiology, environmental chemistry, limnology, oceanography, estuaries and meteorology. Training workshops, based on SOPs and actual case studies, should be held during programme development and implementation. Additional workshops should be held periodically to review inspection procedures.

The co-operation of all involved or interested individuals or organisations is essential if assessments and monitoring programmes are to result in improved water quality and reduced health risks. Representatives from user groups, tourist associations, beach and resort owners, industries and sewage treatment facilities and public health laboratories should be aware of, and invited to participate in, the programme. The complexity and completeness of surveys are likely to increase as programmes develop from local through basic and intermediate to full levels (see Table 3.1). At the most advanced level, for each site, three to five days may be required to prepare for the survey, conduct an inspection and prepare the report, although subsequent re-verification visits may require less time. From these findings public health authorities will be able to classify the beaches within their jurisdiction with ratings ranging from excellent to very poor and to relate these ratings to requirements for monitoring and management. Chapter 9 discusses in depth the development of classification schemes for microbiological quality. In this way, resources can be directed to those beaches that present the greatest risks to public health.

3.5 Data processing and interpretation

If data have been collected in a careful manner in a properly designed study, they will be representative and unbiased and suitable for analysis, interpretation and reporting. Processing, management and storage of data can be collectively described by the term “data handling”. This activity has a central position in any study as shown in Figure 3.1. The data handlers receive data from samplers and interviewers in the field, as well as the results of analyses from laboratories. They then assemble and archive the data in central files and analyse the data as instructed, so that they can be interpreted by those responsible for reporting the results. It is most desirable to prevent bias, from the introduction of personal or political perceptions, by arranging for separate groups of people to carry out the collection, handling and interpretation of data. Data handlers must be appropriately trained and qualified for their duties. It is also desirable that the data handlers do not have any particular interest in the results of the study, apart from carrying out their work with dedication and accuracy. The whole process of the study should be under the control of a study director, who will receive instructions on the objectives and on the programme to be carried out as well as statistical and other scientific advice from appropriate experts. In this way, the requirements for effective data management (Box 3.1) will be fulfilled. The period over which the data may be needed and the purpose for which they could be used should be considered at the outset, particularly for large or continuing studies. Computer hardware and software systems inevitably change over time and future compatibility could be a problem. For example, it is doubtful whether data stored on punched cards or paper tape some 30 years ago could now be retrieved. Moreover, changes in methods of analysis will reduce the compatibility of data from older studies with those of recent studies, thus rendering the raw data (but not necessarily the conclusions) less useful. Consideration should be given to storing data in several places, and in more than one format, in order to forestall the risk of loss and damage.

Figure 3.1 The central relationship of data handling in the conduct of monitoring programmes and scientific studies



Box 3.1 Requirements for effective data management

The requirements of the monitoring programme must be defined. This is the responsibility of the study director, instructed by the regulator or client and as advised by the statistician and scientific experts. In the case of a scientific survey or epidemiological study, the effects under study are framed as conceptual models, requiring the testing of null hypotheses by statistical methods. The data required are determined by the monitoring schedule imposed by the regulatory authority or, in the case of a survey, by the desired statistical power.

Appropriate forms or other recording instruments are produced for collection of the raw data required to meet the above requirement. These must be approved by the collectors and handlers of the data, so that all suppliers of data use a standard, approved format. It is essential that the raw data are in a form that can be transcribed easily and with minimal risk of error to a permanent file, suitable for processing and archiving.

The data are analysed by appropriate methods to produce the desired output. The appropriateness of the methods must be decided earlier in fulfilment of the first requirements. Suitable analytical quality control procedures must be used for input and analysis of data.

For the long term, it is vital to ensure that published reports carry adequate details of the methods of surveys and that the key data and basic statistics, such as means and standard deviations of measurements, are recorded as appendices to the final reports. The basic test of the adequacy of survey recording is whether an independent investigator could understand what was done sufficiently to be able to repeat the work and to be able to test the results and conclusions from the data presented.

The greatest cost of any study is the data collection. These data will be unique and there will usually be no opportunity of repeating a missing or erroneously recorded observation. The raw data sheets should be examined critically by the person in charge of collecting the data as soon as possible after recording, so that discrepancies can be detected and corrected while the events can still be remembered. It may be possible to check suspect meteorological and tidal records with those obtained at nearby, official recording stations. Different sets of data (e.g. from different sites, sampling runs or days) must not be pooled, unless it has been shown that there is no significant statistical difference between the data sets.

It must be remembered that data are collected for a specific purpose, and thus they may not be useful for meeting subsequent objectives. Most data from routine, regulatory monitoring fall into the “data-rich, information-poor” category because other information, necessary for explaining the circumstances or trends, was not recorded at the time. The use of statistical methods to test *a priori* null hypotheses is an obligatory part of scientific method. However, the temptation to carry out further analysis of the complete data set at a later date, in order to detect statistically significant associations or correlations between variables, is strictly invalid unless used solely as a method of suggesting hypotheses for further study. Such “data dredging” is analogous to fitting targets to holes (Jolley, 1993).

The effects studied in behavioural and epidemiological research of water recreation are typically small and are difficult to separate from confounding factors (i.e. factors that affect the responses of the exposed and control groups differently). Potentially confounding factors must be identified at the planning stages of a study and suitable measures should be introduced into the design of the study (e.g. by the matching of exposed and non-exposed subjects, or by random assignment) and in the analysis of the data to detect or nullify them. Confounding can be detected or corrected for in analysis by various methods, such as stratification of the exposed and non-exposed groups (e.g. by common potential confounders such as age, sex or socio-economic class) or by including potential confounders as variables in multivariate analyses, such as logistic regression analyses (for a description, see McCullagh and Nelder, 1989). It is not possible to eliminate the effects of confounding in the data, if it has not been considered, measured and examined in the design of the study (see Chapter 13) (Datta, 1993; Leon, 1993; WHO, 1998).

3.5.1 Database construction

The construction of the database will depend on:

• The level of monitoring as defined in section 3.2 and the intended use of the data, e.g. for compliance monitoring, risk assessment, baseline data, acquisition, new scheme design, epidemiological investigations and/or post-audit evaluations or remediation efficacy.

• The technology available.

• The requirement for data transfer to regulators or to other agencies.

• The requirements for “data audit” and “chain of evidence” procedures.

There are many data storage systems available that have been developed by laboratories and software engineers. Perhaps the most stringent and sophisticated database would be characterised by a data storage system able to accommodate the information suitable for a full “chain of evidence” assessment. This type of database is being developed in drinking water surveillance monitoring. The reason for the high level of laboratory audit stems from the potential litigation that could derive from contaminated waters and associated disease outbreaks. This type of database is emerging in drinking water assessment in the UK following a consultation document produced by the Drinking Water Inspectorate (DWI, 1998). The requirements of such a data storage programme are that the data could be used in a court of law to achieve a criminal prosecution. To achieve this the prosecution is generally required to demonstrate that the data are accurate beyond reasonable doubt, i.e. the analyst can prove that there are systems in place to prevent tendentious or accidental misreporting through laboratory analytical error or database construction mistakes (such as recording a result from the wrong sample).

Laboratory audit control is now routine in laboratories undertaking compliance assessment programmes for recreational and drinking waters in some countries. Here, the recording laboratory must be in a position to prove that the recorded value is correct and that laboratory procedures have been followed. The level of internal control is less than for “chain of evidence” assessments, i.e. the laboratory would have to prove that all reasonable measures had been taken but it would not have to demonstrate that there was no error “beyond reasonable doubt”.

Any environmental sampling programme should implement appropriate intralaboratory and interlaboratory analytical quality control (AQC) procedures. These would normally involve the collection of field and laboratory blank (e.g. sterile) samples to investigate the integrity of aseptic techniques and field duplicate samples to investigate reproducibility. Generally, it is preferable to use split samples from the same bottle in the case of duplicate enumerations. In inter-laboratory trials duplicate samples are split, or spiked samples are prepared and delivered to participating laboratories for analysis. Analytical quality control is described in full in Chapter 4.

There are many commercial data storage software systems available. The developing international standard is a spreadsheet that facilitates storage, rudimentary statistical analyses, graphical representation and data export for external communication and reporting. Spreadsheet packages are commonly used for the storage and recording of recreational water quality data worldwide. Such systems are appropriate for the user familiar with computation methods and standard package use. They may require adaptation for the lay user or to make them effective tools for clerical staff with a low level of information technology (IT) skills. In such circumstances, the spreadsheet can be modified with a Graphical User Interface (GUI). This facilitates rigid, but user friendly, data access and export, appropriate and tightly controlled analysis and, where appropriate, data cleaning and security. Perhaps the most widely used programming language in the production of a GUI is Visual Basic. This language can be used to design forms and screen menus to control data input, data analysis and reporting, as well as to provide security through the use of passwords. Visual Basic programming requires a competent programmer but the advantages in its use for data security and quality are very significant.

The data storage system should be appropriate to the intended purpose of the storage exercise. If the objective is to demonstrate chain of evidence or a future audit of the data for compliance, a bespoke system will probably need to be constructed to facilitate clerical input, data checking by non-technical staff, data cleaning after extreme values have been flagged, and clearly defined data reporting as defined by legislation. Here, linking a GUI with a spreadsheet through Visual Basic is probably the most appropriate route. Where the data are simply being stored for scientific and/or baseline definition purposes, and no immediate audit beyond normal AQC is required, the spreadsheet alone can be a suitable vehicle for data storage.

3.5.2 Preliminary examination of data

It is important, in the interests of consistency, to agree at the outset the procedure to be used for dealing with missing, indeterminate and outlying values, and to adhere to these procedures consistently. Sets of data frequently contain missing values. These values can often be estimated from trends in the data set (where the subsequent analysis requires a complete set) but it must be realised that such “patching” reduces the number of degrees of freedom on which to base statistical decisions. Missing data cannot be estimated or reported when monitoring to assess compliance with a standard for water quality.

Indeterminate values are frequently found, implying that the volume of sample examined, or the concentration of the determinand, was too small or too large to be within the limits of the method. The analysts should be instructed to record the facts, i.e. “less than” or “greater than” the analytical limit, together with the volume examined. This, at least, enables a rank order of values to be established. Procedures for estimating a mean from data with indeterminate values are described below. The particular difficulties encountered in microbiological analyses are explained in Chapter 8.

Detailed examination of data often reveals values that lie outside the normal range of values or trends in the data and which therefore seem improbable (known as “outliers” or doubtful values). Some computer programmes are able to identify outliers, according to defined criteria. Such doubtful values must be investigated by going back to the original records and, if at all possible, to all laboratory personnel and samplers who were responsible for obtaining the value recorded. Only if there are strong technical reasons (e.g. contamination of a batch of culture medium, or a fault in a recording instrument) should such values be deleted from the data set. There are no valid statistical reasons for excluding outliers from sets of data. Their occurrence provides a strong case for investigation of sampling and laboratory procedures. They also indicate the value of carrying out simple checks for the consistency of data in the laboratory at the time when they are first recorded. If no technical problem is found, such values must be accepted. These values could be the first indication of a change in water quality or they may be a random, infrequent event.

Due regard must be taken of the underlying nature of the probability distribution within the data collected, because this will determine the most appropriate way of expressing the central tendency and dispersion of the data. This should be taken into account when standards are derived, because standards will invariably specify their requirements and limiting values in terms of statistics, such as the average, median or geometric mean, or an upper percentile value.

Table 3.3 Examples of the different probability distributions which may be encountered in surveys of recreational water use areas

Distribution	**Properties and examples**
Discrete (whole-number) random variables	
Binomial	Results of a sequence of independent trials, specified in advance and with two possible outcomes (“yes”, “no”) and constant probability of success from trial to trial.
Poisson	Describes occurrence of random events in a continuum (e.g. annual deaths by drowning in a region or counts of randomly distributed particles in independent, identically sized samples); variance and mean are identical.
Continuous random variables	
Normal	Conforms to the normal probability density function, distributed symmetrically about the arithmetical mean (average); mean and median are identical. Distribution is described by a mean and the standard deviation. Heights and weights of individuals, errors of analysis and data for many physical and chemical measurements usually approximate to normality.
Log-normal	The logarithms of the values are normally distributed. Distribution is described by the geometric mean, which is equivalent to the median, and by the standard deviation of log values (the log standard deviation). Generated by random variation

	of the rate constant in natural processes subject to exponential decay or growth; sets of microbiological counts or chemicals diffusing in water, or particle-size analyses of sediments are therefore typically log-normally distributed.
Ordered, categorical variables	
	Data can be arranged and ranked in order of size or categorised, but do not have discrete or continuous values. For example, water users can be arbitrarily ordered according to their observed degree of contact with water (i.e. none, wading and paddling or head immersion). In other cases, the categories do not have a size or implied order (e.g. water-contact recreation may be swimming, surfing, rafting or diving). Data can be tested by appropriate non-parametric methods.

Parametric tests for statistical significance assume that the data conform to a particular model of distribution, such as the normal distribution, and thus are only valid when applied to data that are known to conform approximately to that distribution, or can be transformed appropriately so that they do so. This is a problem where statistical advice must be sought before attempting analysis of data. However, many non-parametric tests of significance, such as those using ranked values instead of actual numbers, are inherently distribution-free, but they lack the statistical power of the parametric tests.

Table 3.3 lists examples of frequency distributions that are commonly encountered in the data collected in surveys of recreational water-use areas, together with their properties. Invariably, there will be data sets that do not conform to the frequency distributions listed in Table 3.3 and, at best, only an approximate fit to the frequency distributions will be possible. A full treatment of probability distributions and their properties can be obtained from textbooks of probability and statistics (e.g. Devore, 1991).

The use of the mean and the standard deviation to describe central tendency and variability in data is appropriate if the data are distributed normally or approximately normally. This is usually the case with physical and chemical data. Microbiological, virological and biological data are almost invariably found to be skewed and distributed log-normally or approximately log-normally. This is thought to occur because of growth, decay and dispersion processes in natural waters, which tend to follow exponential (first-order) reactions, in which the rate constants are subjected to random environmental fluctuations. Skewness caused by log-normality can be detected in several ways. It should be suspected if the data contain many relatively small values and relatively few very large values, or if the average (arithmetical mean) is much larger than the median.

Tests for skewness are given in many statistical textbooks. Log-normally distributed data can be made to conform to the normal distribution before analysis if they are first transformed to logarithms. The geometric mean (i.e. the antilog of the average of the log values) and the standard deviation of the logarithms are the appropriate statistics for such data. The geometric mean cannot be calculated if any of the values are zero (e.g. “undetectable in the volume examined” or “less than the limit of detection”). In such cases, the median can be recorded as an equivalent to the geometric mean, with an explanatory note, or a log (x + 1) transformation can be used, by adding 1 to all the values of x before taking log values. The reverse transformation is used in expressing the geometric mean.

Other transformations are sometimes used. Whole number counts of random events, such as deaths by drowning, usually conform to the Poisson distribution. This should be suspected if the variance (the square of the standard deviation) of the data is similar numerically to the mean. The appropriate transformation is to take square roots of the values, if they lie in the range 10-100, or $\sqrt{(x + 0.5)}$ if the values are less than 10. If most of the numbers are greater than 100, transformation is not needed. In the case of values which are rates or ratios, such as velocities (m s^{-1}), reciprocals should be used and the correct mean of the *n* values of *x* is the harmonic mean:

$$\frac{1}{\sum(1/x)/n}$$

Skewness and the effects of transformation can be detected by constructing frequency distributions. This is most readily done by graphical analysis on the computer, but may also be done with small data sets by constructing cumulative frequency plots on normal probability graph paper. Such plots will approximate to straight lines, with slopes proportional to the standard deviation if the data are normally distributed, or if the transformation is appropriate.

3.5.3 Internal data check mechanisms

Mechanisms for checking the quality of data require links between sampling and AQC (which is achieved through rigorous programme design) and appropriate data checking mechanisms. For purposely built systems, the links can be built into the GUI or Spreadsheet or Graphics system. For example, the implementation of field and laboratory blank (sterile) and duplicate samples, together with participation in inter-laboratory AQC programmes, should ensure the numerical integrity of the data acquired, provided appropriate dilutions are employed. Thus, data reporting unexpected low or high values should provide a true representation of indicator bacterial concentration or physicochemical parameters. However, the entry of the data to the database offers a further opportunity for automated data checking, principally in the form identifying outliers in the data set. The definition of an outlier requires a historical data set. Identification of a specific data item as an outlier value is based on a knowledge of the statistical distribution of the environmental determinand. In the case of microbiological data, it is generally accepted that environmental data measured at recreational waters follow a $\log_{10}$-normal probability density function. Thus an outlier could be defined as a data item where the $\log_{10}$ value was greater than, for example, two standard deviations from the historical $\log_{10}$ mean value.

This above approach is perhaps the most scientifically valid, although, it presupposes historical data and has a significant problem which derives from the choice of a cut-off at which the data item is considered an outlier. It is certainly the case that microbiological concentrations in recreational waters can commonly increase by several orders of magnitude (i.e. 3-4) following rainfall events. Thus, the definition of a numerically low cut-point tied to an automatic data cleaning system designed into a GUI and operated by clerical staff with little scientific insight into the acquired data, might result in very significant data loss from precisely the most high-risk periods against which the system was seeking to provide protection. For this reason, it is essential that data cleaning is closely supervised by competent scientific staff and that any automated systems simply define the items on which a scientific decision is required, rather than being allowed to carry out any automated change to the raw data matrix by deletion of apparent outlier values.

3.5.4 Determining compliance with a standard

Many monitoring programmes are designed to ascertain compliance with a standard or other objective for water quality. The standard should be carefully checked to determine the method that will be needed to assess compliance. Any deviation from the specified method will lead to doubts about comparability of the conclusions between parallel programmes in different regions, nationally or internationally. The compliance method will have three basic components: the design of the sampling scheme, (including the number of samples, their frequency and the period of sampling); the description of water quality (such as the chemical, physical or microbiological variable), the units of measurement and the descriptive statistics used to describe the level attained in the set of observations (such as the range of values, the average, median, or geometric mean, the standard deviation, or a given percentile value); and the criterion for judging passing or failing of the standard, or for classifying the quality at the water recreation area.

Water quality at a particular site is, essentially, a continuous population of measurements of quality, from which a sampling programme can only provide a limited number of discrete measurements. Because of the errors of measurement that occur when a small number of samples are taken, the sampling programme only gives an approximate estimate of the conditions existing over the whole period of the programme. These errors may, or may not, be regarded as important, although they will have most significance when assessing compliance of waters which are of borderline quality in terms of the standard. Once this problem is recognised, it is important to consider the burden of proof required to assess compliance. It is most usual to take results at their face value, without taking account of sampling error. This is justifiable, but is an empirical decision which will always carry a risk of misclassifying waters, particularly if small numbers of samples are taken. Alternatively, allowance may be made for sampling errors by adopting a “benefit of the doubt” or a fail-safe approach, so that when water quality is borderline the risk of incorrectly failing or passing the standard, respectively, is acceptably small and defined in statistical terms. A general description of these problems in the design of water quality monitoring programmes has been given by Ellis (1989).

Most standards specify a limiting value, which may be either a measure of central tendency, such as a mean or median value obtained over a specified period, or an upper limit which must not be exceeded. An upper limit may be absolute, or it may allow for the natural variability that occurs with time and which is caused by such factors as weather and tidal conditions. Such allowance can be made by defining an upper confidence limit, defined statistically by the observed variability or by the percentage of samples (percentile) which must not exceed the upper limit.

3.5.5 Data presentation

In all except the smallest surveys, the study director and data handlers will wish to use electronic computation for speed and accuracy. They should, however, use reliable and proven statistical packages and assure themselves that the calculation routines give the correct output. Manual calculation may be the only method available to small teams, handling small data sets and lacking computation facilities, or for field use. It can also be invaluable as a means of checking the data as they are produced, or for checking the output from computation. Desk calculators that provide a printed output of the operations and calculations are recommended because the printout provides a means of checking errors of input.

Graphical methods can be used manually when the data set is small. Linear probability graph paper, which has ordinates ruled in equal divisions and abscissae ruled proportionally to the percentage points of the standard normal density function, can be used to check that the data (transformed if necessary) conform approximately to the normal distribution. If so, the graph paper can then be used to estimate the median, other percentile points and the standard deviation of a set of data, with two-figure accuracy.

Much data processing is associated with assessments of microbiological quality and this is described in Chapter 8. Water quality data points may be plotted sequentially on a chart to indicate changes of quality with date of sampling. The occurrence of high values may be used to initiate investigation. In these circumstances the plot becomes a control chart. The choice of a limit value and appropriate action, for when the limit is exceeded, will be chosen to suit the need. The limit value can be set to coincide with the value given in a standard. More conventionally, two upper limit values are set on a control chart: at the mean plus twice and at the mean plus three times the sample standard deviation. These represent values that would be expected to be exceeded only once in 20 or 100 samples and which indicate, respectively, a warning and the need for remedial action.

Use of a computer is obligatory for analysing large sets of data. Even when it is possible to use a hand-held calculator or even to use graphical methods, the computer is able to produce results free from error and, in many cases, to reduce the total time expended in data analysis. There are many statistical packages available that can take raw data from a spreadsheet and subject them to statistical analyses and in many cases use them to produce excellent graphics. Such packages are available to suit different needs. For general use, typical packages that are commercially available are MINITAB Statistical Software, SAS (Statistical Analysis System), SPSS (Statistical Package for the Social Sciences) and BMD (Bio-Medical Data programs). Specialised computer programs include GENSTAT (a comprehensive collection of statistical programs available from Numerical Algorithms Group Ltd) and EpiInfo (public domain software for epidemiological investigations). Each of these has graphical capabilities adequate for most users, although they can also produce output appropriate for more sophisticated and internationally available graphics and publishing software.

3.5.6 Data interpretation and communication

Interpretation of data and communication of results are the final two steps in an assessment programme. Correctly interpreted data will not be of much use if they are not disseminated to relevant authorities, to scientists and the public, in a form that is readily understandable by, and acceptable to, the target audience. The form and level of data presentation is, therefore, crucial. Often, it is advisable to produce two types of reports: a comprehensive, detailed report containing all relevant data and associated interpretation and an executive summary (in an illustrated and simple form) which highlights the major findings. Usually, the interpretation of data is undertaken by specialised professionals, such as the relevant scientists, (e.g. microbiologists, chemists or epidemiologists) the data treatment team and professionals from other organisations such as environmental protection agencies, health authorities, national resource agencies. As a courtesy, the results and recommendations should be discussed with all interested groups and individuals before reports are formally released. A contingency plan should also be developed with the assistance of all those with a vested interest to investigate and respond to cases of adverse health effects or to any unforeseen event or conditions that could lead to a deterioration in water quality and possibly increase the risk of illness to bathers.

A very important part of any sampling exercise is to review the extent to which the desired objectives have been achieved. There are many reasons for periodic adjustments to any assessment or monitoring programme for recreational water quality. The initial objectives may have been achieved and the programme may need reorientating from a baseline study to routine monitoring, with the establishment of new objectives and possibly the addition of supplementary monitoring variables or the substitution of existing activities with new activities. Once the samples have been taken, contemporary information is available on distributions and variability. If the required precision has not been achieved, these new data may be used to establish how many extra samples are required, and how the sampling strategy may be further optimised. If necessary, any extra sampling may then be carried out immediately.

3.6 Elements of good practice

The logistical planning of any monitoring programme or study should take account of socio-economic, technical or scientific and institutional capacities, staffing, equipment availability, consumable demands, travel and safety requirements and sample numbers, without compromising achievement of the objectives or scientific validity of the programme or study.

- The hierarchy of authority, responsibility and actions within a programme or study should be defined. All persons taking part in the programme or study should be aware of their roles and interrelationships.

- Staff should be trained adequately and be appropriately qualified, including in respect of health and safety aspects.

- Collection of data and information should use the most effective combination of methods of investigation, including observation, water quality sampling and analysis, interview of appropriate persons and review of published and unpublished literature.

- Frequency and timing of sampling and selection of sampling sites should reflect beach types, use types and density of use, as well as temporal and spatial variations in the recreational water-use area that may arise from seasonality, tidal cycles, rainfall, discharge and abstraction patterns, beach types and usage.

- Sampling should provide a data set amenable to statistical analysis.

• Data handling and interpretation of results should be done objectively without personal or political interference.

• The need for transformation of raw data, before analysis, to meet the conditions for statistical analysis should be agreed with a statistical expert before commencing analysis.

• Data handlers and collectors should agree on a common format for recording results of analyses and surveys and should be aware of the ultimate size of the data matrix. Forms and survey instruments should be compatible with this format. Likewise, data handlers should agree on a format for the output of results with those responsible for interpreting and presenting the data.

• Procedures for dealing with inconsistencies, such as omissions in records, indeterminate results (e.g. indecipherable characters, results outside the limits of the analytical methods) and obvious errors should be agreed in advance of data collection. On receipt from the data collectors, record forms should be examined and the agreed procedure followed. Discrepancies should be referred immediately to the data collector for correction or amendment. Where re-sampling is impossible, estimates are preferable to leaving gaps in the data record, (estimates should always be recorded as such) although they will reduce the statistical degrees of freedom.

• Ideally, arrangements should be made to store data in more than one location and format, to avoid the hazards of loss and obsolescence. Data should be transcribed accurately, handled appropriately and analysed to prevent errors and bias in the reporting.

• The statistical routine should be selected by a statistical expert.

• Data should be handled and stored in such a way to ensure that the results are available in the future for further study and for assessing temporal trends.

• Data should be interpreted and assessed by experts with relevant recommendations for management actions prior to submission to decision makers. Interpretations should always refer to the objectives and should also propose improvements, including simplifications, in the monitoring activities - stressing the needs for future research and guidelines for environmental planning.

• Interpretation of results should take account of all available sources of information, including those derived from inventory, catalogue of basic characteristics, sanitary and hazard inspection, water quality sampling and analysis, and interview, including any historical records available.

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Chapter 4*: QUALITY ASSURANCE

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Quality Assurance (QA) is a management method that is defined as “all those planned and systematic actions needed to provide adequate confidence that a product, service or result will satisfy given requirements for quality and be fit for use”. A Quality Assurance programme is defined as “the sum total of the activities aimed at achieving that required standard” (ISO, 1994).

Any monitoring programme or assessment must aim to produce information that is accurate, reliable and adequate for the intended purpose. This means that a clear idea of the type and specifications of the information sought must be known before the project starts, i.e. there must be a data quality objective. Data quality objectives are qualitative and quantitative specifications that are used to design the system that will limit the uncertainty to an acceptable level within the constraints allowed. These objectives are often set by the end users of the data (usually those funding the project) in conjunction with the technical experts concerned.

Quality Assurance for a recreational water monitoring programme will, apart from helping to ensure that the results obtained are correct, increase the confidence of funding bodies and the public. Quality Assurance extends to all aspects of data collection from sanitary surveys to laboratory procedures. Unless the data can be checked they should not be included in any assessment; unconfirmed observations have little value and can result in misclassification.

4.1 Components of Quality Assurance

The components of a QA programme are often grouped into three levels, variously labelled: the strategic or organisational level (dealing with the quality policy, objectives and management and usually produced as the Quality Manual); the tactical or functional level (dealing with general practices such as training, facilities, operation of QA); and the operational level (dealing with the Standard Operating Procedures (SOPs) worksheets and other aspects of day to day operations).

4.1.1 Setting up the system

There is no single method for establishing a QA system. Each organisation has its own problems that will require special consideration and planning. However, once the decision to implement a QA system has been taken and the necessary funds and facilities have been made available, then a plan must be drawn up. For a new project the QA system can be drawn up before the start but if the project is already established then a QA system can be retrofitted. In the latter situation, existing practices must be evaluated with respect to QA needs and any QA checks and procedures that are already in place. It is better to build on procedures already in place and only to remove them if they are clearly unsatisfactory. If too many changes are imposed too quickly, especially where they are seen to increase work load, they are unlikely to be met with a favourable response and implementation will be poor. The QA programme must be seen to be practical and realistic and not to include trivial or unnecessarily time-consuming or difficult tasks (WHO/UNEP/VKI, 1997).

4.1.2 The Quality Manual

The Quality Manual is composed of the management documents needed to implement the QA programme and includes (ISO, 1990):

- A quality policy statement, including objectives and commitments.

- The organisation and management structure of the project, its place in any parent organisation and relevant organisational charts.

- The relationship between management, technical operations, support services and the quality system.

- Procedures for control and maintenance of documentation.

- Job descriptions for key staff and reference to the job descriptions of other staff.

- Identification of approved signatories.

- Procedures for ensuring traceability of all paperwork, data and reports.

- The laboratory's scope for calibrations and tests.

- Arrangements for ensuring that all new projects are reviewed to ensure that there are adequate resources to manage them properly.

- Reference to the calibration, verification and testing procedures used.

- Procedures for handling calibration and test items.

- Reference to the major equipment and reference measurement standards used.

- Reference to procedures for calibration, verification and maintenance of equipment.

- Reference to verification practices including inter-laboratory comparisons, proficiency testing programmes, use of reference materials and internal quality control schemes.

- Procedures to be followed for feedback and corrective actions whenever testing discrepancies or departure from documented procedures are detected.

- Procedures to be followed for feedback and corrective actions whenever testing discrepancies or departure from documented procedures are detected.

- Complaints procedure.

- Procedures for protecting confidentiality and property rights.

- Procedures for audit and review.

4.1.3 Training

The development of the programme must include all staff. Typically, the management commit resources, establish policy and standards, approve plans, assign responsibilities and maintain accountability. The supervisory staff take responsibility for the development and implementation of the programme and operating personnel provide technical expertise and advice. At all stages, the operating personnel must be consulted about the practicalities of any proposed changes. In turn, they must notify management of any problems or changes that may affect the programme.

4.1.4 Standard Operating Procedures

Standard Operating Procedures (SOPs) are the documents detailing all specific operations and methods, including sampling, transportation, analysis, use of and calibration of equipment, production of reports and interpretation of data. They are the internal reference manual for the particular procedure and should detail every relevant step. Anybody of the appropriate training level should be able to follow the SOP. They should, where necessary, cross-reference other SOPs and refer to them by number. Method SOPs may originate from organisations such as the International Organization for Standardization (ISO), British Standards Institute (BSI), American Standard Technical Method (ASTM) or from the instructions that come with the test kit where a commercially produced method is used. Such SOPs have the advantage of not requiring verification and save time in writing “in-house” SOPs. However, if they are used they must be used without modification. If any modification at all takes place, the alterations must be documented. Sometimes “in house” methods are preferred, and it is vital that such methods are properly verified. This may be done by reference to scientific literature and by “in house” validation.

The procedure should be written in short, clear sentences. Equipment SOPs should include methods and frequency of maintenance, cleaning, calibration and servicing. Method SOPs should include all the information necessary to carry out the procedure without reference to other documents with the exception of fully documented SOPs. Any statements regarding ranges for measurement variables such as temperature, weights, etc. should be within the scope of the facility, i.e. not so wide that they affect the result but not so narrow that they are not practically achievable or necessary. Calculations should include any equations and demonstration of statistical control. Where applicable, criteria for the acceptance of data should be stated and acceptable ranges quoted. Disposal methods for reagents, test materials and other consumables should also be stated.

Some SOPs, such as those for office procedures, will be customised. The person most technically competent to carry out the procedure described should write the SOP. An SOP should have a descriptive title and also have a unique reference and version number. The purpose of the SOP should be stated alongside the variables measured, the expected range of values, the limitations of the method and the expected precision and accuracy. Any documents regarding the source of the method should be stated. Safety notes should include any foreseeable risks involved in the procedure, alongside procedures to minimise risk and procedures in case of an accident. Any special training required for the operator, and special apparatus required for the procedure (including all reagents and materials required) should be stated along with such information as the grade, reference number, size and company of origin. The storage, handling, recording and subsequent disposal of the sample should also be covered in the SOP, including storage temperatures, sample splitting, traceability, and any other issues. The style and format of the final data report should be given where applicable and reporting procedures and archiving requirements should also be included.

4.1.5 The Quality Assurance manager

For larger projects, proper management of QA will require the appointment of a QA manager to liaise with staff, to manage data archives, to conduct regular audits and reviews and to report on any QA issues. The manager is responsible for inspecting all aspects of the system regularly to ensure compliance, for reporting on such inspections and audits to management and for recommending improvements. These activities involve inspecting facilities and procedures regularly, tracing samples and documents back through the system and ensuring that all appropriate records have been kept.

Where QA is the responsibility of a separate section within an organisation many of the management difficulties are minimised. Appointment of a full time QA manager is difficult in a small organisation and in these cases the responsibility for QA should be assigned on a part-time basis, to a suitable member of staff.

4.1.6 Auditing and checking compliance

When all the documentation for the QA system is in place, it should be piloted. During this time, the QA manager should conduct a series of audits covering all aspects of the system. Traceability of data is a key component which can be checked by picking data at random and tracing them back through all relevant paperwork to the sampling procedure. A review of the system with positive and negative areas clearly defined should be written at the end of the pilot phase.

One method of implementation is to apply for accreditation from a recognised QA system. The ISO standard, ISO 9000, is suitable for the monitoring programme as a whole and is available in many countries. These systems are expensive but do allow the QA programme to be assessed independently against an agreed standard. Sometimes formal accreditation is required by regulatory and commercial bodies.

4.1.7 Maintaining Quality Assurance

In order to maintain the QA system, it is necessary to check periodically each area of the system for compliance. This involves auditing the component parts to assess whether they continue to meet the original criteria. This procedure should be formerly documented. Reports on all audits should be made available to management and to the persons responsible for the work concerned. Deviations from required standards must be corrected as soon as possible. The audit must be independent, and should be thorough and unannounced.

4.2 Equipment maintenance and calibration

All equipment, whether site, office or laboratory, must be maintained on a regular basis as documented in the relevant SOPs, codes of practice and manufacturer's guidelines. Laboratories must apply standards within the limits established for the care of a particular piece of equipment. This applies to general equipment, such as glassware, as well as to sophisticated analytical instruments and vehicles. It especially applies to field equipment.

The care and cleaning of equipment is very important to ensure analytical quality. Regular internal and external calibration checks must be performed on equipment such as balances, pipettes and pH meters. The frequency of these checks depends on the stability of the equipment in question but should be based on established practice. The form and frequency of these checks should be documented in the relevant SOPs. Calibration and maintenance records should be kept for all equipment, thus allowing the repair status to be monitored.

4.3 Sampling

Any analysis can only be as good as the sample taken. Variations in sampling procedures can have a marked effect on the results of analysis. It is very difficult to quantify these effects and therefore procedures for sampling operations should be documented carefully so that all relevant information is recorded at the time of sampling by the field worker.

4.3.1 The sampling plan

For any sampling programme, a sampling plan must be prepared to allow full control of the sampling process so that any change seen between two sampling rounds can be attributed to changes in environmental conditions and not to changes in procedure. Items to be considered in preparing a sampling plan include planning issues, fieldwork procedures and field safety issues.

Planning issues

Planning issues include identification of the objective of sampling (e.g. to test compliance with a bathing water regulation), choice of site (location, type of water body), the type and number of samples to be collected (sample types, e.g. water, sediment, the number of samples, appropriate equipment) and timing of sampling (considering the state of tides).

Fieldwork procedures

Consideration must be given to sampling SOPs (for equipment, sampling method, storage, etc.), as well as size of sample and sample containers. Preservation must be decided in consultation with staff from the analysing laboratory, who will advise clients on the volume and type of sample and who will usually provide sampling containers and preservatives where necessary. Ensuring field quality control includes the use of blanks, duplicate samples, replicate samples and spiked samples. Storage and holding time (conditions for storage, such as in an ice box, maximum time before analysis for unstable parameters, etc.) must also be considered.

The laboratory staff must be made aware when samples are due to arrive so that they can make the appropriate arrangements. When choosing an analytical laboratory it is important to be aware of the location of the laboratory in relation to the sampling site, as well as the latest time of day that they are prepared to accept samples.

Other factors include deciding where to carry out analysis, i.e. in the laboratory or on site. Some analyses may be better performed on site, such as dissolved oxygen measurements, calibration of field measuring equipment, flow pumps and thermometers, etc. and sample treatments such as filtration. Some samples need to be split or subsampled. Where this is done, great care needs to be taken because samples are frequently very variable. Contingency plans need to be prepared for situations such as bad weather and vehicle breakdown. Field sampling sheets also need to be prepared. These can be filled in manually on paper forms or on a portable computer providing that the software has been properly validated. When designing field report forms it is important that the place, time and date of sampling, sampling conditions, any field measured variables, equipment used (with an inventory number), any necessary sample preparation and the name of the operator are included in the form. Practical difficulties, such as how many samples the field worker will need to carry, parking and access to the site also need to be considered.

Field safety issues

Field safety can have a bearing on the quality of data generated where field operators may be inclined to use a less than optimum procedure in order to protect themselves. This must be taken into account when writing the sampling procedure. For example, insisting on sampling water at chest height may deter some operators if the conditions in the water to be sampled are rough. Sampling from boats can be especially hazardous in rough weather. Even the 30 cm depth stipulation of the European Union's Bathing Water Directive can be difficult to comply with. When devising a plan, areas of risk may have to be borne in mind, including water depth and sampling conditions, currents, wildlife, traffic and weather. Staff must always be provided with the appropriate protective equipment and SOPs should be developed with the safety of operators of paramount concern.

4.3.2 Field quality assurance

In spite of the difficulties involved in site work, QA is critical at this point. If a good, practical, field QA programme is put into operation, confidence in the data collected should be ensured (WHO/UNEP/VKI, 1997). All equipment must be kept clean and in good order, and records should be kept of all maintenance and of any irregularities that may affect the results. Conditions in the working area should not expose the operator to undue risk of any type.

Standardised and approved methodologies must be used at all times. If a method proves unworkable on site, then an alternative must be found quickly and agreed by all those involved. Operators must not change procedures without referral to the management procedure. Where unavoidable changes are made, for example, in bad weather, they must be fully documented. Nevertheless, a good sampling plan should make provisions for bad weather.

Prevention of sample contamination and losses

It is important that samples are protected from contamination and deterioration before their arrival in the laboratory. This can be ensured by using only recommended sample containers. Where reusable containers are used, it is essential that they have been cleaned properly and, if necessary, sterilised before use. Containers that have been sterilised must remain sterile until the sample is collected. The inner portion of the sample container should not be touched by the operator. If the seal on the bottle is broken (in the case of a commercially purchased microbiological sample bottle), or if the protective paper or foil has been lost from the top (home-made sampling containers), the bottle should be discarded.

Recommended preservation methods must be used. Where this involves chemical preservatives, the chemicals must be of analytical grade, and provided and tested for efficacy by the analytical laboratory.

Field measurements, such as pH and temperature, must be made on a separate subsample which is then discarded in order not to contaminate samples for interlaboratory analysis. Conductivity measurements should not be made with a sample that has been used previously for measuring pH, because potassium chloride from the pH probe may affect the conductivity reading.

All sample containers should be kept in a clean environment, away from dust, dirt and fumes. Petroleum products and fumes may contaminate samples with heavy metals and hydrocarbons. This can be a major problem on boats, where leaks and seepage of petroleum products are common. Samples must be stored in a cool box or portable refrigerator and transported to the laboratory as soon as possible. Cool boxes are more efficient if they contain some water.

Field Quality Control

Quality Control (QC) is an essential part of the field QA programme. It requires the collection of replicate samples to check the repeatability of sampling (see section 4.5.1), and the submission of field blanks and duplicates to check for contamination, handling and storage problems and other errors that may affect the results from the time of sampling to the time of analysis. The timing and frequency of these samples should be documented in the sampling plan.

4.4 Laboratory facilities

Except for any on-site analysis, analysis is usually performed in a laboratory. It is essential that any facilities are adequately equipped to deal with the analyses required and are convenient for the delivery of samples. This should have been ascertained before the start of the monitoring programme (see Chapter 2).

Small-scale organisations responsible for monitoring may find it more convenient to use outside facilities for analysis and sometimes for sampling. In these cases, the use of a laboratory belonging to an accreditation scheme is advisable and, moreover the laboratory should be inspected for compliance by an experienced member of the monitoring programme. An inspection should take into account the following features (ISO, 1984):

- Lines of communication between staff and management.
- Staff training and qualifications.
- Resources.
- Equipment maintenance and calibration.
- Standard Operating Procedures.
- Traceability of results.
- Sample handling and storage.

Where in-house facilities are used, it is essential that the monitoring work does not overload the laboratory. Resources (staff, space, equipment and supplies) must be sufficient for the planned workload. The laboratory must be well managed and must conform to all relevant health and safety guidelines. All analyses performed must be within the remit and expertise of the facility and SOPs must be in operation for all analyses (see Chapter 2).

4.4.1 Sample receipt and storage

Procedures for sample handling, transport and storage prior to analysis should ensure that the quality of the sample is not compromised. The condition of each sample and its storage location should be recorded along with its proposed analyses. If the sample is split, this must also be recorded. All samples must be identified uniquely with a number or code. It is important to ensure that the passage of a sample, and any associated paperwork, through the laboratory is fully documented and, therefore, traceable.

4.4.2 Reporting

The efforts of QA are directed ultimately towards ensuring that any data produced are suitable for their intended use; this applies to the results and any interpretations. The first stage in the reporting process is the examination of the results to see if they are fit to report (although raw data should have been checked prior to this stage). Results must be reported accurately and in a way that aids interpretation. To facilitate this, information may need to be included that has a bearing on interpretation, such as sampling conditions or the method of analysis. All data included must be checked by an experienced analyst with reference to site reports, calibration and QC data. Many laboratories have a system which requires the checking and countersigning of analytical reports (usually by the laboratory manager) to act as a safeguard against erroneous or misleading data leaving the laboratory. This type of system is only effective when conscientiously applied.

4.5 Analytical Quality Control

Analytical Quality Control consists of two elements: internal quality control (IQC) and external quality control (EQC). External quality control or inter-laboratory control is carried out periodically and checked by the laboratory responsible for the monitoring system. Internal quality control consists of the operational techniques used by the laboratory staff for continuous assessment of the quality of the results of individual analytical procedures. The focus is principally on monitoring precision, although accuracy is not ignored. It is necessarily part of the wider QA programme, but differs from it by the emphasis placed on quantifying precision and accuracy. Whereas QA strives to achieve quality by regulating procedures using management techniques, IQC focuses on the individual method and tests its performance against mathematically-derived quality criteria.

4.5.1 Internal quality control in the chemical laboratory

Internal quality control within the chemical laboratory comprises a variety of activities, some of which are described below (Briggs, 1996).

Choice of analytical method

A variety of different analytical methods are usually available for determining the concentration of any variable in a water sample. The choice of method is critical for ensuring that the results of the analysis meet the laboratory's requirements, because different methods have different precisions and sensitivities and are subject to different potential interferences. Consideration must be given to these parameters before a method is chosen. A number of standard methods are available for most of the analytical determinations involved in water quality monitoring, and in some cases the method is named in the regulations. These standard methods frequently include extensive validation data that allow the method to be evaluated easily. In addition, many methods are sanctioned by appropriate international or national organisations. It is important that any method selected meets the individual programme requirements. The performance of a method can be affected unpredictably by many factors.

Before any analytical method is put into routine use it is essential that it is properly validated. A minimum programme of validation includes a number of elements. One of these elements is the determination of linearity - the calibration point should be determined and if possible a linear response curve should be demonstrated. In addition, the limit of detection (the lowest concentration of the variable that can be distinguished from zero with 95 per cent confidence) should be determined. Within-and between-day coefficients of variation should be performed at three concentration levels to determine precision. Analysis of reference materials with known concentrations of the variable, or comparison analysis with existing methods in other laboratories, should be performed where possible.

Validity checking

After a method has been validated, found to be suitable and introduced into routine use in the laboratory, it is necessary to ensure that it continues to produce satisfactory results. Validity checks should be made on every batch of samples, or at frequent, regular intervals if batches are large or if testing is continuous. Validity checking is an extension of the checks carried out before the method was selected and is intended to confirm regularly the conclusions reached at that time.

Calibration check

If a calibration curve is being used, standard solutions should be analysed from time to time within the required range of concentration. The ideal calibration curve is linear within its most useful range, with a regression coefficient of 0.99 or greater. The response of the measuring equipment to the concentration of the variable in a standard solution (in terms of absorbance or some other parameter) should be recorded when it is expected that this parameter will be comparable from assay to assay. In addition, the deviation of individual calibration points from the line of best fit can be used to assess the precision of the calibration, which should be within the mean precision limits for the method.

Use of blanks

Method blanks and, where possible, field blanks should be analysed with each batch of samples. A method blank consists of reagent water, usually double-distilled water. A field blank is reagent water that has been bottled in the laboratory, shipped with sample bottles to the sampling site, processed and preserved as a routine sample and returned with the routine samples to the laboratory for analysis. The analysis of a blank should not yield a value higher than that allowed by the acceptance criteria. This procedure checks interference and the limit of detection of the assay.

Recovery checking

A specimen spiked with a known amount of the variable should be tested in each batch and the closeness of fit to the expected value calculated. In most cases this procedure provides a check on accuracy but, in assays where a variable is extracted from the original matrix (such as in many sample cleanup procedures used prior to chromatographic analysis), it can be used to monitor the extraction step. It is important that the matrix of the spiked specimen matches the real sample matrix as closely as possible. Many laboratories use real samples with low natural values of the variable for this purpose, spiking them with known amounts of the variable and including both the spiked and natural samples in the same assay batch.

Precision and accuracy checks

Precision and accuracy checks are an extension of the validity checking described above. These checks allow the quality of the assay to be monitored over time using techniques such as control charting. The validity checks described above only allow acceptance or rejection of the assay data. Precision and accuracy checking should allow slow deterioration of data quality to be identified and corrected before data have to be rejected. This results in increased efficiency and reduced costs for the laboratory.

Control by duplicate analysis

Use of duplicate analysis as a method of precision checking has two distinct advantages: quality control materials are matrix-matched and the materials are readily available at no extra cost. Because the samples are analysed using the same method, equipment and reagents, the same bias will affect all results. Consequently, duplicate analyses are only useful for checking precision; they provide no indication of the accuracy of the analyses. Results from duplicate analyses can be used to calculate a relative range value, R, by using the equation:

$$R = \frac{(X1 - X2)}{(X1 + X2)/2}$$

where *X1* and *X2* are the duplicate results from an individual sample and *X1-X2* is the absolute difference between *X1* and *X2.* These values are then compared with the mean relative range values previously calculated for the assay during validation. The simplest method of assessment is to use the Upper Concentration Limit (UCL), where UCL = 3.27 × mean R value. When any value is greater than the UCL, the analytical procedure is out of control. This method, although statistically valid, provides no indication of deteriorating precision.

Precision control using pooled reference material

A more sophisticated approach is to use acceptance criteria based on warning and action limits. This method has the advantage of providing some monitoring of accuracy but is a viable control only if the material to be used will be stable in storage for sufficient time. The reference material is normally prepared by taking previously analysed samples with known concentrations of the variable under investigation, mixing them and aliquoting the resultant pool. The aliquots are then stored in readiness for analysis. A small sample of the aliquots is analysed to determine the mean concentration of the variable, and the standard deviation and the coefficient of variance at that concentration level. Data may be used only if they come from analysis that are in control. This approach requires that the new pool materials must be prepared before the old ones are finished.

A typical precision control exercise would involve the analysis of four aliquots from each pool in each of five assays, thus obtaining 20 results. The material from the pool should be analysed at several different times with different batches, because between batch variance is always slightly greater than within batch variance. Once 20 or more analyses have been made on this pool of material, the mean and standard deviations of the results are calculated. Any result that is more than three standard deviations from the mean is discarded and both of the statistics are recalculated. The mean is the “target” value and ideally, will be a close approximation of the true concentration of the variable in the reference material. The mean and standard deviation become the basis of the acceptance criteria for the assay method and may be used to draw up control charts.

At least three separate reference materials with different mean values of variable concentration should be in use at any one time in order to provide control of the analytical method across a range of concentrations. If precision is checked at only one concentration of the variable, it is impossible to detect whether precision is deteriorating at other concentrations. Use of several reference materials also allows their preparation to be staggered so that they become exhausted at different times. This assures greater continuity of control, because two or more old pools will still be in use during the first few assays of a new reference material.

Although the monitoring of accuracy by assessing deviation from the reference material mean (target value) is possible, care must be taken because the target value is only an approximation of the true value. As reference materials become exhausted and new ones are made, there will be a slow deterioration in accuracy. Accuracy can be safeguarded by regular participation in EQC exercises (see section 4.5.5) and by the use of certified reference materials.

Certified reference materials

Certified reference materials (CRMs) are matrix-matched materials with assigned target values and ranges for each variable, reliably determined from data produced by repeated analysis. Target and range values may be generated from data produced by several laboratories using different analytical methods or calculated from data obtained by the use of one analytical method (usually a reference method). Consequently, there may be bias in the target value. The target values assigned to each variable in the matrix in certified reference materials are generally very close to the true value. For some variables, however, there is an appreciable difference in bias between different analytical methods and this may lead to wide assigned ranges. When a laboratory is not using one of the reference methods the “all method” range may be so wide that it is practically meaningless. Certified reference materials are also only practical for variables that are stable in long-term storage.

Certified reference materials are prepared and checked under carefully controlled conditions and, as a result, they are costly to produce, correspondingly expensive to purchase and they may be difficult to obtain in some countries. Some authorities advocate the routine use of CRMs as precision control materials, but it is more cost effective to use them for the periodic checking of accuracy, in combination with a rigorous IQC programme.

Use of control charts

The principle of control charts is that IQC data can be graphically plotted so that they can be readily used and interpreted. Consequently, a control chart must be easy to use, easy to understand and easy to act upon. The Shewhart chart is the most widely used control chart (Shewhart, 1986). It is a graph with time (or assay batch) on the x-axis and the concentration of the variable in the reference material on the y-axis. Target, warning and action lines are marked parallel to the x-axis. Data obtained from precision control using reference materials (as described above) are usually plotted on a Shewhart chart. In this application, the target line is at the mean concentration of the variable for that specific pool of material and warning lines are placed at two standard deviations to either side of the target line. Provided the distribution is normal, 95 per cent of results from assays in control will fall between the two warning lines. Action lines are normally placed at three standard deviations to either side of the target line and 99 per cent of normally distributed results should be between the action lines.

In the regular use of a Shewhart chart, an aliquot from an appropriate reference material is analysed with every batch of samples and the measured concentration of the variable in the aliquot is plotted on the chart. Normally, no more than 1 in 20 consecutive results should fall outside the warning lines. If this frequency is exceeded, or if a result falls outside the action lines, the method is out of control.

The scatter of the assay results for the reference material around the target line provides an indication of the precision of the method, while the mean of the assay results relative to the target value indicates whether there is any bias (consistent deviation) in the results. If the analysis on one or more of the control specimens yields a result that it is outside the warning or action lines on the chart, the following action should be taken:

- A single result outside the warning lines should lead to careful review of data from that analytical batch and two or three subsequent batches.

- Results outside the warning lines more frequently than once every 20 consecutive analyses of control specimens should prompt detailed checking of the analytical method and rejection of the assay data.

- A result outside the action limits should prompt detailed checking of the analytical method and rejection of the assay data.

4.5.2 Internal quality control in the microbiology laboratory

Internal quality control in microbiology laboratories poses special problems of reproducibility due to the naturally wide variation in the number of organisms found between subsamples (see Chapter 8). Apart from method and field blanks (where the method blank should be sterile distilled water and the field blank should be a natural sample either guaranteed free of the test organisms or sterilised natural water), control samples should be analysed which are known to contain appropriate numbers of the micro-organisms that are normally sought. It is possible to purchase sets of freeze dried wild-type bacterial reference cultures for quality control and accreditation requirements. These cultures should be reconstituted and diluted with quarter strength Ringer's solution to give a suitable number of organisms similar to that which would normally be seen in the natural samples. These cultures are expensive and therefore it is not feasible to use a new culture for every batch of samples or media. However, frequent subculture of reference strains is to be discouraged due to problems with contamination and mutation. This is a special problem with coliphage analysis, where mutation of the host species can prevent the detection of viral plaques. This problem can be solved by freezing down the cultures in glycerol broth and either storing in liquid nitrogen or a -70 °C freezer or, more conveniently, on commercially available plastic storage beads and freezing at -20 °C. Alternatively, some media companies supply standardised cultures in an easy to use form. These cultures may be qualitative or quantitative and have the advantage of eliminating the trial and error diluting of suspensions to achieve the desired count.

Shewhart charts can be used in water microbiology despite the problems of natural random variation. However, this means that wide control limits are necessary. For example, if the count reported for the first half of a duplicate sample is 11, then the 95 per cent confidence interval (CI) for the count of the second sample will be 3-23. Tables giving the CIs of counts are available in reference works on water analyses (HMSO, 1994). The microbiology laboratory should carry out several duplicate analyses regularly and plot the results on a control chart. Each half should be treated as a separate sample and analysed routinely. They should be inserted in the sample run in a random fashion without the knowledge of the analyst (if at all possible) and all results should be read by the same person. The first count should be recorded on the control sheet, and the corresponding CI for the second count entered. The second count is then recorded along these figures. If this count falls outside the CI, this fact should be highlighted. If a Shewhart chart is made up of these results, then any trend can be identified. If, over a period of time, the second count falls outside the CI for more than 95 per cent of the time, the reason should be investigated. As with the use of duplicate samples in chemistry, this approach keeps a check on precision and not on accuracy.

In addition to blanks and a manufactured control, the use of a known wild positive can be included. This can be chosen from the last batch of samples run. However, there can be problems with this, due to alteration in bacterial numbers over time and storage.

All prepared media should be checked for performance and sterility and identified by batch reference number. Both negative and positive control strains of bacteria should be included. Manufacturers of dried media will usually recommend control strains if requested. Where a medium is meant to inhibit the growth of a particular organism, this should also be tested.

4.5.3 Summary of an internal quality control programme

A summary of the IQC programme recommended by the GEMS/Water programme is given below. This programme offers a simple but effective introduction to IQC and is described in more detail in the *GEMS/Water Operational Guide* (WHO, 1992). For each variable the following should be applied:

• For chemical variables, analyse five standard solutions at six different known concentrations covering the working range to develop a calibration curve or, when a calibration curve already exists, analyse two standard solutions at different known concentrations covering the working range to validate the existing calibration curve.

• Analyse one method blank per set of 20 samples.

• Analyse one field blank per set of samples.

• Analyse one duplicate of a sample chosen at random from each set of up to 20 samples.

• Analyse one specimen that has been spiked with a known amount of the variable as a recovery check. This specimen should have a matrix similar to those of the samples being processed.

4.5.4 Remedial action

If any of the QC procedures indicate that a method is out of control or that a problem exists, corrective action must be taken. The main checks to make are calculations and records, standard solutions, reagents, equipment and QC materials (Table 4.1).

Table 4.1 Checks to be carried out when a problem is detected with an analytical method

Problem area	Checks
Calculations and records	Check calculations for transposition of digits or arithmetic errors; confirm that results have been recorded in the proper units and that any transfer of data has been made correctly
Standard solutions	Check the standard solutions that are used for calibrating equipment; check their storage conditions and shelf-life (an old solution may have deteriorated or a new one made up incorrectly)
Reagents and media	Check for deterioration of old products; check QC records to see if new reagents performed correctly and if they were properly prepared; check their storage conditions and shelf-life
Equipment	Check calibration and maintenance records for all relevant dispensers and measuring equipment where a method is out of control; items such as automatic pipettes, balances and spectrophotometers should be checked regularly and recalibrated as necessary Ascertain that equipment is being properly used; check that any QC material has not deteriorated and is properly stored; run analyses on several aliquots to determine whether the concentration of the variable remains within the allowed deviation from the target value and close to the mean of the last 20 determinations

Source: Briggs, 1996

4.5.5 External quality control

External quality control is a way of establishing the accuracy of analytical methods and procedures by comparing the results of analyses made in one laboratory with the results obtained by others conducting the same analysis on the same material. This is usually accomplished by one laboratory sending out sets of samples, with known and unknown concentrations of variables, to all of the specified laboratories. Each participant analyses the samples for the specified variables and reports the results to the reference laboratory

(Box 4.1). The results from all participating laboratories are collated by the organisers of the EQC programme and then subjected to detailed statistical analysis. A report to each laboratory is generated, giving a target value for the reference sample or samples (usually consensus mean or median), a histogram illustrating distribution of results for each material, and an individual performance score relating the individual laboratory results to the target value. The calculations for performance indicators are often quite complex because multiple specimens have to be considered and the method variance varies with the concentration of the variable. However, the general principle of providing a method of performance comparison remains the same in all EQC exercises.

Box 4.1 External Quality Assurance: the experience of a microbiology laboratory

The Unit of Microbiology, Faculty of Medicine, University Rovira i Virgili (Reus, Spain) participates in an EQA on microbial recovery where each participating laboratory analyses an external sample and the results from all participating laboratories are analysed statistically.

A critical element of the EQA is that the test sample should be processed in an identical manner to routine samples. If an analyst is aware that an external sample is to be processed the exercise is viewed as a test of their competence and modifications may be made to routine procedures in order to enhance recovery. Analysts may also use the exercise to test potential new methodologies, but this is not the purpose of EQA and statistically unreliable results may be obtained if new methods are deliberately applied.

Although relatively expensive, properly operated EQA exercises can be of great benefit to participating laboratories because they can identify failures in internal quality control and, if undertaken over a period of time, laboratories can use them to evaluate regularly the performance of their methods. Corrective measures can then be applied whenever the methods are found to be producing poor results.

The key issue identified in participating in such EQA exercises was that they must be carried out anonymously so that the samples are dealt with in exactly the same manner as routine samples.

External quality control reports should indicate clearly whether performance is satisfactory or not. If it is not satisfactory, two general actions must be taken. First, the analysis at fault must be examined to determine the cause of poor performance. Secondly, the IQC programme that allowed the deterioration to progress unchecked must be closely examined to establish where inadequacies exist. Both must be corrected.

The general objective of EQC is to assess the accuracy of analytical results measured in participating laboratories and to improve interlaboratory comparability. Wherever possible, laboratories should participate in EQC programmes for each variable that is analysed routinely. This is only worthwhile where IQC is also part of a laboratory's normal procedures. Participation in relevant EQC programmes, and maintenance of adequate performance in those programmes, is often a requirement for laboratory accreditation.

The organisation of an EQC exercise is expensive. Large quantities of stable reference materials must be prepared, these materials must be transported to the participating laboratories, data must be analysed and detailed reports on performance must be prepared. Participating laboratories are usually charged for the service provided.

4.6 Elements of good practice

• Monitoring programmes should include appropriate QA which does not infringe on health and safety and which covers the integrity of all observation, interviews, field sampling and water quality analyses as well as data input, analysis and reporting.

• A QA manager should be appointed who audits all aspects of the operation regularly with special regard to procedures, traceability of the data and reporting.

• Essential elements of QA programmes include:

• The writing and implementation of a Quality Manual and SOPs. All SOPs should be overhauled regularly and updated as necessary, and any deficiencies should be reported and appropriate remedial action taken.

• SOPs should include maintenance and updating of inventories and catalogues; methodologies for all major equipment, all sampling and analytical procedures; sample receipt, screening and storage; and reporting.

• Σαμπλεσ σηουλδ βε ρεγιστερεδ ον αρριϖαλ ατ τηε λαβορατορψ. Τηε αππλιεδ λαβορατορψ προχεδυρεσ σηουλδ χονφορμ το τηε ΣΟΠσ δεφινεδ ατ τηε λαβορατορψ. Ωηερε ποσσιβλε, αλλ αναλψτιχαλ προχεδυρεσ σηουλδ φολλοω δεφινεδ ΙΣΟ ορ Αμεριχαν Πυβλιχ Ηεαλτη Ασσοχιατιον (ΑΠΗΑ) προτοχολσ. Αλλ εθυιπμεντ σηουλδ βε χαλιβρατεδ ρεγυλαρλψ ανδ τηε οπερατιοναλ προχεδυρεσ συβμιττεδ το θυαλιτψ χοντρολ σταφφ ιν ορδερ το γυαραντεε τραχεαβιλιτψ οφ τηε δατα.

• The programme should be evaluated periodically, as well as whenever the general situation or any particular influence on the environment is changed.

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Chapter 5*: MANAGEMENT FRAMEWORKS

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"Beach management seeks to maintain or improve a beach as a recreational resource and a means of coast protection, while providing facilities that meet the needs and aspirations of those who use the beach. It includes the framing and policing of any necessary regulations, and decisions on the design and location of any structures needed to facilitate the use and enjoyment of the beach environment" (Bird, 1996).

In most countries, a single "beach manager" who undertakes all the activities such as monitoring, planning and decision-making does not exist. These activities are generally devolved among different persons and authorities at various levels (national, federal, regional, provincial, local). In order to achieve effective management of recreational water use areas, managers should have background knowledge on a range of aspects concerned with the coastal or freshwater area, such as inflows and outflows, water quality, physical aspects of the water use area and potential health hazards. Beach managers should, therefore, be aware of the social and economic dimension embedded in their decision-making. Of importance to beach managers and decision-makers are:

- The techniques available to measure the impact of tourism.
- Notions and principles of sustainability.
- Local strategies towards sustainability.
- Sustainability indicators and issues related to water quality management.
- Water analysis and water safety.

5.1 Management concerns and approaches

The coastal zone and freshwater bodies are important areas for human habitation, industry and recreation. There are thus competing uses, not only of water for bathing, surfing, sailing, scuba diving, aquaculture and other maritime industries, but also of land use, such as for residential developments, harbours, ports, marinas and tourism industries. Offshore activities such as oil and gas exploration, disposal of sewage or radioactive waste and shipping, are also responsible for the release of contaminants into the aquatic environments. Many of the pressures mentioned above are common throughout the world and many of the threats facing the quality of recreational water bodies have arisen as a direct or indirect consequence of human activities. Such conflict of uses makes management of recreational water use areas a particular challenge.

Water bodies are often used as a repository for waste and the relatively concentrated period of tourism activity in specific areas increases the environmental pressures on such water bodies. Recreational use of water and tourism activities depend highly on the quality of the natural environment for their continued success. Lack of effective management can lead to the loss of habitats, over exploitation of resources and an associated loss of income.

5.1.1 Tourism

The coastline is a major element in the geographic, recreational, commercial and ecological fabric of many countries and provides major destinations for local, national and international tourists. Freshwater areas, such as large lakes, are popular for recreation and in many cases are being developed for tourism. Associated villages and towns grow and develop their economy in accordance with the prevailing and seasonal tourism needs (main commercial streets, hotels, restaurants, clubs, shops and related activities; recreational activities on the beach or on lakes; transportation facilities, etc.). Socio-economic problems usually derive from the seasonality of the pressures on tourism-related facilities. There are a few data available on the contribution of coastal zone tourism to national economies (Grenon and Batisse, 1989) but for smaller coastal countries and island countries, especially those without industrial or agricultural outputs, tourism can be a substantial part of the economy.

There are concerns associated with the development of touristic ventures that can apply to both marine and freshwater areas. These concerns include tourism-associated aquatic transportation and the resulting pollution from vessels (oil, tar or litter on the adjacent area), as well as stress on populations and the environment where tourism is the major factor in the economy (Bird and Nurse, 1988). The development of tourist facilities may have a particular effect in developing countries where food security problems, pressures from recreational lobbies and public opinion may threaten alternative sources of income, such as the local fishery economy. Mariculture and the recovery of oil and gas can compete for the same space as that desired by recreational users. Tourists themselves can contribute to the waste and pollution of the host area, with a degrading influence on the quality of the recreational water use area arising from noise (primarily from transportation), recreational activities such as boating, and from solid and liquid wastes. Biodiversity reduction, resource depletion and human health problems may result from the accumulated environmental effects of tourism, including direct human impact (such as trampling).

There are a number of issues that may deter tourists from a recreational water use area. These include aesthetic and health problems arising from domestic waste disposal into marine and freshwaters that jeopardises the quality of food and the possibility of recreational activities on beaches in developing and developed nations. Eutrophication can degrade beaches and adjacent waters aesthetically through the accumulation of rotting marine plants; this can lead to a significant loss in tourist revenue. Aesthetic factors, such as litter, have a high deterrent value on visitors to bathing water areas (see Chapter 12).

5.1.2 Integrated management

Integrated coastal zone management (ICZM) is understood most simply as management of the coastal zone as a whole in relation to local, regional, national and international goals. It implies a focus on the interactions between the various activities and resource demands that occur within the coastal zone, and between coastal zone activities and activities in other regions. This might mean, for example, the incorporation of coastal environment protection goals into economic and technical decision-making processes or the co-ordination of tourism policies with nature conservation policies. Although ICZM has been promoted widely in recent years, it has not always been implemented successfully. This was mainly because of a lack of understanding of underlying coastal processes and it has only become apparent relatively recently that multidisciplinary land-use planning in the coastal zone is essential. Financial constraints have also been a contributory factor.

The same principles of ICZM can be applied to freshwater management and therefore marine and freshwater zones will be treated as synonymous in this chapter. In general, recreational water use areas, whether fresh or coastal waters, require similar management actions. However, lakes and other freshwater recreational water use areas are generally smaller bodies of water and are, by nature, more fragile than seas and oceans. The impact of human activities are apparent more quickly and failure to ensure adequate management will accentuate any degradation (Box 5.1).

Box 5.1 Problems of management of a freshwater recreational water-use area: Lake Geneva

Lake Geneva provides a unique example of a freshwater recreational water use area. It lies on the border of Switzerland and France and thus requires the integrated management of the two area authorities. A total of 41 beach resorts cover 4.5 per cent of the lake shore. These are artificial beaches with access to the lake, natural beaches and artificial beaches with a swimming pool. Boating and windsurfing are popular with visitors to the Lake and about 35,000 boats are registered on the Lake. To accommodate these, several yachting harbours and boat yards have been constructed. The main activities in the Lake Geneva basin are trade, tourism, banking and insurance and wine growing.

The bacteriological quality of the water in the beach resorts may vary considerably. There are a number of local pollution “blackspots” and restrictions in many areas prohibit swimming. On the Swiss side of the Lake the water is monitored according to the EU Directive on the quality of bathing waters (CEC, 1976) and also according to the procedure described in 'Examen et évaluation de la qualité hygiénique des bains de lac et de rivière' (Eschmann and Lüönd, 1965). Monitoring is only undertaken for Salmonellae and *E. coli.* On the French side, the monitoring is undertaken by the Ministry of Health, in compliance with the EU Directive (CEC, 1976). Domestic and industrial sewage systems have been installed and storm drains are being phased out. The quality of the water is therefore expected to improve. Human activities around the Lake are generally in conflict with the natural environment. There is intensive development around the shores of large private properties and woodland estates, that is having direct physical and aesthetic effects on the landscape.

In terms of management of the shores of Lake Geneva there is very little co-ordination between France and Switzerland despite certain provisions, such as France's Coastal Law and Switzerland's Cantonal Masterplans, because priority is given to the economy. There is, however, more co-ordination between the Swiss and French laboratories in relation to monitoring water quality in the Lake.

There is a real need for integrated lakeside management in order to consider all activities in the lakeside area. Development should be restricted to suit the capacity of the natural environment and specific lakeside provisions need to be drawn up and enforced. A trans-border structure for collaboration and co-ordination is also required which would promote an integrated approach to management. Economic instruments, both as incentives and disincentives, should be developed to integrate the environment into lakeside management. It is of paramount importance to promote environmental awareness and education throughout society if integrated and environmentally sound lakeside management is to be achieved.

Source: Adapted from OECD, 1993

Integrated coastal zone management programmes must address a range of issues, including habitat (loss of habitat or degradation of coral reefs, seagrass beds, wetlands, beaches and dunes, lagoons and estuaries), water quality (sources and nature of pollution, reduction and flow rates); management of natural hazards, and degradation of cultural resources and management of developments (mariculture, extractive industries, tourism, shore front development, major facility siting). Coastal managers must also consider any decline in fisheries, public access, biodiversity protection, sea level rise and degradation of scenic quality.

Integrated management ensures that priorities are given to all users of the water zone. Through policy supported by legislation and regulations, the most appropriate activity or activities can be given preference and investments in the area can concentrate selectively on these activities. Funding is a common problem in environmental management and for this reason some form of classification scheme combining priority for action and type of action required is especially useful. The classification then becomes an important tool in assisting planners in developing a strategy for improving the quality of the bathing water and the beaches. When the problem can be rectified through local efforts (such as beach cleaning), the management process should seek appropriate action from the municipality and reclassify the beach accordingly. Where the cause requires major investments or decisions on regional or perhaps national level, the authority should ensure that health concerns are represented adequately.

It is important to emphasise that improving the bathing water and beach quality strictly with the purpose of increasing the amount of tourists visiting a region or a country, can conflict with interests in protecting ecologically important areas or designated sensitive areas, etc. It must be an important issue for planners at regional, as well as national level to develop a plan for selecting and protecting these areas that are not to be transformed into large intensive tourist resorts. This is a premise for working with a classification scheme (see also Chapter 9).

5.2 Management framework

The WHO *Guidelines for Safe Recreational-water Environments* (WHO, 1998) present a management framework within which different levels of health risk and associated interventions are ordered in four major fields under the umbrella of integrated management (see Figure 1.1). The major fields of intervention are clustered as regulatory compliance, public awareness and information, control and abatement technology, and public health advice and intervention.

5.2.1 Legislation and regulation

Effective coastal or freshwater zone management requires an effective legislative framework to define the roles of different bodies and levels of government, as well as to provide environmental objectives. Management is not restricted to national issues - water quality, pollution control, international tourism and shipping are amongst the activities that affect the coastal zone and that extend beyond the national boundary. It is therefore obvious that a single government or agency cannot be responsible for the wide range of issues that need to be addressed in the coastal or freshwater zone. Effective legislation must provide a framework within which the roles and responsibilities of different organisations or interested groups are defined and must accommodate capacity to act at the international, national and local levels.

At an international level, legislation of particular relevance often relates to the management of international or transboundary waters. Whilst the legislation itself may be "hard law" or "soft law", it may provide for harmonisation or standardisation in data generation and exchange, and create obligations to notify other concerned parties regarding hazards and quality changes.

At a national level, regulatory measures are often considered to be inflexible but they are easy to operate and provide a clear and common framework for all parties concerned. In general, some form of basic water law provides a framework within which specified agencies are empowered to regulate. In the field of recreational water use, the national level is particularly important in establishing common ground for the assessment and reporting of safety and thereby supporting "informed personal choice". However, laws and regulations of relevance to safe beach management may derive from diverse influences, such as public health, social integration and rights of the disabled people, navigation for pleasure purposes, aquatic sports, fishing activities (in the sea and on the shore, e.g. bivalves), relevant flying activities, trade activities on public areas and the concession of State lands.

The diversity of national regulatory structures requires diverse approaches and solutions but, in general, managers concerned with recreational water use areas should consider both common law and statutory law. In most countries under common law, liability and negligence arise from the breach of duty known as "duty of care". This applies to members of the public as well as to operators. The duty specified is to take reasonable care. In the case of the safe management of the beach, the responsibility for taking adequate precautions rests with the operator. Of particular importance to those concerned with beach operation is the standard of care arising from their activities, i.e. that of an ordinary skilled person exercising or professing to have a particular skill. This is of particular relevance to lifeguards, for example, who are expected to conduct themselves as one would expect of the competent qualified lifeguard.

Those who employ staff on a beach may have specific duties to those staff under statutory law. In general these duties cover premises, written operating procedures, general working conditions, training, appropriate health and safety policy statements, consultation with safety representatives, safety procedures, the free provision of appropriate uniform, protective clothing and personal safety equipment and the provision of adequate first aid facilities for employees. For the employers there is an obligation to conduct operations in such a way as to ensure that members of the public are not exposed to risks to their health and safety.

The occupier of premises has a duty of care to any visitor using the premises for the purpose for which he is invited or permitted to be there. In general the operator of a natural beach will not be exposed to liability, although only one attraction, such as a diving platform, would expose the operator to liability if the duty of care is breached. The same applies to any operator deriving income from the provision of services for visiting swimmers. The operator may be exonerated from liability if a danger is brought to the visitors' attention and the operator takes appropriate measures.

5.2.2 Public awareness and information

The concept of reciprocal rights and responsibilities (as implicit in the concept of duty of care) highlights the importance of the capacity of the individual to make healthy or safe choices. In order to participate successfully in healthy recreation, members of the public require awareness (i.e. in this context knowledge regarding hazards and safe behaviours) and access to information to enable them to make informed choices. However, informed personal choice contributes not only to the protection of the individual but creates an incentive for improvement in the quality and safety of recreational water use areas - as users demand safer locations, the economic incentive to provide safe and attractive facilities increases.

Education and awareness

A basic appreciation of the health hazards that may be encountered during recreational water use and regarding safe behaviours is a prerequisite to health protection through the exercise of improved personal choice. A variety of special interest groups, such as lifeguard organisations, are instrumental in promoting education and public awareness activities. Watersport clubs, such as sailing, scuba diving, canoeing and swimming clubs, teach members basic first aid, safety procedures and a respect for the water environment. All watersports are potentially hazardous and participants should be made aware of the particular hazards associated with their sport. A variety of formal courses, usually culminating in an examination, are in existence to ensure that such activities are undertaken in a safe and responsible manner.

Beach classification

One tool to support informed personal choice that has received great attention in recent years is that of beach classification. In order to be effective a beach classification scheme needs to be based upon health and safety and must be of interest to users. It must also be based on reliable comparable data and overseen by a credible and impartial agency. Whilst classification schemes (e.g. Chapter 9) are often designed specifically to support and encourage informed personal choice, other classification schemes may be used for purely management information purposes (see Chapter 6), such as for determining beaches inappropriate for tourism development, or for identifying those suitable for the encouragement of tourism and perhaps those eligible for certain forms of aid or awards from a national or regional authority.

In order to decide to which class a beach belongs, certain criteria for each class need to be defined. The issue of beach classification systems and associated management response is discussed fully in Chapter 9. A classification system allows differences between classes to express differences in the problems to be solved. This means that different programmes or plans are needed depending on what class a specific recreational water use area belongs to. It also implies that the classification of beaches and water is a continuous process. A certain beach or water may belong to one class in one year, but as different measures are taken and problems are solved, the beach or water may change class next year.

On-site and local information

Users of recreational water use areas rely on information about safety, hazards to health and facilities, that comes from the news media, local authority notice boards and signs, environmental groups and tourist publicity, as well as relying on their own perceptions. Users can only control risks actively by acting on knowledge provided to them. Public awareness measures at the local level, in combination with national policies, are thus essential in order to achieve effective management of the recreational water use area and to reduce risks to users (see Chapter 7).

5.2.3 Control and abatement technology

Not all hazards encountered during recreational use of the water environment may be addressed effectively, or their associated adverse health effects averted, through informed personal choice. In some instances removal of the hazard or preventing access to the hazard is the preferred management option. In precluding access to a hazard high intervention approaches (such as fencing) or low intervention such as making access difficult (no development of car parks or public transport access), can be used. Such measures are relevant to a range of hazards, such as areas with strong currents, rocky environments, poor water quality or areas subject to toxic cyanobacterial blooms. To achieve long-term improvements in the quality of recreational waters investment must be made in pollution abatement technologies (see Chapter 9).

5.2.4 Public health intervention

Despite concern for the aesthetic aspects of recreational water use areas, the driving force behind much activity is public health and safety. In many cases circumstances may lead to situations that present an unfamiliar or unacceptable risk to public health. Such circumstances may relate to a breakdown in sewage treatment and disposal infrastructure, to toxic cyanobacterial blooms or to new or transient water uses that are incompatible with existing patterns of use. Under such circumstances the authorities responsible for public health are generally required to take a lead role in determining what actions should be taken and for what period. Such decisions are, in practice, often made under pressure of time and with inadequate information, but may be assisted by the existence of a national point of reference where experience and information on such incidents is maintained.

In addition to emergency response, some countries have, in recent years, instigated some form of accident emergency plans. These deal with, for example, major oil accidents at sea, or a chemical industry or nuclear accident. A structure for alert systems or notification relays, including home numbers for authority staff, may already exist. Unfortunately, the more common pollution accidents, such as failure of a sewage treatment plant or unusual wind direction forcing polluted waters onto a beach which is normally clean, or an algal bloom causing skin irritation, are often not included in this system. They do, however, affect the public more directly. It is therefore preferable to establish warning systems for these kind of events.

Beaches with a full lifeguard or warden service have most of an alert system in place, i.e. a dedicated person with warning signals on site during the bathing season. However, lifeguards are often only responsible for alerting the public to specific hazards, such as high winds, and not to pollution incidents, although in reality they, along with other coastal workers (e.g. rangers, wardens or coastguards) would alert the public to such incidents.

5.3 The role of organisations or individuals with a vested interest

The large number of interested organisations at all levels involved in the coastal and freshwater zones requires particular co-ordination and co-operation (see Figure 1.1). Central and local governments have a particular responsibility to establish standards and regulations to limit the health hazards to users of recreational water environments. Of particular importance is the layout of a management strategy for the achievement of integrated management. This requires addressing the issues of resources, economic development activities and societal needs in recreational water use areas. National government is also instrumental in directing, promoting and co-ordinating all activities relating to the application of laws concerning the coastal zone, including defining general criteria and methodologies for the monitoring of recreational water environments and activities, mode and frequency of sampling and analytical techniques. Organisations such as the World Health Organization Advisory Committee on the Protection of the Sea and other such bodies can provide advice and guidance. Research institutions, universities, non-governmental organisations (NGOs), special interest groups and the tourism industry can aid in the technical assessment of hazards and in monitoring changes. Industry, in particular the tourism industry, is increasingly adopting a more proactive role in monitoring the environment. Local interest groups, NGOs, local authorities, the tourism industry and the media are involved in raising awareness of users to some of the hazards associated with recreational activities (see Chapter 6). Citizens are often instrumental in contributing to remedial measures and are increasingly involved in public participation activities, such as beach cleaning, riverine fly tipping area cleanups and monitoring (see Chapters 6 and 12).

5.3.1 Municipalities

Local authorities are frequently the legal agency of the government. They often take a key role in bringing interested organisations together and gaining their collaboration and co-operation in decision-making, and participation in the implementation of decisions. Local authorities also contribute to the development and enforcement of standards and regulations. In general, public health laws and acts state that a local authority may make bylaws with respect to public bathing and beach management, including public bathing and coastal zone management. Municipalities may therefore be responsible for regulation of the areas and the hours when bathing will be permitted; they may also be responsible for requiring the persons providing accommodation for bathing to provide and maintain lifesaving appliances and lifeguards, as well as being responsible for the regulations for preventing danger to bathers. Municipalities may also enforce regulations regarding the navigation and speed of vehicles for pleasure purposes within any area allotted for public bathing. This is of particular importance because it permits the zoning of pleasure vehicles in relation to bathers. Municipalities may also be responsible for all the inland and adjacent areas above the low water mark where these bylaws have affect; protection of public health and safety, including monitoring of water and adjacent land. Protection, preservation, restoration and enhancement of coastal natural resources (including beaches, floodplains and dunes) as well as the use of beachfront property in a manner compatible with preserving public property may be the responsibility of the municipality. Municipalities may be required to produce coastline management plans as part of their normal planning responsibilities and any local authority with land subject to coastline hazards should plan and manage that land in accordance with its hazard susceptibility. Local authorities are therefore responsible for the investigation, design, construction and maintenance of works and measures to mitigate coastline hazards and also for promoting hazard awareness in their community in an attempt to reduce the social disruption and damage caused by coastline hazards. The latter can be done by supplying information and advice to property owners, residents, visitors, potential purchasers and investors (see Chapter 6). It is also a local authority responsibility to improve and maintain beaches and their amenity.

The structure and responsibilities of local governments differ throughout the world. In the UK, for example, County Councils are responsible for strategic planning, structure plans and waste disposal and the District Councils are responsible for housing, local planning, environmental health, coast protection, waste collection and noise control. In Australia, the Local Councils have general responsibilities for the production of coastline management plans, coastline hazard mitigation, hazard awareness and beach management, as well as specific responsibilities under the Environmental Planning and Assessment Act. In general, however, the fragmented and often duplicated responsibilities in the coastal zone are identified as severe impediments to effective planning and management.

Specific regulations for a beach are disseminated at the local level on the basis of the physical, environmental and social characteristics of the area. Regulations for the management and operation of a beach and for water activities are usually promulgated by the City Council or (as in Italy) by the nearest Harbour Maritime Authority and are addressed to the concessionaires or managers of the maritime State land.

5.3.2 Facility operators

Once a beach manager has a complete picture of the beach characteristics (beach registration and classification) all decision elements are available for the daily operation of the beach and for (mid- to long-term) management plans. It is suggested that the competent authority should designate an operator, or another responsible person, to be on duty when a beach is open for visitors. This operator should take decisions relating to the beach and should take appropriate action when requested by the authority when accidents or spills take place leading to beach contamination or when water quality becomes unacceptable or for safety reasons (such as weather, inadequate lifeguards or safety equipment).

There are also a number of considerations to take into account when designing the facilities to support a public beach. Resources become an important issue and therefore the provision of facilities should be prioritised according to the needs and uses of the area under question. Monitoring for potential health hazards and associated management actions should always be considered a priority over the provision of shops and refreshment kiosks when developing recreational water use areas. However, it is acknowledged that this approach would not always attract tourists and the associated finance to an area. Research has shown that visitors to the coastline place more value on the cleanliness of an area and on the provision of facilities than on unseen human health hazards such as microbiological parameters (Oldridge, 1992; Morgan *et al.,* 1995) (see also Chapter 12). Education and public awareness must, therefore, become an essential part of integrated management, especially where resources are low and prohibit the provision of facilities (see Chapter 6). Consultation between those with a vested interest and local communities is essential if the various conflicts of use are to be resolved.

5.4 Management options

The various different attitudes of visitors to the recreational water use area also determine the necessary level of facilities that are desired. Different cultural contrasts exist in the use of beaches; in many tropical areas the sea is used as a cleaning place or for trade amongst fishermen, whereas throughout Europe beaches are generally used as places of passive activity. Despite many people looking for seclusion at a beach, the pressures to develop recreational water use areas to support growing populations and increases in tourism are so great that it is becoming very difficult to find underdeveloped beaches, particularly in countries with a warm climate.

Management options and preferences vary according to the level of development of an area and the preferences of visitors to that area. Management actions need to take into account local economic needs as well as the desires of the users. In general, two broad categories of user can be identified: those seeking resort areas with facilities, entertainment and easy access, and those seeking secluded or rural areas.

5.4.1 Resort beaches

The following guidelines are provided for management of resort beaches where tourism is a priority. Where resources are a restraining factor careful prioritisation should occur that will minimise public health risks. Topography, including slope and bottom material, need to be considered. Beach cleaning should also address the removal of litter and debris from the lake or seabed where they present a hazard to bathers. Specific regulations may be adopted for the prohibition of potentially hazardous items, such as glass, on the beaches. In addition, adequate litter bins should be provided. Clearly visible depth markers should be provided at the points of maximum depth of all designated areas and at diving boards, platforms and similar facilities. Zoning is an important measure in minimising risk where different user groups coexist within a confined area (i.e. dog-free zones, conservation areas and naturist zones, zones for swimming, sailboarding, powerboating, etc.). Swimming may be limited to a specific area, i.e. the least hazardous, which also facilitates supervision and segregation of incompatible activities (see Chapter 7). Wastewater from toilets and showers should be discharged to the local municipal sewage system. If that is not possible an alternative treatment should be established that is acceptable to local or national standards. Where possible, toilet facilities and showers should be provided in adequate number. To prevent cars and vehicles driving on the beach, access facilities should be provided to beach parking areas. Access to beach areas for emergency vehicles should be provided and appropriately signposted. Easily read and understood information boards should be used to display beach regulations, general information on beach and water quality and facilities. The signs should be located so that they will be seen at the access points before entering the beach, the resort or the swimming area. Where appropriate, more formal regulations, in the form of bylaws, may be adopted to control activities at the coastal zone, particularly noise, fires, dog fouling and litter.

Safety aspects are of particular importance to coastal managers of tourist beaches (see Chapter 7). Clearly identified warning signs should be provided where appropriate indicating, for example, when the beach is closed for swimming, the times when lifeguards are on duty, danger of swimming during heavy storms or after sunset or in dangerous currents.

5.4.2 Rural beaches

Ideally, as with resort beaches, rural beaches should be monitored for potential health risks as a priority. Rural beaches are generally popular with walkers, naturalists, and fishermen and for other kinds of casual enjoyment. Such beaches should be cleaned “as needed” but at least four times a year. Beaches that are particularly frequently used shall be cleaned at least once a week during the summer and each month during the winter.

On rural beaches, safety boards should be displayed at all principal access points to beaches, in car parks (if present) and at particular hazard spots. The hazards of the particular beach should be clearly indicated, together with the times of high and low water, the distance of the nearest telephone and some useful telephone numbers, and the location of the nearest first aid facilities. It is suggested that public rescue equipment should also be in place in the more frequented rural beaches.

5.5 Elements of good practice

- The management framework developed for a bathing water area must take into account the impact of various competing activities, sustainable management processes, water quality issues and associated safety issues.

- Such a framework must reconcile development pressures with socio-economic, cultural and environmental criteria.

- The full range of legislative and regulatory controls that interplay with coastal or freshwater recreational water management must be incorporated into the management framework including duty of care, health and safety legislation, water quality regulations, pollution control and international articles governing international tourism and shipping. Such measures will vary from local bylaws through national to international law.

• The development of an integrated management framework must include a range of issues including nature conservation, water quality, management of coastal development access and environmental degradation.

• The role of a local municipality is central to an effective coastal management framework. Their activities must be co-ordinated within a coherent national context.

• A beach classification scheme can be constructed which provides a discreet hierarchy of categories and concomitant management activities.

• On completion of a full catalogue of the characteristics of a particular recreational water area, the beach manager has the framework from within which to establish the operational activity.

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Chapter 6*: PUBLIC PARTICIPATION AND COMMUNICATION

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Successful beach management requires an understanding of the nature and dynamics of a beach system, i.e. the physical, chemical and biological interactions that take place on and around the beach, the requirements and perceptions of the beach users, economic and tourism interests and environmental protection measures. Inevitably, there are conflicts between these elements, although many of these conflicts can be resolved through effective communication at an early stage, through information and, above all, active participation of all parties, particularly the public.

Large differences exist between the capacity and mechanisms for communication in resort beaches near big urban areas and in rural beaches used only by a limited number of people. Nevertheless, beach managers should consult with, and inform, beach users at all appropriate stages. The success of beach management depends very much on the active participation and involvement of the local population and of beach visitors (Camhis and Coccossis, 1982; Gubbay, 1994). The underlying principle is that the public has a right to know, a right to be heard and a right of co-decision. In keeping with the principles of Agenda 21 (UNCED, 1992), the public should be involved in information gathering and management of recreational water use areas. In resort areas, the management tasks are usually predominantly in the hands of the local government or health authority. Progressively more responsibility lies with the local community or individual user as beaches become more rural.

The public can take an active role in a variety of practical activities concerned with beach management. The participation of the public in monitoring helps to raise awareness of the condition of the recreational water use area and provides a cost-effective method of gathering large amounts of data which can then be acted upon by beach managers (see Chapter 12). Involvement of the public in special interest groups, such as voluntary lifeguard organisations, helps to educate the public for self protection. Beach managers have a responsibility to educate the public about hazards related to the recreational water use area, to provide warnings to the public and to provide other information. There are a variety of methods for communicating with the public, such as flags, signs, literature or beach awards. Whichever method is used, it is imperative that the public understands clearly the message being conveyed.

6.1 Public participation schemes

A number of community participation schemes have been developed worldwide. An example is the “Officer Snook Program” which was initiated in 1992 at Miami Beach and was sponsored by the United States Coast Guard. This scheme includes videos, slide shows, competitions, cleanups and recycling programmes involving 25,000 elementary schools (Sevin and Sevin, 1995). In Glacier Bay, Alaska the prevention of marine debris is an integral part of the visitor management and education programme (Synder, 1995). Some schemes are aimed at specific types of marine debris. In Tasmania, Australia the use of television advertisements was an integral part of a community awareness programme initiated in response to the growing entanglement of marine mammals and seabirds in marine debris (Slater, 1995). In southern Africa, the Dolphin Action and Protection Group launched a national campaign in 1987 entitled “Save our Sealife: Prevent Plastic Pollution”. The scheme targeted shipping and fishing companies, industry, schools and the general public, and involved the distribution of pamphlets, the initiation of beach-cleans and the raising of the issue in Parliament. The scheme has now been extended to Antarctica, Namibia and islands in the Southern Atlantic and Indian Oceans (Rice, 1995).

Education and public awareness are key elements in the reduction of marine debris. Public involvement in beach litter management takes two forms: direct action such as beach cleanups and monitoring; and indirect action, such as education, award schemes and legislation. The involvement of the public in beach monitoring and cleanup programmes has dual advantages in that it allows a large sample size to be achieved, and raises awareness among society which will then translate into effective individual action to reduce litter at source. Involvement of the public in such campaigns has been achieved world-wide. Coastwatch Europe, for example, involves tens of thousands of volunteers each year in monitoring marine debris (Dubsky, 1995).

The largest network organising beach clean events is the Center for Marine Conservation (CMC) based in Washington DC, USA. This centre organises annual beach cleans during “Coastweek” at the end of September and beginning of October. Volunteers use standard recording cards which divide debris into eight major categories according to the fabrication material and a further 65 categories according to type of item. Guidance notes are provided, including an identification guide and information on how the data is used. Volunteers are asked to record the location of the beach and the nearest city, the estimated distance covered and the number of bags filled with debris (Bierce and O'Hara, 1992, 1993).

6.2 Local communication

Chapters 2 and 9 provide examples of schemes involving registration and classification designed to help managers identify the characteristics of their recreational water use area and its different uses. It is extremely important at the stage of beach classification to involve all interested parties. The more people feel they are involved, the easier it is to get active and constructive participation and support for monitoring programmes, development plans and environmental protection measures. When beach management plans are operational, further information updates are important. This information needs to involve aspects of beach safety and water quality.

Baseline surveys, eco-audits, or shore and hinterland surveys are an excellent way to gather data about the bathing water itself, the surrounding aquatic environment and the hinterland. For resort beaches, the baseline survey should be carried out during the main bathing season, when caravan parks are full, local restaurants are running at peak capacity and facilities such as public toilets and showers are being well used. If a survey is undertaken out of season unexpected events, such as seepage from overloaded septic tanks and storm drains, are likely to be missed. It is essential that the baseline survey is augmented with background information concerning seasonal changes. Such information can be gained from local people, such as year-round swimmers or non-governmental organisations.

Local public participation should be part of the whole exercise from survey to subsequent action. However, it must be borne in mind that involving local public participation during baseline survey data gathering might occasionally invite bias into reporting unless care is taken by the survey team. The baseline survey will provide a variety of information that can be used in plans for informing the beach users and visitors on safety and health risks.

6.3 Types of information

6.3.1 Beach safety

Unless users are aware of the hazards and regulations applicable to particular areas, they are unable to make an informed choice about their destination or to react appropriately to management strategies. While there may be resort areas with an abundance of public information and controls, it is not economically viable, nor necessarily desirable, to extend such infrastructure to rural bathing places, used by only a few people. However, it is in everybody's interest that bathing is as safe as possible even in these isolated areas.

Strategies for accident prevention should first address the removal of hazards. If this is not possible, steps should be taken to reduce the level of risk. Information is particularly important where less can be done physically to reduce the risk. In this regard, all available techniques should be used to convey safety messages, such as the provision of safety signs and notices, flags and brochures (see Chapter 7).

6.3.2 Water quality

The primary reason for monitoring bathing water quality and for informing the public is to protect public health. Members of the public are unlikely to want to know the details of sample treatment in the laboratory, although they would need to know whether the water quality is safe. It is essential that information provided to the public is presented in a clear, unambiguous and easily understood way. Some of the cheapest and quickest approaches to assembling and presenting summary data are often the most effective.

6.4 Award schemes

Award schemes are often used as an incentive programme to involve all parties concerned in participating in optimising beach safety, water quality and education activities. Awards are generally the recognition of effort, or of standard achieved. Most award schemes look at only a few of the parameters associated with beach classification. They often fail to take account of the beach user's perception of the environment. The ideal scheme should consider physical, biological and human parameters. The first two are relatively easy to measure (see Chapters 8, 10 and 11), the latter is more difficult to assess.

Beach award and evaluation systems are valuable tools for the promotion and management of beaches and tourism. Annual and systematic surveys of a variety of parameters, including beach litter, have been undertaken for a number of award schemes. Beach awards can be important agents for change, integrating a variety of factors, including water quality, safety, litter, and beach management practice in general. Resorts, in particular, want these awards and manage their beaches to ensure that they comply with the requirements of the award.

6.4.1 Blue Flag

Probably the most widely known beach award within the European and Mediterranean context is the Blue Flag Award. The Blue Flag scheme is organised by the Federation of Environmental Education in Europe (FEEE) (FEEE, 1998). The Blue Flag Campaign was started in 1987 as one of the many activities of the European Year of the Environment. It is a Europe-wide initiative involving more than 1,000 beaches and 500 marinas in 19 European countries. Within the European Union, only “identified” bathing waters within the terms of the Bathing Waters Directive 76/160/EEC are eligible for the award. Outside the European Union, almost any beach could apply for the award via the national operator. Qualification is based on a wide variety of criteria (some of which are obligatory, others are guideline criteria) divided into four groups: environmental education and information, environmental management, water quality, and safety and services. In terms of the environmental education criteria “the aim of the campaign is to increase the public's environmental awareness and to create a platform for active participation in the protection of the environment” (FEEE, 1998). Co-operation between FEEE and the United Nations Environment Programme, Industry and Environment (UNEP IE) office resulted in a pilot project for implementing the Blue Flag concept in non-European regions.

To combine monitoring of Blue Flag holders with gathering extra information, an “In Season Beach Award” was run in Ireland in 1992 for 100 beaches. Points were allocated for a range of criteria and each beach was visited and checked thoroughly by one of a volunteer team (see Box 6.1).

Box 6.1 The In-Season Beach Award

The presence of a Blue Flag indicates that the visitor should find dependable water quality, cleaning, toilets and other facilities on a managed beach. It does not relate to wind shelter, diving facilities, beautiful scenery, etc., nor does it allow the visitor to predict when swimming might be safe or unsafe and on which days lifeguards are supposed to be on duty. Nevertheless, such questions would be asked by more concerned tourists before booking their holiday or before heading off to any particular beach from several possible beaches that are at an equal distance away.

In order to combine the checking of Blue Flag winners with the gathering of extra information, an in-season beach award was designed and run in Ireland in 1992 for the 100 top beaches. A national weekly newspaper sponsored the award. A list of the top 100 beaches was prepared from Blue Flag entries, augmented by further beaches known from local community notes. A detailed questionnaire was designed and tested on different beaches before being adapted. A volunteer team with a good environmental background was established and trained together. Each beach was visited in the peak July and August bathing season and checked thoroughly by a member of the team. Where possible, local people were interviewed. Photographs and sketches augmented the reports filed by the team.

Points were allocated, on a predetermined scale, for natural assets and facilities provided, with an option of bonus points for special quality. In the health category, for example, a stream was considered something positive as a natural asset and allocated a point. Sixty-seven beaches had such an asset. Unfortunately, the majority of the streams turned out to be polluted when checked for faecal streptococci. A clean stream with good invertebrate diversity was thus a rare quality, and was awarded an extra six points. The display of water quality results and minimal frequency monitoring was also awarded a point. Moreover, when members of the public questioned at random found the information clear and understandable, an extra three points were awarded. Winners were announced in each of the following sections: water quality, other facilities, natural beauty and wildlife value.

The scheme received very good publicity and initiated a lot of local activity to remove accumulations of litter. Those beaches shortlisted as final award winners were revisited over a two-day period by the sponsor's helicopter. While a beach with undependable water quality could not become a winner, it could get a very high number of points for other assets. The results could then be used to argue for improvement of the weakest feature, e.g. water quality.

6.4.2 Costa Rica

Chaverri (1989) devised a rating system to identify beaches suitable for governmental and private tourist development in Costa Rica under the authority of the Marine and Terrestrial Act (Ley Marítimo Terrestre). Up to 113 factors, classed as either positive and negative, were given a score between zero and four, with the final rating score for the beach obtained by subtracting the sum of the “negative” scores from the sum of the “positive” scores. The factors comprised six groups. Some selected factors were water, beach, sand, rock, general beach environment and the surrounding area.

No attempt was made to attribute quantitative values to scores for any of the factors, so that the beach score for any factor was based purely on the subjective judgement of the particular assessor. In addition, no attempt was made to assess the importance attached by beach users to any of the factors in the checklist, to assess which factors were of importance for various types of beaches (apart from a differentiation between sand and rock areas), or to attach weightings to the various factors. Even the rigid division of factors into "positive" and "negative" categories could be considered to be subjective.

6.4.3 Black Sea Environment Programme

The Black Sea Environment Programme aims to strengthen and create regional capacities for managing the Black Sea, in particular by developing policies and legislative frameworks relating to pollution, health, biodiversity, and to attract investment. The programme emphasises the importance of harmonisation of methodologies and standards for evaluation of bathing beaches and beach quality. It provides guidelines for assessment of bathing beaches and bathing water quality, and on how to implement assessment programmes and to evaluate the results.

The programme suggests a questionnaire for registering beach quality that takes into account details concerning beach facilities, physical characteristics of the beach, usage, accessibility, water quality and designation. It does not involve "scoring" the beaches. The final classification is based on the following definition: *"a good beach is a safe beach as well as a beach with good water and beach quality"* (WHO, 1995). The beach is classified according to any problems discovered and, using this classification, an action programme can be identified. The objective of the programme is then to encourage the use of data to refine the action programmes to solve problems that have been highlighted through the monitoring programmes.

6.4.4 Schemes developed for Turkey

Morgan *et al.* (1995) used a questionnaire based on beach users preferences and priorities linked with a 47 factor checklist for five Turkish beaches; Oludeniz beach scored the highest with 87 per cent. Additionally, Morgan *et al.* (1996) carried out further studies on Turkish, Spanish and Maltese beaches by testing beach user perception for 50 beach aspects. Williams and Morgan (1995) have also assessed 28 Turkish beaches in terms of 50 physical, biological and human parameters based on the views of a range of international coastal experts; Dalaman beach rated the highest at 93 per cent. Beaches were scored for each parameter on a scale of one (poor) to five (good). Williams *et al.* (1993b) and Leatherman (1997) have used a similar scale for 182 beaches in the south west peninsula of the UK and 650 beaches in the USA respectively. These checklists could be readily improved because many aspects of the beach environment were classified as good or bad without regard to the varying preferences of different types of users, and various uses, of the beach environment. Many factors were judged on a subjective basis with no weightings attached. In addition no attempt was made to resolve the problem of different views and preferences of visitors to different types of beach.

6.4.5 Local quality schemes

Various local schemes exist to assess beach and water quality, such as the Solent Water Quality Awards, which were established in 1992 and are administered through the Solent Water Quality Conference, a consortium of local authorities and interest groups in Hampshire, UK. All bathing waters in the Solent region (identified and non-identified) that are used regularly for bathing can enter the scheme. The criteria for achieving an award are:

- At least one representative sampling point must be selected for each beach.

- Imperative standards of the EU bathing water directive (CEC, 1976) must be met.

- The water must not contain any gross pollution by faeces or other sewage-related debris, or suffer from persistent occurrence of oil, tar or a significant smell.

- Supporting information, such as water quality results from the previous years must be given.

The main criticism of these awards is that they do not consider the beach itself and are restricted to the water quality.

6.4.6 Other schemes

Recent studies suggest strongly that people with different personalities and demographic variables have different requirements for the beach environment and prefer to visit different types of beaches (Morgan *et al.,* 1993; Williams *et al.,* 1993b; Williams and Morgan, 1995). This poses a problem for beach ratings, but it can be overcome by dividing the beaches into a number of categories on the basis of degree of commercialisation, i.e. presence or absence of particular facilities. For example, Williams *et al.* (1993a,b) and Morgan *et al.* (1993, 1995) used questionnaire surveys as a basis for establishing preferences and priorities of beach users at various beach types, and to weight the various factors in a beach quality rating scale. The scheme was carried out in two main stages. Firstly, an assessment was made of the preferences for various beach features (such as pocket, log spiral or linear beaches) and facilities (such as toilets) and the attributes of the visitors to different types of beaches. This enabled the various factors in the beach quality rating scale to be optimised and correctly weighted. This was followed by the introduction of a checklist for the beach quality rating scale containing classifications and categories of 48 beach aspects closely matched to those in the questionnaire. As many beach aspects as were reasonably possible were assigned classifications based on quantifiable values. Weighting and scoring of the various beach aspects on the checklist was generated by analysis of questionnaire responses.

6.4.7 Standardising grades and categories

The standardisation of grades and categories for describing and informing the public of the quality of recreational water use areas is complicated by the variety of aims that exist amongst different schemes. Recently, Earll *et al.* (1997) put forward the idea of a standardised litter pollution category, i.e. the “ABCD” grading system used in the “Code of Practice on Litter and Refuse" developed by the 1990 Environmental Protection Act (DoE, 1991) and the Thames Clean Project (Lloyd, 1996). Litter categories suggested by Earll *et al.* (1997) are:

Grade A Absent, no evidence of litter anywhere.
Grade B Trace, small items only.
Grade C Unacceptable, widespread distribution with minor accumulations.
Grade D Objectionable amount, area heavily littered with accumulations along the boundaries.

Litter categories of concern to the general public include sewage-related debris, litter accumulations and harmful litter such as medical waste. The number of items listed in Table 6.1 relates to a 100 m stretch of beach at the high water strand line. A recreational water use area would receive a grading based on one of the categories falling into the worst grade, i.e. if one of the categories scores a "D" then the beach is graded a "D" beach. The actual numbers proposed in Table 6.1 are subject to further research. A constant strand line length of 100 m has been advocated but this could cause problems for small pocket beaches of less than 100 m in length.

Table 6.1 Proposed classification scheme for the assessment of aesthetic quality of coastal and bathing beaches

Category/type	A	B	C	D
Sewage-related debris				
General	0	0	1-5	6+
Cotton buds	0	1-9	10-49	50+
Litter				
Gross	0	1-5	10-24	25+
General	0-49	50-99	100-999	1,000+
Harmful	0	0	1-3	4+
Accumulations				
Number	0	0	1-3	4+
Total items	0	1-5	4-49	50+
Oil	Absent	Trace	Some	Objectionable
Faeces	0	1	2-9	10+

Source: Earll *et al.*, 1997

6.5 Education

Awareness on water safety may be achieved through community education. This can be by means of talks to groups and schools, information sheets and posters, videos or practical activities. Public participation and education can be promoted through government advisory committees, citizens advisory committees, interest group representatives, public hearings, broad dissemination, information gatherings, community meetings, media campaigns, brochures, newsletters, school programmes, community exhibitions and user group training.

6.5.1 School education

School education differs greatly between countries and also between regions and school types. Most students never see a County Council or Parliament debate and have never asked their local representative to pose a written question for them (such as why the local beach is not designated). In addition, environmental law is rarely taught in schools. Although water quality experiments might be carried out in chemistry and biology classes, and field-work might be undertaken, the results are rarely compared with real data generated by official monitoring programmes.

The involvement of school groups in awareness campaigns such as Coast-watch Europe (Dubsky, 1995) (see Box 6.2), in beach cleans such as those organised by the CMC in the USA (Bierce and O'Hara, 1993) and in other community participation programmes (Box 6.3) (see also Chapter 12) is becoming more widespread. Understanding provides the ability to make informed decisions. Bathing is practised so widely as a form of recreation that information relating to its safe enjoyment should be widely disseminated beyond swimming classes. A basic understanding of water pollution, water quality and dangers on the shore may be taught beneficially in school such that the knowledge gained is applied early.

Box 6.2 Coastwatch Europe survey

Coastwatch Europe is an international network of universities and environmental groups co-operating on coastal zone management issues, as well as public information, participation and training schemes. The core Coastwatch project, shared by 23 participating countries, is the Coastwatch Europe survey. A single set of questions is agreed internationally by all co-ordinators in order to give baseline information about all sections of the coast. The questionnaire is translated into national languages, and may be augmented by extra national questions and, where financially possible, with water quality testing. Local baseline surveys are undertaken by local volunteers on 500 m stretches of shore from the water's edge, covering the splash zone and immediate hinterland. The volunteers are recruited through newspaper publication of the questionnaire or through associations (schools, scouts, ladies clubs, divers or sea anglers). Since 1989 there have been over 10,000 sites surveyed every autumn, making it the largest volunteer data set for the coast of Europe.

In many countries the scheme does not just involve environmental groups, universities and local volunteers, but also local authorities. Before the survey starts, surveyors are provided with a local contact and are equipped with coded maps of their area, questionnaires, survey notes and test kits. The survey often leads to follow-up actions, such as experienced in County Louth (Ireland). The County Council asked surveyors to return questionnaires to the authority before submitting them to Coastwatch, with the pledge that officials would look through the data and act on broken pipes, illegal dumping, etc. within weeks of receiving the information. As promised, within a month of receiving the data a big coastal clean up was started by the Council, which invited local people to join in. Such co-operation in management builds good will and translates into better coastal quality.

In running the survey and various forms of follow-up action, Coastwatch co-ordinators have found that Europe-wide, specific volunteer subsets, such as fishermen and yachtsmen, have excellent knowledge based on their experiences of living and working in the locations. In most cases, simply raising the polluters' awareness of the consequences of their actions and bringing people together in the common cause of making their local water safe, brings about the required change. Sometimes, lack of finance is clearly the limiting factor.

Increasingly, it is the local people who ask for guidelines to gather baseline information and draw up a management plan for their area. The survey has often resulted in cases of co-operation between local public and officials for common aims and quality control, such as the joint management of litter, introduction of recycling campaigns or nature trails. If sewage treatment is inadequate, for example, a combined effort in lobbying the government to supply the necessary funding can be much more effective than either local people or a local authority asking alone. In cases where officials cannot be persuaded to join in, scientifically qualified environmental groups can be an alternative.

Box 6.3 Community participation schemes

Negril Coral Reef Preservation Society (NCRPS) based in Jamaica is a nonprofit non-governmental organisation that was formed by a group of diving operators in 1990 because of concern over the state of the reefs. At the time of its inception, the main goal was to install reef mooring buoys on frequently dived reefs in a growing tourist town that was once a fishing village. Thirty-five state-of-the-art reef mooring buoys were installed in 1991 with the help of “REEF RELIEF”, a partner organisation in Key West, Florida. Although the reef mooring buoys prevented over 20,000 anchorages annually, it was decided that the project should be expanded.

Deteriorating water quality was identified as the biggest threat to Negril's (and Jamaica's) reefs. Lack of proper sewage treatment, deforestation, poor agricultural and solid waste management practices allowed nutrient-laden effluents to enter coastal waters. The nutrients were stimulating the growth of nuisance algae, which were smothering the reefs. As a result, the coastal waters of Negril are now in the advanced stages of eutrophication and live coral coverage is less than 10 per cent, while algae dominate more than 65 per cent of the reef. The NCRPS has as one of its primary concerns the restoration of water quality so that coral reefs can, hopefully, someday return to their previous state, or at least become recovering reefs. In 1997, a small water quality monitoring laboratory was established at the NCRPS Headquarters. A water quality monitoring programme was initiated, measuring nutrient levels in rivers, streams, ground, and coastal waters throughout the Negril Environmental Protection Area and National Marine Park. Monthly samples were collected by the NCRPS rangers and analysed in the local laboratory, while some samples were sent to outside laboratories for analysis.

An aggressive public education campaign targeting schools, communities and the hospitality industry involved raising awareness of water quality issues. Annual workshops entitled “Protecting Jamaica's Coral Reef Ecosystem” allow open discussion and participatory planning of management initiatives. A Junior Ranger training programme, involving hundreds of children between the ages of 10 and 17 years, gets students and teachers within the local schools involved in learning about water quality issues and taking part in the monitoring programme. In the context of establishing a management structure for a Marine Park, the water quality initiatives are included in an overall coral reef monitoring programme. In partnership with the Jamaican government, through locally established “Resort Boards”, NCRPS has also designed a watersports and recreational zoning programme. Demarcation buoys set 300 feet from shore mark a safe swimming zone, and there are plans to expand this programme by adding additional buoys for demarcation of non-motorised craft and environmental zones. The Society is responsible for the installation and maintenance of these demarcation buoys, and the rangers patrol them together with the police, to ensure that rules and regulations are adhered to.

Source: Negril Coral Reef Preservation Society, Pers. Comm.

6.5.2 Special interest groups

Swimming, lifesaving and other local interest groups play an important part in the education and awareness of the public towards recreational water-use quality and safety (Box 6.3). Recognition of beach hazards has led to the introduction of various beach safety regulations and the establishment of lifesaving clubs at many resorts, particularly in the USA, Australia and New Zealand. Surf Life Saving, Australia, for example, is a national organisation co-ordinating 255 Life Saving Clubs and professional lifeguards who patrol 300 beaches and make over 10,000 rescues each year. This organisation has also sponsored the Beach Safety Management Programme, documenting coastal hazards and their impacts on public safety on more than 7,000 Australian ocean beaches. It has developed a database for every beach, showing location, access, nature, physical characteristics, facilities, use, and beach and surf conditions, together with an assessment of risk levels (on a scale of 1 to 10) and a prediction of the cost required to maintain adequate levels of public safety on each beach (Short *et al.,* 1993).

6.6 Elements of good practice

• The findings of any monitoring programmes should be discussed with the appropriate local, regional and/or national authorities and others involved in management (including integrated water resource management), such as the industrial development or national planning boards.

• The results of monitoring programmes should be reported to all concerned parties, including the public, legislators and planners. Any information relating to the quality of recreational water use areas should be clear, concise and should integrate microbiological, aesthetic and safety aspects.

• In issuing information to interested parties (the public, regulators, NGOs, legislators, etc.), it is essential that their concerns are kept in mind.

• Reports addressing the quality of recreational water use areas should be accompanied by references to local and visitor perceptions of the aesthetic quality and risks to human health.

• The deleterious impacts of human health hazards and aesthetic pollution, and of control measures to avoid or reduce such impacts, should be introduced into environmental health education programmes in both formal and informal educational establishments.

• The usefulness of the information obtained from monitoring is severely limited unless an administrative and legal framework (together with an institutional and financial commitment to appropriate follow-up action) exists at local, regional and international level.

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Chapter 7*: PHYSICAL HAZARDS, DROWNING AND INJURIES

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Physical hazards are generally perceptible and discernible. Physical hazards, unlike many microbiological, biological and chemical hazards do not require laboratory analysis for their recognition or description. The hazards that can lead to drowning and injury may be natural or artificial. By definition a hazard is a set of circumstances that may lead to injury or death. The term “risk” is used to describe the probability that a given exposure to a hazard will lead to a certain (adverse) health outcome.

In the context of this chapter, hazards are best viewed as both the potential causes of ill health and the absence of measures to prevent exposure or mitigate against more severe adverse outcomes. Thus, an area of dangerous rocks against which swimmers may be drawn by prevailing currents or wind, the absence of local warnings, the absence of general public awareness of the types of hazards encountered in the recreational water environment, and the absence of local capacity to recognise and respond to a person in danger, may all be readily conceived as part of the hazard. The number of injuries can be reduced by elimination of the actual hazard, by restricting access to the hazard, by members of the public recognising and responding appropriately to the hazard, and by ensuring deployment of effective management actions.

The severity of the adverse health outcomes considered in this chapter differs markedly from that described elsewhere in this book. The severity of the outcomes varies widely but includes death (for example through drowning) and lifelong disability through quadriplegia, as well as blindness arising from retinal displacement. It also addresses less severe health outcomes such as cuts and lacerations that are nevertheless important in determining the pleasure derived from recreational use of water environments. Whilst the overall frequency of severe outcomes may be low they are of considerable importance for public health.

Despite the importance of the health outcomes addressed in this chapter, methods for assessment of the associated hazards and mitigating factors are relatively poorly developed and have attracted limited research when compared with, for example microbiological pollution of bathing waters (see Chapters 8 and 9). Nevertheless, assessment may be rapid and simple and may be readily and rapidly associated with short-and medium-term actions of immediate relevance to the protection of public health. This chapter draws heavily on the corresponding chapter of the *Guidelines for Safe Recreational Water Environments* (WHO, 1998) in which the issue of physical hazards and drowning is also discussed. This chapter summarises the key components of that chapter and provides a practitioners guide to the various issues.

7.1 Health outcomes

The most prominent health outcomes resulting from recreational use of water are:

• Drowning and near-drowning.

• Major impact injuries, especially spinal injuries, resulting in quadriplegia and less frequently, paraplegia, as well as head injuries.

• Slip, trip and fall injuries (including bone fracture and breaks).

• Cuts, lesions and punctures.

• Retinal dislocation resulting in near blindness or blindness.

7.1.1 Drowning and near drowning

Drowning and near drowning are important health issues and merit special consideration in the development and management of water recreational facilities. Informal peer supervision in more densely-used areas may contribute significantly to the prevention of drowning and, conversely, the desire for greater seclusion may be a significant contributory factor. Private pools (including ornamental, swimming and paddling pools) contribute significantly to drowning statistics, but are not addressed in this volume.

Males are more likely to drown than females (WHO, 1998) and this is, in part, associated with higher exposure to the aquatic environment (through occupational and recreational uses). In many countries, alcohol consumption is one of the most frequently reported contributory factors associated with drownings. Amongst children, lapses in parental supervision are the most frequently cited contributory factor in drownings and near drownings. Drowning and near drowning may often be associated with recreational water uses with low water contact, such as use of water craft (yachts, boats, canoes) and fishing (from water craft and from the waters edge or solid structures). Where these recreational water uses occur during cold weather, immersion cooling may be a significant contributory factor (Keatinge, 1979; Poyner, 1979). However, non-use of life jackets, even when they are readily available, is a significant contributory factor in all cases. The availability of cardiopulmonary resuscitation (CPR) and rescue skills have been reported to be important in determining the outcome of accidental immersions. However, attempted rescue represents a significant risk to the rescuer.

Most drownings occur in non-swimmers and the value of swimming lessons as a preventative measure appears logical. However, there is significant debate regarding the age at which swimming skills may be acquired safely, and the role of swimming skills in preventing drowning and near drowning is unclear. Whilst evidence does not suggest that water safety instruction increases the risk of young children drowning, their increased skills do not decrease the need for adult supervision; the impact of training on decreasing parental vigilance has not been assessed (Asher *et al.,* 1995).

Pre-existing diseases are associated risk factors and higher rates of drowning are reported amongst those with seizure disorders and paediatric seizures. Further documented contributory factors in drownings include water depth and poor water clarity (Quan *et al.,* 1989). Studies of “near drowning” show that the prognosis depends more on the effectiveness of the initial rescue and resuscitation than on the quality of subsequent hospital care. The principal contributory factors and preventative and management actions for drowning and near drowning are similar and are summarised in Table 7.1.

7.1.2 Spinal injury

Diving accidents have been found to be responsible for a variable percentage of traumatic spinal cord injuries. However, in diving accidents of all types, injuries are almost exclusively located in the cervical vertebrae and typically cause quadriplegia or, less commonly, paraplegia. In Australia, for example, diving accidents account for approximately 20 per cent of all cases of quadriplegia (Hill, 1984). The financial cost of these injuries to society is high, because those affected are frequently healthy young persons, principally males under 25 years of age (Blanksby *et al.,* 1997).

Data from the USA suggested that diving into a wave at a beach and striking the bottom was the most common cause of spinal injury, and 10 per cent of spinal injuries occurred when the person dived into water of known or unknown depth, particularly from high platforms, including trees, balconies and other structures (CDC, 1982). As with drowning, alcohol consumption may contribute significantly to the frequency of injury. Special dives, such as the swan or swallow dive are particularly dangerous because the arms are not outstretched above the head but to the side.

The role of water depth in determining the outcome of diving injuries has not been ascertained conclusively and the minimum depths for safe diving are often greater than expected. Technique and education appear to be important in preventing injury and inexperienced or unskilled swimmers require greater depths for safe diving. Most diving injuries occur in relatively shallow water (1.5 m or less) and a few in very shallow water (e.g. less than 0.6 m) where the hazard may be more obvious. The typical injurious dive occurs into a water body known to the individual.

Table 7.1 Drownings[1] and near-drownings - contributory factors and principal preventative and management actions

Contributory factors	Principal preventative and management actions
Alcohol consumption	Continual adult supervision (infants)
Cold	Provision of lifeguards
Ice cover	Availability of resuscitation skills/facilities
Waves (coastal, boat, chop)	Wearing of lifejackets when boating
Underwater entanglement	Provision of rescue services (lifeboats)
Pre-existing disease	Local hazard warning notices
Sea current (including tides, undertow and rate of flow)	Development of rescue and resuscitation skills amongst general public and user groups
Offshore winds (especially with flotation devices)	Development of general public (user) awareness of hazards and safe behaviours
Bottom surface gradient and stability	Access to emergency response (e.g. telephones with emergency numbers)
Impeded visibility (including coastal configuration, structures and overcrowding)	Co-ordination with user group associations concerning hazard awareness and safe behaviours
Water transparency	

[1] In most countries males and infants constitute a disproportionate number of drownings

Source: WHO, 1998

The principal contributory factors and preventative and management actions for spinal cord injury are summarised in Table 7.2. Evidence suggests that preventative education and awareness-raising offer most potential for diving injury prevention, partly because people have been found to take little notice of signs and regulations. However, because of the young age of many injured persons, awareness raising and education about safe behaviour is required early in life.

Table 7.2 Major impact injuries - contributory factors and principal preventative and management actions

Contributory factors	Principal preventative and management actions
Poor underwater visibility Conflicting uses in one area Bottom surface type Water depth Diving into a wave or into water of unknown depth Jumping into water from trees, balconies or other structures	Access to emergency services Use separation/segregation Provision of lifeguards Local hazard warnings Development of general public (user) awareness of hazards and safe behaviours Early education in diving hazards and safe behaviours

Source: WHO, 1998

7.1.3 Impact, slip, trip and fall injuries

Accidents involving limb fractures or breaks of different types have many causes and may occur in a variety of settings in or around water. The principal contributory factors and preventative and management actions are summarised in Table 7.3.

Table 7.3 Slip, trip, fall and minor impact injuries - contributory factors and principal preventative and management actions

Contributory factors	Principal preventative and management actions
Diving into shallow water Underwater objects (e.g. walls, piers) Adjacent surface type (e.g. water fronts, jetties) Poor underwater visibility	Selection of appropriate surface type Use of adjacent fencing (e.g. around docks and piers) Development of general public (user) awareness of hazards and safe behaviours

Source: WHO, 1998

7.1.4 Cuts, lesions and punctures

There are many reports of injuries sustained as a result of stepping on glass, broken bottles and cans. Discarded syringes and hypodermic needles may present more serious risks and may attract greater public outcry. Simple measures, such as the use of footwear on beaches, as well as adequate litter bins and cleaning operations may contribute significantly to prevention, as may educational policies to encourage users to take their litter home. The principal contributory factors and preventative and management actions are summarised in Table 7.4.

Table 7.4 Cuts, lesions and punctures - contributory factors and principal preventative and management actions

Contributory factors	**Principal preventative and management actions**
Presence of broken glass, bottles, cans, and medical wastes Walking and entering water barefoot	Development of general public (user) awareness of hazards and safe behaviours Development of general public (user) awareness regarding litter control Local availability of first aid Provision of litter bins Beach cleaning Adequate solid waste management

Source: WHO, 1998

7.1.5 Retinal dislocation

Impact to the head, resulting from diving and jumping into the water from height have been known to cause detachment of the retina in the eye. The principal contributory factors and preventative and management actions are summarised in Table 7.5.

Table 7.5 Retinal dislocation - contributory factors and principal preventative and management actions

Contributory factors	**Principal preventative and management actions**
"Bombing" (jumping onto other water users) Diving into water Jumping into water from height	Development of general public (user) awareness of hazards and safe behaviours

Source: WHO, 1998

7.2 Interventions and control measures

Control of physical hazards may involve their removal or reduction, if possible, or measures to prevent or reduce human exposure or to minimise the adverse effects of exposure. As described at the beginning of this chapter the term hazard is generally used in relation to the capacity of a substance or event to affect human health adversely. However, in the context of this chapter, the absence of appropriate control measures may be treated as a component of the chain of causation. For example, the lack of guards, rescue equipment, signs and other remedial actions can contribute to a variety of health outcomes.

The roles of various interventions and control measures in preventing human injury are discussed in the *Guidelines for Safe Recreational Water Environments* (WHO, 1998). The principal measures include public warnings and information (signs, flags, public information), lifeguarding, use separation (zoning, lines, designated areas), and infrastructure and planning, such as for emergency communication, rescue and resuscitation and emergency vehicle access. Whilst the requirement for each of these measures is largely determined locally by a variety of factors, it is important to note that most measures may be more or less effective; their effectiveness may decline after periods of limited or non-use and all are amenable to simple inspection. Importantly, for many measures, replacement or improvement may be within the capacity (financial or practical) of local authorities and, in some circumstances, user groups.

7.3 Monitoring and assessment

The assessment of hazards at a beach or water is critical to ensuring safety. The assessment should take into account several key considerations, which include:

- The presence and nature of natural or artificial hazards.
- The severity of the hazard in relation to health outcomes.
- The availability and applicability of remedial actions.
- The frequency, density and type of use of the area.
- The level of development.

The investigation of hazards in or near present or potential recreation areas, including land and water (natural and artificially constructed) results from a visual inspection procedure. The investigation of physical hazards involves an understanding of the process of causation leading to injury. Because of the importance of individual behaviours in causation, and of awareness in prevention, the involvement of the public, and of interest and user groups in particular (see Chapter 6), is especially important.

The assessment of hazards should take into account the severity and likelihood of health outcomes and the extent and density of use of the recreational area. Health risks that might be acceptable for a recreational area that is used infrequently and is undeveloped may result in immediate remedial measures at other areas that are widely used or highly developed.

Physical hazards vary greatly between sites. Monitoring of a site for existing and new hazards should be undertaken on a regular basis. The inspection and further investigation of hazards requires an understanding of the elements involved in such a programme. The identification of physical hazards, and the subsequent monitoring of any changes to the hazards depend upon potential and present water recreation areas and the hazards encountered. The purposes of inspection and investigation are to provide a routine, systematic, periodic and relevant verification of events, structures, conditions or other situations that represent hazards, whether “theoretical” or “actual” and under “real” conditions.

The following steps have been identified to evaluate an inspection process for hazards in recreational areas:

1. Determine what is to be inspected and how frequently.

2. Monitor changing conditions and use patterns regularly.

3. Establish a regular pattern of inspection.

4. Develop a series of checklists suitable for easy application throughout the system. Checklists should reflect national and local standards where they exist.

5. Establish a method for reporting faulty equipment and maintenance problems.

6. Develop a reporting and monitoring system that will allow easy access to statistics that record “when”, “where”, “why” and “how”.

7. Investigate the frequency of positive and negative results of inspections.

8. Motivate and inform participants in the inspection process through in-service training.

9. Use outside experts to review critically the scope, adequacy and methods of the inspection programme.

7.3.1 Inspection forms and checklists

Because hazards vary greatly and because of the importance of social and behavioural factors (in causation and in prevention), it is important that checklists and inspection forms are developed, tested and refined according to local priorities and experience. Based upon Tables 7.1 to 7.5 some of the factors that may be included in an inspection protocol are described in Table 7.6. Many factors of importance described above are not included in this list because they are not amenable to an inspection-based approach.

Table 7.6 Factors to consider when designing an inspection programme relevant to physical hazards and drownings in recreational waters

Hazard	Factors
Drownings and near-drownings	Sea current (including tides, undertow and rate of flow) Offshore winds (especially with flotation devices) Possibility of underwater entanglement Bottom surface gradient and stability Waves (coastal, boat, chop) Water transparency Impeded visibility (including coastal configuration, structures and overcrowding) Lifeguard provision Provision of rescue services (e.g. lifeboats) Access to emergency response services (e.g. telephones with emergency numbers) Local hazard warning notices Availability of resuscitation skills and facilities Rescue and resuscitation skills amongst user groups Co-ordination with user group associations concerning hazard awareness and safe behaviours Wearing of lifejackets when boating
Cuts and lacerations	Presence of broken glass, bottles, cans and medical wastes Frequency of beach cleaning Solid waste management Provision of litter bins Local availability of first aid
Spinal injuries	Bottom surface type Water depth Conflicting uses in one area Jumping into water from trees, balconies or other structures Underwater visibility Local hazard warnings General public (user) awareness of hazards and safe behaviours Early education in diving hazards and safe behaviours Level of separation/segregation Lifeguard supervision Access to emergency services
Slip, trip and fall accidents	Underwater objects (e.g. walls, piers) Underwater visibility Adjacent surface type (e.g. water fronts, jetties) Surface type selection Adjacent fencing (e.g. around docks and piers)

7.3.2 Timetabling of inspections

The frequency of inspection will vary according to the intensity of use of the area and the speed with which the hazards encountered and the remedial actions in place change at a specific location. The timing of inspections should take account of periods of maximum use (e.g. inspection in time to take remedial action before major holiday periods) and periods of increased risk. The frequency of inspection therefore has to be predicted based on the size of the facility, the number of features in the facility, and the extent of past incidents or injuries. The criteria for inspections and investigations may vary from country to country. There might be legal requirements and/or voluntary standard-setting organisations.

7.3.3 Reporting and notifications

The importance of co-ordination and participation of all interested individuals or organisations is emphasised in Chapters 5 and 6. Except where minimum legal requirements are specified, action to address deficiencies identified in inspections depends upon the goodwill of local authorities (local government, user groups and other interested parties). Maintaining co-ordination with such persons and authorities contributes greatly to the overall success of a monitoring programme in containing hazards and preventing adverse health effects.

Whilst much reporting is necessarily of a local nature, it is worthwhile to interpret and report findings at regional or national levels (where this is possible). Moreover, some approval schemes (see Chapter 5) stipulate either general requirements that management plans should be developed and implemented or that specific safety-related requirements should be met. Safety-related data may, therefore, contribute to informed personal choice (and thereby assist individuals in contributing directly to the protection of their own health) and also encourage local authorities to support safety-related improvements.

In addition to the benefits of reporting mentioned above, the availability of information concerning the existence of hazards and the deployment of remedial or preventative, measures may help to generate new insight into the effectiveness of those, and other, measures. Information on this aspect is limited at present.

7.4 Elements of good practice

• The nature and extent of any risk to, or potential hazard to, human health or well-being must be identified and characterised fully. The individual hazards must be related to a likely adverse health outcome.

• To assess the extent of risk, a suitable inspection protocol must be adopted. Such a protocol must define the components of a bathing area that may pose a risk to human health, cataloguing the water conditions, substratum, effects of climatic factors, infrastructure, management and regulatory regime, etc.

• The end-use of the bathing area, including carrying capacity and density of bathers, influences the outcome of the risk assessment.

• On completion of the initial assessment, appropriate control measures including management responses, must be defined, such as zoning, warning mechanisms and public information schemes, lifeguard provision and bathing area infrastructure.

• All situations that may give rise to adverse health outcomes at a bathing area should be reported in a consistent format and stored in an incident database that can be used to inform the level and nature of future management procedures.

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Chapter 8*: SANITARY INSPECTION AND MICROBIOLOGICAL WATER QUALITY

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Sanitary inspection, water quality determination and data analysis and interpretation are essential elements in characterising the microbiological safety of water in recreational areas. Sanitary inspection is a necessary adjunct to water microbiological analysis. A well-conducted sanitary inspection can identify sources of microbiological hazards, microbiological water quality data confirm the presence of hazards, and the two together allow an estimation of the risk of illness to bathers and other users. In assessing the microbiological quality of recreational waters, it will normally be necessary to conduct:

- An intensive sanitary inspection (only once as part of an assessment or annually in monitoring programmes).

- Periodic appraisal visits in which water quality analysis and shortened inspections are undertaken.

- Follow-up appraisals to investigate abnormal events, new sources of pollution and extreme values of pollution indicators.

One of the most important aspects of aquatic microbiology is related to several human diseases transmitted via water. The design and development of epidemiological surveillance studies described in Chapter 13 have led to the awareness of the magnitude of human morbidity and mortality associated with waterborne infectious diseases. The most relevant micro-organisms and the associated waterborne infectious diseases are summarised in the *WHO Guidelines for Safe Recreational Water Environments* (WHO, 1998). The derivation of guideline values for microbiological quality are also discussed in the *WHO Guidelines for Safe Recreational Water Environments* (WHO, 1998).

This present chapter deals with sanitary inspection, microbiological analytical methods and data handling and reporting. Strategies to implement sanitary inspections and recommendations for selection of the site and frequency of water sampling are given in Chapter 9. Specific methods for sampling and analysis are detailed in the following sections together with the different statistical procedures to express the overall microbiological water quality at a specific recreational water use area. It should be noted that a single beach or recreational area may vary widely in relation to microbiological measures of health risk within relatively short periods of time and thus the commonly used methods of defining a recreational water as passing or failing a defined microbiological standard has inherent limitations; these are discussed in this chapter and also in Chapter 9.

8.1 Sanitary inspection and sampling programmes

A sanitary inspection is a search for, and evaluation of, existing and potential microbiological hazards that could affect the safe use of a particular stretch of recreational water or bathing beach. It provides the foundation required to design and implement an effective water quality sampling programme and provides valuable information to assist in the interpretation of water quality data. In particular, it provides public health authorities with information to aid the selection of sampling locations, times and frequencies, in order to estimate more accurately water quality and therefore to allow for sound risk management decisions (see Chapter 9).

A comprehensive sanitary inspection of an existing recreational area should be conducted annually, just prior to the bathing season. The annual inspection should not only look for new sources of microbiological hazards but also review the adequacy of any sampling programme and corrective measures in place to deal with existing hazards. Further inspections should be conducted along with routine sampling during the bathing season, in order to identify recent events and their impact on water quality. During the peak bathing season additional inspections at different days and times of the day may provide a more complete picture of the bathing area.

Comprehensive inspections should be conducted prior to any major new or proposed activity which could significantly alter the microbiological quality of the water in an existing recreational water use area. A sanitary inspection should therefore be carried out as part of, or in response to, any proposal to expand or develop a new recreational bathing area. The findings of the inspection should receive prime consideration in any decision to proceed with development. A comprehensive annual sanitary inspection consists of four steps:

- Pre-inspection preparations.

- An on-site visit.

- The preparation of a preliminary report including recommendations on location of sampling sites and changes to the sampling frequency if necessary.

- The preparation of a final assessment report often in combination with water quality data.

While sampling, important field data can be obtained at each bathing area by inspecting specific sources of pollution. Microbial contamination may be suspected, for example, when inspection reveals abnormal colouration or odour of the water at the bathing site. In the Mediterranean coastal area where the influence of tides is minimal, changes in microbiological quality are mainly due to riverine and direct, especially urban, discharges at the bathing site. The microbiological contamination produced by long sea outfalls, if well designed, is normally diluted and should not influence the microbiological quality of the bathing area. Land-based sources of contamination are normally associated with smaller discharges or with the likelihood of heavy rain events, characteristic of the Mediterranean climate at the end of the summer period, where a great amount of water falls in a very short period of time. Heavy rain may wash out faeces from pastures or other agricultural land and directly influence microbiological water quality. Studies in other regions also document pulses of poor water quality associated with rainfall events (O'Shea and Field, 1992; Vonstille *et al.,* 1993; Armstrong *et al.,* 1997; Wyer *et al.,* 1994, 1995, 1997). In inland recreational waters the main sources of pollution are water inlets (PHLS, 1995). Therefore, influences from rivers, natural watercourses and, particularly around populated areas, combined sewer overflows, produce important changes in the microbiological quality of bathing waters. Sporadic malfunctioning of sewerage systems can produce similar problems (Davis *et al.,* 1995; Marsalek *et al.,* 1996). These events, if recent, can sometimes be recognised visually at the recreational site by changes in the appearance of the water. In marine recreational waters, a field analysis of the salinity can indicate the discharges of freshwater at the bathing site. Such measurements indicate indirectly, that land-borne discharges are occurring.

8.1.1 Pre-inspection preparations

The collection and review of any existing data or reports on the area, including reports of previous inspections, will allow a thorough and efficient on-site evaluation. Topographical maps and aerial photographs are useful tools for locating activities and features that could affect water quality and for establishing sampling sites. Historical data on tides, currents, prevailing winds, rainfall and discharges of sewage, storm overflows and combined sewer overflows, and urban and agricultural effluents should be collected and reviewed to determine the impact of these events, (either singly or collectively) on water quality. Depending on the availability of water quality data, experts conducting the annual inspection may need to collect samples for microbiological analyses. Therefore, adequate numbers of sterile sample bottles and sampling equipment should be readily available and prior arrangements should be made with the microbiology laboratory to process samples promptly after collection. Arrangements should be made to meet with user groups and with individuals in charge of any facility or activity that affects, or has the potential to affect, water quality in the recreational area. It will be essential to obtain the trust and co-operation of the groups or individuals if the survey is to provide an accurate assessment of water quality and to identify and remedy unacceptable water quality (see Chapter 6).

8.1.2 On-site visit

The purpose of the on-site visit is to identify and evaluate all existing and potential sources of microbiological contamination that could affect the safe use of the area. Attention should be paid to the presence of sewage disposal facilities, including long sea outfalls, industrial outfalls, seabird colonies, sanitary sewers, and rivers, tributaries, streams or ditches receiving sewage, storm water or agricultural runoff. All data recorded should be added to the catalogue of basic characteristics to form a catalogue of inspections that would enable the tracking of trends and influences (see Chapter 2).

Visual faecal pollution (including sanitary plastics), sewage odour and suspicious water colour should also be considered as an immediate indication of unacceptable water quality. Adjacent industries should also be identified and their impact assessed. The impact of local geography and meteorological conditions on water quality should also be evaluated. In most cases it will be necessary to collect representative water samples to confirm the presence of faecal pollution, to establish its variability and to identify the source. Non-toxic fluorescent tracer dyes, bacteriophages (such as PDR-1) or faecal sterol biomarkers (coprostanol and 24-ethylcoprostanol) may also be helpful to identify sources of contamination.

Epidemiological studies have shown that bathers can be a significant source of pathogenic micro-organisms (Seyfried *et al.,* 1985; Calderon *et al.,* 1991; Cheung *et al.,* 1991). In small bathing areas with a lot of bathers and a low rate of water turnover, the person to person disease transmission has to be considered, even if there is no source of faecal pollution from the outside. The assessment may therefore need to consider measures to control microbiological water pollution by bathers in the area. This is especially important in shallow, enclosed areas used by young children where water circulation and flushing rates are low. Intensive studies to locate sources of pollution and to propose remedial actions have been undertaken successfully (Wyer *et al.,* 1994; Tsanis *et al.,* 1995; Marsalek *et al.,* 1996). A specially designed form can assist in the process of comprehensive sanitary inspection (Box 8.1).

Box 8.1 Sanitary inspection form

Background information

Area name and code number: ______________________________
Location: ____
Type of water: Fresh ☐ Marine ☐ Estuarine ☐
Responsible authority: ____________________________________
Address: ____
Tel. _________ Fax. _________ E-mail _______________________

Laboratory of analysis:
Name: ____
Distance (km) _____ Sample transport time (h) ________________
Person responsible for samples during transport: _______________

What land or human activity surrounds the bathing area? (check all that apply)
Forest ☐ Fields ☐ Desert ☐ Hills ☐ Swamp ☐ River/stream/ditch ☐
Agriculture (specify) ☐ __________ Urban ☐ Commercial ☐
Residential ☐ Industry (specify) ☐ ____________ Hotel ☐
Harbour ☐ Airport ☐ Road/rail ☐ Military ☐ Waste tip ☐ Other ☐
Is the area surrounding the bathing area urban? ____

Additional details (historical information, reason for assessment, other contacts, etc.):
Size of bathing area: Area (m^2) ___ Length (m) ___ Mean width (m) ___
Is there a beach? ___ Average area (m^2) ___ Length (m) ___
Width (m) at high tide ___ Width (m) at low tide ___
Prevailing onshore winds: Direction ____ Typical speed (km/h) ____
Prevailing water currents: Direction ____ Typical speed ($m\ s^{-1}$) ____
Shoreline configuration ____ Presence of sandbars ____

Average wave heights: ____
Rainfall: Total annual ____ Seasonal patterns _______
Temperature:

Water: Average ___ Annual low __ Annual high ___
Air: Average ___ Annual low __ Annual high ___
Public facilities: No. of toilets ___ Showers ___ Drinking water fountains ___
Litter bins ___ (are they animal and/or bird-proof? ___)
Are methods in place to warn the public of danger? ____
Are the above facilities adequate?

Accessibility: Road ☐ Path ☐ No access ☐
Is there an adequate parking area? _____
Additional details ______________________________________

Microbiological hazards

a) Sewage and animal wastes. Is the water quality affected, or likely to be affected, by discharges from:

On-site or other private sewage disposal systems ☐ Communal sewage disposal or treatment facilities ☐ Long sea outfalls ☐ Agricultural activities ☐ Aquacultural activities ☐ Unconfined domestic or wild animals and birds ☐ Confined animals or birds (i.e. feedlots) ☐

Are discharges continuous or sporadic? ______

Is wastewater from toilets, showers, etc. likely to contaminate the bathing area? _____

Will typical bather densities impair water quality? ______

b) Storm water runoff. Is the water quality affected or likely to be affected by non-point discharges from:

Municipal storm drains or combined sewer overflows? ☐ Agricultural fields? ☐ Natural drainage? ☐
Are onshore winds likely to carry polluted water into the bathing area? _____
Are currents likely to carry polluted water into the bathing area? _____
Are tides likely to affect water quality in the bathing area? _____
Microbiological water quality data or additional information:__________

Note: Any of the above with a "yes" answer require a detailed investigation and risk analysis. This investigation should include:

- Proximity of potential contamination source to bathing area.

- Background and contamination incident flow rates.

• Effective rainfall which triggers contamination events (and typical duration of contamination).

• For discharges from sewage systems or treatment facilities, include what type of treatment is used, the system capacity, flow rates and variability, and indicator standards.

• For animals/birds, stocking densities and types of animals, indicator data will be necessary to support and supplement this information.

Chemical and other hazards

Water quality

Is the water likely to be affected by: Discharges from industrial sources? ☐ Agricultural drainage? ☐ Water craft mooring or use? ☐ Urban surface runoff? ☐
Are onshore winds likely to carry polluted water into the bathing area? ____
Are currents likely to carry polluted water into the bathing area? ____
Are tides likely to affect water quality in the bathing area? ____

Sand quality

Is the sand likely to be affected by: Discharges from industrial sources? ☐ Agricultural drainage? ☐ Water craft mooring or use? ☐ Urban surface runoff? ☐
Are plastic residues present? ___ Are tar residues present? ___
Are algae present? ___ Are other residues present? ___
Supporting chemical water quality data or additional comments: _______

Note: Any of the above with a "yes" answer require a detailed investigation and risk analysis. This investigation should include:

• Proximity of potential contamination source to bathing area.

• For boats, densities and pumpouts.

• For urban surface runoff, the effective rainfall.

• For discharge from industrial sites, the type of discharge, treatment being used, flow rates and variability, system capacity and chemical/indicator standards.

Please attach a map of the beach area included in this sanitary inspection, with possible contamination sources (rivers, storm drains, outfalls, etc.) marked. If possible, maps of the entire catchment area indicating land-use, topography, and infrastructure networks (i.e. wastewater and storm drain systems, etc.) should also be attached.

Reporting systems

Are there formal mechanisms for reporting waste discharges, spills, treatment bypasses, etc. to the local health authorities? ___

Is there an illness or injury reporting mechanism in place that would be effective for epidemiological investigations? ___

Sampling or posting recommendations. This section should describe circumstances which indicate the need to post warning notices or close beaches and provide information such as sampling locations, times and frequencies.

8.1.3 Sampling location, time and frequency

The first step in planning a sampling activity is to define clearly the objective; in most cases the objective will be either exploratory (assessment) or monitoring (surveillance). While the former is designed to provide preliminary or “one-off information about a site, the latter is undertaken for regulatory or non-regulatory purposes” (Keith, 1990). For a recreational water use area, both objectives may initially coincide. Exploratory sampling will be required to define subsequent sampling. Special requirements for epidemiological studies will be necessary as highlighted in Chapter 13.

The selection of sampling sites, time and frequency of sample collection should attempt to capture the overall microbiological quality of the water at the recreational water use area. These choices should be based upon the information gathered during the sanitary inspection. The selection of sampling stations and time of sampling should take into consideration, variables known to affect water quality, such as the length of the bathing area, presence and periodicity of point and non-point sources of faecal contamination, influences of local weather, the physical characteristics of the bathing area and the presence of bathers. For example, at bathing areas with no detectable sources of external faecal contamination, samples should be collected at the places with the greatest bather densities. Bathing areas known to be influenced by direct or indirect faecal contamination will require additional sampling sites to help define the degree and extent of pollution. The time of day can be an important source of variation (Brenniman *et al.,* 1981; Fleisher, 1985; Tillett, 1993; PHLS, 1995) especially at beaches with significant tides (Cheung *et al.,* 1991). Consideration should also be given to collecting samples at times when bather densities are greatest for example, afternoons at weekends (Cheung *et al.,* 1991; APHA, 1995). Chapter 9 gives an example of an approach to a sampling programme.

Sampling frequency can also influence the acquisition of reliable information on microbiological pollution in a bathing area (Fleisher, 1990; Tillett, 1993). For those laboratories with limited economic or human resources it is better to direct efforts towards increasing sampling frequency instead of confirming presumptive results for *Escherichia coli* and faecal streptococci. The sampling frequency adopted in many programmes and assessments is fortnightly during the bathing season. Some authors have advocated more frequent samplings such as weekly or more, especially in the peak season in temperate climates (Figueras *et al.,* 1997) and others maintain lower intensity monitoring (e.g. monthly) outside the bathing season. Evidence suggests that once an understanding of quality behaviour has been developed through relatively intensive monitoring and sanitary surveys, then reduced sampling frequencies may be justifiable and can contribute to reducing the burden of monitoring (Chapter 9). For colder climates where the bathing season is restricted by weather, water sampling should be concentrated in that period where historical data show a higher probability of favourable weather conditions for recreational activities. If abnormal favourable weather conditions appear, more frequent sampling should be carried out, especially in freshwater resources with poor water circulation that may be overcrowded under those circumstances.

Monitoring a bathing area or a site to reconfirm repeated failure to meet a guideline or poor water quality has little value. Equally, sampling frequency can be reduced when an area is known, through historical microbiological data, to have consistently good microbiological quality and when it is known from the catalogue of basic characteristics that it will not be subjected to pollution influences because potential sources of contamination are absent. In these situations only occasional confirmatory sampling will be required. Such an approach will direct resources to those beaches known to have variable water quality (see Chapter 9).

Resampling and new sanitary inspection, following the detection of unexpected peak values, is essential to establish the cause of the observed peak. An exhaustive investigation, including an inspection of the site and possible collection of additional samples to locate the source or sources of pollution, is also essential where the cause is known not to be due to a sporadic event. The effect of episodic events, such as heavy rainfall, on the water quality of bathing beaches, and the management response to such events, is discussed in Chapter 9.

8.2 Sampling

8.2.1 Sampling procedures

Sampling in chest depth water, typically 1.2-1.5 m depth, represents areas of greatest bather density although sampling at ankle depth may be appropriate to determine risk to young children.

Microbiological counts from surface samples have been shown to have a tendency to be higher than those beneath the surface (PHLS, 1995), but the epidemiological significance of this has yet to be studied. Therefore samples should be collected from beneath the surface. Precise sampling recommendations vary, for example 30 cm below the surface is indicated by the American Public Health Association (APHA) (APHA/AWWA/WPCF, 1992) and the European Community (EC) Directive (EEC, 1976), while the World Health Organization (WHO) and the United Nations Environment Programme (UNEP) (WHO/UNEP, 1994a) have proposed 25 cm. Every sample within a monitoring programme should be taken as near as possible to the defined sampling location.

Care must be taken to avoid external contamination during sample collection. Sterilised sample bottles should be opened with the opening facing downward and should be held by the base and submerged in the water. At the appropriate depth, the bottle should be turned upwards with the mouth facing the current (if any). After retrieving the bottle, some water should be discarded to leave an air space of at least 2.5 cm to allow mixing by shaking before examination (APHA/AWWA/WPCF, 1989; Bartram and Ballance, 1996). The utmost care must be taken at all times not to touch the top of the bottle during removal or replacement of the cap.

The sample volume should be sufficient to carry out all the required tests. In practice, 300-500 ml are adequate. If *Salmonella, Vibrio cholerae* or enteroviruses are to be analysed, as required by some authorities or under certain circumstances, greater volumes of water will be necessary (1.5 litres, 10 litres and 10 litres respectively). Bottles of borosilicate glass or suitable autoclavable plastic (PHLS, 1994; Bartram and Ballance, 1996) are recommended. They should have screw caps that withstand repeated sterilisation at 121°C or 180°C. Quality assurance procedures, as described in Chapter 4, should be followed. All sampling bottles should be correctly labelled with the reference of the sampling point. Additional information of the time of collection, temperature of water and other observations should be recorded on sample record sheets designed for this purpose.

8.2.2 Sample storage

There is little published information available that gives a consensus on the time limit for storage of samples to avoid changes in the concentrations of indicator organisms (Gameson and Munro, 1980; Tillet and Benton, 1993). Storage times should be as short as possible and it is recommended here that samples should be analysed as soon as possible, preferably within 8 hours of collection. If samples cannot be analysed within 24 hours field analysis should be considered. Immediately after collection, the samples should be stored in insulated boxes with cooling packs (prefrozen packs) and/or ice. Samples should be kept in the dark and the temperature of the cooling box maintained below 10°C where possible (APHA/AWWA/WPCF, 1995). This temperature may be difficult to reach and so in practice samples should be kept as cold as possible, but not frozen. In practical terms these storage conditions can have at best only a limited effect on reducing variations in bacterial populations. It is generally accepted that changes in microbial populations in water samples will begin to occur around 2 hours after collection; within 6 hours the samples are likely to have altered significantly particularly if no cooling mechanism was available and the samples were exposed to light. The key factor to consider in storage and transport of samples is time between collection and analysis rather than the time between collection and receipt at the laboratory. Ideally, the temperature of the insulated box should be controlled and recorded, as should the storage time. This information should be considered in the interpretation of results. Storage under these conditions should be as short as possible, and samples should be analysed promptly after collection.

8.3 Index and indicator organisms

Natural waters are subject to important changes in their microbial quality that arise from agricultural use, discharges of sewage or wastewater resulting from human activity or storm water runoff. Sewage effluents contain a wide variety of pathogenic micro-organisms that may pose a health hazard to the human population when the effects are discharged into recreational waters. The density and variety of these pathogens are related to the size of the human population, the seasonal incidence of the illness, and dissemination of pathogens within the community (Pipes, 1982). Appropriate indicators of faecal contamination under various conditions are discussed in Chapter 9 and in the *Guidelines for Safe Recreational Water Environments* (WHO, 1998).

Many waterborne pathogens are difficult to detect and/or quantify and the specific methodology to detect them in environmental water samples has still to be developed

(Borrego, 1994). While faecal streptococci are suggested as the recommended indicator for salt water, either faecal streptococci or *Escherichia coli* can be used for monitoring freshwaters. Additional variables can be investigated if they are considered relevant, such as the spores of *Clostridium perfringens* in tropical waters where the traditional indicators may increase in number in soil and water (Hardina and Fujioka, 1991; Anon, 1996). Staphylococci are generally assumed to serve as indicators of water pollution deriving from bathers themselves (i.e. by shedding from the body surface). The epidemiological significance of the recovery of Staphylococci remains unclear.

8.3.1 Thermotolerant coliforms and *E. coli*

Thermotolerant (faecal) coliforms constitute the subset of total coliforms that possess a more direct and closer relationship with homeothermic faecal pollution (Geldreich, 1967). These bacteria conform to all the criteria used to define total coliforms (all are aerobic and facultatively anaerobic, Gram-negative, non-spore forming rod-shaped bacteria that ferment lactose with gas and acid production in 24-48 hours at 36 ± 1°C), but in addition they grow and ferment lactose with production of gas and acid at 44.5 ± 0.2°C within the first 48 hours of incubation. For this reason, the term “thermotolerant coliforms” rather than “faecal coliforms” is a more accurate name for this group (WHO, 1993). The physiological basis of the elevated temperature phenotype in the thermotolerant coliforms has been described as a thermotolerant adaptation of proteins to, and their stability at, the temperatures found in the enteric tracts of animals (Clark, 1990). Thermotolerant coliforms include strains of the genera *Klebsiella* and *Escherichia* (Dufour, 1977). The thermotolerant coliform definition is not based on strictly taxonomic criteria, but on specific biochemical reactions or on the appearance of characteristic colonies on selective and/or differential culture media. Certain *Enterobacter* and *Citrobacter* strains are also able to grow under the conditions defined for thermotolerant coliforms (Figueras *et al.,* 1994; Gleeson and Gray, 1997). *E. coli* is, however, the only biotype of the family Enterobacteriaceae that is almost always faecal in origin (Bonde, 1977; Hardina and Fujioka, 1991). Therefore, the thermotolerant coliform group when used should ideally be replaced by *E. coli* as an indicator of faecal pollution. For the purpose of water testing, most *E. coli* can be confirmed by a positive indole test and by their inability to use citrate (as the only carbon source) in the culture medium. Alternatively, *E. coli* can be distinguished easily enzymatically by the lack of urease or presence of β-glucuronidase enzymes. The enzymes can be recognised easily using culture media that contain specific substrata (Gauthier *et al.,* 1991; Brenner *et al.,* 1993; Walter *et al.,* 1994).

However, several studies have indicated the limitation of both the thermotolerant coliform group and *E. coli* as ideal faecal indicators or pathogen index organisms. Several thermotolerant *Klebsiella* strains have been isolated from environmental samples with high levels of carbohydrates in the apparent absence of faecal pollution (Dufour and Cabelli, 1976; Knittel *et al.,* 1977; Niemi *et al.,* 1997). Similarly, other members of the thermotolerant coliform group, including *E. coli,* have been detected in some pristine areas (Rivera *et al.,* 1988; Ashbolt *et al.,* 1997) and have been associated with regrowth in drinking water distribution systems (Lechevallier, 1990). The principal disadvantages of this organism as an indicator in water are: (i) its detection in other environments without faecal contamination (Hazen and Toranzos, 1990; Hardina and Fujioka, 1991), and (ii) its low survival capability in aquatic environments when compared with faecal pathogens (Borrego *et al.,* 1983; Cornax *et al.,* 1990).

8.3.2 Faecal streptococci and enterococci

Faecal streptococci have received widespread acceptance as useful indicators of faecal pollution in natural aquatic ecosystems. These organisms show a close relationship with health hazards (mainly for gastrointestinal symptoms) associated with bathing in marine and freshwater environments, (Cabelli *et al.,* 1982, 1983; Dufour, 1984; Kay *et al.,* 1994; WHO, 1998). They are not as ubiquitous as coliforms (Borrego *et al.,* 1982), they are always present in the faeces of warm-blooded animals (Volterra *et al.,* 1986), and it is believed that they do not multiply in sewage-contaminated waters (Slanetz and Bartley, 1965). Enterococci, however, have been shown to grow in freshly stored urine (Höglund *et al.,* 1998). Nonetheless, their die-off rate is slower than the decline in coliforms in seawater (Evison and Tosti, 1980; Borrego *et al.,* 1983) and persistence patterns are similar to those of potential water-borne pathogenic bacteria (Richardson *et al.,* 1991). Reviews of all these aspects have been carried out by Sinton *et al.,* (1993a,b).

The group called faecal streptococci includes species of different sanitary significance and survival characteristics (Gauci, 1991; Sinton and Donnison, 1994). In addition, the proportion of the species of this group is not the same in animal and human faeces (Rutkowski and Sjogren, 1987; Poucher *et al.,* 1991). The taxonomy of this group, comprising species of two genera *Enterococcus* and *Streptococcus* (Holt *et al.,* 1993), has been subject to extensive revision in recent years (Ruoff, 1990; Devriese *et al.,* 1993; Janda, 1994; Leclerc *et al.,* 1996). Although several species of both genera are included under the term enterococci (Leclerc *et al.,* 1996), the species most predominant in polluted aquatic environments are *Enterococcus faecalis, E. faecium* and *E. durans* (Volterra *et al.,* 1986; Sinton and Donnison, 1994; Audicana *et al.,* 1995).

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 minutes, growth at pH 9.6 and at 6.5 per cent NaCl, and the ability to reduce 0.1 per cent methylene blue. The most common environmental species fulfil these criteria and thus in practice the terms faecal streptococci, enterococci, intestinal enterococci and *Enterococcus* group can be considered synonymous.

8.3.3 Alternative faecal indicators

The lack of a strong relationship between faecal indicators and health outcomes in a number of epidemiological studies in warm tropical waters may, in part, relate to the inappropriate nature of *E. coli* or faecal streptococci as indices of waterborne pathogens in these recreational waters. In this context an alternative index group, sulphite-reducing clostridia or spores of *Clostridium perfringens,* have been proposed and are used in Hawaii (Anon, 1996).

Spores of *C. perfringens* are largely faecal in origin (Sorensen *et al.,* 1989), they are always present in sewage (about 10^4-10^5 colony forming units (cfu) per 100 ml), they are highly resistant in the environment and appear not to reproduce in aquatic sediments (which appears to be the case with thermotolerant coliforms) (Davies *et al.,* 1995). It is interesting to note, however, that dog faeces may have some 9×10^8 cfu *C. perfringens* per gram dry weight (dw), whereas pig faeces are similar to humans (4.8×10^5 cfu *C. perfringens* per gram dw). *C. perfringens* is generally less common or absent in other warm blooded animals. Hence, although dogs have a similar number of thermotolerant coliforms and faecal streptococci to that found in humans, the relatively higher ratio of *C. perfringens* spores found in dog faeces may be a useful indicator when fresh faecal contamination is being investigated (Leeming *et al.,* 1998).

It is important to note that spores of *C. perfringens* do not act as an indicator for non-sewage or animal faecal contamination in general, and therefore they are only suitable as indicator organisms for parasitic protozoa and viruses from sewage-impacted waters (Payment and Franco, 1993; Ferguson *et al.,* 1996). Their resistance to disinfectants may also be an advantage for indexing disinfectant-resistant pathogens. Simple anaerobic culture is possible for *C. perfringens* spores after a short heat treatment to remove vegetative cells. Confirmation of their presence may be assisted by the addition of a methylumbelliferyl phosphate substrate to the growth medium (Davies *et al.,* 1995).

Other indicator organisms for sewage, but also specific for human sewage are the bacteriophages to *Bacteroides fragilis* HSP40. These *B. fragilis* phages appear to survive in a manner that is similar to the hardier human enteric viruses under a range of conditions (Jofre *et al.,* 1995; Lucena *et al.,* 1996). Their numbers in sewage such as the F-specific RNA bacteriophages may be an order of magnitude lower than various coliphages. Furthermore, only 1-5 per cent of humans may excrete these phages (Leeming *et al.,* 1998), and thus they may be unsuitable pathogen indicator organisms for small communities. The International Office for Standardization (ISO) standard methods for these phages are under final review (ISO, 1999c).

The ratio between thermotolerant coliforms and faecal streptococci has been proposed by Geldreich (1976) as a means of distinguishing between human and animal-derived faecal matter. However, this method is no longer recommended (Howell *et al.,* 1995) and none of the currently-used bacterial indicators distinguish different sources of faecal matter confidently when used alone (Cabelli *et al.,* 1983), although genetic typing of *E. coli* shows some potential (Muhldorfer *et al.,* 1996). Identification of human enteric viruses can identify specifically the presence of human faecal material although the necessary procedures are difficult and expensive, and not readily quantifiable. Other microbiological options include specific identification of phenotypes of *Bifidobacterium* spp. (Gavini *et al.,* 1991), *Bacteroides* spp. (Kreader, 1995), serotypes of F-specific RNA bacteriophages (Osawa *et al.,* 1981) or, as previously discussed, the bacteriophages to *Bacteroides fragilis* (Puig *et al.,* 1997). However none of these organisms are suitable for quantifying the proportion of human faecal contamination. Moreover, no one indicator or single approach is likely to represent all the facets and issues associated with faecal contamination of waters.

Recently, Leeming *et al.* (1994, 1996) demonstrated the ability to distinguish human from herbivore-derived faecal matter using a range of faecal sterol biomarkers (Table 8.1). The distribution of sterols found in faeces, and hence their source-specificity, is caused by a combination of diet, the animal's ability to synthesise its own sterols and the intestinal microbiota in the digestive tract. The combination of these factors determines “the sterol fingerprint”. The principal human faecal sterol is coprostanol (5β(H)-cholestan-3β-ol), which constitutes about 60 per cent of the total sterols found in human faeces. The C29 homologue of coprostanol is 24-ethylcoprostanol (24-ethyl-5β(H)-cholestan-3β-ol). In large quantities (relative to coprostanol), this faecal sterol is indicative of faecal contamination from herbivores. It is possible to determine the contribution of faecal matter from these two sources relative to each other by calculating the ratio of coprostanol to 24-ethylcoprostanol in human and herbivore (sheep and cow) faeces (Leeming *et al.,* 1996) and comparing these to ratios obtained for water samples (Leeming *et al.,* 1998). Other animals that are ubiquitous in urban areas such as dogs and birds, either do not have coprostanol in their faeces or have it in trace amounts only (Leeming *et al.,* 1994).

Table 8.1 Examples of faecal sterol biomarkers

Systematic name	Common name	Comments
C_{27} sterols		
5β-cholestan-3α-ol	Coprostanol	Human faecal biomarker; high relative amounts indicate fresh human faecal contamination
5β-cholestan-3α-ol	Epi-coprostanol	Present in sewage sludges; high relative amounts suggest older faecal contamination
cholest-5-en-3β-ol	Cholesterol	C_{27} precursor to 5α- and 5β-stanols
5α-cholestan-3β-ol	Cholestanol	The thermodynamically most stable isomer is ubiquitous; if the ratio of coprostanol to cholestanol is < 0.5, origin of 5β-stanols may not be faecal
C_{29} sterols		
24-ethyl-5β-cholestan-3β-ol	24-ethylcoprostanol	Herbivore faecal biomarker; high relative amounts indicate herbivore faecal contamination
24-ethyl-5β-cholestan-3α-ol	24-ethyl-*epi*-coprostanol	Present in some herbivore faeces
24-ethylcholest-5-en-3β-ol	24-ethylcholesterol	C_{27} precursor to 5α- and 5β-stanols
24-ethyl-5α-cholestan-3β-ol	24-ethylcholestanol	The thermodynamically most stable isomer is ubiquitous

Faecal sterols generally associate with particulate matter, and can be concentrated from 1-10 litres of water by simply filtering the water through a glass fibre filter (such as type OFF, Whatman). The lipids are extracted by acetone, concentrated, derivatised and quantified by gas chromatography. Thus the method requires a suitable chemistry laboratory and may cost ten times more than that for the analysis of *E. coli* and enterococci. Nonetheless, it is an appropriate method for specific studies investigating the proportion of human and animal faecal contamination.

8.4 Analytical methods

8.4.1 Most Probable Number

Most Probable Number (MPN) analysis is a statistical method based on the random dispersion (Poisson) of micro-organisms per volume in a given sample. Classically, this assay has been performed as a multiple-tube fermentation test. Although the technique is rather time consuming (taking between five and seven days), several laboratories prefer it to other methods of water analysis because it is applicable to all sample types.

The MPN technique is generally conducted in three sequential phases (presumptive, confirmatory, and complete), each phase requiring 1 to 2 days of incubation. In the initial or presumptive phase, three volumes of samples (usually 10, 1, and 0.1 ml) (Table 8.2 and Figures 8.1 and 8.2) are inoculated into 3, 5, or 10 tubes containing the appropriate medium to allow the target bacteria to grow (Figure 8.2; Tables 8.3 and 8.4). In this test, it is assumed that any single viable target organism in the sample will result in growth or a positive reaction in the medium.

Table 8.2 Recommended serial dilutions for water samples in relation to the degree of microbiological contamination and type of indicator

Type of water	Serial dilutions for thermotolerant coliforms[1]					Serial dilutions for faecal streptococci				
Sewage	10^{-2}	10^{-3}	10^{-4}	10^{-5}	10^{-6}	10^{-2}	10^{-3}	10^{-4}	10^{-5}	10^{-6}
Secondary effluent	10^{-1}	10^{-2}	10^{-3}	10^{-4}	10^{-5}	1	10^{-1}	10^{-2}	10^{-3}	
Contaminated bathing water	10	1	10^{-1}	10^{-2}	10^{-3}	10	1	10^{-1}		
Clean water	10	1	10^{-1}			100	10	1		

[1] E. Coli

Source: Anon, 1983

Figure 8.1 Preparation of a series of dilutions

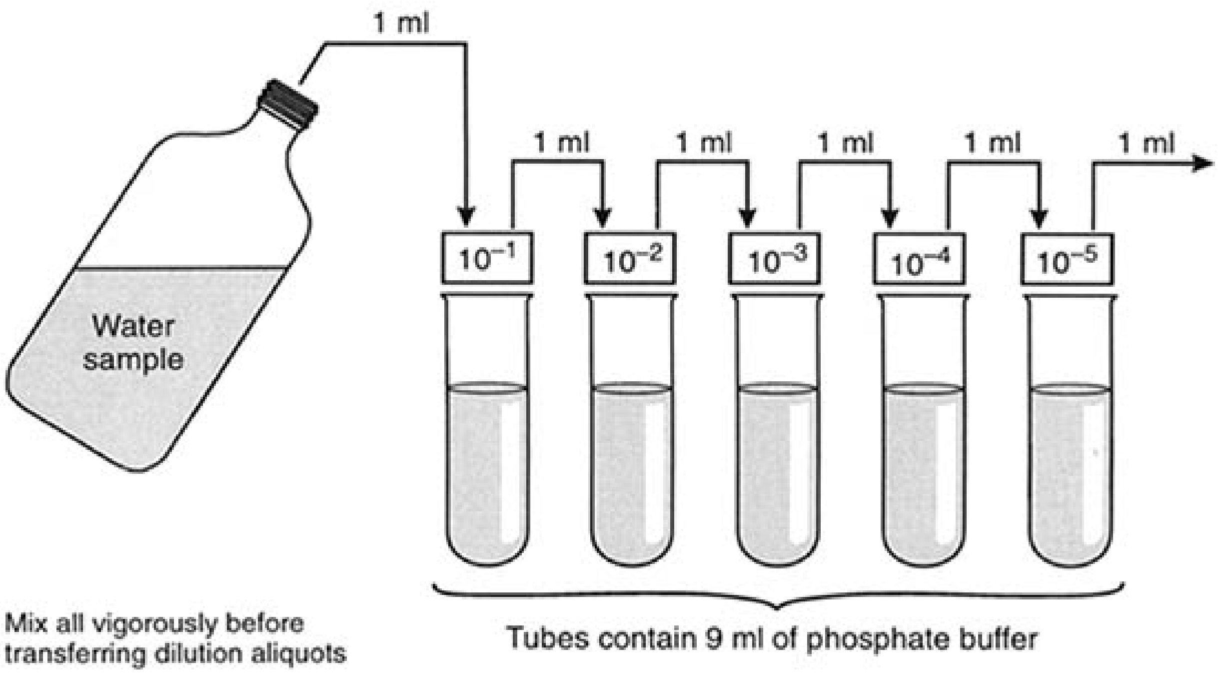
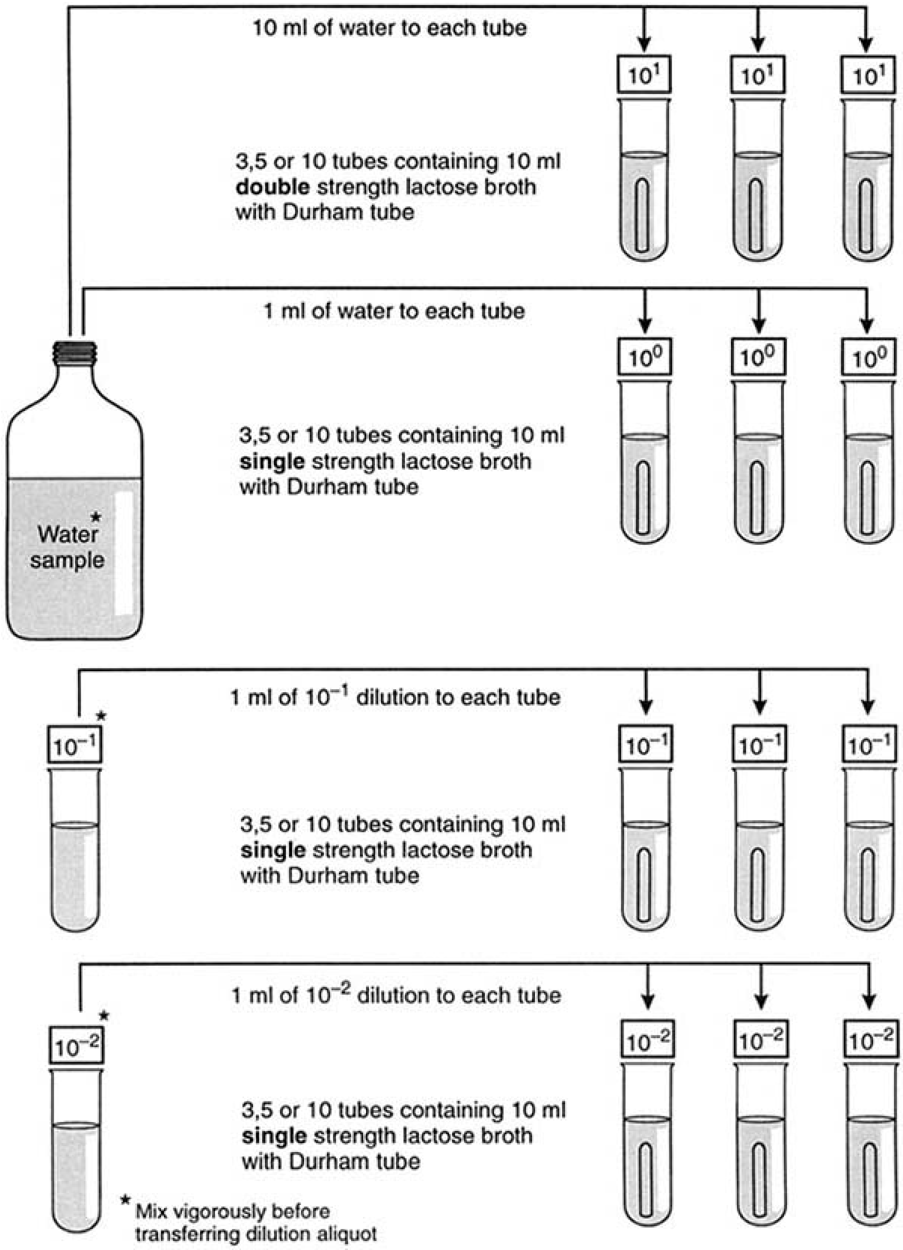


Figure 8.2 Inoculation scheme for the multiple test tube method



After the incubation period, all the inoculated presumptive positive tubes must be inoculated into a more selective medium to confirm the presence of the target bacteria (confirmatory phase). The confirmation test is reliable evidence, but not proof, that the target bacteria have been detected. Therefore, subsamples of the confirmed positive reactions should be inoculated onto a selective agar medium and several verification tests (Gram stain, and biochemical, serological or enzymatic tests) should be carried out (Tables 8.3 and 8.4). This completed test is generally conducted on 10 per cent of the positive tubes as a quality control measure. For practical purposes, the number of positive and negative tubes in the confirmatory phase of the technique is generally used to determine the MPN of the target bacteria by using tables of positive and negative tube reactions (WHO/UNEP, 1994a; APHA/AWWA/WPCF, 1995).

The major advantages of the MPN technique are (Fujioka, 1997):

- It will accept both clear and turbid samples.

- It inherently allows the resuscitation and growth of injured bacteria.

- The results may be recorded by personnel with minimal skill.

- Minimal preparation time and effort are required to start the test, and therefore processing of samples can be initiated at any time of the day.

By contrast the MPN technique may also have several disadvantages, such as:

- The total time, labour, material and costs required to analyse one sample.

- The substantial increase in reagents, tubes, incubation space and cleanup requirements when multiple samples need to be analysed or when the sample volume must be increased to 100 ml.

- The multiphase nature of the technique, each phase requiring a 24 hour or 48 hour incubation period.

- The fact that MPN is a simple estimated number, while the true number (95 per cent confidence limit) may show extreme variation from the MPN.

The choice of precision level of the technique (using 3, 5 or 10 tubes of each dilution) depends on the required detection sensitivity, because the total volumes analysed by each are 33.3, 55.5, and 111 ml, respectively. Miniaturised MPN methods with 96 incubation wells (e.g. ISO 1996a,b) are more precise than traditional five-tube tests with three descending decimal dilutions and equivalent to membrane filtration (Hernandez *et al.,* 1991, 1995). The existing standardised procedures for the MPN technique are given in Tables 8.3 and 8.4.

Table 8.3 Standard methods for the determination of thermotolerant coliforms (*E. coli* presumptive) - MPN methods

ISO 9308-2 (ISO, 1990c)[1]	**ISO 9308-3 (96 wells)**[2] **(ISO, 1998)**	**APHA**
Isolation media		
a) Lactose broth b) MacConkey broth c) Lauryl tryptose (lactose) broth d) Formate lactose glutamate medium	Tryptone salicine triton MUG broth (MU/EC)	a) EC medium, or b) A-1 medium
Incubation conditions	*Incubation conditions*	*Incubation conditions*
24-48 h at 35 ± 1 °C or 37 ± 1 °C	36-72 h at 44 ± 0.5 °C	a) 24 ± 2 h at 44.5 ± 0.2 °C b) 3 h at 35 ± 0.5 °C followed by 21 ± 2 hat 44.5 ± 0.2 °C
Reaction	*Reaction*	*Reaction*
Turbidity = (+)	Blue fluorescence = (+) for *E. coli*	a) and b) Gas production = (+) for thermotolerant coliforms
Confirmatory media tests		
Two confirmatory methods can be used:	Confirmatory tests are not required	If using EC medium, verify with the following test:
A. With two steps		
1. a) EC medium b) Brilliant green lactose (bile) broth		Brilliant green lactose (bile) broth
Incubation conditions		*Incubation conditions*
24 h at 44 ± 0.25°C or 44.5 ± 0.25°C		24 ± 2 h at 44.5 ± 0.2 °C
Reaction		*Reaction*
Gas production = (+) for thermotolerant coliforms		Gas production = (+) for thermotolerant coliforms
2. Tryptone water		If using A-1 medium, a confirmatory test is not required
Incubation conditions		
24 h at 44 ± 0.25°C or 44.5 ± 0.25°C		
Reaction		
Indol production with indol reagent Kovacs = (+) for *E. coli*		
B. With one step		
Lauryl tryptose mannitol broth with tryptophan		
Incubation conditions		
24 h at 44 ± 0.25 °C or 44.5 ± 0.25 °C		
Reactions		
Gas production = (+) and indol = (+/-) for thermotolerant coliforms; gas production = (+) and indol = (+) for *E. coli*		

MPN Most probable number

APHA American Public Health Association

MUG 4-methylumbelliferyl-β-D-glucoside

[1] ISO 9308-2 is at an early stage of revision by an ISO working group

[2] Not suitable for drinking water - lower limit of detection is 15 counts per 100 ml

Table 8.4 Standard methods for the determination of faecal streptococci (enterococci) - MPN methods

ISO 7899-1 (ISO, 1984a)[1]	APHA
Isolation media	
Azide dextrose broth	Azide dextrose broth
Incubation conditions	*Incubation conditions*
22 ± 2 h at 35 ± 1 °C or 37 ± 1 °C; negative tubes may be re-incubated for 22 ± 2 h	24 ± 2 h at 35 ± 0.5 °C; negative tubes may be re-incubated until 48 ± 3 h
Reaction	*Reaction*
Turbidity = (+)	Turbidity = (+)
Confirmatory media tests	
Two tests are recommended:	Two tests are recommended:
1. BEAA	1. PSE agar
Incubation conditions	*Incubation conditions*
44 ± 4 h at 44 ± 0.5 °C	24 ± 2 h at 35 ± 0.5 °C
	Reaction
	Brownish-black colonies with brown halos (+) for faecal streptococci
2. Catalase	2. BHIB containing 6.5% NaCl
	Incubation conditions
	24 h at 45°C
Reaction	*Reaction*
Dark brown to black colonies surrounded by black halos are (+) BEAA, with a (-) catalase test = faecal streptococci	Turbidity = (+) A (+) PSE with a (+) BHIB (6.5% NaCl) = enterococcus group

MPN Most probable number

APHA American Public Health Association

BEAA Bile esculin azide agar

PSE Pfizer selective enterococcus

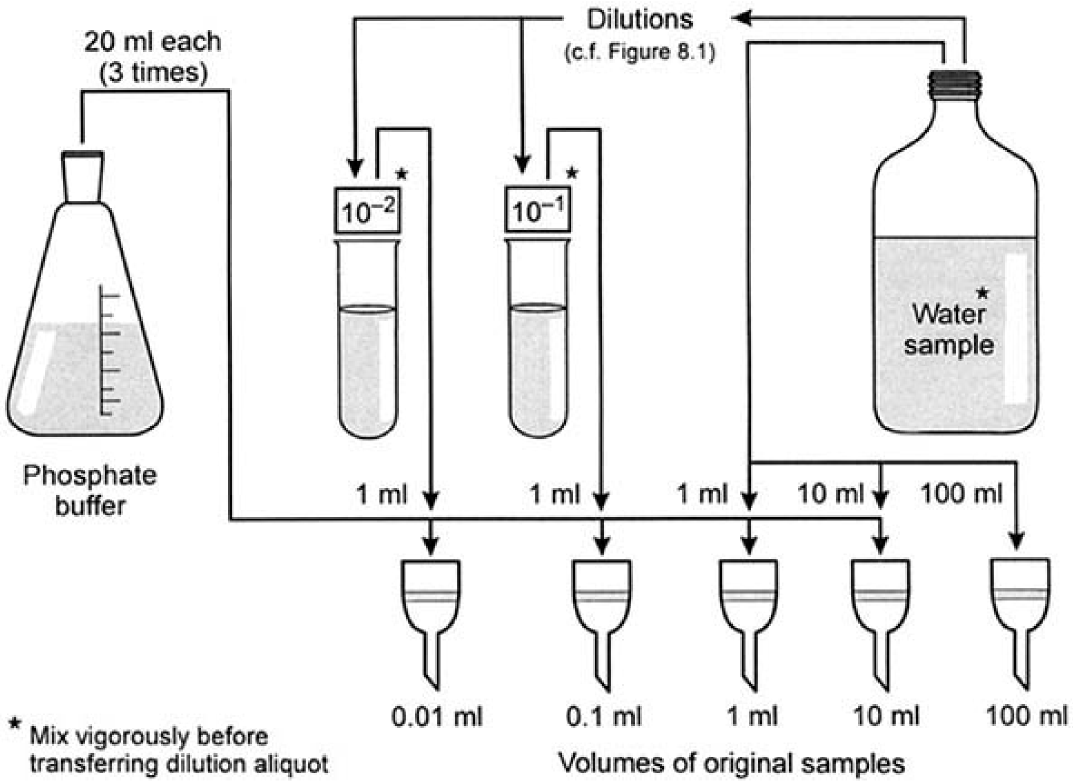
BHIB Brain heart infusion broth

[1] ISO 7899-1 has been replaced recently with a new methodology proposed under the same ISO reference as in Table 8.7

8.4.2 Membrane filtration

The membrane filtration (MF) technique is based on the entrapment of the bacterial cells by a membrane filter (pore size of 0.45 µm) (Figure 8.3). After the water is filtered, the membrane is placed on an appropriate medium and incubated (Tables 8.5, 8.6 and 8.7). Discrete colonies with typical appearance are counted after 24-48 hours, and the population density of the target bacteria, usually described as cfu per 100 ml in the original sample, can be calculated from the filtered volumes and dilutions used. This technique is more precise than the MPN technique, but the MF test can only be used for low-turbidity waters with low concentrations of background micro-organisms.

Figure 8.3 Preparation of dilution series and procedure for the membrane filtration method



The advantages of the MF technique include (Fujioka, 1997):

- Savings in terms of time, labour, and cost compared with the MPN technique.

- Direct determination of the concentrations of bacteria with high precision and accuracy.

- The formation of the target bacteria as colonies which can be purified for further identification and characterisation.

- The ability to process large volumes of water samples to increase greatly the sensitivity of this method.

Several disadvantages of the MF technique have also been reported:

• Inapplicability of the method to turbid samples which can clog the membrane or prevent the growth of the target bacteria on the filter.

• False negative results due to the inability of viable but non-culturable bacteria present in environmental waters to grow with standard MF methods.

• False positive results when non-target bacteria form colonies similar to the target colonies (Figueras *et al.,* 1994. 1996; Hernandez-Lopez and Vargas-Albores, 1994).

Table 8.5 Standard methods for the determination of thermotolerant coliforms (*E. coli* presumptive) - MF methods

ISO 9308-1 (ISO, 1990b)[1]	**APHA**
Isolation media	
a) TTC agar with Tergitol-7 or Teepol b) Lactose agar with Tergitol-7 or Teepol c) Membrane enrichment Teepol broth d) m-FC medium with 1% rosolic acid in 0.2N NaOH added e) Laurylsulphate broth	m-FC medium with 1% rosolic acid in 0.2N NaOH added (if there is interference with background growth)
Incubation conditions	*Incubation conditions*
18-24 h at 44 ± 0.25°C or 44.5 ± 0.25°C; a pre-incubation of 4 h at 30°C is recommended	24 ± 2 h at 44.5 ± 0.2°C
Reaction	*Reaction*
Depends on the media selected	Blue colonies = (+)
Confirmatory media tests	
Verify a representative number of colonies. Two confirmatory methods can be used:	Verify by picking at least 10 typical colonies; by two tests:
A. With two steps	
1. Lactose peptone water	1. Lauryl tryptose broth
Incubation conditions	*Incubation conditions*
24 h at 44 ± 0.25°C or 44.5 ± 0.25°C	24-48 h at 35 ± 0.5°C
Reaction	*Reaction*
Gas production = (+) for thermotolerant coliforms	Gas production = (+)
2. Tryptone water	2. EC broth
Incubation conditions	*Incubation conditions*
24 h at 44 ± 0.25°C or 44.5 ± 0.25°C	24 h at 44.5 ± 0.2°C
Reaction	*Reaction*
Indol production with indol reagent Kovacs = (+) for *E. coli*	Growth (+) and gas production = (+) for thermotolerant coliforms
B. With one step	
Lauryl tryptose mannitol broth with tryptophan	
Incubation conditions	
24 h at 44 ± 0.25°C or 44.5 ± 0.25°C	
Reaction	
Gas production = (+) and indol = (+/-) for thermotolerant coliforms; gas production = (+) and indol = (+) for *E. coli*	

MF Membrane filtration

APHA American Public Health Association

TTC Triphenyl-tetrazolium chloride

[1] ISO 9308-1 is under revision. Only one culture medium (a) has been chosen and is proposed under the same ISO reference in Table 8.7

Table 8.6 Standard methods for the determination of faecal streptococci (enterococci) - MF methods

ISO 7899-2 (ISO, 1984b)[1]	**APHA**
Isolation media	
a) KF streptococcus agar with 1% sterile solution of TTC added to cooled basal medium b) Slanetz-Bartley agar[2] with 1% sterile solution of TTC added to cooled basal medium	a) m-E agar for enterococci, or b) m-Enterococcus agar[2] for faecal streptococci
Incubation conditions	*Incubation conditions*
44 ± 4 h at 35 ± 1 °C or 37 ± 1 °C; however, if other types of micro-organisms are expected use 5 ± 1 h at 37 ± 1 °C followed by 44 ± 0.5°C until 48 h	a) 48 h at 41 ± 0.5 °C; transfer membrane fil- to esculin iron agar for 20 min ± 2 h at 41 ± 0.5 °C b) 48 h at 35 ± 0.5 °C
Reaction	*Reaction*
a) and b) Colonies: red, brown or pink (+)	a) and b) Colonies: pink to red (+)
Confirmatory media tests	
Verify a representative number of colonies by two tests:	Verify at least 10 well-isolated typical colonies by sub-culturing on:
1. BEAA	BHIA
Incubation conditions	*Incubation conditions*
48 h at 44 ± 0.5 °C	24-48 h at 35 ± 0.5 °C
2. Catalase	Transfer a loop-full of growth to:
Reaction	BHIB
Dark brown to black colonies surrounded by black halos are (+) BEAA, with a (-) catalase test = faecal streptococci	*Incubation conditions*
	24 h at 35 ± 0.5 °C
	A series of five tests are recommended for confirmation:
	1. Catalase
	2. Gram
	3. BEA
	Incubation conditions
	48 h at 35 ± 0.5 °C
	Reaction
	Growth = (+)

4. BHIB

Incubation conditions

48 h at 45 ± 0.5 °C

Reaction

Growth = (+)

5. BHIB containing 6.5% NaCl

Incubation conditions

48 h at 35 ± 0.5 °C

Reaction

Turbidity = (+)

Final reaction

A (+) BEA with a (+) BHIB 45 °C (test number 4) = faecal streptococci; a (+) BEA with (+) BHIB 45 °C (test number 4) and (+) BHIB 6.5% NCl (test number 5) = enterococci

MF Membrane filtration

KF KF streptococcus agar

BEAA Bile esculin azide agar

BHIB Brain heart infusion broth

APHA American Public Health Association

TTC Triphenyl-tetrazolium chloride

BHIA Brain heart infusion agar

BEA Bile esculin agar

[1] ISO 7899-2 is currently under revision; the new proposed version under the same ISO reference is given in Table 8.7

[2] Slanetz-Bartley has the same formulation as m-Enterococcus but the latter already includes TTC

Table 8.7 Recently proposed modifications to ISO standard methods

MF methods for conforms and *E. Coli* ISO DIS 9308-1 (ISO, 1997)[1]	**MF methods for intestinal enterococci ISO DIS 7899-2 (ISO, 1999b)[2]**	**MPN methods for intestinal enterococci ISO DIS 7899-1 (96 wells) (ISO, 1999a)[3]**
Isolation media	***Isolation media***	***Isolation media***
For a standard test use a lactose TTC agar with Tergitol-7, incubate for 21 ± 3 h at 36 ± 2°C[4]; typical colonies will turn the medium yellow For a rapid test use tryptone soya agar and incubate for 4-5 h at 36 ± 2°C[4]	Use a m-Enterococcus agar (Slanetz-Bartley) with a 1% sterile solution of TTC added to the cooled basal medium, incubate for 44 ± 4 h at 36 ± 2°C[4]; typical colonies are light and dark red	Use a medium with tryptose, nalidixic acid, TTC thallium acetate and MUD (MUD/SF medium); incubate for 36-72 h at 44 ± 0.5°C; fluorescence indicates intestinal enterococci

Confirmatory media tests	***Confirmatory media tests***	***Confirmatory media tests***
In the case of the standard test, verify all or a representative number of typical colonies (at least 10), using the following series of tests: 1. Non selective agar (i.e. tryptone soya agar); incubate for 21 ± 3 h at 36 ± 2°C[4] 2. Oxidase test; the non-appearance of a dark purple colour within 5-10 s indicates a negative result; a (-) oxidase = coliform bacteria 3. Tryptophane broth, incubate for 21 ± 3 h at 44 ± 0.5°C[4] Add indol reagent; indol production (i.e. a red ring) indicates a positive result A (-) oxidase and (+) indol = *E. coli* For a rapid test, transfer the membrane filter to tryptone bile agar, incubate for 19-20 h at 44 ± 0.5°C[4]; place the membrane filter on a filter paper saturated with indol reagent; the appearance of red colonies = *E. coli*	Transfer the membrane filter to a bile esculin azide agar, preheated at 44°C; incubate at 44 ± 0.5 °C for 1 h The appearance of dark brown to black colonies surrounded by black halos = intestinal enterococci	Tests are not required

MF Membrane filtration

MPN Most probable number

TTC Triphyl-tetrazolium chloride

MUD 4-methylumbelliferyl-β-D-glucoside

[1] Suitable for drinking water with low background growth

[2] Suitable for drinking water, swimming pools and other water with low intestinal enterococci

[3] Not suitable for drinking water; lower limit of detection is 15 counts per 100 ml

[4] These conditions substitute and standardise those from the previous ISO 9308-1 and 7899-2

E. coli has been demonstrated to be a more specific indicator for the presence of faecal contamination than the thermotolerant coliform group (Dufour, 1977). Improvements in both MPN and MF techniques have been carried out for the rapid and selective enumeration of *E. coli.* Barnes *et al.* (1989) designed a rapid seven hour membrane filter test for quantification of thermotolerant coliforms from drinking water samples and other freshwaters and salt waters, although it is not suitable for salt water due to the high proportion of false positives obtained. Fluorogenic and chromogenic tests using 4-methylumbelliferyl-β-D-glucuronide (MUG) have been applied in MPN and MF techniques, for the detection of β-glucuronidase that is specific to *E. coli* (Manafi and Kneifel, 1989; Balebona *et al.,* 1990; Gauthier *et al.,* 1991; Rice *et al.,* 1991). A miniaturised MPN method with a 96-well microplate has been developed for *E. coli* (Hernandez *et al.,* 1991; ISO, 1996b) (Table 8.3). Based on this principle a number of different media have been developed for the use in MF and MPN techniques (Frampton *et al.,* 1988; McCarty *et al.,* 1992). Commercially available media include Colisure (formerly Millipore, now IDEXX) (McFeters *et al.,* 1995), Colilert (IDEXX) (Edberg *et al.,* 1988; Palmer *et al.,* 1993), m-ColiBlue (Hach), ColiComplete (BioControl), Chromocult (Merck) and MicroSure (Gelman). Similar media for the detection of *E. coli* in water have also been described (Sartory and Howard, 1992; Brenner *et al.,* 1993; Walter *et al.,* 1994). Molecular methods have also been designed to detect specifically *E. coli* from water samples, such as PCR-gene probes for the *uid* gene (Bej *et al.,* 1991a,b; Tsai *et al.,* 1993; McDaniels *et al.,* 1996). In addition, other alternative techniques, i.e. enzyme capture (Kaspar *et al.,* 1987) and radioisotopes (Reasoner and Geldreich, 1989) have been proposed.

8.5 Laboratory procedures

8.5.1 Faecal streptococci and enterococci

Early attempts to quantify faecal streptococci relied on enrichment tube procedures and the MPN technique; Rothe Azide Dextrose broth followed by a confirmation in Ethyl Violet Azide (Litsky) broth being the procedure most widely accepted by researchers. A rapid system for enumeration of faecal streptococci or enterococci in water samples using a miniaturised fluorogenic assay based on a 96-well microplate MPN system has been described by several workers (Hernandez *et al.,* 1991; Poucher *et al.,* 1991; Budnicki *et al.,* 1996) and the technique has recently been proposed as an ISO method (Table 8.7). In addition, Enterolert (IDEXX) is available for the MPN technique with up to 100 ml of sample, and has been shown to be reliable (Fricker and Fricker, 1996).

The enumeration of faecal streptococci by a MF procedure using a selective medium was first reported by Slanetz and Bartley (1957). Since then, several media have been proposed, including Thallous Acetate agar (Barnes, 1959), KF agar (Kenner *et al.,* 1961), PSE agar (Isenberg *et al.,* 1970), Kanamycin Aesculin Azide (KEA) agar (Mossel *et al.,* 1973), mSD agar (Levin *et al.,* 1975), and mE agar (APHA/AWWA/WPCF, 1989). The accepted standardised procedures for the MF method are given in Table 8.6. Other media formulations and incubation procedures for faecal streptococci have been proposed for specific situations (Lin, 1974), such as increasing the membrane incubation period from 48 hours to 72 hours to recover stressed faecal streptococci. Rutkowski and Sjogren (1987) developed a medium, designated M2, to distinguish between human and animal pollution sources.

The methods for enumeration of faecal streptococci from natural waters have been compared by different authors (Volterra *et al.,* 1986; Yoshpe-Purer, 1989). Dionisio and Borrego (1995) compared eight methods for the specific recovery of faecal streptococci from natural freshwater and marine waters on the basis of the following characteristics: accuracy, specificity, selectivity, precision and relative recovery efficiency. The results obtained indicated that none of the tested methods showed perfect selectivity. The methods that showed the best performance characteristics were the MPN technique (with Rothe and Litsky media) and the m-Enterococcus agar in conjunction with the MF technique. The latter is the only technique recommended in the “Standard Methods” for faecal streptococci in conjunction with membrane filtration (APHA/AWWA/WPCF, 1995). A rapid confirmation technique, based on the transplantation of the membrane from m-Enterococcus agar after incubation for 48 hours at 36 ± 1°C to Bilis-Esculin-Agar (BEA) for 4 hours additional incubation, improves the low specificity of the m-Enterococcus agar, enabling the confirmation of 100 per cent of the colonies (Figueras *et al.,* 1996). A similar procedure has been proposed by ISO (Table 8.7).

Recently, Audicana *et al.* (1995) designed and tested a modification of the KEA agar, named Oxolinic acid-Aesculin-Azide (OAA) agar, to improve the selectivity in the enumeration of faecal streptococci from water samples by the MF technique. The OAA agar showed higher specificity, selectivity and relative recovery efficiencies than those obtained when using m-Enterococcus and KF agars. In addition, no confirmation of typical colonies was needed when OAA agar was used, which shortens the time taken significantly and increases the accuracy of the method. The excellent performance of this culture medium was recently reconfirmed in a routine monitoring programme for bathing waters (Figueras *et al.,* 1998). A Europe-wide standardisation trial demonstrated that the m-Enterococcus agar with total confirmation of the colonies (Figueras *et al.,* 1996), the OAA medium (Audicana *et al.,* 1995), and the miniaturised MPN method (Hernandez *et al.,* 1991) produced the best results (Hernandez *et al.,* 1995).

8.5.2 Thermotolerant coliforms and *E. coli*

The presumptive detection of thermotolerant coliforms can be considered sufficient to give an estimation for the presence *of E. coli.* The EC and A-1 media are the most widely recommended for the presumptive detection of thermotolerant coliforms with the MPN technique (APHA/AWWA/WPCF, 1992). The differences between the two approaches are based on the incubation periods: 44.5 ± 0.2 °C for 24 hours for the EC medium, and 36 ± 1 °C for 3 hours and transfer to 44.5 ± 0.2 °C for 21 hours for the A-1 medium. The tubes containing gas and acid in EC medium are confirmed in the same medium by subsequent incubation at 44.5 ± 0.2 °C for 24 hours. The A-1 medium does not require a confirmation test. Table 8.3 details accepted media. Thermotolerant coliform density and the 95 per cent confidence limits can be estimated with the use of MPN tables (APHA/AWWA/WPCF 1995; Bartram and Ballance, 1996).

The mFC agar is the most frequent medium used to quantify thermotolerant coliforms in water samples when the MF technique is used. Petri dishes containing filters are incubated at 44.5 ± 0.2 °C for 24 hours. Typical thermotolerant coliform colonies appear various shades of blue, atypical *E. coli* may be pale yellow, and non-thermotolerant coliform colonies are grey to cream in colour. Table 8.5 details accepted standardised media.

The existing ISO methods are now under revision. Table 8.7 shows the proposed modifications. The unification of temperature precision has been introduced by ISO as 36 ± 2 °C and 44 ± 0.5 °C.

8.6 Field analyses

Field analysis embraces all the tests that can be performed completely or partially at the site of sampling. Several field analysis techniques have been developed for drinking waters, where the principal requirement is the absence of indicator organisms (Manja *et al.,* 1982; Bernard *et al.,* 1987; Dange *et al.,* 1988; Dutka and El-Shaarawi, 1990; Smoker, 1991; Ramteke, 1995; Grant and Ziel, 1996). Quantitative on-site analysis using the MF technique is also possible (Bartram and Ballance, 1996).

The microbiological quality of bathing waters is presently assessed by the techniques described previously for indicator organisms. Field analyses may be preferred when the time between the collection of the sample and its examination will be long. Field laboratory equipment with filtration and incubation devices are being marketed. The time taken to obtain presumptive results will be the same as at a standard laboratory.

On-site filtration with a delayed incubation is another possibility when conventional procedures are impractical, i.e. when it is not possible to maintain the desired temperature during transport, and when the time between sample collection and analysis will exceed the optimum time limit. With this procedure, filters are placed in water tight plastic Petri dishes with a transport medium, and in conditions that maintain viability but will not allow visible growth. The test is completed at the laboratory by transferring the membranes to appropriate selective media and incubating them for the period of time required. It has to be recognised that growth will start if high temperatures are encountered during transport. Delayed incubation has been found to produce results consistent with those from immediate standard tests (Chen and Hickey, 1983, 1986; APHA/AWWA/WPCF, 1995; Brodsky *et al.,* 1995).

The continuous demands for more rapid techniques that can be performed on site and provide direct results have yet to be satisfied, despite the advances in analytical methods, particularly those based on DNA chips or arrays (Eggers *et al.,* 1997). By contrast, a one-hour assay for thermotolerant coli-forms has been demonstrated for marine bathing beaches, based on MUG detection of β-glucuronidase activity with a portable fluorometer (Davies and Apte, 1996, 1999).

8.7 Data recording, interpretation and reporting

Analysis may be performed as part of a regulatory monitoring programme, as part of a survey of an area used for water recreation, or as part of an epidemiological study in which water quality is related to risks to health from infectious diseases. Each approach has its own requirements which are specified at the outset by the regulatory authority or the study director. One of the most important functions of the analyst is to provide reliable and accurate results in a form that can be recorded for statistical interpretation and reporting, as described in Chapter 3. The following guidance will help the microbiological analyst achieve this aim.

8.7.1 Forms and records

An individual record must be produced for each site inspection or sample and should include the location and reference of the sampling site (that should ideally be equal to the code number of the sample), the date, the time, weather conditions, tide, water temperature, results of visual inspection for abnormal conditions, sources of contamination and the name of the inspector or sampler. Sampling records should also list the laboratory procedures and results (method of analysis, dilutions or volumes analysed, time of analysis, results for each step, any anomalies in the analysis of results and the name of the analyst). Ideally, record forms should occupy a single page. Forms should be conveniently archived, because they will be used later by the data handlers for transcription to the database and they will be analysed for the purposes of preparing the report of the monitoring or survey programme. Great care should be taken in preparing the report and its contents and format should be agreed by those responsible for analysis, data handling and for reporting results and, if necessary, by those responsible for co-ordinating results of regional, national and international programmes. For quality assurance, it should be possible to conduct an “audit trail” through the whole process of visiting the site, analysing the sample and filing the results on the database (see Chapter 4). An example of a record form for site inspections is given in Box 8.1.

8.7.2 Recording results of microbiological analyses

The results of microbiological analyses of water quality must always be regarded as an estimate of the water quality at the time and site of sampling, rather than as an absolute determination (PHLS 1994, APHA/AWWA/WPCF, 1995). All enumeration methods depend on the assumption that bacteria and other micro-organisms are randomly distributed in water samples and that the samples conform to the Poisson distribution (see Chapter 3). In reality, the clumping of bacteria and their aggregation on particles cause samples to depart from the Poisson distribution, thereby introducing additional error. There is little that can be done to reduce this error, apart from taking representative samples (free of sediment and other solid matter) and mixing the contents of the sampling bottles vigorously before taking sub-samples for analysis. It has been shown that two halves of the same sample can vary widely in the counts observed (PHLS, 1994).

A historical record of water quality in a bathing area, in normal and extreme situations, enables the selection of the most appropriate dilutions and facilitates the correct enumeration of the final density of indicator organisms that has to be reported as the total number of cfu per 100 ml.

Great care should go into the counting and recording of analytical results in order to avoid recording results wrongly, leading to errors that can result in statistical misinterpretations of water quality at the recreational area. Typical sources of error in the laboratory are caused by operator fatigue and mistakes, such as mislabelling of bottles, Petri dishes and tubes, and errors in preparing and transferring volumes and dilutions of samples. Because labels attached to the lids of Petri dishes, tubes and bottles can be transposed, the labels should be placed on the dish, tube or bottle itself, because these contain the culture medium. Although it may not be obvious, operators vary in the accuracy with which they count colonies and, in addition, unsuspected partial colour blindness can interfere with the interpretation of biochemical reactions of target colonies or in tubes of diagnostic media. Mistakes and errors can be minimised by proper training and supervision of samplers and analysts in their duties. In addition, laboratory quality controls and the careful application of standard procedures for analysis are essential. Standard procedures should be written correctly and copies should always be available for reference in the laboratory.

The multiple tube method is very sensitive for the detection of a small number of indicator organisms, but the MPN is not a precise value. Confidence intervals, i.e. most probable range (MPR), are often published with the MPN and are meant to indicate the imprecision of the method (Tillett, 1995). However, it should be stated clearly that the range applies to the sample and not to the water source (PHLS, 1994).

For thermotolerant coliforms and faecal streptococci membrane filters with 20-60 typical colonies are recommended for counting, with the provision that filters with no more than 200 colonies of all types should be considered; if the counts of colonies on the membranes are all below the minimum recommended, they should be totalled (APHA/AWWA/WPCF, 1995). Counting colonies on all filters has been shown to improve precision (Gameson, 1983) and is the only method specified in other standard procedures (ISO, 1988, 1990b). The count, in cfu per 100 ml, becomes the sum of all colonies counted multiplied by 100 ml and divided by the total volume (in ml) of water filtered.

If confirmation tests have been applied to a number of typical and atypical colonies, the initial count should be adjusted by multiplying it by the percentage of verified colonies. This percentage will be calculated by dividing the number of verified colonies by the total number of colonies subject to verification and then by multiplying the result by 100 (PHLS, 1994; WHO/UNEP, 1994a; APHA/AWWA/WPCF, 1995). This procedure has a considerable effect in reducing the precision of the count, depending on the total number of presumptive colonies selected and the fraction confirming (PHLS, 1994). Nevertheless, all colonies should be selected if there are ten or fewer colonies on a membrane. New proposed ISO methods try to overcome the imprecision by proposing verification methods for all the colonies grown on the filter (Table 8.7).

If the total number of bacteria colonies, including the specific target colonies, exceed 200 per membrane or are not distinctive enough to enable counting, results should be reported as “too numerous to count” (APHA/AWWA/WPCF, 1995) or “count too high to be estimated at the dilution employed” (PHLS, 1994). A new sample should be requested immediately if possible and more appropriate volumes should be selected for filtration (Table 8.2). The resultant data however, represent the results from a different sample; nevertheless the data may help to investigate the event. If this approach is not possible, it is preferable to try to count a sector of the original filter, before the information is lost thereby estimating the total number of colonies (even though these counts may lack precision). This technique is facilitated by the grid printed on the membranes. The details of the estimate should be recorded, for example “Count in n squares = x; diameter of filtration circle = d squares; estimated count = $\pi d^2 x/4n$ colonies”. If 135 colonies were counted in 10 squares and the diameter of the filtration area was 11.5 squares, the count would be estimated to two significant figures as $3.142 \times 11.5^2 \times 135/40 = 1,400$ colonies.

Although the statistical reliability of membrane filter results is higher than that of the MPN procedure, membrane counts are not absolute numbers; 95 per cent confidence limits can be calculated using a normal distribution equation (Fleisher and McFadden, 1980; PHLS, 1994; APHA/AWWA/WPCF, 1995).

8.7.3 Statistical procedures

A single water sample from a recreational area gives very little useful information. However, when individual results are accumulated and analysed statistically, then the trend in water quality will become apparent. Statistical analysis also enables evaluation of the improvement of water quality after remedial actions have been applied (e.g. to sewage contamination sources) and enables achievement of comparability between different regions within the same country or across countries. To establish comparability for the concentrations of thermotolerant coliforms from different regions, it is essential to agree what will be analysed (i.e. all the thermotolerant coliforms or only *E. coli*) otherwise comparison is impossible (Figueras *et al.,* 1994, 1997). The type of data analysis needed depends on the nature of the study (see Chapter 3).

For regulatory monitoring programmes, the objective of data analysis is to demonstrate compliance with a standard. The definition of the standard specifies the type of statistical analysis required. Most recreational water quality standards derive from those of the US EPA (Dufour and Ballentine, 1986), UNEP/WHO (1985) or the European Bathing Water Directive (EEC, 1976). More recently, WHO has published the *Guidelines for Safe Recreational Water Environments* (WHO, 1998). Microbiological standards typically specify the frequency of analysis and the number or proportion of samples that must not exceed given limiting values of the target organism. The rules for interpreting compliance differ with each standard. For microbiological surveys or epidemiological studies, the type of data analysis is decided at the planning stage. The procedures and statistics that are most often used to assess compliance during or after a bathing season are described below, together with worked examples from two sets of data from the two different bathing areas shown in Table 8.8, with the calculation of their basic statistics in Table 8.9.

Table 8.8 Comparability of methods for assessing compliance with microbiological quality criteria in two bathing areas

Method of assessing compliance	Interim criteria of quality[1] = 100 cfu per 100 ml 50%	
	Bathing area A	**Bathing area B**
Percentage compliance	Non-compliance (only 1 result complies)	Compliance (10 results comply)
Ranking method	590 non-compliance	8 compliance
Geometric mean[2] (95% confidence intervals)	597 non-compliance (216, 1,646)	16 compliance (4, 66)
Log normal distribution method[3]	680 non-compliance	13 compliance
50 percentile point 90 percentile point	597 non-compliance	15 compliance
50th percentile 90th percentile	595 non-compliance	20 compliance
Method of assessing compliance	**Interim criteria of quality[1] = 1,000 cfu per 100 ml 90%**	
	Bathing area A	**Bathing area B**
Percentage compliance	Non-compliance (only 8 results comply)	Compliance (11 results comply)
Ranking method	3,390 non-compliance	140 compliance
Geometric mean[2] (95% confidence intervals)	597 non-compliance (216, 1,646)	16 compliance (4, 66)
Log normal distribution method[3]	4,700 non-compliance	530 compliance
50 percentile point 90 percentile point	4,618 non-compliance	288 compliance
50th percentile 90th percentile	5,707 non-compliance	1,162 non-compliance

The sets of data for faecal coliforms obtained consecutively from the two bathing areas are as follows (in units of cfu per 100 ml):
A (16; 170; 3,390; 450; 450; 590; 740; 190; 1,180; 6,700; 2,800; 600) and

B (92; 1,600; 36; 0; 140; 4; 0; 36; 4; 8; 0; 32)

[1] According to the UNEP/WHO (1985) Interim Criteria for Recreational Waters, the concentrations of thermotolerant coliforms in at least 10 water samples should not exceed 100 cfu per 100 ml in 50% of the samples and 1,000 cfu per 100 ml in 90% of the samples

[2] The regulation that applies the geometric mean has only one standard in the interim criteria and not two as in the example and the geometric mean is calculated from at least 5 samples equally spaced over a 30 day period (running geometric mean) (US EPA, 1986)

[3] Data extracted from WHO/UNEP (1994b) and Anon (1983)

Percentage compliance

To assess percentage compliance (EEC, 1976) the regulatory percentage of the total number of data "n" obtained from a sampling station has to be calculated. The individual results of the set of data that comply with the established standards have to be counted in order to see if they are higher or lower than the compliance level. This is the approach used in the European Union. For thermotolerant coliforms the guideline standard is 100 cfu per 100 ml in 80 per cent of the samples and the mandatory standard is 2,000 cfu per 100 ml in 95 per cent of the samples (EEC, 1976). This approach is very easy to calculate but does not take into account the absolute values of all the microbiological counts and does not produce any average numerical value of the concentration of micro-organisms in the bathing area. For the worked example of Table 8.8, 50 per cent and 90 per cent of 12 samples is 6 and 10.8 (=11) respectively, which signifies the number of samples that must have counts under or equal to the standards associated to those percentages. In the case of bathing area A, only one result complies with the 50 per cent standard instead of the six needed, and only eight results comply with the 90 per cent standard when there should be 11. This bathing area is therefore failing the compliance assessment. Bathing area B complies with both the 50 and 90 per cent standards (Table 8.8).

Table 8.9 Basic statistics of worked examples from two sets of data obtained for thermotolerant coliforms from two bathing areas

Sample	Bathing area A			Bathing area B		
	Count[1]	Rank[2]	Log count	Count	Rank	Log (count + 1)[3]
1	16	1	1.20	92	10	1.97
2	170	2	2.23	1,600	12	3.20
3	3,390	11	3.53	36	8	1.56
4	450	4	2.65	< 1	1	0.00
5	450	5	2.65	140	11	2.15
6	590	6	2.77	4	4	0.70
7	740	8	2.87	< 1	2	0.00
8	190	3	2.28	36	9	1.57
9	1,180	9	3.07	4	5	0.70
10	6,700	12	3.83	8	6	0.95
11	2,800	10	3.45	< 1	3	0.00
12	600	7	2.78	32	7	1.52
Total	17,276	78	33.31	1,952	78	14.32
Average	1,440	-	2.7758	162.7	-	1.1933
SD[4]	1,965	-	0.6934	455	-	0.9883

SD Standard deviation

[1] cfu per 100 ml

[2] Ranks are given in ascending order

[3] This transformation has been used to enable the geometric mean and the log standard deviation to be calculated, given that three of the values are below the limit of detection, i.e. < 1 cfu per 100 ml

[4] Calculated as $s = \sqrt{\{[\Sigma x^2 - (\Sigma x)^2/n]/(n - 1)\}}$

Ranking method

The ranking method (WHO/UNEP, 1994b) is a very simple method because it involves ordering and multiplication operations, making the use of any complex formulae or laborious graphical analysis unnecessary. The interim UNEP/WHO Mediterranean criteria for recreational waters specify that thermotolerant coliform counts in at least ten samples taken during the bathing season must not exceed 100 cfu per 100 ml in 50 per cent of samples and 1,000 cfu per 100 ml in 90 per cent of samples (UNEP/WHO, 1985). The "*n*" values obtained are first ranked in ascending order of concentration (by definition, the order number, "*i*", takes values of 1 to *n*) (Table 8.9). Then the appropriate order numbers for a given percentage, P (i.e. 50 and 90 per cent) are calculated as $i = nx\mathrm{P}/100$. If ten samples have been taken, then the 50 per cent is measured directly against the fifth value of cfu per 100 ml in the ranking and the 90 per cent against the ninth value. If the number of samples taken does not give a whole number value, the result should be rounded to the nearest whole number to obtain the order. This concentration has to be lower than or equal to the specified standards to comply with the interim criteria. The order point in the rank for the 50 per cent criterion of 12 samples (examples of Table 8.8 and 8.9) is 50 × 12/100 = 600/100 = 6. The order point for the 90 per cent criterion is 90 × 12/100 = 1,080/100 = 10.8 = 11th position. Thus for bathing area A, the sixth point in rank order corresponds to a concentration of 590 cfu per 100 ml while the eleventh corresponds to a concentration of 3,390 cfu per 100 ml. For bathing area B, the corresponding values are 8 and 140 respectively. Whereas A fails both standards, B complies with both (Table 8.8).

Geometric mean

The other systems of interpreting water quality, i.e. the geometric mean with confidence intervals (US EPA, 1986) and the log-normal distribution method (WHO/UNEP, 1994b), are based on the fact that sets of microbiological data from sampling a recreational area are found to conform to a skewed positive distribution, because normally there are many low values and only a few high values.

The transformation of the microbiological counts obtained into decimal logarithms often produces a more symmetrical distribution. The proper descriptive statistic for central tendency is the geometric mean (equal to the median in the case of a normal distribution) with two associated measures of dispersion: the standard deviation of the logarithms of the values (the log standard deviation) and the 95 per cent confidence interval of the geometric mean.

The geometric mean is equal to the antilogarithm of the arithmetic mean of the logarithms of individual concentrations. In the USA it is considered to be the best estimate of the central tendency and the preferred statistic for summarising microbiological results (APHA/AWWA/WPCF, 1995). If there are values less than 1 cfu per 100 ml (i.e. 0 cfu per 100 ml) in the data set it will be impossible to calculate the geometric mean, because the logarithm of zero does not exist. In this instance one has to be added to all the results (+1) before their logarithmic transformation and then after the average of the logarithms is calculated and the antilog has been taken, the added value has to be subtracted. The calculation of the measures of dispersion are given in Box 8.2.

Box 8.2 Calculation of the measures of dispersion

The logarithmic standard deviation, s_l is calculated by entering the log values, x, into a scientific calculator, programmed to calculate the sample standard deviation. This can also calculated manually as $s_l = \sqrt{\{[\Sigma x^2 - (\Sigma x)^2/n]/(n - 1)\}}$. The 95 per cent confidence intervals of the geometric mean are calculated in two stages by the method below:

• The standard error (se) of the logarithmic mean "*m*", is calculated as $s_l/\sqrt{n}$.

• The 95 per cent confidence intervals are defined as $m \pm t_{(0.025)} \times$ se, where $t_{(0.025)}$ is the value of Student's t for $\alpha = 0.025$ and for *n* - 1 degrees of freedom (from statistical tables).

• For bathing area A the antilog of the log mean is antilog 2.7758 = 597 cfu per 100 ml (Tables 8.8 and 8.9), the standard error (se) of the logarithmic mean 2.7758 is 0.6934/√12 = 0.2001 and the 95 per cent confidence intervals of the log mean where $t_{(0.025)}$ is the value of Student's t for α = 0.025 and for *n* - 1 degrees of freedom (from statistical tables; for 11 degrees of freedom = 2.201) is 2.7758 ± 2.201 × 0.2001, giving 2.7758 - 0.4405 = 2.3353 and 2.7758 + 0.4405 = 3.2163 respectively. The antilogs of these values are 216 and 1,646 respectively.

• For bathing area B three counts are below the limit of detection, so the transformation count +1 had been applied to all counts before taking logarithms. The antilog of the log average 1.1933 is 15.6; and the estimated geometric mean is obtained by subtracting 1 from this, giving 14.6, rounded-off to 15. The standard error is 0.9883/√12 = 0.2853 and the 95 per cent confidence intervals of the log mean are thus 1.1933 ± 2.201 × 0.2853, giving results of 1.1933 - 0.6279 = 0.5654 and 1.1933 + 0.6279 = 1.8212. The antilogs of these are 4 and 66 respectively.

Log-normal distribution method

The log-normal distribution method involves the ranking of results and the transformation of data into logarithms to determine the log-normal distribution that fits most closely the experimental results. This can be done by hand fitting the data directly onto log-normal probability paper together with their corresponding cumulative frequencies. The concentration of micro-organisms corresponding to certain specified percentile points on the frequency distribution cumulative frequencies (50 per cent or 90 per cent) can be deduced by the graphic representation (WHO/UNEP, 1994b). This approach is quite similar to the calculation of percentile points on a continuous distribution of an infinite number of samples.

Percentile points, "*p*", on the distribution of the *n* data values from the mean "*m*", (or log mean) and the standard deviation "*s*" (or log standard deviation), as $p = m + zs$, where *z* is the standard normal variable for the desired percentile, obtained from tables of the quantiles (percentage points) of the standard normal distribution. The values of *z* for the 80-, 90- and 95-percentage points are 0.8416, 1.2816 and 1.6449 respectively. The value for the 50-percentage point will be equal to the mean “*m*”. For completeness, it can be noted that the standard normal distribution is symmetrical, so that the values of z for the 20, 10 and 5 percentile points are respectively -0.8416, -1.2816 and -1.6449. This approach can only be applied when the data follow a normal distribution, whereas calculation of classical percentiles does not require normality. For bathing area A, in the example, the 50 percentile point corresponds to the value of “*m*” (597), while the 90 percentile point is estimated by the log-normal distribution method from the log mean 2.7758 and the log standard deviation 0.6934 (Table 8.9). Hence, the log 90 percentile point is 2.7758 + 1.2816 × 0.6934 = 3.6645 and the antilog is 4,618 cfu per 100 ml. This value is quite similar to that obtained by plotting the data on the log-normal-normal distribution probability paper (4,700 cfu per 100 ml). For bathing area B, the 50 percentile point corresponds to a value of *m* = 15 cfu per 100 ml while the 90 percentile point is 1.1933 + 1.2816 × 0.9883 = 1.1933 + 1.2666 = 2.4599 and its antilog is 288 cfu per 100 ml.

The classical statistical calculation of percentiles estimates the variability of the distribution of a set of results (after ordering them in ascending order) independently of whether they are normally distributed and indicates the concentration of micro-organisms that embraces a specific percentile. For example, 80 per cent and 95 per cent of the data will be below the value (cfu per 100 ml) of the 80th and 95th percentiles, respectively, and 20 per cent and 5 per cent of the data will be above those concentrations, respectively. Some computer programs will not calculate the 95th percentile unless 20 records are available, although they will do so if the individual records are specifically considered the midpoint of an interval. An example of the calculation by hand is given in Box 8.3.

Box 8.3 Calculation of percentiles

$Pr = x_i + (j - i)(x_{i+1} - x_i)$

where:

Pr is the percentile required (i.e. P_{50}, P_{80}, P_{90} or P_{95})
x_i is the concentration that corresponds with an i position in the ranking corresponding to that Pr
j is the next position in the ranking (calculated as j = r (n + 1)/100)
x_{i+1} is the next concentration in the ranking
• For bathing areas A and B of Table 8.8 and 8.9, the calculation of j is:
50 (12 + 1)/100 = 650/100 = 6.5 = j for P_{50}, and
90 (12 + 1) = 1,170/100 = 11.7 = j for P_{90}
• For bathing area A:
x_i = 590 is the concentration at position 6 while 600 is the concentration at position x_{i+1} (= 7)

Then

P_{50} = 590 + (6.5 - 6) (600 - 590) = 590 + 5 = 595 cfu per 100 ml, and
P_{90} = 3,390 + (11.7 - 11) × (6,700 - 3,390) = 3,390 + 2,317 = 5,707 cfu per 100 ml

• For bathing area B, applying the same criteria:
P_{50} = 8 + (6.5 - 6) × (32 - 8) = 8 + 12 = 20 cfu per 100 ml, and
P_{90} = 140 + (11.7 - 11) × (1,600 - 140) = 140 + 1,022 = 1,162 cfu per 100 ml

8.7.4 Interpretation and reporting

Interpretation of results and reporting do not normally involve the analyst. Nevertheless, strict adherence to analytical control procedures make it possible for queries about unusual or anomalous results to be referred back to the analyst and sampler, through an audit trail.

Although the absolute values may differ with different approaches described, there is a high level of agreement on the water microbiological quality qualification of the beach in relation to compliance. Notice from Table 8.8 how, in bathing area A, only the geometric mean allows compliance in relation to a 1,000 cfu per 100 ml standard. However, in the regulations that govern the use of the geometric mean there is only one standard and the mean is calculated from five consecutive results taken over a period of one month (i.e. the running geometric mean). In this approach there is a strong influence from the most frequent values (8 of the 12 samples rank from 16 to 740 cfu per 100 ml). In the second example, bathing area B, the 90th percentile is the most restrictive, because this approach is highly influenced by the single high value obtained.

One of the features of microbiological studies of water quality at recreational areas is the wide variations in results, temporally and spatially (Fleisher and McFadden, 1980; Gameson, 1982; Tillett, 1993; PHLS, 1994) that are much greater than those caused by laboratory procedures. With effective quality control in the laboratory, variability caused by procedures, such as sub-sampling from the same bottle or by the enumeration procedures themselves, are little greater than expected from the assumption of random distribution of bacteria in the sample and of the Poisson theory, particularly when the mean number is low.

A single limit standard leads to water of borderline quality and with low variability, consistently passing, whereas water which is usually of high quality, but is occasionally affected by intermittent pollution, would fail, even though the former arguably poses a greater risk to health. More detailed study of the results in the latter case might identify the causes of failure and enable advice to be given to the public not to use the water when poor conditions are expected, or enable remedial action to be taken. Bathing area A shows consistently bad quality, with a higher geometric mean (597) and a lower log standard deviation (0.69) than bathing area B (15 and 0.98 respectively). The failure of bathing area B is caused by a single, unusually high count (1,600 cfu per 100 ml) that may be due to a rain effect on the second sampling date.

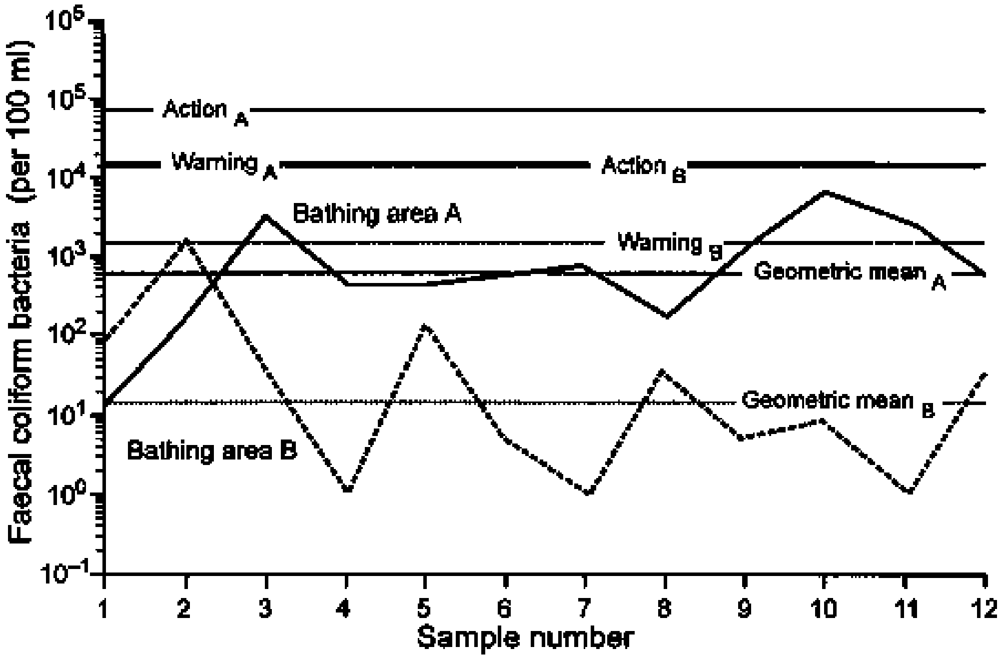
8.7.5 Control charts

One way of identifying systematic changes in water quality, or of pinpointing sudden deterioration of water quality at a recreational area, is to create control charts (see Chapter 4). Water quality data points are plotted sequentially, as they are obtained, against time on a chart. Any existing historical data from specific bathing areas can be used to create control charts and helps to identify patterns of behaviour. The occurrence of high values is used to initiate investigation. Such values may be set to coincide to a guideline standard. More conventionally, two upper limit values are set on a control chart, at the mean plus twice and three-times the sample standard deviation, i.e. at $m + 2s$ and $m + 3s$ respectively. These represent values that would be expected to be exceeded only once in 20 or 100 samples respectively, and that would indicate a warning (for example, the need for checking the results and/or for resampling) and then the need for taking remedial action (such as closing the beach until conditions improve and identifying and removing the source of pollution). The bacteriological data in Table 8.8 are presented as two control charts in Figure 8.4, showing the generally poor quality at bathing area A and the greater variability at B, which fails the 90 percentile criterion solely through the high count at the second sampling, even though it is otherwise of good quality.

8.7.6 Technical assessment report

Once the investigation (generally of sanitary conditions and water quality) or monitoring programme is completed, the information is assembled into a comprehensive report. The main body of the report should state the objectives; the manner in which the programme was conducted, with a full description of the recreational area; a historical account of problems and developments; the strategy and reports of inspection, sampling and analysis; significant results obtained; a discussion of the results; and conclusions and recommendations for action. Many of the readers will not have a technical background and therefore an easily readable and accurate "executive summary" should be provided at the front of the report. This gives such readers the main points of the report and invites them to follow-up areas of interest. The report itself should enable the technical reader to understand fully the way in which the study was carried out. Larger bodies of data should be placed in an appendix, so as not to interrupt the flow of the report. A typical report of a sanitary inspection and microbiological analysis includes the following: a description of the survey area(s) and sampling stations, and of any identified hazard(s) and source(s) of pollution (photographs and maps would be useful); the results of the study, including those of the sanitary inspections and microbiological analysis; an in-depth assessment of the risks associated with identified hazards and/or poor water quality; recommendations about the suitability of the area for recreational water use; description and evaluation of various options for improving conditions and thereby for reducing aesthetic and health hazards to users; and recommendations for action, including modifications if necessary, to the monitoring programmes.

Figure 8.4 Counts of faecal coliform bacteria per 100 ml at bathing areas A and B (see Table 8.8) displayed as control charts, with warning and action limits at two and three logarithmic standard deviations respectively above geometric mean counts



The results and recommendations should be discussed with any interested parties before the report is released formally. A contingency plan should also be developed, with the assistance of any interested parties, to investigate and respond to cases of illness or to any unforeseen event or condition that could lead to a deterioration in water quality and possibly increase the risk of illness or danger to bathers. Consideration should also be given to preparing a nontechnical report for the general public.

8.8 Quality control

All laboratories should guarantee that the results of the microbiological analysis of a water sample actually originated from the sample and were not introduced accidentally during sampling or analysis. To support this guarantee, internal and external quality controls should be implemented (Tillett and Lightfoot, 1995). Quality control is described in detail in Chapter 4. Internal quality control includes constant monitoring of equipment (pH meters, balances, pipettes, sterilising equipment, incubators, etc.) and reagents (membrane filters, culture media, buffer solutions, etc.) using controls and reference materials (PHLS, 1994; APHA/AWWA/WPCF, 1995; Janning *et al.,* 1995). Working practice should also be included in this quality control, as well as the precision of the techniques (MPN and MF techniques). In addition, controls have to be made at regular intervals (PHLS, 1994; WHO/UNEP, 1994c). External quality controls are meant to establish good performance by comparing laboratory results with those of other laboratories testing the same artificially prepared sample (Tillet *et al.,* 1993; PHLS, 1994).

8.9 Presenting information to the public

Aspects of public information are considered in Chapter 6. It is sufficient to note here that the quality of recreational water is of great public concern and is often used in publicity to attract visitors to recreational areas. Several countries have developed regular information services, using television, teletext, newspapers and radio (EEC, 1996) to supplement bulletins in municipal buildings and on public notice boards at the recreational areas. The implementation of a monitoring programme with these characteristics is described by Figueras *et al.* (1997). Generally, the public simply want to know if it is safe to use the water and most people have little understanding of the meaning of bacterial counts, let alone their variability. The information presented to the public, therefore, should be direct and unequivocal, up-to-date and not open to misinterpretation.

8.10 Elements of good practice

• Sanitary inspection should be undertaken as a necessary adjunct to microbiological analysis of waters to identify all real and potential sources of microbiological contamination. It should assess the impact of any microbiological contamination present on the quality of the recreational water and on the health of bathers. During the inspection, the temporal and spatial influences of pollution on water quality should receive full consideration.

• An exhaustive sanitary inspection should be carried out immediately prior to the main bathing season. Inspections of specific conditions should be conducted in conjunction with routine sampling during the bathing season. Pertinent information should be recorded on standardised checklists and used to update the catalogue of basic characteristics. If a problem is identified, it may be necessary to collect supplementary samples or information to characterise the problem.

• Visual faecal pollution or sewage odour should be considered a definite sign of elevated microbiological pollution and the necessary steps should be taken to prevent health risks to bathers.

• Standard operating procedures for sanitary inspections, water sampling (including depth) and analyses should be well described to ensure uniform assessments.

• Sample point location and the distance between each location should reflect local conditions (overall water quality, bather usage, predicted sources of faecal pollution, temporal and spatial variations due to tidal cycles, rainfall, currents, onshore winds and point or non-point discharges) and may vary widely between sites.

• Sterile sample containers should be used for microbiological samples. Scrupulous care should be taken to avoid accidental contamination during handling and during sample collection. Every sample should be identified clearly with the time of collection, date and location.

• The most appropriate depth for sampling should be selected and adhered to consistently.

• The sample should be kept in the dark and maintained as cool as possible within a chilled insulated container and returned to the laboratory promptly after collection. Samples should be analysed as soon as possible and preferably within 8 hours of collection. It is recommended that samples should not be stored for more than 24 hours at 5°C.

• Additional information should be collected at the time of sampling, including: water temperature, weather conditions, water transparency, presence of faecal material, abnormal colouration of the water, floating debris, cyanobacterial or algal blooms, flocks of sea birds and any other unusual factors. All information should be recorded on standardised checklists.

• The minimum microbiological variables that should be investigated are faecal streptococci or enterococci and thermotolerant coliforms or *E. coli.* While the former is a recommended indicator for salt water both can be used for freshwater. Additional variables should be investigated if considered relevant and if resources allow.

• The influence of specific events, such as the influence of rain on the recreational water use areas, should be established particularly in relation to the duration of the peak contamination period.

• Extreme events, such as epidemics and natural disasters, may require additional measures to ensure there is no additional risk associated with recreational water use areas.

• The procedures to be used for transformation of raw data, to meet the statistical requirements, should be agreed with the statistical expert prior to analysis. It is usually necessary to transform bacterial counts to logarithms and to convert their approximately log-normal frequency distribution to normality.

• When unexpectedly high microbiological results are obtained, resampling should be carried out to determine whether the unexpected results were due to sporadic events or persistent contamination. In the latter case, the source of pollution should be established and appropriate action taken.

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Chapter 9*: APPROACHES TO MICROBIOLOGICAL MONITORING

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* *This chapter represents the conclusions of a meeting of experts organised by WHO in co-operation with the US EPA and held in Annapolis, MD, USA, in November 1998 (for further details, see the Acknowledgements section). The US EPA has not conducted a policy and legal evaluation of this chapter.*

Despite evident successes in the protection of public health, present approaches to the regulation of recreational water quality suffer several limitations. A modified approach to regulation of recreational water quality could provide for improved protection of public health with the minimum necessary monitoring effort and could provide greater scope for interventions, especially for those within the resources of local authorities. This chapter describes the principal issues discussed at a meeting of experts, who concluded that such an alternative was possible and should be tested and promoted.

9.1 Issues

9.1.1 Current regulatory schemes

Recreational water standards have had some success in driving cleanups, increasing public awareness, contributing to informed personal choice and contributing to a public health benefit. These successes are difficult to quantify, but the need to control and minimise adverse health effects has been the principal concern of regulation. Present regulatory schemes for the microbiological quality of recreational water are primarily or exclusively based on percentage compliance with faecal indicator counts. Examples of compliance criteria currently in use are given in Table 9.1. A number of constraints are evident in the current standards and guidelines:

• Management actions are retrospective and can only be deployed after human exposure to the hazard.

• The risk to health is primarily from human excreta, the traditional indicators of which may also derive from other sources.

• There is poor inter-laboratory and international comparability of microbiological analytical data.

• While beaches are classified as safe or unsafe, there is a gradient of increasing severity, variety and frequency of health effects with increasing sewage pollution and it is desirable to promote incremental improvements prioritising "worst failures".

Table 9.1 Examples of guidelines and standards for microbiological quality of water (number of organisms per 100 ml)

Country	**Primary contact recreation**			**Shellfish harvesting**		**Protection of indigenous organisms**		**Reference(s)**
	TC	**FC**	**Other**	**TC**	**FC**	**TC**	**FC**	
Brazil	80% < 5,000[1]	80% < 1,000[1]			100% < 100			Ministerio del Interior, 1976
Colombia	1,000	200						Ministerio de Salud, 1979
Cuba	1,000[2]	200[2] 90% < 400						Ministerio de Salud, 1986
Ecuador	1,000	200						Ministerio de Salud Publica, 1987
Europe, EEC[3]	80% < 500[4] 95% < 10,000[5]	80% < 100[4] 95% < 2,000[5]	Faecal streptococci 100[4] *Salmonella* 0 per litre[5] Enteroviruses 0 PFU per litre[5] Enterococci 90% < 100 Faecal streptococci < 100					EEC, 1976
France	< 2,000	< 500						CEPPOL/UNEP, 1991
Israel	80% < 1,000[6]							
Japan	1,000							WHO/UNEP, 1977
Mexico	80% < 1,000[7] 100% < 10,000[9]							INCYTH, 1984
Peru	80% < 5,000[7]	80% < 1,000[7]		70		1,000		Environmental Agency, 1981
Poland			*E. coli* < 1,000	70[8] 90% < 230		10,000[8] 80% < 10,000 100% < 20,000		SEDUE, 1983
Puerto Rico		200[10] 80% < 400		80% <	80% < 200	80% < 20,000	80% < 40,000	Ministerio de Salud, 1983

				1,000	100% < 1,000	
USA California	80% < 1,000[11,12] 100% < 10,000[9]	200[2,12] 90% < 400[13]				WHO, 1975
US EPA			Enterococci 35[2] (marine), 33[2] (fresh) *E. coli* 126[2] (fresh) *E. coli* < 100	70[10] 80% < 230		JCA, 1983
Former USSR				70[8]		California State Water Resources Board, undated
UNEP/WHO		50%<100[14] 90% < 1,000[14]			14[2] 90% < 43	US EPA, 1986; Dufour and Ballentine, 1986
Uruguay		< 500[14] < 1,000[15]				WHO/UNEP, 1977
Venezuela	90% < 1,000 100% < 5,000	90% < 200 100% < 400			80% < 10 100% < 100	WHO/UNEP, 1978
Yugoslavia	2,000					DINAMA, 1998
				70[2] 90% < 230	14[2] 90% < 43	Venezuela, 1978
						INCYTH, 1984

TC Total coliforms
FC Faecal or thermotolerant coliforms

1 "Satisfactory" waters, samples obtained in each of the preceding 5 weeks

2 Logarithmic average for a period of 30 days of at least 5 samples

3 Minimum sampling frequency - fortnightly

4 Guide

5 Mandatory

6 Minimum 10 samples per month

7 At least 5 samples per month

8 Monthly average

[9] No sample taken during the verification period of 48 hours should exceed 10,000 per 100 ml

[10] At least 5 samples taken sequentially from the waters in a given instance

[11] Period of 30 days

[12] Within a zone bounded by the shoreline and a distance of 1,000 feet from the shoreline or the 30 foot depth contour, whichever is further from the shoreline

[13] Period of 60 days

[14] Geometric mean of at least 5 samples

[15] Not to be exceeded in at least 5 samples

Source: Adapted from Salas, 1998

Table 9.2 Outbreaks of disease associated with recreational waters in the USA, 1985-1994

Etiological agent	No. of cases	No. of outbreaks
Shigella	935	13
E. coli	166	1
Leptospira	14	2
Giardia	65	4
Cryptosporidium	418	1
Norwalk virus	41	1
Adenovirus 3	595	1
Acute gastro-intestinal infections	965	11

Sources: Morbidity and Mortality Weekly Report, 1988, 1990, 1991, 1993; Kramer, *et al*., 1996

The present form of regulation tends to focus upon sewage treatment and outfall management as the principal or only effective interventions. Because of the high costs of these measures, local authorities may be disenfranchised and few options for effective local intervention in securing bather safety from sewage pollution may be available. The limited evidence available from cost-benefit studies of pollution control alone rarely justifies the proposed investments. The costs may be prohibitive or may detract resourcing from greater public health priorities (such as securing access to a safe drinking water supply), especially in developing countries. If pollution abatement on a large scale is the only option available to local management, then many will be unable to undertake the required action.

Considerable concern has been expressed regarding the burden (cost) of monitoring, primarily but not exclusively to developing countries, especially in light of the precision with which the monitoring effort assesses the risk to the health of water users and the effectiveness with which it supports decision-making to protect public health.

9.1.2 Pathogens

There is a broad spectrum of illnesses that have been associated with swimming in marine and fresh recreational waters. Table 9.2 is a list of microbes that have been linked to swimming-associated disease outbreaks in the USA between 1985 and 1994. Two bacterial pathogens, *Escherichia coli* and *Shigella,* and two pathogenic protozoans, *Giardia* and *Cryptosporidium,* are of special interest because of the circumstances under which the associated outbreaks occurred. These outbreaks usually occurred in very small, shallow bodies of water that were frequented by children. Epidemiological investigations of the outbreaks found that the source of the etiological agent was usually the bathers themselves, most likely children. Each outbreak affected a large number of bathers, as might be expected in unmixed, small bodies of water containing large numbers of pathogens.

Table 9.3 Serological response to Norwalk virus and rotavirus in individuals with recent swimming-associated gastro-enteritis

Antigen	No. of subjects	Age range	No. with 4-fold titer increase
Norwalk virus	12	3 months - 12 years	4
Rotavirus	12	3 months - 12 years	0

Outbreaks caused by *Leptospira,* Norwalk virus and Adenovirus 3 were more typical because the sources of pathogens were external to the beaches and, except for *Leptospira,* associated with faecal contamination. *Leptospira* are usually associated with animals that urinate into surface waters. Swimming-associated outbreaks attributed to *Leptospira* are very rare. Conversely, outbreaks of acute gastro-intestinal infections with an unknown aetiology are more common. Although the cause is unknown, the symptoms associated with the illness are frequently similar to those observed in viral infections.

Very few studies, other than those associated with outbreaks, have been conducted to determine the etiological agents related to swimming-associated illness. Some previously unpublished data shown in Table 9.3 do confirm that viruses are candidate organisms for the gastro-enteritis observed in epidemiological studies conducted at bathing beaches. The data in the table are from acute and convalescent sera obtained from swimmers who suffered from acute gastro-enteritis after swimming at a very contaminated beach in Alexandria, Egypt. The sera were obtained from 12 subjects, all of whom were less than 12 years old, on the day after the swimming event and again about 15 days later. The sera were tested with Norwalk virus and rotavirus antigens. None of the subjects showed a fourfold increase in titre to rotavirus antigen. However, 33 per cent did show a fourfold increase in titre to the Norwalk virus antigen. This reactivity indicated that Norwalk virus is a pathogen that has the potential to cause swimming-associated gastroenteritis. These data also show a possible approach for linking specific pathogens to swimming-associated illness.

The types and numbers of various pathogens in sewage vary depending on the incidence of disease in the contributing population and known seasonality in human infections. Hence, numbers vary greatly across different parts of the world and times of year, but a general indication of incidence is given in Table 9.4.

Table 9.4 Examples of pathogens and indicator organisms commonly found in raw sewage

Pathogen or indicator[1]	Disease or role	No. per litre
Bacteria		
Campylobacter spp.	Gastro-enteritis	37,000
Clostridium perfringens[2]	Indicator organism	6×10^5-8×10^5
E. coli	Indicator organism	10^7-10^8
Salmonella spp.	Gastro-enteritis	20-80,000
Shigella	Bacillary dysentery	10-10,000
Viruses		
Polioviruses	Indicator	1,800-5,000,000
Rotaviruses	Diarrhoea, vomiting	4,000-850,000
Parasitic protozoa		
Cryptosporidium parvum oocysts	Diarrhoea	1-390
Entamoeba histolytica	Amoebic dysentery	4
Giardia lamblia cysts	Diarrhoea	125-200,000
Helminths		
Ascaris spp.	Ascariasis	5-110
Ancylostoma spp.	Anaemia	6-190
Trichuris spp.	Diarrhoea	10-40

[1] Many important pathogens in sewage have yet to be adequately enumerated, such as adenoviruses, Norwalk/SRS viruses and Hepatitis A

[2] From Long and Ashbolt, 1994

Source: Adapted from Yates and Gerba, 1998

9.1.3 Indicators

The risk of exposure to pathogens in recreational waters has been well described in the literature (WHO, 1998) and this information has been noted and used by risk managers. However, it is very difficult to detect pathogens, especially viral and protozoan pathogens, in water samples obtained from bathing beaches. Methods for detecting and identifying infectious viruses or parasites are either very difficult to perform or do not exist at all. Bacterial pathogens can be detected, but their fastidious nutritional requirements and susceptibility to different types of environmental stress also can make the task very difficult. One hundred years ago, Escherich and other creative bacteriologists who were concerned with cholera and typhoid fever, proposed the use of a harmless organism that was always found in faeces and which was easy to detect on simple bacteriological media, as an indicator of the presence of faecal material in water. By implication, these indicator organisms would signal the potential presence of organisms that cause gastro-intestinal disease.

The indicator concept has been used successfully for a long time. The faecal bacteria most commonly used today are thermotolerant coliforms, *E. coli* and enterococci or faecal streptococci, which are described in detail in Chapter 8. However, there are still many questions concerning the effectiveness of the way in which water quality is measured and monitored; a number of environmental and physical factors may influence the usefulness of faecal bacteria as indicators. No single indicator or approach is likely to represent all the facets and issues associated with contamination of waterways with faecal matter. Table 9.5 provides an overview of possible indicators, together with the strengths and drawbacks of each.

Die-off in marine and freshwater environments

The differential die-off of indicators in marine and freshwater environments is illustrated for coliforms in Figure 9.1. The figure, which was adapted from Chamberlain and Mitchell (1978), shows that in marine waters the mean T_{90} (the time taken for 90 per cent of organisms to die) for total coliforms is about 2.2 hours, whereas in freshwaters the mean T_{90} is about 58 hours. These results were obtained from *in situ* studies at wastewater outfalls where die-off was determined after accounting for dispersion and dilution. Similar behaviour is exhibited by thermotolerant coliforms and *E. coli.* Although similar studies have not been conducted with enterococci, laboratory studies by various investigators, particularly Hanes and Fragala (1967) suggest that enterococci also die-off more rapidly in sea water than in freshwater environments (Table 9.6). The differential die-off for enterococci is not as great as that for *E. coli,* which may account for their superior effectiveness as indicators of health risk. Very few similar studies have been conducted for viral indicators. One study, conducted in marine and freshwaters in Italy (Table 9.7), showed that Polio, ECHO and Coxsackie viruses decayed at approximately the same rate in these two environments (Cioglia and Loddo, 1962). If, as appears likely, indicators have different die-off characteristics in marine and freshwater, whereas viral indicators die-off at similar rates in these two environments, then viral pathogens may be present at higher levels in these waters relative to the bacterial indicator numbers. The conclusion that can be drawn is that higher levels of exposure to viral pathogens may occur in marine waters at similar bacterial indicator levels and may require reconsideration of guideline levels in the two environments.

Solar radiation

The effect of sunlight on *E. coli* and enterococci is shown in Figure 9.2. The rate of *E. coli* die-off increases rapidly as solar radiation increases. Conversely, the rate of die-off of enterococci does not increase as the intensity of sunlight increases. Other investigators have observed similar effects of sunlight on indicators. Although human viruses have not been examined under similar experimental conditions, viruses of *E. coli* (coliphages) have been tested and they react very much in the same manner as enterococci. If human viruses react to sunlight in a manner similar to bacterial viruses (phages) this would provide yet another explanation why enterococci are superior to *E. coli* as a predictor of human health risk at bathing beaches.

Table 9.5 The relative merits of selected indicators of sewage contamination

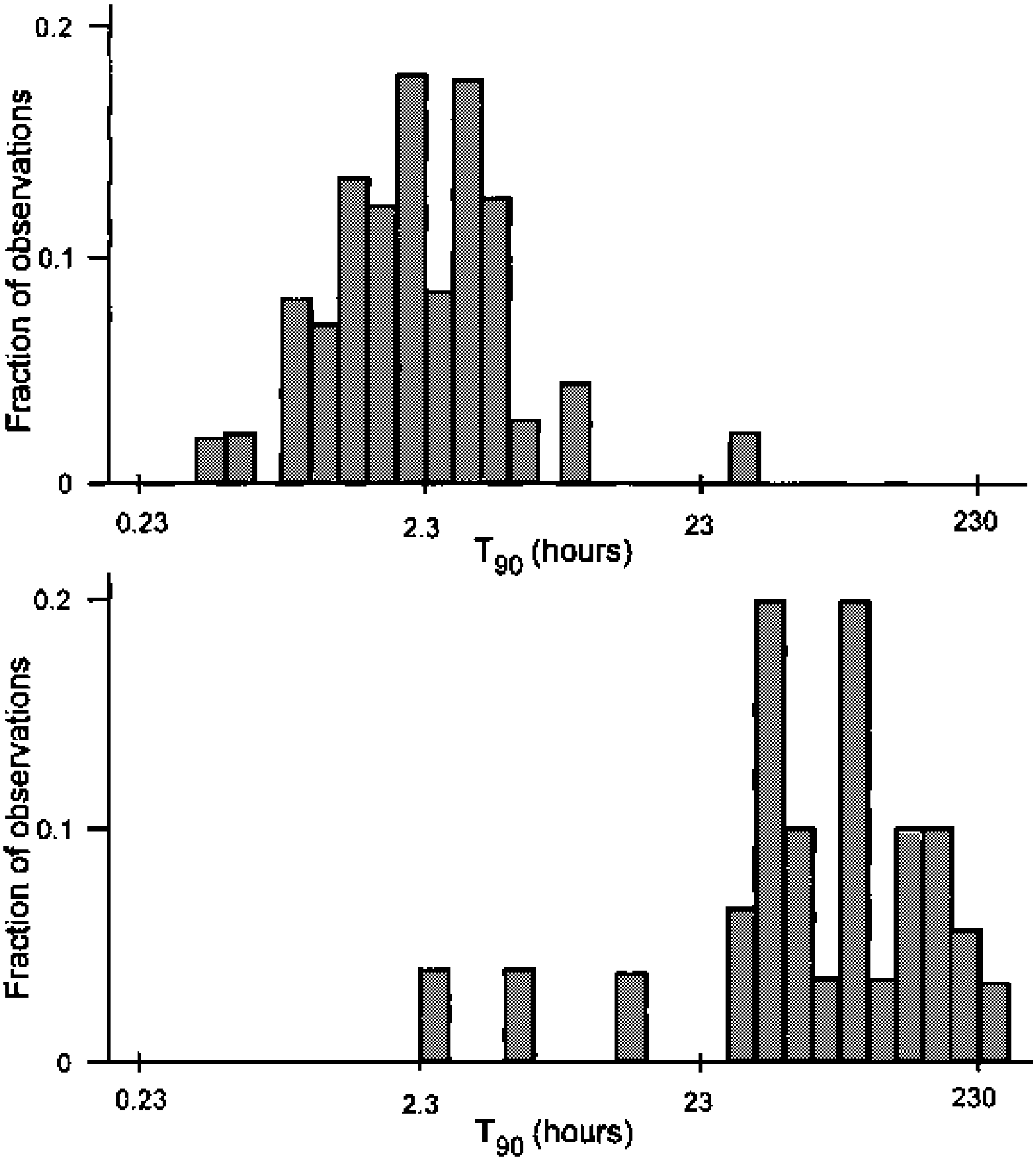
Indicator	Advantages	Disadvantages
Faecal streptococci/ enterococci	Marine and potentially freshwater human health indicator More persistent in water and sediments than coliforms Faecal streptococci may be cheaper than enterococci to assay	May not be valid in tropical waters due to potential growth in soils
Thermotolerant conforms	Indicator of recent faecal contamination	Possibly not suited to tropical waters due to growth in soils and waters Confounded by non-sewage sources (e.g. *Klebsiella* spp. in pulp and paper wastewaters)
E. coli	Potential freshwater human health indicator Indicator of recent faecal contamination Potential for typing *E. coli* to aid in sourcing faecal contamination Rapid identification possible if defined as β-glucuronidase-producing bacteria	Possibly not suited to tropical waters due to growth in soils and waters
Sanitary plastics	Little training of staff required and immediate assessment can be made for each bathing day Can be categorised	May reflect old sewage contamination and thus be of little health significance Subjective and prone to variable description
Preceding rainfall (12, 24, 48 or 72 h)	Simple regressions may account for 30-60% of the variation in microbial indicators for a particular beach	Each beach catchment may need to have its rainfall response assessed Response may depend on the period before the event
Sulphite-reducing clostridia[1]	Inexpensive assay with H_2S production Always in sewage impacted waters Possibly correlated with enteric viruses and parasitic protozoa	Enumeration requires anaerobic culture May also come from dog faeces May be too conservative an indicator
Somatic coliphages	Standard method well established Similar physical behaviour to human enteric viruses	Not specific to sewage May not be as persistent as human enteric viruses May grow in the environment
F-specific RNA phages	More persistent than some coliphages Standard ISO method available Host does not grow in environmental waters below 30°C	WG49 host may lose plasmid (although F-amp is more stable) Not specific to sewage Not as persistent in marine waters
Bacteroides fragilis phages	More resistant than other phages in the environment and similar to hardy human enteric viruses Appears to be specific to sewage ISO method recently published	Because numbers in sewage are lower than for other phages and most do not excrete this phage, it is of limited value for small populations Requires anaerobic culture
Faecal sterols	Coprostanol largely specific to sewage	Requires expensive gas

	Coprostanol degradation in water similar to die-off of thermotolerant coliforms A ratio of 5β: 5α stanols > 0.5 is indicative of faecal contamination; i.e. a ratio coprostanol: 5α-cholestanol of > 0.5 indicates human faecal contamination, while C_{29} 5β(24-ethylcoprostanol): 5α stanol ratio of > 0.5 indicates herbivore faeces Ratio of coprostanol: 24-ethylcoprostanol can be used to indicate the proportion of human faecal contamination, which can be further supported by ratios with faecal indicator bacteria (Leeming *et al*., 1996)	chromatography (about US$ 100 per sample) Requires up to 10 litres of sample to be filtered through a glass fibre filter to concentrate particulate stanols
Caffeine	May be specific to sewage, but unproven to date Could be developed into a dipstick assay	Yet to be proven as a reliable method
Detergents	Relatively routine methods available	May not be related to sewage (e.g. industrial pollution)
Turbidity	Simple, direct and inexpensive assay available in the field	May not be related to sewage; correlation must be shown for each site type
Cryptosporidium[2]	Required for potential zoonoses, such as *Cryptosporidium* spp., where faecal indicator bacteria may have died out, or not present	Expensive and specialised assay (e.g. Method 1622, US EPA); human/animal speciation of serotypes is not currently defined

[1] *Clostridium perfringens*

[2] Animal-sourced pathogens

Figure 9.1 Survival of coliforms in marine and fresh waters (Adapted from Chamberlain and Mitchell, 1978)



Effects of chlorine

Enterococci and *E. coli* are both sensitive to chlorine, although enterococci are somewhat more resistant to this disinfectant than *E. coli.* For example, to achieve a two-log removal (i.e. 99 per cent removal), reported calculated CT values for *E. coli* (Conc. of disinfectant (mg l^{-1}) × Contact time (mins)) are in the range of 5 mg min l^{-1} compared with 120 mg min l^{-1} for *S. faecalis.* Enterococci survival may therefore be more similar to that exhibited by faeces-carried pathogens than that of *E. coli.* This differential resistance to disinfection is another factor that influences the effectiveness of indicator bacteria in surface waters where disinfection of wastewaters by chlorine is practised.

Table 9.6 Decay rate estimates for *E. coli* and enterococci in sea water and fresh water

Indicator	Decay rate[1] (days)		Reference
	Fresh water	Sea water	
E. coli	3.9[2]	0.8[2]	
	6.3		Bitton *et al*., 1983
	2.7		McFeters and Stuart, 1974
	3.1		Keswick *et al.,* 1982
	4.6	0.8	Hanes and Fragala, 1967
		0.7	Omura *et al*., 1982
Enterococci	4.4[2]	2.5[2]	
	34.7		Bitton *et al*., 1983
	4.2		McFeters and Stuart, 1974
	4.5		Keswick *et al*., 1982
	3.0	2.4	Hanes and Fragala, 1967
		2.6	Omura *et al*., 1982

[1] Time required for 90% of the population to die off, in days
[2] Median values

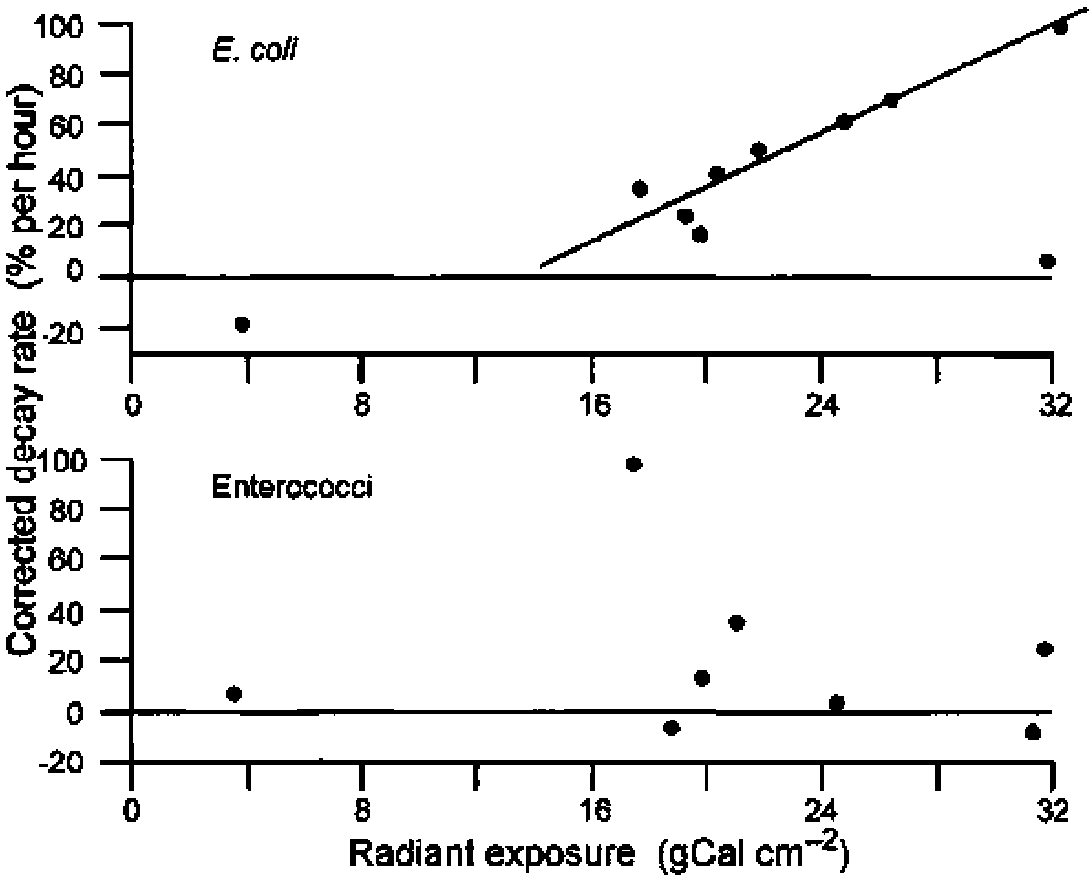
Table 9.7 Survival of enteroviruses in sea water and river water

Virus strain	Die-off rate (days)[1]	
	Sea water	River water
Polio I	8	15
Polio II	8	8
Polio III	8	8
ECHO 6	15	8
Coxsackie	2	2

[1] Maximum number of days required to reduce the virus population by 3 log values

Source: Adapted from Cioglia and Loddo, 1962

Figure 9.2 The effect of solar radiation on the die-off of *E. coli* and enterococci (Adapted from Sieracki, 1980)



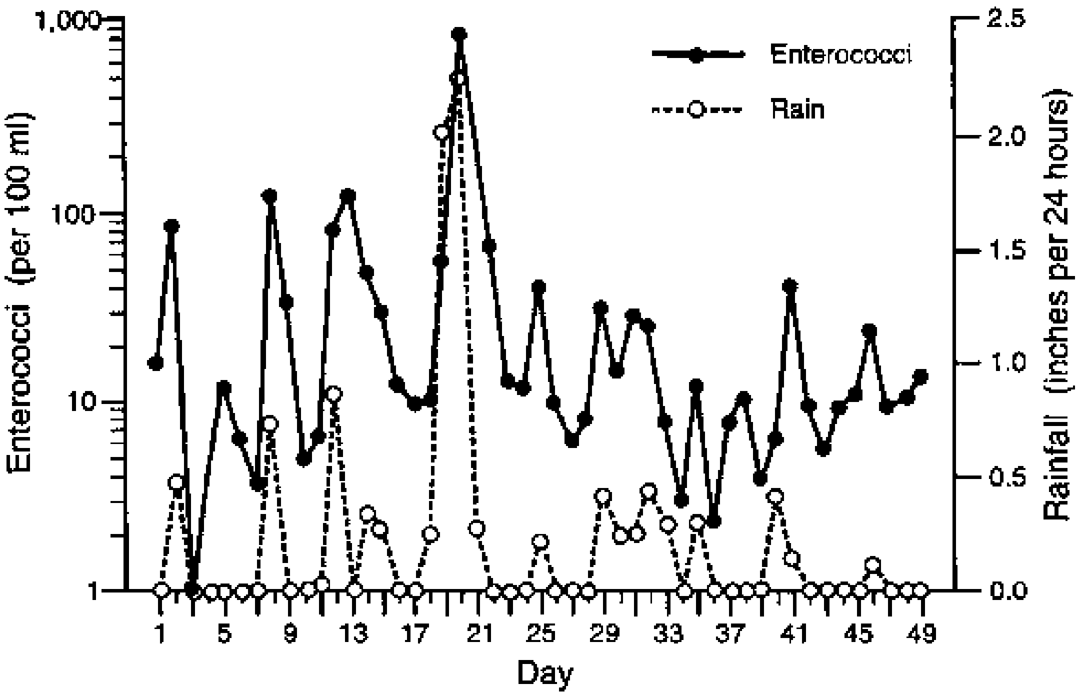
Rainfall

Rainfall can have a significant effect on indicator densities in recreational waters increasing the densities to high levels because animal wastes are washed from forest land, pasture land and urban settings, or because treatment plants are overwhelmed causing sewage to by-pass the treatment process. In either case, the effect of rainfall on beach water quality can be quite dramatic (Figure 9.3) (Calderon, 1990 Pers. Comm.). The effect, illustrated in Figure 9.3, on a beach surrounded by forests, was very rapid and usually persisted for 1-2 days. The highly variable effect of rainfall on water quality can result in the frequent closing of beaches. The important question is whether high indicator levels that result from animal wastes carried to surface waters by rain water run-off, indicate the same level of risk to swimmers as would exist if the source of the indicators was a sewage treatment plant. There are conflicting reports in the literature with regard to risk associated with exposure to recreational water contaminated by animals.

Sources of indicators

Coliforms and thermotolerant coliforms are known to have sources other than mammalian enteric systems. These two indicator groups can grow to very high densities in industrial wastewaters, such as those discharged by pulp and paper mills. *E. coli* and enterococci are not usually associated with industrial wastewaters, but some investigators believe that these indicators can grow in soil in tropical climates. Under any of these conditions, where the source of the indicator is other than the faeces of warm-blooded animals, it is questionable whether the indicator would have any value as a measure of faecal contamination of recreational waters.

Figure 9.3 The effect of rainfall on enterococci densities in bathing beach waters (After Calderon, Pers. Comm.)



The most commonly used indicators for surface water quality, *E. coli* or faecal coliforms and enterococci or faecal streptococci, can readily be detected in the faeces of humans, other warm-blooded animals and birds. The broad spectrum of animals in which enterococci can be found is shown in Table 9.8. This list is not exhaustive, but helps to illustrate that there are many non-human sources of enterococci. This issue is closely related to rainfall because, if it can be shown that the risk of exposure to water contaminated by animals is significantly less than that contaminated by humans, the way in which water quality is currently measured may have to be changed considerably. Methods for distinguishing human from animal-derived faecal matter are described in Chapter 8.

9.1.4 Pollution abatement and water quality

Beaches, especially near urban areas, are often subject to pollution from sewage and industrial discharges, combined sewer overflows (CSO) and urban run-off. Pollution abatement is, therefore, a key part of coastal zone management aimed at minimising health risks to bathers and ecological impacts. Pollution abatement measures for sewage may be grouped into three wastewater disposal alternatives: treatment, dispersion through sea outfalls and discharge to non-surface waters (i.e. reuse, in which wastewater is stored and then used for agricultural or other purposes, or groundwater injection). In practice, there are numerous anomalies to these general categories. In addition, CSOs and sanitary sewer overflows (SSO) usually occur as a result of excessive rainfall events and can result in high human health risks for certain beach zones. Pollution abatement alternatives for these overflows, such as holding tanks, separate storm overflow submarine outfalls, over-design of sewer systems for extreme storm events, etc., are often prohibitively expensive and difficult to justify. In view of the costs of control, it may be preferable for integrated beach zone management to focus on restricting beach use or, at the very minimum, warning the public of the potential health risks during and after high risk events.

Table 9.8 Occurrence of enterococci in human faeces and in faeces of other warm-blooded animals

Species	Total number of subjects	% of subjects with samples containing	
		E. faecalis	***E. faecium***
Humans	32	41	88
Dogs	21	29	76
Puppies	2	100	100
Cats	1	-	-
Kittens	2	100	100
Pigs	22	77	100
Piglets	3	33	100
Horses	6	50	33
Sheep	4	100	100
Cows	15	-	73
Chickens	13	92	100
Goats	2	100	100
Beavers	3	-	-

Treatment

For large urban communities, at least secondary or tertiary sewage treatment plants with disinfection are necessary for onshore or near-shore discharges to protect nearby recreational areas. Public health risks can vary depending on the operation of the plant and the effectiveness of disinfection. Smaller communities with lesser population densities usually apply treatment by means of septic tank systems, latrines, etc. The ground acts as a filter for pathogenic organisms and, therefore, such disposal systems result in a very low health risk to recreational areas except where Karst topography occurs leading to the possibility of direct contamination.

The general removal levels of the major pathogen groups by conventional primary, secondary and tertiary sewage treatment are summarised in Table 9.9. The advent of new detection methods for a range of hardier enteric viruses may change views on the persistence of viruses that cannot be enumerated by culture-based methods. For example, identification of hepatitis A virus by antigen capture polymerase chain reaction (AC-PCR), followed by hybridisation on membranes, indicated their presence in raw sewage and secondary treatment effluent in 80 per cent and 20-30 per cent of samples respectively (Divizia *et al.,* 1998). Advanced sewage treatment based on ultra- and nanofiltration methods can also be an effective barrier to viruses, with over 10^6 removal (Otaki *et al.,* 1998), and other pathogens (Jacangelo *et al.,* 1995; Madireddi *et al.,* 1997). Additionally, revaluation of ultra violet (UV) (Oppenheimer *et al.,* 1997), ozone (Perezrey *et al.,* 1995) and disinfection kinetics (Haas *et al.,* 1996; Gyurek and Finch, 1998) are also changing the way in which engineers are evaluating disinfection and treatment processes.

Oxidation pond treatment may remove significant numbers of pathogens, particularly the larger protozoan cysts and helminth ova. However, short circuiting due to poor design, thermal gradients or hydraulic overloading may all reduce considerably the residence time from the typical 30-90 days. In addition to removal by sedimentation during long resident times, inactivation by sunlight and temperature, and predation by other micro-organisms may reduce faecal bacterial numbers by 90-99 per cent (Yates and Gerba, 1998). Inactivation of viral and parasitic protozoa is also influenced heavily by temperature. For example, poliovirus type 1 may be inactivated by 99 per cent in 5 days in summer but may take 25 days in winter (Funderburg *et al.,* 1978). The cysts and oocysts of *Giardia* and *Cryptosporidium* may take at least 37 days to achieve a 99.9 per cent reduction (Grimason *et al.,* 1992, 1996b), whereas the larger ova of helminths may be totally removed in 12-26 days (Grimason *et al.,* 1996a).

Long sea submarine outfalls

Long sea outfalls are assumed to be properly designed and of sufficient length, diffuser discharge depth and design to ensure a low probability of the sewage plume reaching designated beach zones. As such, the long sea outfall is a very low human health risk alternative because the bather is unlikely to come into physical contact with the sewage, whether treated or untreated. Modern diffusers are usually designed to achieve minimum, near-field, immediate dilutions of 100 to 1 that would reduce the concentration of organics and nutrients in the sewage to levels that would have no adverse ecological effects in an open ocean situation. Higher dilutions are achieved most of the time, depending on the current structure. Under stratified conditions, complete sewage plume submergence can occur and can reduce further the possibility of sewage reaching designated beach zones. The diffuser length, depth and orientation, as well as the area and spacing of the discharge ports, are the key design considerations (Roberts, 1996). For pathogenic and indicator organisms, additional order-of-magnitude reductions may be required to meet established bathing beach water quality criteria, depending on the degree of treatment and disinfection. This far-field "dilution" is achieved through additional physical dilution and mortality in the ocean environment subsequent to discharge. The design distance required, i.e. length of the outfall, to achieve the additional far-field reduction is determined by the dominant current structure and mortality rates (T_{90}).

Table 9.9 Pathogen removal during sewage treatment

Level of treatment	Enteric viruses	*Salmonella*	*C. perfringens*	*Giardia*
No treatment (raw sewage)				
No. remaining (per litre)	10^5-10^6	5,000-80,000	10^5	9,000-200,000
Primary treatment[1]				
% removal	50-98.3	95.5-99.8	30	27-64
No. remaining (per litre)	1,700-500,000	160-3,360	70,000	72,000-146,000
Secondary treatment[2]				
% removal	53-99.92	98.65-99.996	98	
No. remaining (per litre)	80-470,000	3-1,075	2,000	
Tertiary treatment[3]				
% removal	99.983-99.9999998	99.99-99.9999995	99.9	98.5 to 99.99995
No. remaining (per litre)	0.007-170	0.000004-7	100	0.099-2,951

[1] Physical sedimentation

[2] Primary sedimentation, trickling filter/activated sludge and disinfection

[3] Primary sedimentation, trickling filter/activated sludge, disinfection, coagulation-sand filtration and disinfection; note that tertiary treatment does not involve coagulation-sand filtration am the second disinfection step in the case of *C. perfringens*

Sources: Long and Ashbolt, 1994; Yates and Gerba, 1998

Pre-treatment with milliscreens with apertures of 1-1.5 mm, is considered to be the minimum treatment required to remove floating matter and thus avoid aesthetic impacts on the designated beach zones. For the same aesthetic considerations, removal of grease and oil should be implemented at source, especially if effluent concentrations are high and not reduced sufficiently after initial dilution. To avoid possible ecological impacts in the vicinity of the discharge, more advanced treatment may be justified.

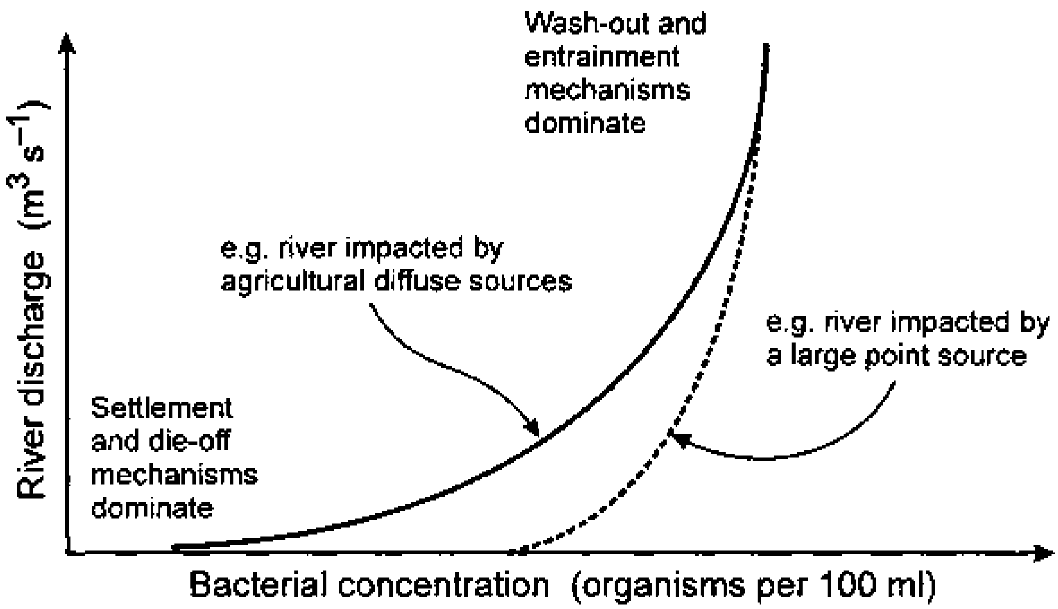
Discharge to non-surface waters

Reuse of wastewater and groundwater recharge are two methods of sewage disposal that have minimal impact on recreational waters. In arid regions, sewage (after appropriate treatment) can be an important resource for agricultural purposes such as crop irrigation. Reuse has the dual benefit of the productive use of sewage while avoiding wasteful discharges to the marine environment with the inherent pollution potential. Direct injection of sewage below ground for groundwater recharge is practised in some regions of the world, usually in combination with advanced treatment. Groundwater injection is a no (or very low) human health risk option for designated beach zones except in areas with Karst topography.

9.1.5 Hydrological considerations

Rivers contribute a significant proportion of the bacterial load to coastal beach areas. In some regions, significant numbers of freshwater beaches are directly affected by river water quality. The bacterial concentration in river water is determined by faecal pollution from point and non-point or diffuse sources. Major point sources include sewage effluents, CSOs, industrial effluents and confined animal sources such as feedlots. Non-point sources relate directly to agricultural activity within the watershed, and are influenced primarily by the type of livestock and its density. A significant contribution is also derived from urban surfaces.

Figure 9.4 The relationship between river discharge and bacterial concentration



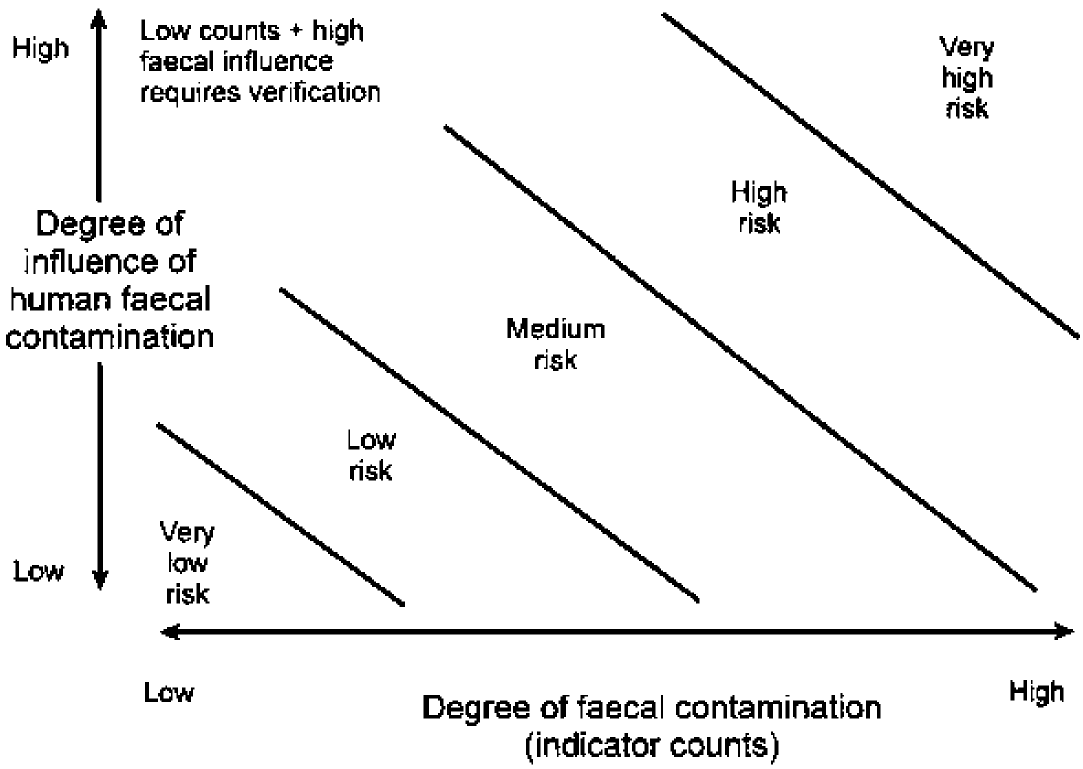
The transport of microbial contaminants through the watershed to the river and subsequently through the river system to the marine environment is controlled by the flow of water and therefore, rainfall is a key influence on concentrations (see section 9.1.3). Faecal material is transported from the watershed surface to the river and changes in flow are determined by rainfall and by the hydrological characteristics of the basin (soils, bedrock, etc.) which therefore have a significant impact on the total flux of microbes transported. In river water, the decrease in bacterial concentrations downstream of a source, conventionally termed "die-off, largely reflects the settlement or sedimentation of organisms to the river bed. In riverbed sediments, survival times are increased significantly and the bacteria are readily resuspended when the river flow increases. All rivers demonstrate a close correlation between flow and bacterial concentration due to the increased supply of bacteria from watershed surfaces and some point sources (e.g. CSOs) during rainfall events (Figure 9.4). The two curves represent hypothetical examples. In reality, all rivers will exhibit individual relationships depending on their hydrological characteristics and bacterial sources. The shape of the flow relationship will be variable between different catchments and may also break down during prolonged high flows if the store of organisms in the bed-sediment (or the catchment surface) is exhausted. This phenomenon, however, has only been documented for small streams dominated by diffuse inputs and is less likely to occur on major rivers with multiple point and non-point sources. The processes controlling transport and fate of bacteria in watersheds are now well understood and river water bacteria concentrations can be modelled and predicted (see section 9.4.1).

9.2 Alternative approaches to monitoring and assessment programmes

The experts who met in Annapolis in November 1998 (see Acknowledgements) agreed that an improved approach to the regulation of recreational water, reflecting health risk more reliably and providing enhanced scope for effective management intervention, was necessary and feasible. The major output of the meeting was the development of such an approach, which is described in this section. Because this approach is so different to established practice it includes elements that require substantial testing. The description provides sufficient detail to enable field testing but should be amended to take account of specific local circumstances. The approach will be refined further as experience with implementation accumulates. This chapter also sets out principles for the design of an intensive assessment for evaluating the modified approach and studying relationships between factors that affect beach water quality and the ability of monitoring schemes to detect these changes. The Annapolis group would like to encourage pilot testing of this approach, and is interested in receiving the results of any such studies. The proposed approach leads to a classification scheme through which a beach would be assigned to a class (i.e. very poor, poor, fair, good or excellent) based upon health risk. By enabling local management to respond to sporadic or limited areas of pollution and thereby upgrade a beach's classification, it provides a significant incentive for local management actions as well as for pollution abatement. The classification scheme provides a generic statement of the level of risk and indicates the principal management and monitoring actions likely to be appropriate. The advantage of a classification scheme, as opposed to a pass or fail approach, lies in its flexibility. A large number of factors can influence the condition of a given beach. A classification system reflects this, and allows regulators to invoke mitigating approaches for beach management.

The most robust, accurate and feasible index of health risk is provided by a combination of a measure of a microbiological indicator of faecal contamination with an inspection-based assessment of the susceptibility of an area to direct influence from human faecal contamination. This reflects two principal factors. Firstly, high counts of faecal indicator bacteria may be caused by either human faecal contamination or contamination from other sources. In general, sources other than human faecal contamination present a significantly lesser risk to human health, and by adopting a combined classification it is possible to reflect this modified risk. Secondly, any microbiological analytical result provides information on only a moment in time, whereas microbiological quality may vary widely and rapidly even within a small area (see section 9.1). It is possible to perform a large number of analyses to obtain an improved evaluation of the situation, but with concomitant cost. However, information concerning the existence of sources of contamination and their likely influence upon the recreational water use area provides a robust and rapid means of increasing the reliability of the overall assessment. This would lead to a series of classes of relative risk as presented schematically in Figure 9.5. The strengths of such an approach are demonstrated by the case study presented in Box 9.1.

Figure 9.5 Schematic representation of classes of health risk



Box 9.1 A risk management approach to beach closure in Southern California

During February 1992, a severe winter storm battered the southern California coastline. Winds, high surf and the deluge of rain led to much damage. One casualty of the storm was a pipe that carried treated wastewater from 200,000 homes and businesses to the ocean for disposal. Following the storm, divers confirmed that the 48-inch diameter pipe was broken and about 250,000 gallons per day of non-disinfected secondary treated wastewater were leaking into 10 feet depth of water approximately 90 feet from shore. Water samples were collected directly above the broken pipe and at the adjacent swimming and surfing beach which was used all year round. Coliform concentrations in the samples directly above the pipe break exceeded State standards for recreational water contact, whereas the samples at the beach did not.

In spite of the relatively low coliform densities at the beach, the local Health Officer closed the beach because of the discharge of non-disinfected waste-water. The Health Officer was of the opinion that even though State coliform standards were not exceeded at the beach it did not mean viruses that cause gastro-intestinal illness, hepatitis or polio were not present. The Health Officer's concern stemmed from the fact that activated sludge treatment alone is only between 90 and 95 per cent efficient in removal of human enteric viruses. Sampling at two local treatment facilities had demonstrated human enteric virus levels in secondary treated wastewater to be between 5 and 50 infectious units per gallon. Even with dilution and dispersion of the indicator bacteria to below State standards, a discharge known to contain human enteric viruses constituted an unacceptable risk to this particular Health Officer. In closing the beach, the Health Officer took a risk management approach to swimmer health protection, namely to prevent contact with waters known to contain faecal contamination, regardless of the density of wastewater "indicator bacteria" measured during water testing.

Variation in water quality may occur in response to events (such as rainfall) with predictable outcomes, or the deterioration may be constrained to certain areas or sub-areas of a single beach. It is possible to discourage use of areas that are of poor quality, or to discourage use at times of increased risk. In addition, if success in discouraging bathing at times of risk can be demonstrated, it might be reasonable to up-grade the classification of a beach. Because measures to predict and discourage use at certain times or in areas of elevated risk may be inexpensive, greater cost-benefit and greater possibilities for effective local management intervention are possible.

Figure 9.6 illustrates the process for assigning a classification to a given beach. The two principal components of the scheme are:

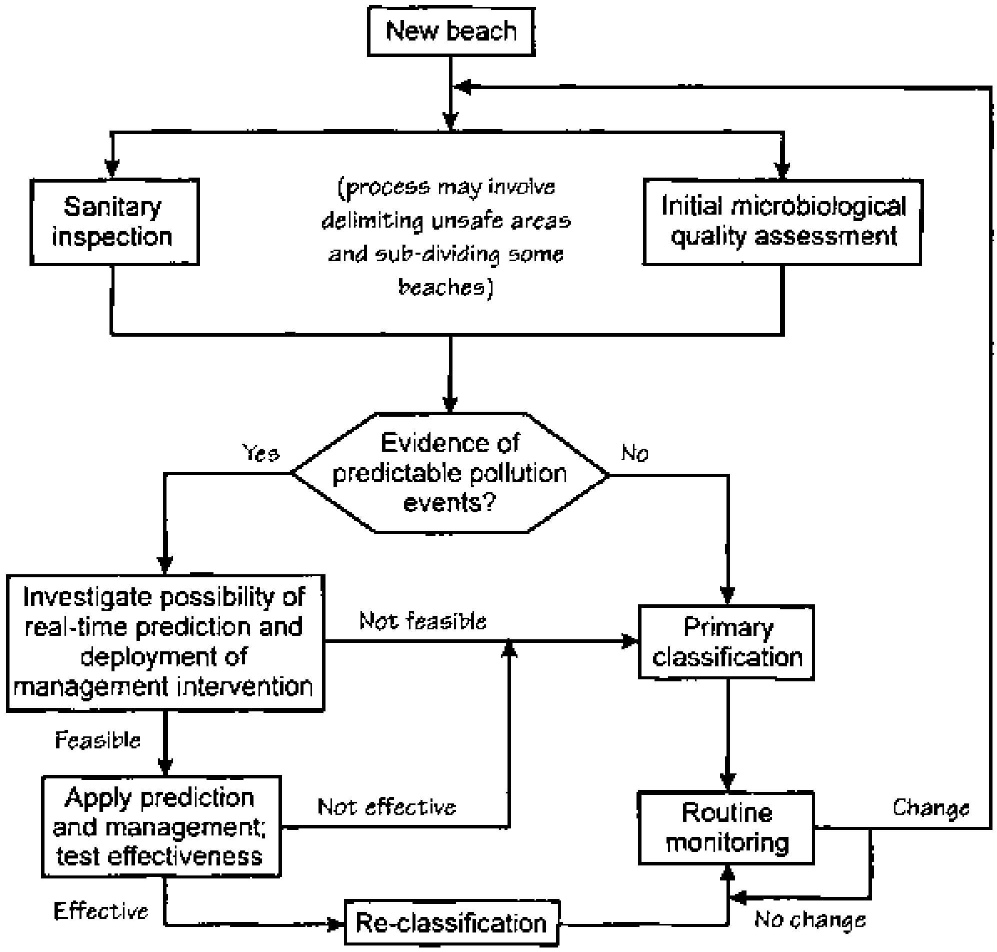
• A primary classification based upon the combination of evidence for the degree of influence of human faecal material (a sanitary inspection) alongside counts of suitable faecal indicator bacteria (a microbiological quality assessment).

• The possibility of "reclassifying" a beach to a higher (better) class if effective management interventions are deployed to reduce human exposure at certain times or in places of increased risk.

9.3 Primary classification

The primary classification is based upon the combination of an inspection-based assessment of the area's susceptibility to influence from human faecal contamination and a microbiological indicator measure of faecal contamination.

Figure 9.6 The steps to be taken for assigning a classification to a new beach or location



9.3.1 Sanitary inspection

Sanitary inspection is the evaluation of the principal sources of faecal pollution. The three most important sources of human faecal contamination of bathing beaches for public health purposes are:

- Sewage, including CSO and storm water discharges.

- Riverine discharges, where the river is a receiving water for sewage discharges and is used directly for recreation or discharges near a coastal or lake area used for recreation.

- Bather contamination, including excreta.

All of these sources lead to the presence of faecal indicators that may be recovered and that can provide a semi-quantitative estimate of health risk, as has been demonstrated by many epidemiological studies (WHO, 1998).

Sources of faecal indicators other than human sewage also exist, such as drainage from areas of animal pasture and intensive livestock rearing. In general, due to the "species barrier", the density of pathogens of public health importance is assumed to be less in aggregate in animal excreta than in human excreta and may therefore represent a significantly lower risk to human health. As a result, the use of faecal indicator bacteria alone as an index of risk to human health may overestimate risks significantly where the indicators derive from sources other than human excreta. Nevertheless, the human health risk associated with pollution of recreational waters from animal excreta is not zero and some pathogens, such as *Cryptosporidium,* can be transmitted through this route.

The experts at the Annapolis meeting ranked qualitatively the relative risk to human health through direct sewage discharge, riverine discharge contaminated with sewage and bather contamination. In doing so they took account of the likelihood of human exposure and the degree of treatment of sewage. In taking account of sewage discharges to recreational areas and of rivers, account was also taken of the pollutant load, using population as an index. While in many circumstances several contamination sources would be significant at a single location, the approach adopted was to categorise a beach according to the single most significant source of pollution. Even two sources of similar magnitude would, on aggregate, increase exposure by a factor of two which, in microbiological terms, is of very limited significance. The methodology for designing and conducting a sanitary inspection is described in Chapters 2 and 8.

Sewage discharges

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

- Discharges directly to the beach (above low water level in tidal areas).

- Discharges through "short outfalls", where the discharge is into the water but sewage-polluted water is likely to contaminate the beach area.

- Discharges through long sea outfalls, where the sewage is diluted and dispersed and is unlikely to pollute bathing areas.

Although the terms "short" and "long" are often used in relation to outfalls, length is generally less important than proper location and effective diffusion that will ensure the pollution is unlikely to reach bathing areas. A short outfall is assumed to be a discharge to the inter-tidal zone, with a significant probability of the sewage plume reaching the designated beach zone. For short outfalls, the relative risk is increased, based upon the size of the contributing population. An effective outfall is assumed to be properly designed, with sufficient length and diffuser discharge depth to ensure low probability of the sewage plume reaching the designated beach zone. Urban storm water run-off and outputs from CSOs are included within the scheme under the category of direct beach outfalls.

The classification is based upon a qualitative assessment of risk of contact or exposure under "normal" conditions with respect to operation of sewage treatment works, hydrometeorological and oceanographic conditions. The potential risk to human health through exposure to sewage can be categorised as shown in Table 9.10. The sewage effluent treatments listed in this table are classified as no treatment (raw sewage); preliminary (filtration with milli- or micro-screens); primary (physical sedimentation); secondary (primary sedimentation and high rate biological processes such as trickling filter or activated sludge); secondary with disinfection; tertiary (advanced waste-water treatment, including primary sedimentation, trickling filter or activated sludge, and coagulation or sand filtration); tertiary with disinfection; and lagoons (low rate biological treatment). Septic tank systems are assumed to be equivalent to primary treatment.

Table 9.10 Potential human health risks arising from exposure to sewage

Level of treatment	Discharge type		
	Directly on beach	**Short outfall**[1]	**Effective outfall**[2]
None[3]	Very high	High	NA
Preliminary	Very high	High	Low
Primary (including septic tanks)	Very high	High	Low
Secondary	High	High	Low
Secondary plus disinfection	Medium	Medium	Very low
Tertiary	Medium	Medium	Very low
Tertiary plus disinfection	Very low	Very low	Very low
Lagoons	High	High	Low

[1] The relative risk is modified by population size; relative risk is increased for discharges from large populations and decreased for discharges from small populations

[2] 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)

[3] Includes combined sewer overflows

Riverine discharges

Riverine discharges are categorised with respect to the sewage effluent load and the degree of dilution, as illustrated in Table 9.11. Effluent load is characterised by the total human population in the watershed or catchment above the beach or estuary. The population of relevance is the peak population which, in many recreational water use areas, will be significantly greater than the resident population and is likely to occur during weekends and local holidays during the summer season. Dilution is defined by the "dry weather" river flow or discharge during the bathing season. Use of dry weather flow is a "worst case" approach and coincides with reality where the bathing season is also the season of reduced flow. In many circumstances, the most significant sewage discharges are near to the coast and die-off during riverine travel is likely to be of limited significance for the travel times encountered in many rivers. Removal of pathogens through sedimentation may be of some significance but could not be accounted for reliably in a simple way. Resuspension of sediments and CSO discharges can be important during pollution episodes and in this context may be predictable (see section 9.4.1). Episodic input can dominate in areas subject to frequent summer rainstorms such as Northwest Europe.

Table 9.11 Potential human health risks arising from exposure to sewage through riverine flow and discharge

Dilution effect[1,2]	Treatment level				
	None	Primary	Secondary	Secondary plus disinfection	Lagoon
High population with low river flow	Very high	Very high	High	Low	Medium
Low population with low river flow	Very high	High	Medium	Very low	Medium
Medium population with medium river flow	High	Medium	Low	Very low	Low
High population with high river flow	High	Medium	Low	Very low	Low
Low population with high river flow	High	Medium	Very low	Very low	Very low

[1] The population factor takes account of all the population upstream from the beach and assumes no instream reduction in the hazard factor used to classify the beach

[2] Stream flow is the 10 per cent flow during the period of active beach use; stream flow assumes no dispersion plug flow conditions to the beach

In practice, several discharges into a single river course are likely to occur and where larger discharges are treated to a higher level, the smaller sources (including septic tank discharges) and CSOs may represent the principal source of concern. It is assumed that the discharge travels in a consolidated manner, with little mixing or dilution by the river water or little dispersion. The overall riverine discharge risk category is that accorded by the most significant single pollution source.

The classification can be used directly for freshwater beaches on the river and for beaches in estuarine areas or which are dominated by riverine pollution. For marine beaches the same classification may be used but it should be varied depending on the proximity of the river to the beach.

Table 9.12 Potential human health risks arising from exposure to sewage from bathers

Bather density/dilution factor	Risk category	Bather density/dilution factor	Risk category
High density		Low density	
High dilution[1]	Low	High dilution	Very low
Low dilution[1,2]	Medium	Low dilution[2]	Low

[1] Move to the next highest risk category if no sanitary facilities are available at beach site

[2] If no water movement

Bather shedding

While bather shedding is generally of lesser importance than sewage or riverine discharge, the resulting pollution is direct and fresh, and therefore potentially of great public health significance. Several studies (see section 9.1.2) have demonstrated accumulation of faecal material (as indicated by recovery of faecal indicator bacteria) during the course of a day, despite potentially enhanced die-off due to sunlight. Small volume areas of limited turnover are especially affected, such as bays and coastal and estuarine areas constrained by sandbars. The two principal factors of importance are therefore bather density and degree of dilution (Table 9.12). Low dilution is assumed to represent no water movement (such as occurs in lakes and lagoons and coastal embayments). The likelihood of bathers defaecating into the water is substantially increased if toilet facilities are not readily available. Where bather densities are high, the classification should therefore be increased to the next higher class if no sanitary facilities are available at the beach.

9.3.2 Microbiological quality assessment

Sewage contamination may be identified by a range of microbial, chemical or visual parameters as described in Table 9.5 and Chapter 8. Each gives a different view of the possible source(s) of contamination and should be used appropriately in a staged approach for assessing sewage contamination of bathing beaches. Hence, in addition to identifying which indicators to use, it is also important to identify action levels for the primary indicators selected to assess beaches. A further issue is the number of samples required to make an assessment, taking into account the variability of the beach site under study.

A basic selection of sewage indicators called "primary indicators" is proposed as an essential first step in the evaluation of bathing water. Table 9.13 tabulates primary indicators for marine and fresh water. "Secondary indicators" are described for follow-up analysis to assist in the assessment and management of faecal contamination at beaches.

Table 9.13 A beach categorization scheme based on the concentrations of primary indicators of sewage contamination in marine and fresh waters

Water source	Indicator(s)	Category	95th percentile
Temperate marine water	Faecal streptococci	A	< 10
	Enterococci[1]	B	11-50
		C	51-200
		D	201-1,000
		E	> 1,000
Alternative for tropical marine waters[2]	Sulphite-reducing	A	< 1
	Clostridia	B	1-10
	Clostridium perfringens	C	11-50
		D	51-80
		E	> 80
Temperate fresh water[3]	Faecal streptococci	A	< 10
	Enterococci[1]	B	11-50
		C	51-200
		D	201-1,000
		E	> 1,000
	E. coli	A	< 35
		B	36-130
		C	131-500
		D	501-1,000
		E	> 1,000
Optional for tropical fresh water[2]	Sulphite-reducing	A	< 1
	Clostridia	B	1-10
	Clostridium perfringens	C	11-50
		D	51-80
		E	> 80

[1] Source for faecal streptococci/enterococci 95th percentile ranges: WHO, 1998

[2] Based on preliminary data

[3] While studies suggest that there is a differential die-off rate for microbial indicators in marine and fresh waters (see section 9.1.3), current data are not sufficient to derive separate 95th percentiles for freshwater environments; the above faecal streptococci/enterococci percentiles are therefore based on data obtained from marine studies, but may be reconsidered when further freshwater studies have been conducted

Primary indicators

The minimal non-microbial, primary indicators of faecal contamination in marine environments are sanitary plastics and grease. Although somewhat crude indicators, they have been used as aesthetic health indicators because they are associated with faecal contamination. In freshwaters, sanitary plastics may also act as non-microbial primary indicators, whereas grease will not fulfil such a role.

The primary microbial indicators identified are faecal streptococci and enterococci (temperate marine and freshwaters), *E. coli* (temperate fresh-waters) and sulphite reducing clostridia, i.e. *Clostridium perfringens* (temperate and tropical marine and freshwaters). Table 9.13 provides an example of beach categorisation, with "A" representing excellent water quality and "E" designating a beach with unacceptable water quality. A single sample result greater than the unacceptable 95th percentile requires follow-up action, such as a sanitary inspection, to verify that it is a statistical occurrence and not due to a real change in exposure.

Secondary indicators

Secondary indicators aimed at identifying the source of faecal contamination should include sulphite reducing clostridia *(Clostridium perfringens)* in temperate waters. Consideration must be given to the fact that dog excreta from surface run-off may be a source of these organisms, moreover it may be the only significant source other than humans. Other secondary indicators in temperate marine waters include faecal sterols and bacteriophages, such as the F-RNA serogroups I and IV for humans or phages to *Bacteroides fragilis* HSP40. In freshwaters, secondary indicators include faecal sterols and phages as above, but further potential secondary indicators include turbidity and phosphate and ammonium levels.

Measurement of indicators

Although the detail in the available literature varies considerably, the incidence of swimming related illness generally increases with the level of sewage contamination suggested by traditional bacterial indicators. There are few consistent relationships between individual indicator organisms and sewage load, and even fewer consistent relationships between individual indicators and particular pathogens. However, poorer quality water as indicated by total and thermotolerant coliforms, *E. coli,* faecal streptococci and enterococci is consistently associated with increased risk to the health of those using the water for recreational purposes (WHO, 1998).

Various statistical procedures for analysing microbiological indicator counts are discussed in Chapter 8. Regulatory standards are based on the use of these statistical methods. Most regulatory approaches have adopted a percentage compliance approach, in which a given percentage (e.g. 95 per cent) of the sample measurements taken must lie below a specific value in order to meet the standard. This simple percentage does not incorporate within its derivation the probability density function that describes the distribution of indicator organisms at a particular sampling location. The most important weakness in such an approach is that it fails to take account of the overall body of data. Some other approaches, such as use of the geometric mean or percentile values, are less affected by individual data.

The statistic most commonly used as a measure of compliance in the USA has been the geometric mean. By definition a mean is a measure of central tendency. As such, the mean is a statistic around which individual measurements tend to cluster. In the context of water quality monitoring, use of the mean will result in a situation in which the higher indicator organism measurements become obscured by the properties inherent in the calculation of the mean. Use of the geometric mean will further obscure extreme values. The median, another measure of central tendency, has an even greater effect on obscuring the higher levels of individual measurement contained within its derivation.

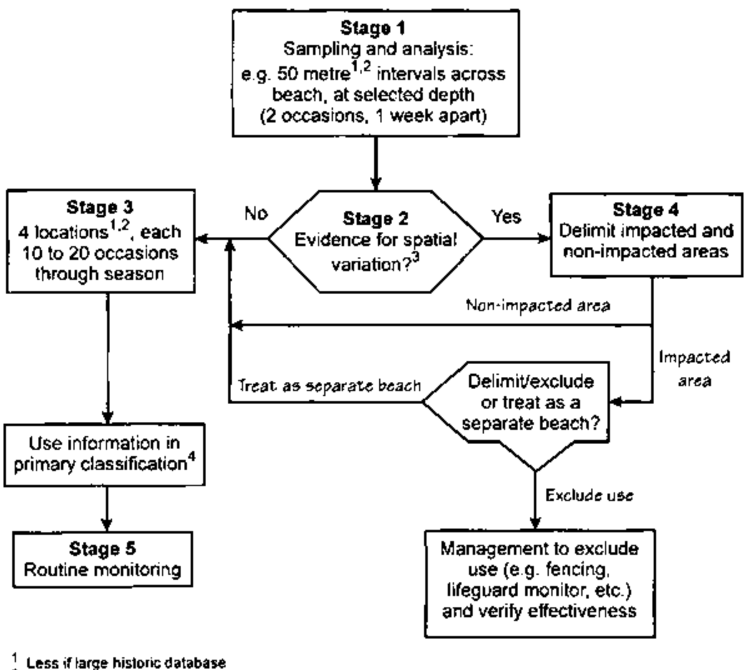
In contrast, a percentile value may be calculated by using the probability density function that describes the series of measurements taken. In this manner, the percentile value describes the distribution of indicator organism measurements at a particular location. Therefore, inherent in the calculation of the percentile value is the distribution of the entire series of measurements taken, resulting in a more accurate description of indicator organism densities at a particular location. Chapter 8 contains an example of a percentile calculation, using a log normal distribution (see section 8.7.3).

The categorisations in Table 9.13 are based on a minimum of 20 samples of the suggested microbial indicator(s). As the 95th percentile values were derived from limited studies, they are provisional and are meant to serve as a general guideline rather than as a standard. The categorisations should be treated as examples, and individual beaches should be evaluated based on site-specific conditions.

Microbiological categorisation sampling protocol

Figure 9.7 and Box 9.2 illustrate the steps necessary to assign a primary microbiological categorisation to a given beach.

Figure 9.7 An example sampling protocol for primary microbiological categorisation



1 Less if large historic database
2 Modified by sanitary inspection
3 For example, across full bandwidth of microbiological categories
4 If variation in quality is recognised then re-classification as described in Section 9.4 may be applicable.

Box 9.2 Example of the practical application of the primary microbiological categorisation protocol

STAGE 1

1. Full width of beach intended for recreational use delimited.

2. Along this full width, collect samples at a selected depth at 50 m intervals on two occasions one week apart at the start of the bathing season. The timing of the sampling should take into account the likely period of maximum contamination from local sewage discharges and bather shedding (i.e. the day after peak numbers of visitors).

3. Concurrently collect sanitary inspection data as described in Chapter 8.

STAGE 2

1. Use Stage 1 data to assess spatial variation.

2. If no significant spatial variation, move to Stage 3.

3. If spatial variation indicated, move to Stage 4.

STAGE 3 (if no spatial variation observed in Stage 2)

1. Select four evenly distributed sampling locations at no greater than 500 m intervals. If the beach is in excess of 2 km in length, include further sampling locations.

2. Conduct microbiological sampling at each of the four locations on 10-20 occasions at equal time intervals throughout one bathing season.

3. At the end of the year, assess Stage 3 data in conjunction with Stage 2 data plus outcomes of sanitary assessments to determine whether there is any significant variation (e.g. in response to rainfall).

4. If significant variation, then assess possibility of reclassification (see Section 9.4). Otherwise, confirm primary classification and proceed to routine monitoring (Stage 5).

STAGE 4 (if spatial variation is found in Stage 2)

1. If spatial variability is exhibited, affected and unaffected zones should be treated as separate bathing areas and each should be classified separately.

2. Determine the potential source and extent of the affected zone.

3. Delimit the unaffected zone; treat unaffected zone as in Stage 3 with one of the four identified sample locations at the poorer limit of the affected zone.

4. For the affected zone:

• A monitoring regime for a zone exhibiting spatial variability and likely to be affected by sewage contamination depends on the extent of the zone.

• It may be that the affected zone has to be managed by exclusion and that no monitoring is required, particularly if the zone is small in extent. Exclusion management action would apply where increased risk is restricted to a specific area. This implies, for example, fencing combined with general and site warning notices or general and site warning notices plus pro-active individual advice (such as from life-guards) not to use areas. The effectiveness of such management would need to be verified.

• If the affected zone warrants monitoring, then the Stage 3 process must be replicated. In such a case, if the zone is relatively small in area, fewer sample locations may be selected but sampled more frequently to provide a minimum of 20 data points.

5. At the end of the year, all data from a given zone are used to determine the primary classification to be applied.

STAGE 5

In the following year, microbiological monitoring is confined to five samples at each of the four identified locations within an individual zone (zones in excess of 2 km will require further sample locations). The five sampling occasions will be distributed evenly throughout the bathing season. A sanitary inspection should also be conducted. Routine monitoring requirements in subsequent years may vary, depending on the classification of the beach (section 9.5.5).

The individual data sets for the sampling locations will be further analysed to ensure that there is no significant difference between them. Assuming that no such variation is recognised, treat the data from all years as a single statistical body.

9.3.3 Determination of primary classification

Obtaining a primary classification for a given beach incorporates the results of both the sanitary inspection and the initial microbiological quality assessment described above. Once the appropriate categories for each of these criteria have been determined, a lookup table such as that in Table 9.14 can be used to determine the primary classification for the beach.

9.4 Reclassification

Microbiological contamination varies widely and rapidly. In addition, the risks to human health are associated principally with periods of high contamination. Thus:

• where a bathing area is subject to elevated faecal contamination for a limited proportion of the time or over a limited area of the potential bathing areas; and

• where the times of contamination can be predicted in some way; and

• where management interventions can be applied which effectively reduce or prevent exposure at these times,

it is reasonable to modify the beach risk evaluation to take account of the reduction in risk. This approach requires a database that allows an estimation of whether the significant faecal influence is constrained in time and whether "predictors" can be used to determine when such conditions are likely to occur. In addition, a locally applicable early warning system and subsequent management action that can be deployed in real time, must be determined. Finally, in order for a reclassification to be applied, evidence of the effectiveness of management action is required. Consequently a reclassification should be provisional; although it may be confirmed if the efficacy of management interventions is verified during the initial season of provisional reclassification. As the outcome of this process is of significant economic importance, it should be a requirement to ensure independent audit and verification wherever feasible, in order to satisfy the conflicts of interest that may arise.

Note that it may be appropriate to add an additional dimension to the resulting risk assessment to take account of special groups with increased risk, either because of the activity in which they engage or because they seek out areas not used by traditional bathers. Surfers represent such a special group. Alternatively, this may require an additional "commentary" element to the classification.

Table 9.14 Primary classification matrix

Sanitary Inspection Category	Microbiological Assessment Category (indicator counts)				
	A	**B**	**C**	**D**	**E**
Very low	Excellent	Excellent	Good	Good[2]	Fair[2]
Low	Excellent	Good	Good	Fair	Fair[2]
Moderate	Good[3]	Good	Fair	Fair	Poor
High	Good[3]	Fair[3]	Fair	Poor	Very poor
Very high	Fair[3]	Fair[3]	Poor[3]	Very poor	Very poor

[1] Reflects susceptibility to faecal influence

[2] Implies non-sewage sources of faecal contamination (e.g. livestock) and this should be verified

[3] Indicates an unexpected result which requires verification

9.4.1 Simple predictive approaches

It is impossible to predict every type of event that may leave an effect on every beach, because the variation is enormous. However, using one key issue that consistently affects bathing water quality, it is possible to delineate the principles that apply when dealing with such events. The objective is to define the conditions under which increased detection of sewage contamination (and, by inference, risk to human health) can be predicted. Exposure to risk at these times may be reduced by direct interventions. If such interventions can be demonstrated to be effective, then upgrading the classification of the beach to reflect the reduced health risk can be justified.

The issue selected to illustrate this predictive approach is rainfall. To provide appropriate information for this process, rainfall data (real time and historic) must be available. The location of existing rain gauges can be surveyed to determine the optimal position from which to predict effects on the beach. In addition, to determine the effect that a rainfall event may have on a bathing water, sources of contamination to the beach must be categorised; primary inputs of concern are CSOs, riverine and storm drains. Examples of the type of information required for each input are given in Box 9.3.

A protocol can be adopted to investigate whether deterioration in water quality at recreational beaches is predictable and hence subject to appropriate management action. The assumption is that a local administration wishes to contend that a beach has experienced water quality deterioration and that this deterioration is predictable. A number of study designs have been adopted and could be of use. All assume a sanitary inspection of the types of sources listed in Box 9.3. Although the use of simple predictive approaches requires additional work to plan and implement, such approaches are not highly expensive.

Box 9.3 Discharge sources associated with rainfall

Generic factors associated with combined sewer overflows (CSOs), riverine and storm drain inputs

Predictive outputs should be evaluated by examining a set of historical data to determine whether the predictor would previously have accurately predicted exposure events. Basic data requirements include: rainfall history, rainfall intensity (a function of amount and duration), sewage flow, location of discharges and definition of the zone of influence. Catchment and population equivalent loadings also need to be defined. Here location and zones of influence (resulting in both inputs and outputs) need to be defined. The zones of influence should lead to the delineation of impacted bathing areas. It is essential to undertake at least one intensive run of monitoring associated with an event or series of events. This monitoring should include a determination of the estimated extent of the impacted area linked to the various baseline data collected. Thus the rainfall intensity leading to a defined impacted area may be determined. If resources do not enable extended feedback monitoring to differentiate between different event intensities, then the predicted worst case zone of impacted area should be defaulted to. These data and their interpretation will provide the predictive base for estimating thresholds for subsequent events.

In some circumstances, a combination of other factors associated with the rainfall event may be used to determine the predictive capacity. These will include climatic and hydrographic conditions - specifically tide current and wind. Such factors could affect the occurrence/non-occurrence of an event, the likely zone of impact, and the duration of the event outcome.

Discharge source: combined sewer overflows

Background information

Combined sewer overflow discharges are derived from localised urban catchments. There are none of the 'softening' effects characteristic of riverine systems, typified by peaks and troughs of contamination. Effects are manifested rapidly. There is a simple, direct relationship between rainfall and discharge. Storage capacity exists on many current systems, and small events may therefore be contained. A widely applied "rule of thumb" is that effects may become obvious when dry weather flow is exceeded threefold. This is already incorporated into many systems. When an event triggers the threshold, the effect is rapid, with a potential for high microbial load and high public health risk.

Utility in prediction

Low rainfall may be accommodated; typically there is a threshold that will trigger an increased risk outcome. The best predictor may not be rainfall itself - it is the actual flow within the system. A relationship between rainfall and flow through the system that will trigger an alert must be determined. While good practice dictates that they should discharge below low water, CSOs may discharge directly onto the beach. Direct measure of the CSO operation forms the process; when they are operating, the risk is real.

Discharge source: riverine

Background information

Rainfall in a catchment affects all its contributing inflows in a complex way over a wide area: delays, complex flow characteristics (including non-"plug" flow) and a series of small plugs may result. Riverine inputs are potentially the sum of multiple discharges from sewage systems, CSOs, storm drains and other industrial and rural sources. Where riverine pollution is dominated by a single pollution source, which may manifest as a plug, rainfall is a likely predictor in a relatively simple relationship. A significant increase in flow after a relatively long low-flow period could lead to sediment remobilisation and associated contamination. All likely influencing factors in a catchment must be categorised and identified (e.g. CSOs, storm drains, likelihood of sediment resuspension and surface run-off from grazing land). Effects of multiple CSOs and storm drain events contributing to a major riverine outcome are very complex to predict. They may lead to delays and staggered loadings, varying in intensity. The resuspension of sediments is related to extended dry periods and river flow. Complex and multiple sources of contamination can result and the health risk is difficult to predict.

Utility in prediction

Outcomes are difficult to predict; generally there is a variable delay in events manifesting themselves. There may be multiple, overlapping sources resulting in an unpredictable duration. Predisposing weather conditions, particularly the first major event after a period of low rainfall or low river flow, should signal a potential risk outcome. In terms of run-off, predictors will include rainfall intensity likely to lead to a threshold effect. Agricultural practices will modify the nature and extent of run-off and, in turn, may vary the threshold.

Discharge source: storm drains

Background information

Storm drain discharges are associated with localised, generally urban sources. In principle storm drains should not be connected to the sewage system, and therefore should not have a high sewage loading. Provided that there is no sewage connection, the likely discharge is generally of low significance to public health. However, the discharge may be associated with high total coliform (and sometimes high thermotolerant coliform) counts, which are a poor predictor of health risk. Generally storm drains discharge directly onto the beach and as a result, if they are connected to the sewage system, there will be an increased health risk.

Utility in prediction

Storm drains respond directly to rainfall. There is no storage capacity and therefore no delay in the outcome of the event. In effect, there is no threshold before a discharge occurs. Thus the system response is almost instant. A flushing effect means that the most significant (albeit generally low) health risk is at the start of the event. There is no simple relationship between the amount of discharge and risk burden; as time progresses the contamination load may be exhausted. The first rainfall releases a discharge contaminant plug, while subsequent rainfall leads to a discharge with little contaminant loading.

9.4.2 Advanced predictive approaches

More advanced studies have been developed to provide data on: the reasons for short-term elevated microbiological indicator counts; the timing of such elevated analytical results; the time taken for water quality to return to "baseline" conditions; the potential for prediction of water quality change; and the potential for remediation of poor water quality. Although these studies were designed initially for use under percentage compliance based regulatory structures, they are also very valuable tools for the classification approach suggested in this chapter.

Studies of this type from the UK suggest that well founded scientific studies (i.e. "compliance" modelling, budget studies, diffuse source modelling and near-shore modelling) would require between ten thousand and hundreds of thousands of USA dollars, depending on the complexity of the study. Where a full site study is required, the beach authority wishing to claim that prediction of elevated microbiological indicator counts is a feasible management tool for public health maintenance, should plan for and appropriately resource a potentially costly 12 month study.

Compliance modelling

This type of investigation was initially designed to understand the causes of occasional "high values" leading to a failure to comply with percentage compliance based standards. These investigations require reliable microbiological data covering several years and possibly several locations, as well as a set of variables that have been proved to predict microbiological concentrations at the study sites.

Multivariate statistical methods, such as multiple regression, can be applied to the data set to predict faecal indicator concentrations. The success of modelling should be judged on the basis of the explained variance (R^2) of the predictive multivariate model assuming statistical significance. Values for R^2 over 60 per cent for a particular beach year have been achieved. Nevertheless, this approach should not be adopted if there are insufficient sampling periods for each year (e.g. less than 20). In addition, careful control on variable inclusion (and hence multicolinearity) is required in model construction and constant input from a professional statistician is also essential.

The initial modelling study is an exploratory tool. It suggests predictability, which should be confirmed by further sampling of inputs through a budget investigation.

Budget studies

Budget studies can be undertaken if the initial modelling proves the possibility of a relationship with predictable inputs. This type of investigation requires the characterisation of inputs to a bathing water. It is vital that low flow and high flow inputs are measured, together with quantity and quality measurements. Potential sources of pollution include sewage effluent, CSOs and SSOs, rivers, avian inputs, bather loading, septic tanks, industrial discharges, private discharges and lagoon outlets. For these sources, data are required on the type of source and pollution input, frequency of episodic inputs, magnitude of all inputs, (e.g. base flow and episodes, duration of inputs, the flow volume of all inputs), and the microbiological quality of all inputs.

Budget studies provide information that is known to be episodic. Clear evidence that, during specific events, beach microbiological concentrations are commonly dominated by predictable (but non-sewage) sources of faecal indicators would provide local managers with evidence that elevated counts associated with such events would not pose a large risk to public health provided effective management action is taken to limit bather exposure during this time period.

Diffuse source modelling

If riverine inputs to a bathing water are derived from diffuse or non-point source areas, remediation of a beach with poor quality bathing water would require "catchment area" or watershed management. Lumped and distributed models have been applied to predict episodic catchment-derived sources of pollution. The construction of a diffuse source model of the upstream catchment can offer evidence of the contamination being derived from non-sewage sources. Information can therefore be provided by these studies to aid decisions on remediation strategies.

Such modelling requires the definition of sub-catchment units and the implementation of an intensive and targeted data collection exercise to characterise water quality from each characteristic sub-catchment unit. The intensity of agricultural land use and stocking density are of particular importance. Both stochastic multivariate and deterministic modelling have been applied, with good prediction of faecal indicator delivery based on agricultural land use types.

Near-shore hydrodynamic modelling

When the inputs to the beach have been identified and characterised as above, the next question becomes the impact of these constant and episodic inputs at different locations on a specific beach site. One tool applied to this problem is the use of near-shore hydrodynamic modelling. This type of modelling requires tidal information, water quality dynamics (e.g. T_{90} values for microbiological indicators), wind speed and direction and sampling regimes. Significant data inadequacy exists in the currently available T_{90} values, which describe decay rates. Thus new scientific information is required. In addition to these data, elements such as wave height and sedimentary resuspension may be important predictors of microbiological contamination. However, they are not specifically addressed in current modelling systems.

The near-shore hydrodynamic modelling approach requires complex, finite element modelling. A high level of expertise is necessary to use this approach successfully to predict compliance in shallow near-shore waters. However, such approaches can accommodate both constant and episodic inputs to bathing waters, dynamic change in the near-shore waters, and impact under different tidal states and hydrometeorological conditions.

9.5 Management actions and routine monitoring

Key elements in protecting human health from potential risks associated with recreational or bathing waters are the identification of pollution sources (continuous and intermittent), assessing their impact on the target area and undertaking remedial or management action to reduce their public health significance. Depending on the circumstances, there may be a number of actions that can be taken to reduce public health risk. Such actions would therefore have an impact on the overall classification of the bathing water.

Routine monitoring should be undertaken to determine if the classification status of a beach changes over time. If management actions are shown to be effective and a beach can be reclassified as a result, monitoring requirements may be substantially reduced. Examples of classifications and their associated management and monitoring actions are given in Table 9.15.

9.5.1 Direct action on pollution sources

Direct action should be the principal management action because, if successfully undertaken, it provides a permanent and verifiable reduction of potential health risks. Remedial actions can include: diversion of sewage discharges away from the target area by the construction of long sea outfalls, provision of higher levels of sewage treatment, and increasing storm water retention to reduce frequency of discharge and/or relocation of intermittent discharges. These actions may, however, be outside the control of local communities or regional authorities and an alternative approach of local intervention may be more applicable.

9.5.2 Managing intermittent pollution events

Where there is clear evidence that water quality varies at certain predictable periods, such as following significant rainfall events, it may be possible for local management to undertake verifiable interventions that would reduce public health risks. Interventions would include passive non-verifiable actions, such as advising local residents and tourists not to bathe in the affected zone of the intermittent discharge for a given period following heavy rainfall. Active and verifiable interventions could include posting warnings around the affected zone following a rainfall event, advising bathers not to swim for a given period of time. In addition, advice could be given about the location of alternative bathing waters and transportation could be provided to and from those locations. Lifeguards, if present, could re-enforce the message. More restrictive measures could be the closure of relevant car parks and service industries (but not sanitary facilities).

Table 9.15 Examples of classification outcomes and their associated management and monitoring actions

Primary classification	Reclassification	Generic statement for public (non-verifiable, passive action)	Generic management advice[1] (verifiable, active action)	Monitoring requirements[2]
Excellent	-	Excellent beach.	NA	Annual sanitary inspection to ensure no change. Microbiological quality assessment every five years to verify status.
Good	Excellent[3]	This beach is of good quality.	No action needed on health grounds. Action may be warranted for local tourist promotion.	As above.
Fair	Good[4]	Inform public through advice at beach and tourist locations that bathing at location X is discouraged.	Post beach (i.e. bathing discouraged between specified posts). Restrict access (i.e. do not allow car parking). Discourage service industries. Fence area off. Encourage alternatives via car parks, bus stops and service industries.	Annual sanitary inspection to verify no change. Low-level microbiological quality assessment (4 samples on 5 occasions, equally spaced throughout the bathing season). Abnormally high samples need further verification and additional monitoring and possible review of impacted zone. Annual verification of management intervention effectiveness.
Fair	Good[5]	Inform public through advice at beach and tourist locations that bathing is discouraged after periods of heavy rainfall.	Post notice at bathing water. Use lifeguards to warn bathers. Close car parks and service facilities. Stop tourist buses. Encourage use of alternative beaches by providing free transport.	As above.
Poor	Good/fair	This area is of periodic poor quality and bathing is discouraged at certain locations/times.	Advice similar to that for "Fair"	As for "Fair".
Very poor	Not affected by local management	This area may be polluted with (nature of pollution) from (type of source). This may be unpleasant for bathers and presents some risk to human health.	Post generic warning notices similar to the risk statement at access points to the beach. Use posters to inform of alternative locations. Do not allow development of service industries. Make access difficult (e.g. no provision of car parks). Encourage use of alternative bathing areas. Encourage pressure for remedial action.	Annual sanitary inspection to confirm no changes to primary pollution source. Microbiological quality assessment every five years to verify status.

[1] The level of action depends on the likely health impact of the event

[2] Includes requirements for sanitary inspections and microbiological quality assessments

[3] As defined by the conditions of contamination

[4] As defined by the area of contamination

[5] Increased contamination occurs under certain conditions

9.5.3 Management interventions on spatial pollution

It is possible for a bathing water to be only partially affected by a source of human sewage. For example, a riverine input containing sewage from upstream communities may flow across a bathing water causing significant elevation in microbial indicator concentration. Unless direct action can be taken as outlined in section 9.5.1, various alternative options exist for reducing public health exposure. These options can range from the passive provision of information to the general public that bathing at the location was not advised, to actively dissuading bathing, such as by not providing public transportation or car parking near the affected area or by fencing off the area. As suggested in section 9.5.2, the policy of dissuasion should be reinforced by information about alternative bathing areas together with some encouragement, in the form of transport, easier parking or provision of service industries, etc., to entice bathers away from the polluted area.

9.5.4 Management of polluted zones

Where the whole extent of the bathing area is considered to pose a potential health risk and interventions along the lines of those described in section 9.5.1 are not feasible, management actions are needed to reduce the use of the bathing area. As before, information can be given to the public informing them of the water quality problems associated with the bathing water and this can be re-enforced by actions such as making access difficult by controlling car parking facilities, and by closing service industries. Additionally, information regarding alternative bathing waters of a similar nature, but with acceptable water quality, should be provided.

9.5.5 Routine monitoring

Under the classification scheme, routine monitoring always requires an annual sanitary inspection, to confirm that no changes in the primary pollution source(s) have occurred over the course of the year. In addition, microbiological quality assessments should be carried out, although the level of monitoring required for a given beach may depend largely upon its classification, as shown in Table 9.15. Beaches classified as very high or very low quality (i.e. "excellent" or "very poor"), for example, may only need a microbiological quality assessment every few years, to verify that their status has not changed. Mid-level ("good", "fair" and "poor") beaches may require an annual, low-level microbiological quality assessment, with 20 samples being taken at a minimum of four sites on five occasions evenly spaced throughout the bathing season. Beach zones greater than 2 km in length may require additional sampling sites. Further sampling may be necessary if abnormally high microbiological levels are found. If a beach has been reclassified, annual verification of the effectiveness of management interventions would also be required. When results of this routine monitoring suggest that the status of a beach has altered, the classification of the beach should be revised following a process similar to that described in Figure 9.6.

9.6 Evaluation and validation of the proposed approach

A classification scheme of the type proposed in this chapter is of value if it accomplishes one or more of the following goals:

- Contributes to informed personal choice (e.g. individuals, by using the information provided, can and do modify their exposure). This implies inter-location comparability and an informed public.

- Contributes to local risk management (e.g. by excluding or discouraging access to areas or at times of increased risk, thereby reducing overall exposure).

- Assists in making maximum use of the minimum necessary monitoring effort.

- Assists local decision making regarding safety management.

- Encourages incremental improvement and prioritises areas of greatest risk.

In order to evaluate whether the goals have been reached, both field testing and evaluation of the scientific validity of the approach is required. A limited number of intensive studies would be necessary to test the scientific validity of the approach; in recognition of the importance of this, the participants at the Annapolis meeting developed a protocol for such a study. This protocol requires extensive sampling of study sites, as described in the following sections, and should not be confused with the less rigorous microbial assessments necessary for classifying a beach under the scheme described in this chapter.

9.6.1 Validation protocol

Many countries around the world are interested in establishing uniform recreational water monitoring protocols that would provide accurate assessments of water quality in a timely manner. Scientists and public health officials recognise the need for monitoring approaches, such as that proposed in this chapter, that would characterise a bathing water at reasonable cost and within the constraints of limited resources (personnel and equipment and supplies). To establish such protocols, it is important to determine the essential parameters that must be considered in the monitoring programme, e.g. temporal, spatial and environmental considerations. The sampling of a recreational water must be adequate to capture all of these factors to ensure the likelihood that samples portray the water quality at the time they are taken.

The establishment of a robust set of data from multiple, contrasting locations and conditions is essential to determine general sampling requirements that are transferable to most locations world-wide. It is desirable that all parties interested in improved monitoring approaches participate collectively in conducting studies to develop the data for determining the minimum sampling requirements (at least for typical beach environments), in freshwater, estuarine and marine settings. In order to develop such a database, a standard sampling protocol which can be used (and adhered to) by everybody is required, whereby the data derived from each study would be compatible with data from the other sampling studies. The following is a recommended approach to identify the major elements, parameters and conditions to be developed by the sampling protocol that would be applied to beach studies intended to describe the important monitoring features for recreational waters. This protocol should be implemented in conjunction with a sanitary inspection, as described in Chapter 8.

Microbiological parameters

Two microbial indicators of faecal contamination were selected for this sampling study protocol: faecal streptococci or enterococci and sulphite reducing *Clostridium* or *Clostridium perfringens.* The protocol can apply equally to other indicator organisms described in Table 9.5, such as *E. coli* in freshwaters. The indicators proposed in this protocol development were chosen because the methods for their detection and enumeration have been well described and field tested by a number of investigators in numerous recreational water studies as well in other environmental testing. There is a large database that describes the precision, accuracy and coefficients of variation for these methods. These methods were also chosen because they are considered applicable for both marine and freshwater testing.

Temporal study conditions

The studies should be performed at least over the period of a typical bathing season, which can range from several weeks to all year round, depending on latitude and local customs. A three-month sampling period or longer is considered best to obtain a robust set of data to analyse for temporal effects under most circumstances. Under most conditions a minimum of 50 days of sampling is considered a robust study. This should provide satisfactory data to establish important factors or conditions at a study site and that will allow the assessment of important locations for sampling, when to sample, and to establish factors that contribute to microbiological water quality variability. This amount of study data should allow assessment of critical factors that may trigger sampling (e.g. regression, multivariate regression, trends, etc.) when applied to a beach. It should also allow the combination of data from various studies to make the assessments more robust, so that guidance may be derived for dissemination to all persons concerned with public safety at beaches.

Sampling should encompass daily periods and should be conducted at least several times a week. Pollution varies in response to the density of users and the local population who may be discharging to the sewage system (e.g. peak uses may often occur at weekends and holidays). In addition, local events may occur routinely with resultant effects on waters serving recreational areas. The sampling protocol should take account of these factors, so as not to introduce a bias to the data set. Sampling should be carried out hourly over a 12 hour period, for example from 0700 hours to 1900 hours for all sampling locations comprising the beach study site.

Event sampling

Many studies to date have demonstrated that one of the most significant factors leading to increased faecal pollution in recreational waters is rainfall. While the general sampling protocol described above should pick up the effects of rainfall events over a long recreational season, it may not do so for short-term evaluations. For locations subject to rainfall events, the general sampling protocol could, therefore, lead to a lack of data covering these event and their contribution to microbial pollutant loading at a beach. If feasible, it is recommended that at least 20 per cent of the study sampling days should be during and after rainfall events where there is, or there is likely to be, local run-off.

Spatial sampling conditions

It is very important in sampling studies (for establishing uniform monitoring guidelines) to characterise the water at a beach from the swash zone (i.e. the sand area that is covered with waves on an intermittent or occasional basis during the sampling period) out to the most distant locations confining the beach (but at least to chest height), at the depths where exposure is likely to occur, and also along the designated width of the beach (parallel to the shore line). This becomes the designated area of water that a single sampling event, from off-shore sampling periods in a single day, should characterise. The designated area comprises a grid of sampling locations that are sampled for each period.

Sampling grid

The spacing of the length between grid samples running parallel along the beach should be uniform at 20 m and with a minimum of three locations (resulting in a minimum of 60 m total distance). Beaches shorter than this recommended length cannot be considered for incorporation into the sampling validation study.

Sample site distances perpendicular to the shore should be located from the 20 m grid transects. These locations should be:

- Ankle deep (0.15 m from grid transect on shore).
- Knee deep (0.5 m from grid transect).
- Chest deep (1.3 m from grid transect).

Samples should be taken at the following depths:

- Ankle depth sample ~ 0.075 m below the water surface.
- Knee and chest depth samples ~ 0.3 m below the water surface.

Although sand samples are not an absolute requirement for this sample validation study, they are considered to be desirable. Sand samples should be taken from the swash zone and from the top 2 cm of the sand. Enough sand should be taken to enable one portion to be used to establish the dry weight and another portion to be used to elute microbial components for quantification.

Sampling and analysis

A single, discrete sample should be taken from each location at each period. Each sample must be labelled with location, day and time taken, and any other distinguishing characteristics needed to identify the sample. Samples must be iced, packaged and transported by surface or air to the laboratory for processing and analysis. Sample analysis must be initiated within 8-12 hours and all discrete samples must be assayed in triplicate for each dilution (i.e. three dilutions, although this may be reduced to two if, or when, the water becomes well characterised for the presence of indicators under various sampling conditions). Other test or observational variables for each sampling period are:

Physical and chemical variables

- pH (daily).

- Salinity - estuarine (hourly); marine, if no significant riverine influence (daily).

- Turbidity (hourly).

- Water and air temperature (hourly).

Other observations and measurements

- Rainfall - magnitude, duration, time relative to sampling (every 6 hours).

- Wave height (hourly).

- Current direction and speed - fresh and estuarine (hourly).

- Total light or radiation (hourly).

- Tidal state and magnitude (hourly).

- Wind direction and speed relative to beach (hourly).

- Per cent cloud cover (hourly).

- Bather population at each transect point (e.g. by means of photographs) (hourly).

- Animal population - presence and number of horses, donkeys, dogs, shore birds (hourly).

- Boats anchored or moored within 1 km of the beach (hourly).

- Beach debris and sanitation: sanitary plastics, visible grease balls, algae (daily).

- Location of freshwater, storm water, sewage outfall or other intrusion to beach.

- Location of bather facilities (showers, lavatories) and relevance of input from these sources (shower run-off, sewage overflow) to beach.

Database requirements are:

- All raw data should be provided to a computerised database system.

- All data should be entered into a spreadsheet compatible with universal spreadsheet formats.

- All data entered should be validated for accuracy.

- All data should be duplicated in separate computer files for future access.

Analysis of the data should generate the following descriptive statistics:
- Number of samples taken.

- Geometric means per sample, per replicate, etc.

- Standard deviation.

- Quality assurance and quality control results.

- Coefficients of variation, precision and accuracy of methods used by the laboratory.

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Chapter 10*: CYANOBACTERIA AND ALGAE

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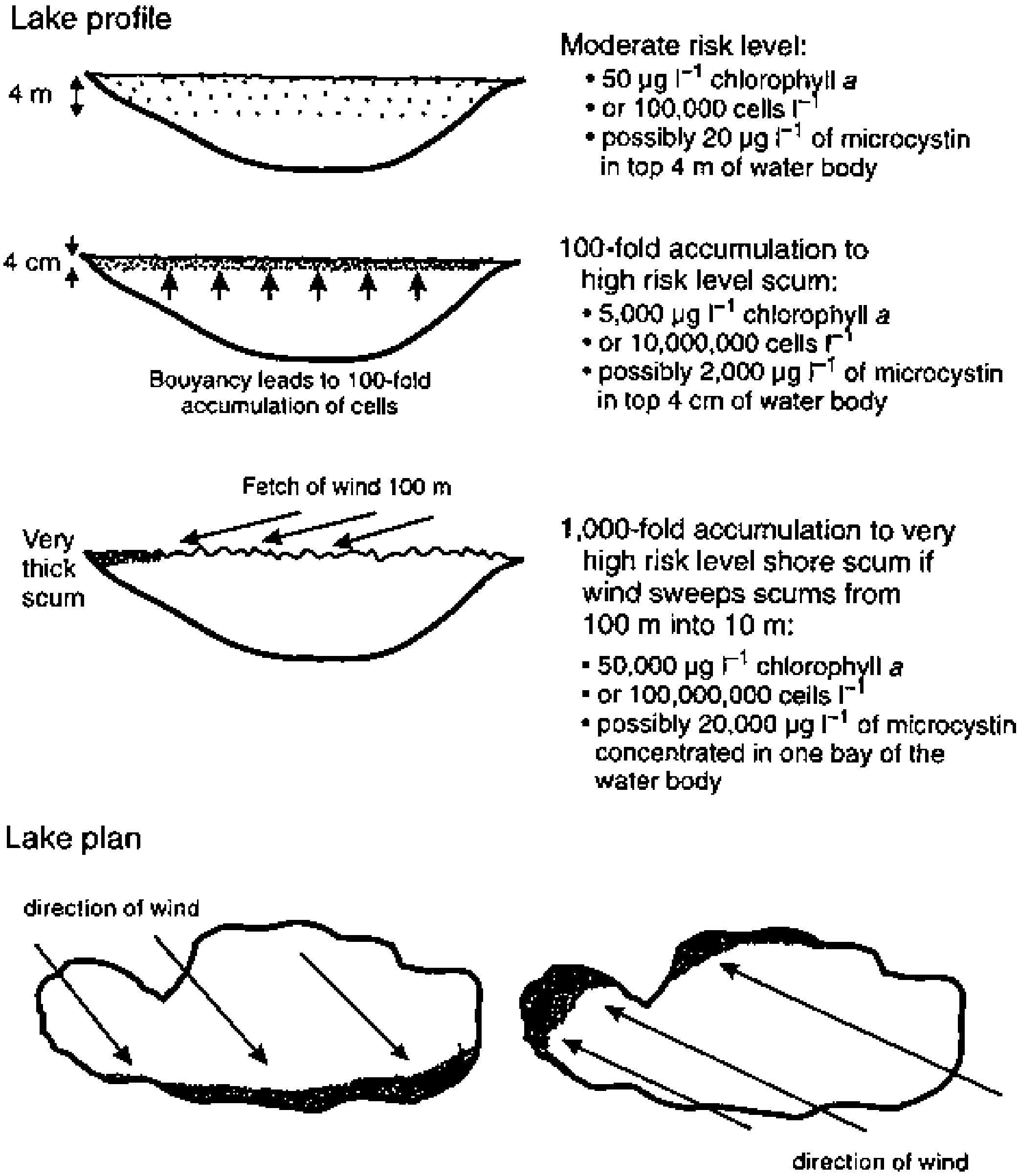
In freshwaters, scum formation by cyanobacterial phytoplankton is of concern to human health. Freshwater algae proliferate quite intensively in eutrophic waters and may contain irritative or toxic substances. Nevertheless, incidents of impairments of human or animal health caused by algae are rarely reported. One example was the closure of a number of bathing sites in Sweden because of mass occurrences of the flagellate *Gonyostomum semen* which causes skin irritations and allergies (Cronberg *et al.,* 1988). Incidents attributed to cyanobacteria are far more numerous and, in most cases, have been caused by species of cyanobacteria that may accumulate to surface scums of extremely high cell density. As a result, the toxins they may contain ("cyanotoxins") reach concentrations likely to cause health effects.

Surface aggregations of planktonic cyanobacteria occur because of their capability to regulate their buoyancy, enabling them to seek water depths with conditions optimal for their growth. Regulation of buoyancy is a slow process, and cells adapted to ambient turbulence may take several days to adapt their buoyancy when conditions change (e.g. turbulence is reduced). Thus, cells or colonies may show excessive buoyancy and accumulate at the water surface. Light winds drive such accumulations to leeward shores and bays, where the resulting scums become thick. In extreme cases, such agglomerations may become very dense, with cells frequently concentrated by a factor of 1,000 or more, eventually reaching in some cases, one million-fold concentrations with a gelatinous consistency. More frequently, surface accumulations are seen as streaks or slimy scums that may look like blue-green paint or jelly. Such situations can change rapidly within hours with changes in the wind direction. Monitoring strategies must take into account this highly dynamic variability of cyanotoxin occurrence in time and space.

Scum formation is influenced by the morphological conditions of the water body, such as the water depth from which cyanobacteria can rise to the surface (i.e. the thickness of the stratum in which they are dispersed) and the length of wind fetch over which surface aggregations can be swept together to form shoreline scums (Figure 10.1). Accumulated scum material may take a long time to disperse especially in shallow bays. Dying and lysing cells within the scum release their contents, including the toxins, into the water. However, toxin dissolved in the water is rapidly diluted and probably also degraded. Cell-bound cyanotoxin concentrations usually are the greater cause for concern in recreational waters.

Figure 10.1 Schematic illustration of scum-forming potential changing the cyanotoxin risk from moderate to very high (After Falconer *et al.,* 1999)



Most of the problems reported with nuisance and toxin-containing aquatic cyanobacteria in freshwaters have involved planktonic species, i.e. those distributed in the water body or forming surface scums. However, benthic (i.e. bottom-dwelling), species have occasionally surfaced, and been washed ashore where they have caused the death of dogs scavenging upon the material. Benthic cyanobacteria can grow as mats on sediments in shallow water. Some mats become detached and are driven onshore where they result in acutely toxic accumulations (Edwards *et al.,* 1992).

Severe illness due to direct dermal contact with such mats has been reported from tropical marine bathing sites (Kuiper-Goodman *et al.,* 1999). In coastal marine environments many toxic species of dinoflagellates, diatoms, nanoflagellates and cyanobacteria occur, and have led to several forms of human health impacts mainly after consumption of shellfish and fish, i.e. syndromes such as Paralytic Shellfish Poisoning, Diarrhetic Shellfish Poisoning, Amnesic Shellfish Poisoning, Neurotoxic Shellfish Poisoning and Ciguatera. Nevertheless, there exists very little scientific evidence that marine toxic algal blooms cause health problems for recreational users of water. This evidence is reviewed in the WHO *Guidelines for Safe Recreational-Water Environments* (WHO, 1998).

As a result of the lack of evidence for effects of toxic algae on recreational users of marine waters, this chapter is concerned principally with monitoring and assessment of toxic cyanobacteria in freshwaters. Nevertheless, the methods given for cyanobacteria may be employed to assess the development of other planktonic algae. Although various toxic marine algae have been associated occasionally with human health effects, concern for human health centres on toxic cyanobacteria, and therefore these are the subjects of the remainder of this chapter. Further information concerning coastal phytoplankton blooms and associated monitoring strategies are available in Franks (1995), Smayda (1995) and in the UNESCO manual on this subject (UNESCO, 1996).

For lakes, reservoirs and rivers different levels of "alert" (for cyanobacterial cell concentrations and their toxin contents) have been proposed in *Toxic Cyanobacteria in Water* (Chorus and Bartram, 1999) and in the *Guidelines for Safe Recreational-water Environments* (WHO, 1998). These documents present a series of guideline values and situations associated with incremental severity and probability of greater effects in relation to cyanobacterial occurrence (Table 10.1).

10.1 Design of monitoring programmes

Many cyanobacterial species frequently forming mass developments may contain hepatotoxins or neurotoxins, and all of them contain lipopolysaccharides (LPS) in their cell wall. Lipopolysaccharides may be the cause of irritations of the skin, digestive tract, respiratory membranes, eyes and ears that are frequently associated with cyanobacteria. Research in pharmacology and ecotoxicology indicates that cyanobacteria contain a variety of substances not yet identified, but that may have a potential impact on people. The implications of the present state of knowledge for surveillance and management are that any mass development of cyanobacteria may be a potential health hazard. If the cyanobacterial cells contain hepatotoxic microcystins, cause for concern may be higher because of the chronic effects of this potent toxin. Therefore, monitoring should address primarily the occurrence of cyanobacterial mass developments, whereas microcystin analysis may be adequate in specific situations.

Table 10.1 Phase 3 monitoring - guidelines for safe practice in managing recreational waters according to three different levels of risk

Level of risk[1]	Health risks	Recommended actions
20,000 cells cyanobacteria per ml or 10 µg l^{-1} chlorophyll a with a dominance of cyanobacteria	Short-term adverse health outcomes (e.g. skin irritations and gastro-intestinal illness, probably at low frequency)	Post on-site risk advisory signs Inform relevant authorities
10^5 cells cyanobacteria per ml or 50 µg l^{-1} chlorophyll a with a dominance of cyanobacteria	Potential for long-term illness with some species Short-term adverse health outcomes (e.g. skin irritations and gastro-intestinal illness)	Watch for scums Restrict bathing and further investigate hazard Post on-site risk advisory signs Inform relevant authorities
Cyanobacterial scum formation in bathing areas	Potential for lethal acute poisoning Potential for long-term illness with some species Short-term adverse health outcomes (e.g. skin irritations and gastro-intestinal illness)	Immediate action to prevent contact with scums; possible prohibition of swimming and other water-contact activities Public health follow-up investigation Inform relevant authorities

[1] Expressed in relation to cyanobacterial density and given in order of increasing risk

Visual inspection of a bathing site is of crucial importance because it shows immediately whether cyanobacteria occur in potentially hazardous densities. However, as scum formation and dispersion may occur within hours and thus too frequently for monitoring, assessment of the risk of cyanobacterial exposure during recreational activities is greatly enhanced by an understanding of the population development of these organisms in a given water body (i.e. through background monitoring of variables that enable their proliferation). Good knowledge of the local growth conditions for cyanobacteria can greatly enhance predictability of bloom formation. As knowledge and understanding of a given site accumulate, regular patterns of cyanobacterial growth may be noticed, and surveillance may as a result be focused upon critical periods. Involving limnological expertise may be very useful, particularly during the development of monitoring programmes and for their periodic assessment for efficacy.

10.1.1 Monitoring strategy for freshwater cyanobacteria

A structured, quantitative investigation approach aims at focusing surveillance efforts upon those sites that are likely to present a risk. It further provides a scheme for immediate assessment and action by discerning three steps of action (Table 10.2).

- Determination of the carrying capacity of the ecosystem for cyanobacteria.

- Site inspection to detect mass developments.

- Quantitative assessment of biomass as a basis for risk assessment when mass developments occur.

Preliminary identification of water bodies potentially harbouring high densities of cyanobacteria is possible on the basis of simple transparency measurements with a Secchi disc. If transparency is high (greater than 2 m) and no significant discolouration of the water can be seen, cyanobacterial densities are unlikely to be high. It should be noted, however, that in large and deep lakes with a large volume from which cells can accumulate to scums, high densities in some parts of the lake cannot be excluded. In water bodies with Secchi disc readings of less than 2 m, the following three steps may be undertaken to assess the risk of toxic cyanobacteria.

Step 1. Determination of the carrying capacity of the ecosystem for cyanobacteria

Algal and cyanobacterial population growth requires phosphorus and nitrogen. The concentrations of these nutrients determine the maximum amount of algae and cyanobacteria that can develop in a given water body, or the "carrying capacity" of an ecosystem for these organisms. The carrying capacity is more often limited by the availability of phosphate but sometimes, particularly in marine ecosystems, it may also be limited by nitrogen. If total phosphorus is not limiting, it may be worthwhile to analyse nitrogen to check whether the carrying capacity may be lower than assumed from the total phosphorus concentrations. Total phosphorus should be measured rather than dissolved phosphate (soluble reactive phosphate (SRP), also known as orthophosphate) because algae and cyanobacteria can store sufficient amounts of phosphate to increase their population 10-fold, even if no dissolved phosphate can be detected. Thus, cell-bound phosphate (which is included when measuring total phosphorus but is missed when measuring only dissolved phosphate) is more meaningful for the assessment of carrying capacity.

Total phosphorus should be assessed several times during the cyanobacterial growth season in order to check temporal variability. If variability of the concentrations is low (less than 50 per cent), assessment twice a year may prove sufficient (in subtropical and temperate climates in spring at total overturn and in summer during the main bathing season). If total phosphate concentrations are below 0.01-0.02 mg l^{-1} P, mass developments of cyanobacteria are unlikely and high turbidities, (if present) may have other causes. If total phosphate concentrations are higher, monitoring should move to Step 2 in order to check for the presence of phytoplankton mass developments.

Table 10.2 Parameters to be measured or assessed for each of the three phases of the recommended monitoring strategy

Phase or activity	Rationale for monitoring	Variables
Monitoring phases		
Phase 1 (background)	Potential for cyanotoxin problems	Nutrient concentrations (total phosphorus, nitrate and ammonia) Transparency (Secchi disc) Hydrophysical conditions (e.g. flow regime and thermal stratification) Other biological complex interactions
Phase 2 (basic)	Site inspection for indicators of toxic cyanobacteria	Transparency (Secchi disc) Discolouration Scum formation Hydrophysical conditions Temperature Weather conditions (e.g. winds, light) Changes in turbulence (i.e. mixing) Other biological complex interactions
Phase 3 (cyanobacteria)	Qualitative/quantitative assessment of potentially toxic cyanobacterial assemblages	Transparency (Secchi disc) Qualitative microscopic analysis to identify dominant taxa (genus is sufficiently precise) Quantitative microscopic analysis (only as precise as needed for management)[1]
	Determination of cyanobacterial biomass	Chlorophyll a analysis (provides an estimate of cyanobacterial biomass in the case of rather monospecific blooms)
Additional activities		
Toxicity	Presence of toxicity	Bioassays
Toxin analysis	Presence of specific toxins (qualitatively and quantitatively)	Chemical analyses

[1] The ratio of the concentration of algal to cyanobacterial cells (or filaments or colonies) in the water sample (as determined by enumeration) may be converted to biomass values

Step 2. Monitoring to detect possible mass developments of phytoplankton (algae plus cyanobacteria) by visual inspection of bathing sites and immediate actions to prevent health hazards

Monitoring should generally be performed at fortnightly intervals. The areas most likely to be affected should be assessed first, such as the downwind shorelines. Information on changes in wind direction or strength in the preceding 24 hours may be valuable for understanding the movement of surface scums in the water body.

Visual inspection should consider the three following conditions. Each condition may be considered a prerequisite for the following condition.

1. Determine if Secchi transparency is less than 1 m or, in absence of a Secchi disc, if the bottom of the lake cannot be seen at 50 cm depth along the shore line; if so

2. Determine if cyanobacteria are visible as a greenish discolouration of the water or at the water's edge or as green or blue-green streaks on the water surface (note: evenly dispersed greenish discolouration may also be due to algal phytoplankton rather than to cyanobacteria, and microscopic assessment is necessary to determine the causative organism, but surface scums and streaks may be attributed to cyanobacteria); if so

3. Determine if a green or blue-green scum is visible on the water surface in any area.

If cyanobacteria are visible and if their population density exceeds one of the guideline values given in Table 10.1, then action appropriate for the given location should be taken, such as informing responsible authorities, initiating the posting and publication of warning notices, regular monitoring according to Step 3 below, and deciding whether or not to initiate immediate action such as marking or enclosing affected areas (if sufficiently small) and prohibiting access to such areas by water users; restricting access to the water edge that is affected, other than for launching boats; and regulating or restricting access by all recreational users where cyanobacterial blooms cover the general waters.

High nutrient inputs favour the development of cyanobacteria and algae. Therefore, if cyanobacteria are present, inspect the catchment area for signs of sewage outlets, excessive fertilisation close to the shoreline, erosion, or other potential sources of phosphate input. The identification of such sources provides the basis for measures addressing the cause of the problem.

Step 3. Quantitative assessment of cyanobacterial biomass and further actions to prevent health hazards

Upon detection of cyanobacteria at bathing sites, their quantification may be desirable for risk assessment. Two quantitative measures are equally valuable: microscopic cell counts or determination of chlorophyll *a* concentration as a simple measure for algal (including cyanobacterial) density. The choice of methods depends on equipment, expertise and personnel available. If chlorophyll *a* is used and its concentrations remain below 10 $\mu g\ l^{-1}$, hazardous densities of cyanobacteria are unlikely. At higher concentrations, chlorophyll *a* analysis must be supported by qualitative microscopic investigations for dominance of cyanobacteria.

If the results of the measurements exceed the guideline values, immediate management action are suggested (Table 10.1). Furthermore, the detection of cyanobacteria at potentially hazardous concentrations should initiate planning of measures for restoration of bathing water quality. Because of the complexity of factors leading to cyanobacterial proliferation, flexible approaches are important, involving further development of monitoring and of protection measures as information on a given water body is accumulated.

Regular measurements of transparency (Secchi readings) in Step 3 can greatly enhance understanding of the system. If transparency is high (greater than 2 m) and no significant discolouration of the water can be seen, cyanobacterial densities are unlikely to be high. However, samples should be taken to determine the phytoplankton community and the water chemistry, in order to estimate the capacity of the water body for cyanobacterial bloom formation. In deep and stratifying lakes, samples from different levels within the vertical stratification are required because some cyanobacteria accumulate near the thermocline. If transparency is low (less than 1-2 m) and accompanied by a greenish to bluish discolouration, high cyanobacterial densities are likely. In addition, greenish streaks formed by buoyant cyanobacteria (e.g. *Microcystis, Anabaena*) during warm and calm weather may be visible on the water surface. Inspection of downwind areas of the water body is essential when these characteristics are observed, because wind action can readily lead to accumulations of these buoyant organisms. Samples for further analysis (taxonomical and toxicological) should be taken.

Assessment of toxicity or chemical analysis for specific cyanotoxins is not generally recommended for two reasons: (i) the results of toxin analysis provide only a partial basis for risk assessment because only some of the substances causing health outcomes are known and can be analysed, and (ii) the results of an epidemiological study indicate that some health outcomes are not due to known cyanotoxins (Pilotto *et al.,* 1997). Assessment of toxicity using a bioassay may circumvent this problem, but the results are not easily interpreted in terms of human exposure during swimming, particularly with respect to skin reactions. Many cyanotoxin survey studies in different parts of the world have shown that more than half of the field populations investigated did contain toxins, particularly microcystins. Therefore, cyanobacteria are likely to be toxic.

Toxin analysis or toxicity assays may be useful under specific circumstances, because if concentrations of the known cyanotoxins (particularly microcystins) prove to be low, some critical health outcomes (particularly liver intoxication due to microcystins) may be excluded. Warning notices regarding potential irritative effects of cyanobacteria (on the skin, gastrointestinal tract, ears, eyes, respiratory membranes) should nonetheless be posted while the cyanobacterial population density is above the guideline values (Table 10.1). Toxin analysis or toxicity assays may be required in advance of a sports event to improve understanding of the potential risk due to cyanobacteria. If cyanotoxin concentration is low, a local authority may decide to proceed with the event in spite of high cyanobacterial density. Recreational facilities may also choose to invest in monitoring of the known cyanotoxins in order to avoid temporary closure of the facility whenever cyanobacteria proliferate.

10.1.2 Sample site selection

In addition to the general guidance provided in Chapter 2, site selection for cyanobacteria sampling should take account of specific incidents, that are suspected to be associated with exposure to cyanobacteria or algae. Additional factors to be considered in site selection are: the history, if available, of cyanobacterial and algal population development and occurrence of toxins at the water body or coastal strip; local characteristics of the catchment and water body which (may) influence the development and fate of cyanobacterial populations, or even their current location in parts of the water body; and the wider knowledge of the characteristics of cyanobacterial and algal population development and fate and the production and fate of cyanobacterial and algal toxins.

The heterogeneous and dynamic nature of many planktonic population developments may require sampling several sites, such as locations that are prone to accumulation or scum formation, particularly if these are in areas used for recreation; locations at beaches or in the open water body which are used for immersion sports or those involving accidental immersion (such as sail-boarding or water-skiing); a central reference site in open, mixed water (experience may indicate if this can be used as a representative site for the main water mass to assess the total population size and thus estimate accumulation or scum formation potential); and decaying accumulations for dissolved toxins. A useful approach for large water bodies is to sample individual zones that differ in some physical, chemical, geomorphological or biological features. Preliminary sampling may help to define these zones, based on their spatial variability and gradients in environmental and plankton properties.

Sampling to assess the total population size may require information on the horizontal distribution of cyanobacteria within the water body, as well as on their distribution with depth. Particularly in thermally stratified water bodies, some cyanobacteria form pronounced population maxima at the depth with optimal light intensity and/or nutrient concentrations. Stratification leads to a water body functioning as two separate masses of water (the epilimnion and the hypolimnion) with different physico-chemical characteristics, with a transitional layer (metalimnion) sandwiched between. Thermal stratification can be determined by measuring vertical profiles of temperature within the water body. In temperate climates, thermal stratification generally occurs seasonally in water bodies of appropriate depth, whereas in tropical climates it often follows diurnal patterns. Thermal stratification has important implications for the distribution of the concentrations of nutrients and the interpretation of phosphorus and nitrogen data. Usually, shallow (2-3 m), wind exposed lakes do not stratify, whereas in temperate climates deeper lakes usually exhibit a stable stratification from spring to autumn. Lakes of intermediate depth (e.g. 5-7 m) may develop transient thermal stratification for a few calm and sunny days; the stratification is then disrupted by the next event of rain or wind. However, even if temperature is uniform throughout the water column, stratification of organisms can occur on calm days. Depth gradients of oxygen concentration and pH are good indicators of such stratification. Depth-integrated samples are more adequate than surface samples for the assessment of population size and nutrient concentrations in such situations. Approaches to optimising depth-integration are discussed by Utkilen *et al.* (1999) (see also section 10.3).

10.1.3 Monitoring frequency

Monitoring at time intervals should aim (i) to give warnings of developing cyanobacterial and algal populations and associated toxin levels; (ii) to provide information on the duration of cyanobacterial and algal populations and toxin levels that exceed guideline values; (iii) to provide information on the decline of cyanobacterial and algal populations and toxins due to natural processes; or (iv) to enable assessment of the persistence or reduction in cyanobacterial populations and toxin levels due to intervention, such as eutrophication control.

In recreational waters where bloom formation is suspected, the frequency of monitoring should be sufficient to provide data to enable an appropriate Alert Levels Framework system to operate (Chorus and Bartram, 1999). For example, monitoring may begin on a fortnightly basis and be increased to twice-weekly whilst alert levels are exceeded, before being reduced again after alert levels and guideline values for cyanobacterial cells and toxins are no longer exceeded. A scheme of suggested frequencies according to the steps of monitoring is presented in Table 10.3.

Table 10.3 Monitoring frequency and parameters for each of the three phases of the recommended monitoring strategy

Phase or activity	Parameters	Frequency
Monitoring phases		
Phase 1 (background)	Transparency (Secchi disc)	At least once a month
	Nutrient concentrations	At least twice a year (spring overturn and summer)
	Hydrophysical conditions (e.g. flow regime, thermal stratification)	At least once a month
Phase 2 (basic)	Transparency (Secchi disc)	Fortnightly
	Discolouration	Fortnightly
	Scum formation	Fortnightly
	Hydrophysical conditions	At least once a month
	Temperature	Possibly continuous
	Weather conditions (e.g. winds, light)	Continuous
	Changes in turbulence (i.e. mixing)	Possibly continuous
Phase 3 (cyanobacteria)	Transparency (Secchi disc)	Twice weekly
	Qualitative microscopic analysis	Twice weekly
	Quantitative microscopic analysis	Twice weekly
	Chlorophyll a analysis	Twice weekly
Additional activities		
Toxicity	Bioassays (to confirm the presence of toxicity in cells and/or released into the water)	Possibly at first appearance of situations (as in Phase 3) and in all cases when health problems are suspected or reported
Toxin concentration	Chemical analyses (to confirm the presence of specific toxins both qualitatively and quantitatively)	As above

If a water body prone to cyanobacterial mass development is to be used for water-contact sports on a seasonal basis, or for a single event, monitoring should begin not less than two weeks before the beginning of the season or the event. Monitoring should then continue, with the frequency adjusted to enable decisions to be made about access to the facility throughout the season, or whether to proceed with the event.

10.2 Laboratory and staff requirements

Monitoring for cyanobacterial and algal health hazards makes a range of demands upon analytical resources, some of which are different from those made by other aspects of water quality monitoring. Although a higher level of sophistication will provide more information, cyanobacterial and algal monitoring can be highly effective at a very low level of demand on facilities (Table 10.4).

Background monitoring of physical and chemical variables reflecting bloom-forming potential (such as transparency, nutrient concentrations and hydrophysical conditions) makes limited demands upon analytical resources and capacities and may be readily decentralised. While any capable chemical laboratory can carry out laboratory analysis, some limnological or oceanographic expertise is necessary for planning field work, quality control of data, and interpretation of results.

Basic monitoring for indicators of toxic cyanobacteria focuses mainly on critical site inspection and requires almost no facilities. If performed by local staff with observation skills and increasing experience, regular monitoring and recording of simple variables such as transparency, discolouration and scum formation, provides much information for management.

Monitoring of populations of cyanobacteria and algae requires a microscope and some skill in its use. Health authority staff with experience in microscopy can easily learn to recognise the most important toxin-producing cyanobacteria and algae in the waters under their responsibility, provided occasional training by experts is provided.

10.3 Sampling

Samples taken directly by immersion of a sample bottle or sampling device are termed "grab" samples; they are also known as "spot" or "snap" samples. For sampling cyanobacteria, grab samples are often taken from the surface. Composite or integrated samples consist of several sub-samples collected separately (e.g. from different parts of the water body) and then mixed together. They are taken for quantitative, representative samples when the variables to be assessed are unevenly distributed (but information on distribution is not required), for example, when assessing the total content of a substance in a water body (e.g. total phosphorus potentially available for phytoplankton growth) or the total population of an organism (e.g. taking into account the horizontal or vertical variations in distribution of cyanobacterial populations due to the presence of physico-chemical gradients). If knowledge of the precise distribution is required, each sub-sample can be evaluated individually. Composite samples may be:

• *Depth-integrated.* These are most commonly made up of two or more equal sub-samples collected with a sampler at predetermined depth intervals from the surface to just above the bottom. Selection of depths for subsampling must be adequate to account for stratification of temperature, substances and organisms in the water body (Utkilen *et al.,* 1999). Continuous depth integration can be obtained with a tube sampler or a pumping system (see Figures 10.2-10.4).

• *Area-integrated.* These are made by combining a series of samples taken at various sampling points spatially distributed in the water body (usually all at one depth or at predetermined depth intervals).

• *Time-integrated.* These are made by mixing equal volumes of water collected at a sampling station at regular time intervals.

• *Discharge-integrated.* These are integrated over time intervals adapted to the discharge at regular intervals over the period of interest. A common arrangement is to sample every 3 hours over a 24-hour period. The composite sample is then made by mixing portions of the individual samples that are proportional to the rate of discharge at the time the sample was taken.

Table 10.4 Monitoring approaches, their requirements and options for their organisation

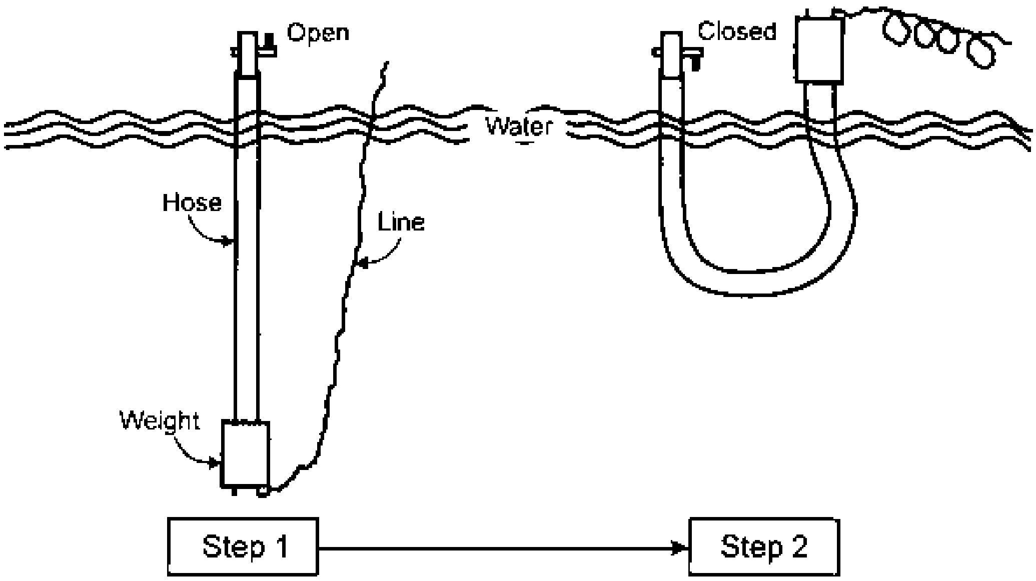
Monitoring type	Parameters of interest	Analytical demands	Who	Where	Notes
Background[1]	Nutrient concentrations (i.e. total phosphorus, nitrate and ammonia); flow regime; thermal stratification; transparency	Low, basic (i.e. photometer, boat, depth sampler and Secchi disc)	Environmental officers or consultants with limnological expertise	Local, regional	Readily incorporated into water resource monitoring
Basic[2]	Transparency; discolouration; scum formation	Minimal (i.e. Secchi disc and regular site inspection by trained staff)	Environmental or health officers	Local	Very high return in relation to input
Cyanobacteria	Dominant taxa - quantity (often determination of genus is sufficiently precise; quantification only as needed for management)	Low, basic (i.e. microscope; photometer is useful)	Environmental or health staff; consultants with limnological expertise	Local, regional	Specific training is required, but quite easily achieved; very high return in relation to input
Toxicity	Toxicity	Low, but skilled (i.e. biotests)	Toxicologists	Central	Demands on skills rather high
Toxin concentration	Toxin content	High[3]	Analytically skilled staff	Central	High return in relation to effort; enables de-warning if bloom proves to be non-toxic

[1] Potential for cyanotoxin

[2] Site inspection for indicators of toxic cyanobacteria

[3] Methods with lower demands are currently under development, for example ELISA

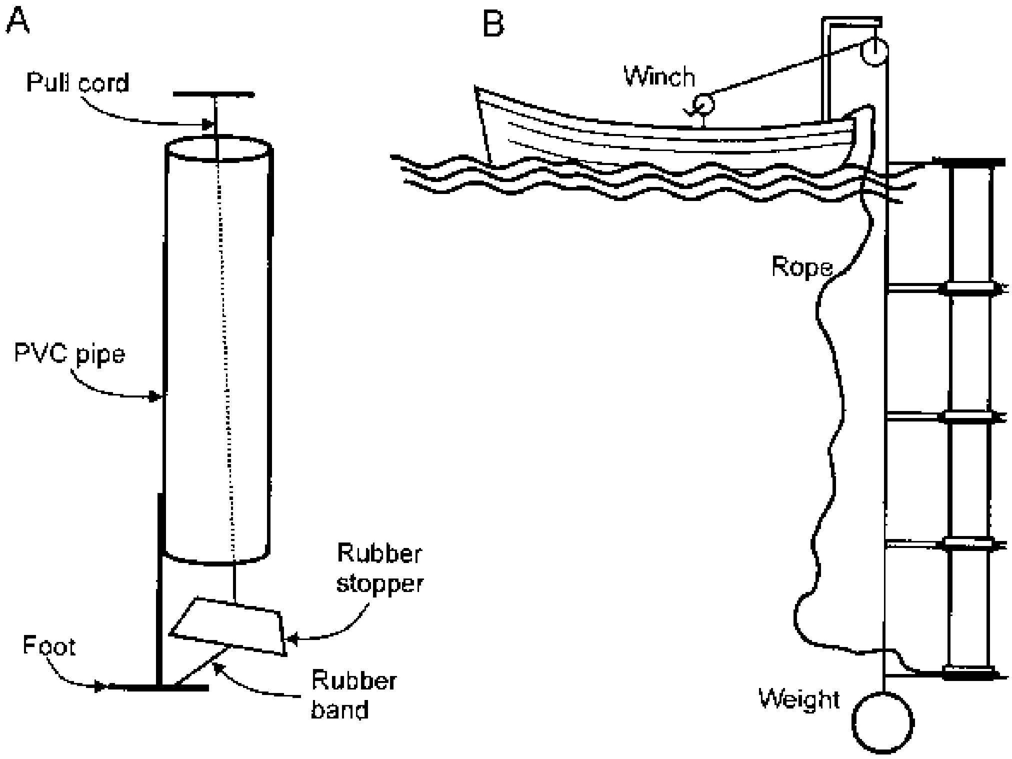
Figure 10.2 A hosepipe sampler for deep waters



A depth sampler, sometimes called a grab sampler or bottle (Van Dorn or Niskin type), is designed in such a way that it can retrieve a sample from any predetermined depth (Venrick, 1978). It consists of a tube that can be closed at its ends by spring-loaded flaps that are triggered by dropping a weight (called a messenger) down the lowering rope. A sample obtained in this way can be used for all chemical analyses except dissolved oxygen. This common sampler is relatively inexpensive, robust and can be deployed from almost any vessel. It gives samples for quantitative analysis.

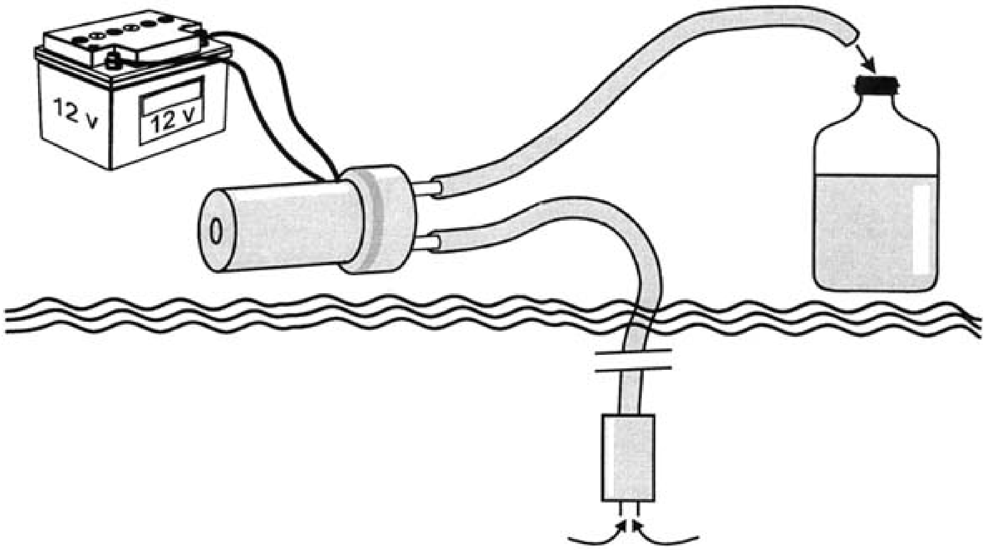
The hosepipe sampler is a piece of flexible plastic piping of several metres in length, weighted at the bottom, and provides a simple mechanism for collecting and integrating a water sample from the surface to the required depth in a lake. The hosepipe is lowered with its upper end open, trapping a water column as it is lowered (Figure 10.2, Step 1). The upper end is closed before hauling up the lower (open) end by means of an attached rope (Figure 10.2, Step 2). The total length of this tube can be up to 30-35 m and it is suitable for relatively calm waters. Another device suitable for taking depth-integrated samples for shallow water columns (less than 5 m deep) or surface waters of deeper water bodies is a simple pipe sampler (Figure 10.3A). The segmented tube sampler (Lindahl, 1986; Sutherland *et al.,* 1992) is a similar alternative for relatively shallow and calm waters. It consists of lengths (1-3 m each) of PVC (polyvinylchloride) pipe linked with valves, with a total length of up to 20 m (Figure 10.3B).

Figure 10.3 A. The pipe sampler: a simple device for depth integrated samples from shallow water bodies; B. The segmented tube sampler (After Sutherland *et al.*, 1992)



Integrated samples can also be obtained using a battery-operated water pump (electric diaphragm pumps are effective) and flexible plastic piping (Figure 10.4) which is operated at a steady pumping rate while the water inlet is drawn upwards between the desired depths at a uniform speed. This apparatus may also be operated to sample many litres of water from the same depth or to filter, for example through a plankton net, large quantities of water from a fixed depth for qualitative and for quantitative (volumes can be measured) analyses. General discussions are given in Beers (1978) and Powlik *et al.* (1991) for peristaltic pumps, and Voltolina (1993) and Taggart and Legget (1984) describe diaphragm pumps.

Figure 10.4 A pump sampling system



Sampling scums

Scums usually occur near shorelines at low water depths and therefore working with a grab sampler or a plankton net may be difficult. Sampling scums is carried out more easily with a wide-necked plastic or glass container. When sampling scums their heterogeneous density must be taken into account. Two different approaches may be developed. The first aims to assess the maximum density of cyanobacteria and/or highest toxin level by taking a sample where the scum is thickest (move the bottle mouth along the surface to collect the dense mats of buoyant cyanobacteria). The second approach aims to simulate conditions where shallow waters are mixed by bathers and playing children (agitate the scum before submerging the bottle). Both types of approach may be used for comparison.

Plankton net sampling

Sampling with a plankton net (Tangen, 1978) is mainly performed when large quantities of cell material are required (e.g. for toxicity testing) or when only qualitative analysis of phytoplankton is necessary. Sampling with a net causes a bias according to the size and shape of the organisms, i.e. the finest mesh size of 10 µm will miss small cells, such as unicells of *Microcystis* spp. and picoplankton. Furthermore, filamentous cyanobacteria may be under-represented because some filaments may slip through the net.

The sampling depth is dependent on the taxa of algae and cyanobacteria present. Floating cells *(Microcystis, Anabaena)* are harvested within the upper few metres, whereas sampling of well mixed or stratified water bodies showing a depth distribution of cyanobacteria (e.g. *Planktothrix*) may include deeper water layers. For sampling the water column (or parts of it) lower the plankton net (25-50 µm mesh) to the desired depth, wait until the rope is taught, and then draw it back slowly to the surface. The net should be drawn very slowly out of the water to allow the water to run through it because large nets are sometimes heavily loaded with water when the pores are clogged with plankton. Rinsing plankton off the netting can be assisted by shaking the net slightly while raising it out of the water slowly.

For sampling surface blooms horizontal net hauls are more appropriate in order to filter floating cells. The plankton net should be moved parallel to the water surface. It can also be towed behind a boat moving slowly.

A disadvantage of collecting material with a plankton net is that the water volume filtered through the net cannot be determined precisely. Calculations based on the area of the net opening and length and distance hauled are not recommended because they overestimate strongly the amount of water actually filtered (due to clogging of the pores, only a fraction of the water volume will actually have passed through the net).

10.3.1 Determination of phosphorus and nitrogen

Water samples are collected with a clean sampler, a water pump or directly with the sample bottle. In shallow, unstratified lakes the sampling depth is less important than in deep, stratified lakes, where at least one sample from the epilimnion, one from the metalimnion and one from the hypolimnion should be taken. If this is not possible, a single surface sample will provide useful information, but only an incomplete picture of the growth conditions for cyanobacteria and algae.

A 100 ml sample container (see section 10.3.4) should be filled, immediately closed and stored cool. If analysis aims at differentiating between the different forms of phosphorus and nitrogen, the storage time should be as short as possible, and no more than 24 hours if the samples can be stored cool. Risks during extended storage involve transformations between dissolved and particulate fractions as well as between nitrate and ammonia. Preferably, the samples should be filtered in the field using membrane filters (0.45 µm pore diameter) pre-washed with a few millilitres of sample, and the filtered fractions should be stored separately. Alternatively samples should be filtered within 4 hours after sampling.

During filtration of samples for analysis of the dissolved nutrient fractions it is particularly important to avoid contamination. Filtering devices may be contaminated with higher concentrations from previous samples, especially if nutrient-enriched deep water layers had been filtered previously. This is especially important for phosphate, because it tends to adsorb to materials. Rinsing the equipment with double-distilled water between localities or with the sample to be filtered is recommended. It is helpful to take samples with low concentrations first (e.g. usually surface water) and to move on to samples in which higher concentrations are expected (e.g. deep waters).

10.3.2 Quantitative and qualitative determination of cyanobacteria and algae

For microscopic determination of cyanobacteria and algae, and for their microscopic quantification grab samples of 50-200 ml are put into a brown glass bottle and fixed immediately with Lugol's iodine solution or formaldehyde solution. Lugol's solution renders cells heavier, thus facilitating enumeration. Addition of 1-2 ml of Lugol's iodine per 100 ml of sample results in a 1 per cent final concentration (note: very hypertrophic waters may require more preservative). Formaldehyde should be avoided because it presents health risks to the user (it is a potent allergen), or it should be used only under conditions of excellent ventilation in the laboratory and at the microscope. It has the advantage of not causing discolouration of the sample, but the disadvantage of not enhancing settling in counting chambers as effectively as Lugol's solution. Additional fresh, unpreserved samples are useful for microscopic identification because the iodine in the Lugol's solution covers the characteristic colour of the cyanobacteria. Such samples need not be quantitative and may be collected with a plankton net (10 µm mesh) or as a grab sample at a site with high cyanobacteria and algal density. Unpreserved samples for identification may be stored for several hours without appreciable deterioration if kept cool during transportation to the laboratory.

If biomass is to be quantified by chemical analysis of chlorophyll *a* concentration, samples of 1 litre (or less if cell densities are high) of water are taken and filtered as soon as possible. Filtration in the field involves problems of filter transport, which is possible either on ice in an icebox, or submerged in ethanol used for extraction. Direct sunlight must be avoided during filtration and transport.

Materials and method

✔ Brown (or white) glass bottles: 50-200 ml, preferably pre-stocked with a few drops of Lugol's iodine solution.

✔ Brown glass bottles or dark plastic bottles of 1 litre (for chlorophyll a)

✔ Lugol's solution (Willén, 1962).

For chlorophyll a analysis, apparatus for field or laboratory filtration of the water samples includes:

✔ Electric vacuum pump (if filtration is to be performed in the field, a system using a 12V power supply or hand vacuum pump is necessary).

✔ Filtration device.

✔ Glass fibre filters (average pore size 0.7 µm, filter diameter 47 mm).

✔ Either ice and icebox or ethanol and glass vessels for filter transport.

Lugol's solution

Dissolve 20 g potassium iodide into 200 ml of distilled water, mix and add 10 g of sublimated iodine (the solution must not be supersaturated with iodine because this can result in crystal formation with consequent interference in counting). Supersaturation can be tested by diluting 1 ml of stock solution to 100 ml with distilled water to give concentrations similar to those in preserved samples. If iodine crystals appear after standing, more potassium iodide (approximately 5 g) should be added and the test repeated. If no crystals appear, 20 ml of glacial acetic acid must be added. Store stock solution in a dark bottle and use within one year.

10.3.3 Cyanotoxin analysis

For toxicity testing, a large amount of cell material may be collected (without determining the water volume from which it originates) with a plankton net as described in section 10.3. If the concentration of cell-bound toxin is to be related to the water volume from which the cells were collected, the best approach is to filter a defined volume as described by Lawton *et al.* (1994) through a 0.45 µm mesh membrane filter. The volume chosen should be sufficient to provide a pronounced greenish layer of material on the filter without clogging the filter. Care must be taken to stir the sample to disperse the cells evenly every time immediately before a sample is poured onto the filter. The filtrate may be used for the analysis of toxins dissolved in the water.

10.3.4 Sample containers

The laboratory that conducts the analyses should ideally provide containers and bottles for the transport of samples. These should be pre-labelled and well-arranged in suitable containers (if cooling is not necessary, soft drink crates, with subdivisions for each bottle, are cheap and very practical). For routine sampling of the same sites, it is advisable always to use the same bottle for each site and each variable. This avoids cross-contamination, which is a particular concern for phosphorus analyses. For most samples glass containers are appropriate, but often plastic containers (which are more stable than glass) can be used instead. The following containers are recommended for the transport of cyanobacteria and related samples:

- *Phosphorus analysis.* 100 ml glass bottles pre-washed and stored with sulphuric acid (4.5 mol l^{-1}) or hydrochloric acid (2 mol l^{-1}) until use, rinse well before use.

- *Nitrate, ammonia and total nitrogen.* Glass or polyethylene bottles (100 ml).

- *Microscopic identification of cyanobacteria.* Wide-mouth polyethylene bottles (100 ml) are appropriate for studying living material in a fresh grab or net sample.

- *Microscopic identification and quantification of cyanobacteria.* Brown glass bottles (100 ml) already containing preservative. Clear (plastic or glass) bottles may be used if the samples can be stored in the dark.

- *Chlorophyll a analysis.* 1 litre brown (plastic or glass) bottles are recommended to avoid degradation of chlorophyll by sunlight. Clear (plastic or glass) bottles may be used if the samples can be stored in the dark. If filtration for chlorophyll *a* analysis is performed in the field, moist filters are best transported either on ice, by folding them with the cell layer inside and wrapping them in aluminium foil, or by immersing them in ethanol immediately after filtration. For the latter, 50 ml wide-mouth brown glass bottles or 10 ml tightly sealing test tubes may be used.

For toxin analysis, cell material is either collected with a plankton net or by filtration. Material enriched with a plankton net is best transported in wide-mouth plastic containers, in which it may be frozen for subsequent freeze-drying. Material on filters is best transported either dry, if filters can be dried rapidly without direct sunlight, or on ice by folding them with the cell layer inside and wrapping them in aluminium foil. Samples for analysis of dissolved cyanotoxins are acquired by filtration either in the laboratory or in the field. They are transported (filtered or unfiltered) in 1 litre plastic containers.

10.3.5 Sample transport and preservation

Samples must be labelled clearly with sampling site (station), date and time of sampling and depth of sampling. Preservation of samples, filtered volumes and any irregularity should be noted in the field record. In general, samples should be stored cool and dark in a storage box (if necessary, with solid coolant) during sampling and transportation. Storage of samples between 2°C and 5°C may preserve many types of samples, but checks should be made to confirm this with each sample type. Preferably, the samples should be analysed immediately after sampling. If a storage time longer than 12 hours is necessary, quick-freezing of samples to -20°C is recommended. Samples to be filtered must be filtered before freezing.

For the analysis of ammonia, storage is particularly critical. Samples can be cooled in a refrigerator but should be analysed within three hours of collection. Preservation for longer periods is not recommended. Filtration of samples should be avoided. It is nearly impossible to obtain filters free of ammonia and filtration may also evaporate the ammonia contained in the sample.

If samples for chlorophyll *a* are filtered in the field, filters must be either transported frozen, or submerged in ethanol. The ethanol should be boiling (at 75°C) when put onto the filters. If this is not possible in the field, transporting filters submerged in cold ethanol and heating the ethanol later in the laboratory may be preferable to the risk of degradation occurring on filters transported dry but poorly cooled. The suitability of this approach should be checked for any given situation.

Samples for microscopic enumeration preserved with Lugol's iodine at the time of collection (see section 10.3.2) are relatively stable and no special storage is required, although they should be protected from extreme temperatures and strong light. Nevertheless, it is recommended that samples are examined and counted within a few weeks because some species of phytoplankton are sensitive to prolonged storage and Lugol's iodine solution and may degrade if stored for many months or even years. Unpreserved samples for microscopic quantification require immediate attention, either by addition of preservative or by following alternative counting methods that do not use preserved cells. Where unpreserved samples cannot be analysed immediately they should be stored in the dark with the temperature kept stable, at about +4 °C.

10.4 On-site analysis

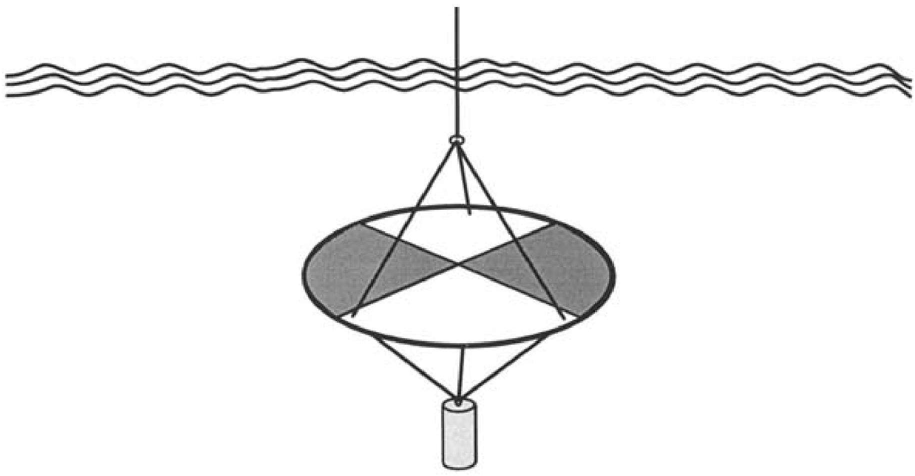
A significant advantage of on-site testing is that tests are carried out on fresh samples which have not been contaminated or the characteristics of which have not otherwise changed as a result of storage. Some analyses such as temperature and transparency can only be carried out in the field.

10.4.1 Transparency

The transparency of water is strongly influenced by turbidity due to particles such as phytoplankton (cyanobacteria and algae) or suspended silt. It can be measured easily in the field. If transparency is low (less than 1-2 m) and accompanied by greenish to bluish discolouration, streaks, or even scums, high cyanobacterial densities are likely. Such results may initiate immediate inspection of further downwind sites and the collection of samples for cyanobacterial analysis.

Transparency can be obtained approximately with a Secchi disc (see Figure 10.5). The disc is made of rigid plastic or metal, but the details of its design are variable. It may be 20 or 30 cm or even larger in diameter and is usually painted white. Alternatively, it may be painted with black and white quadrants. The disc is suspended on a light rope or chain so that it remains horizontal when it is lowered into the water. The suspension rope is graduated at intervals of 0.1 and 1 m from the level of the disc itself and usually the rope does not need to be more than 5 m in length. A weight fastened below the disc helps to keep the suspension rope vertical while the measurement is being made. Transparency estimated in this way (submersible photometers are also available for these measurements) is taken to be the mean of the depths at which the disc disappears when viewed from the shaded side of the boat and at which it reappears upon raising after it has been lowered beyond visibility. Observation of the disc through a tube (painted black inside) with a transparent pane at the lower end and held just below the surface improves precision, particularly if the water surface is perturbed (Wetzel and Likens, 1990).

Figure 10.5 The Secchi disc for measuring transparency



If cyanobacteria occur as floating streaks or mats on the water it is difficult to obtain representative transparency data. Depending on the measuring site, values can vary from 0 to greater than 2 m. It may be useful to determine transparency in areas without floating cells as well as within scums. The Secchi disc has to be lowered very carefully to prevent destroying the formation of accumulated cyanobacterial cells, and before taking the Secchi disc measurement the surface scums should be given time to return to their original water coverage again.

An improvised transparency determination may be recommended by local authorities for users of recreational sites known to be affected frequently by cyanobacterial mass developments. If bathers cannot see their feet while standing knee-deep in the water because of greenish turbidity, bathing should be avoided.

10.4.2 Temperature

Temperature is best measured *in situ* with a probe because water samples gradually reach the same temperature as the surrounding air. If this is not possible, it may be measured with a thermometer or probe in a water sample of at least 1 litre, immediately after taking the sample. Graduations of 0.1 °C are appropriate.

Procedure

1. If measuring in a sample, immerse the thermometer in the water until the temperature reading is constant. Record the reading to the nearest 0.1 °C.

2. If using a probe, lower the probe to the required depth. Hold it at that depth until the reading on the meter is constant. If a complete profile of temperatures is to be taken, measurements should be made at 1 m intervals from the surface to the bottom. Near the surface, or in areas of large thermal discontinuities, measurements should be made at intervals of less than 1 m.

10.4.3 *In situ* fluorometric analysis of chlorophyll *a* and remote sensing

Submersible fluorometers exist which can provide fine scale profiles vertically and horizontally. This is valuable, particularly for monitoring large water bodies and coastal areas for highly variable patterns of chlorophyll concentration (algal and/or cyanobacterial biomass). This approach has been used successfully to monitor variability in phytoplankton biomass and species composition, as well as surface temperature, salinity and nutrient concentrations, in the Baltic Sea. Fully automated analyser systems are installed on three passenger ferries. The system allows high frequency sampling with a spatial resolution of about 100 m and a temporal resolution of 1-3 days. The project uses, especially during the cyanobacterial bloom period, satellite images to detect the extent of the algal surface accumulations. The data are used to provide information on the Baltic Sea phytoplankton on the Internet at http://www.fimr.fi. Another example of the use of a flow-through system deployed on ferries has been reported from Japan (Harashima *et al.,* 1997).

Real-time data for chlorophyll *a* distribution and concentrations, and potentially for cyanobacterial phycobiliprotein pigments in freshwaters, can be generated by the remote sensing of water optical properties by high resolution airborne scanners (Cracknell *et al.,* 1990; Jupp *et al.,* 1994). However, flight times may be infrequent and data collection depends on factors such as cloud cover. Nevertheless, the remote sensing of cyanobacterial populations as a contribution to water body management has excellent potential.

Visible satellite imagery provides a synoptic perspective but, with few exceptions, does not yet have the ability to discriminate between different phytoplankton taxa nor is it effective during inclement weather. New techniques like the use of Advanced Very High Resolution Radiometers (AVHRR) on board the polar orbiting National Oceanic Atmospheric Administration (NOAA) satellites have been applied to monitor large-scale algal blooms, for example in North America (Gower, 1997) and the Baltic Sea (Kahru, 1997).

Table 10.5 Differentiation of phosphorus fractions

Fraction	Definition
1. Soluble reactive phosphorus	Filtered sample
2. Dissolved organic phosphate	Digested filtered sample
3. Particulate phosphorus	Total phosphorus less the dissolved organic phosphate fraction (i.e. 4 minus 2)
4. Total phosphorus	Digested unfiltered sample

10.5 Determination of nutrients in the laboratory

Whereas for phosphorus, the method given in section 10.5.1 is widely accepted and easy to perform with common laboratory equipment, several methods may be considered appropriate for nitrate, depending on available equipment. The method given in section 10.5.2 demands the least equipment but, due to an evaporation step, reproducibility may be poorer than for the method given in section 10.5.3. Both methods use hazardous chemicals which require appropriate safety protection and hazardous waste collection. Ion chromatography, if available, evades this problem and may be the preferred option.

10.5.1 Phosphorus

Some widely accepted digestion methods for dissolving particles to release all of the phosphate achieve this aim only incompletely. The procedure of Koroleff (1983a) for determining total phosphate has proved to be simple and efficient and is the basis of the ISO/FDIS protocol 6878 (ISO, 1998). This method is applicable to many types of water including seawater, in a concentration range of 5-800 µg l^{-1} of P, or higher if the samples are diluted. Differentiation by the fractions shown in Table 10.5 is possible through filtration. Further information is available in Wetzel and Likens (1990) and APHA/AWWA/WPCF (1995).

Principle

Digestion or mineralisation of organophosphorus compounds to SRP (also known as orthophosphate) is performed in tightly sealed screw-cap vessels with persulphate, under pressure and heat in an autoclave (in the absence of which good results have also been obtained with a household pressure cooker), or simply by gentle boiling. Polyphosphates and many organophosphorus compounds may also be hydrolysed with sulphuric acid to molybdate-reactive orthophosphate. Many organophosphorus compounds are converted to SRP by mineralisation with persulphate. Orthophosphate ions are reacted with an acid solution containing molybdate and antimony ions to form an antimony phosphomolybdate complex. The complex is reduced with ascorbic acid to form a strongly coloured molybdenum blue complex. The absorbance of this complex is measured to determine the concentration of SRP present. An overview of the procedure, necessary equipment and chemicals is provided below (see also ISO, 1998).

Reagents

Only reagents of recognised analytical grade, and only distilled water having a phosphate content that is negligible compared with the smallest concentration to be determined in the samples, should be used.

✓ Sulphuric acid (1.84 g ml^{-1}) (Caution, eye protection and protective clothing are necessary).

✓ Sulphuric acid, solution, $c(H_2SO_4)$ = 9 mol l^{-1}. Add 500 ± 5 ml of water to a 2 litre beaker. Cautiously add, with continuous stirring, 500 ± 5 ml of sulphuric acid (1.84 g ml^{-1}) and mix well. (Caution, eye protection and protective clothing are necessary).

✓ Sulphuric acid, solution, $c(H_2SO_4)$ = 4.5 mol l^{-1}. Add 500 ± 5 ml of water to a 2 litre beaker. Cautiously add, with continuous stirring, 500 ± 5 ml of sulphuric acid solution (9 mol l^{-1}) and mix well. (Caution, eye protection and protective clothing are necessary)

✓ Sulphuric acid, solution, $c(H_2SO_4)$ = 2 mol^{-1}. Add 300 ± 3 ml of water to a 1 litre beaker. Cautiously add, with continuous stirring and cooling, 110 ± 2 ml of sulphuric acid solution (9 mol l^{-1}). Dilute to 500 ml ± 2 ml with water and mix well. (Caution, eye protection and protective clothing are necessary)

✓ Ascorbic acid, 100 g l^{-1} solution. Dissolve 10 g ± 0.5 g of ascorbic acid in 100 ml ± 5 ml of water. The solution is stable for two weeks if stored in an amber glass bottle in a refrigerator and can be used as long as it remains colourless.

✓ Sodium hydroxide, solution, $c(NaOH)$ = 2 mol l^{-1}. Dissolve 80 g of sodium hydroxide pellets in water, cool and dilute to 1 litre with water. (Caution, eye protection and protective clothing are necessary).

✓ Acid molybdate, solution. Add the molybdate solution (I) to 300 ml ± 5 ml of sulphuric acid 9 mol l^{-1} with continuous stirring. Add the tartrate solution (II) and mix well. (Caution, eye protection and protective clothing are necessary).

✓ Molybdate solution (I). Dissolve 13 g ± 0.5 g of ammonium heptamolybdate tetrahydrate $[(NH_4)_6Mo_7O_{24}4H_2O]$ in 100 ml ± 5 ml of water.

✓Tartrate solution (II). Dissolve 0.35 g antimony potassium tartrate hemihydrate $[K(SbO)C_4H_4O_6½H_2O]$ in 100 ml ± 5 ml of water. The reagent is stable for at least two months in an amber glass bottle.

✓Sodium thiosulphate pentahydrate, 12 g l^{-1} solution. Dissolve 1.2 g sodium thiosulphate pentahydrate ($Na_2S_2O_35H_2O$) in 100 ml water. Add about 50 mg of anhydrous sodium carbonate (Na_2CO_3) as preservative. This reagent is stable for at least four weeks if stored in an amber glass bottle.

✓Potassium peroxodisulphate solution. Add 5 g potassium peroxodisulphate ($K_2S_2O_8$) to 100 ml water, stir to dissolve. The solution is stable for at least two weeks, if the supersaturated solution is stored in an amber borosilicate bottle, protected from direct sunlight.

✓Soluble reactive phosphate (orthophosphate), stock standard solution corresponding to 50 mg of P per litre. Dry a few grams of potassium dihydrogenphosphate to constant mass at 105 °C. Dissolve 0.2197 g of KH_2PO_4 in about 800 ml of water in a 1,000 ml volumetric flask. Add 10 ml of 4.5 mol l^{-1} sulphuric acid and make up to the mark with water. The solution is stable for at least three months if stored in a well stoppered glass bottle. Refrigeration to about 4 °C is recommended.

✓Soluble reactive phosphate (orthophosphate), standard solution corresponding to 2 mg of P per litre. Pipette 20 ml of SRP stock standard solution into a 500 ml volumetric flask.

Make up to the mark with water. Prepare this solution each day it is required. One millilitre of this standard solution contains 2 µg of P.

Apparatus

✓Ordinary laboratory apparatus and filter assembly with membrane filters, 40-50 mm diameter with 0.45 µm pore size.

✓Pre-cleaned glass bottles for filtered samples.

✓Spectrometer, suitable for measuring absorbance in the visible and near infrared regions. Capable of accepting optical cells with pathlengths from 1 cm to 5 cm. The most sensitive wavelength is 880 nm, but if a loss of sensitivity is acceptable, absorbance can be measured at the second maximum of 680-700 nm. The detection limit of the method is lower if a spectrometer capable of accepting 10 cm pathlength optical cells is available.

✓Autoclave (or pressure cooker): used for digestion of samples at 115-120 °C.

✓Borosilicate flasks, 100 ml, with glass stoppers tightly fastened by metal clips (heat resistant polypropylene bottles or conical flasks (screw capped) are also suitable). Before use, clean the bottles or flasks by adding about 50 ml of water and 2 ml of 1.84

gm l^{-1} sulphuric acid. Place in an autoclave for 30 minutes at 115-120°C, cool and rinse with distilled water. Repeat the procedure several times and store filled with distilled water and covered.

Before use, all glassware should be washed with 2 mol l^{-1} hydrochloric acid at 45-50 °C and rinsed thoroughly with water. Do not use detergents containing phosphate. Preferably, the glassware should be used only for the determination of phosphorus. After use it should be cleaned as above and kept covered until use. Glassware used for the colour development stage should be rinsed occasionally with sodium hydroxide solution to remove deposits of the coloured complex that has a tendency to stick (as a thin film) on the walls of glassware.

Procedure

If filtration is necessary for the determination of total soluble phosphorus and/or dissolved phosphate, filter the sample within 4 hours after sampling. If the sample was cooled, bring it to room temperature before filtration. Wash a 0.45 µm membrane filter to ensure it is free of phosphate by passing through it 200 ml of water, previously heated at 30-40 °C. Filter the sample discarding approximately the first 10 ml of filtrate and collecting 5-40 ml depending on the concentrations expected. The filtration time should not exceed 10 minutes. If necessary a larger diameter filter should be used. Add 1 ml of 4.5 mol l^{-1} sulphuric acid per 100 ml of test sample. The acidity should be about pH 1, if not, adjust with NaOH 2 mol l^{-1} or H_2SO_4 2 mol l^{-1}. Store in a cool dark place until analysis is possible.

The mineralisation method using potassium peroxodisulphate is described here. This method will not be efficient in the presence of large quantities of organic matter. In this case oxidation with nitric acid-sulphuric acid is necessary. This latter procedure must be carried out in an efficient fume cupboard.

1. Pipette up to a maximum of 40 ml of the test sample (appropriately prepared) into a 100 ml conical flask. If necessary, dilute with water to about 40 ml.

2. Add 4 ml of potassium peroxodisulphate solution and boil gently for 30 minutes. Periodically, add sufficient water so that the volume remains between 25 ml and 35 ml.

3. Cool, adjust pH to between 3 and 10, transfer to a 50 ml volumetric flask, and dilute with water to about 40 ml.

Thirty minutes is usually sufficient to mineralise phosphorus compounds; some polyphosphoric acids need up to 90 minutes for hydrolysis. Alternatively mineralise for 30 minutes in an autoclave at between 115 °C and 120 °C. Most ordinary kitchen pressure cookers are adequate if laboratory equipment is not available. Any arsenate present will cause interference. If arsenic is known or suspected to be present in the sample, eliminate the interference by treating the solution with sodium thiosulphate solution immediately after the mineralisation step. In the case of seawater mineralised in an autoclave, free chlorine must be removed by boiling before the arsenate is reduced by thiosulphate. Iron concentrations above 600 mg l^{-1} (e.g. in mining lakes) and sulphide (detectable by its smell) will also interfere.

Table 10.6 Selection of appropriate volumes for test portions in relation to the concentration of soluble reactive phosphorus

SRP concentration range ($mg\ l^{-1}$)	Volume of test portion (ml)	Thickness of optical cell (cm)
0.0 to 0.2	40	4 or 5
0.0 to 0.8	40	1
0.0 to 1.6	20	1
0.0 to 3.2	10	1
0.0 to 6.4	5	1

4. Test portion after the mineralisation or filtration step. The maximum volume of test portion to be used is 40 ml. This is suitable for the determination of SRP concentrations of up to 0.8 $mg\ l^{-1}$ when using an optical cell of 1 cm pathlength to measure the absorbance of the coloured complex formed by the reaction with acid molybdate reagent. Smaller test portions may be used as appropriate in order to accommodate higher phosphate concentrations (Table 10.6). Phosphate concentrations at the lower end of the calibration ranges are best determined by measuring absorbance in optical cells of 4, 5 or 10 cm pathlength.

Carry out a blank test in parallel with the determination, by the same procedure, using the same quantities of all the reagents as in the determination, but using the appropriate volume of water instead of the test portion.

5. Calibration solutions. (A1) To prepare the set of calibration solutions transfer, by means of a pipette, 1, 2, 3, 4, 5, 6, 7, 8, 9 and 10 ml of the SRP standard solution to 50 ml volumetric flasks. Dilute with water to about 40 ml. These solutions represent SRP concentrations from 0.04 $mg\ l^{-1}$ to 0.4 $mg\ l^{-1}$. If total phosphate or total soluble phosphate is being determined, proceed according to the mineralisation method chosen. Then proceed to colour development. Proceed accordingly for other ranges of phosphate concentration (Table 10.6). Typically, the test portion volume will be in the range of 5-10 ml.

6. Colour development. (A2) Add to each 50 ml flask, while swirling, 1 ml of ascorbic acid 100 $g\ l^{-1}$ and, after 30 seconds, 2 ml of acid molybdate solution I. Make up to the mark with water and mix well.

7. Spectrometric measurements. (A3) Measure the absorbance of each solution at 880 nm after 10-30 min, or if loss of sensitivity can be accepted at 700 nm. Use water in the reference cell.

8. Plotting the calibration graph. (A4) Plot a graph of absorbance against the phosphorus content (in $mg\ l^{-1}$) of the calibration solutions. The relationship between absorbance and concentration is linear. Determine the reciprocal of the slope of the graph. Check the graph from time to time, especially if new packages of chemicals are used. Run a calibration solution with each series of samples.

9. Determination. (B1) Colour development - proceed as in (A2) using the test portion appropriately processed. (B2) Spectrometric measurements - proceed as in (A3).

Expression of results

The concentration of total phosphorus expressed in mg l^{-1} is given by the equation:

$$P_{tot} = \frac{(A - A_0)V_{max}}{fV_s}$$

where:

A is the absorbance of the test portion
A_0 is the absorbance of the blank test
f is the slope of the calibration graph (e4), in litres per milligram
V_{max} is the reference volume of the test portion (50 ml)
V_s is the actual volume, in ml, of the test portion.

The test report should contain complete sample identification, reference to the method used, the results obtained and any further details likely to influence the results.

10.5.2 Spectrometric method for nitrate using sulphosalicylic acid

This method does not require sophisticated equipment and is suitable for surface and potable water samples (ISO, 1998). The method may be used up to nitrate-nitrogen concentrations of 0.2 mg l^{-1} using the maximum test portion volume of 25 ml, and can be expanded by using smaller test portions. The limit of detection lies within 0.003 and 0.013 mg l^{-1}, using cells of path-length 40 mm and a 25 ml test portion volume. A nitrate-nitrogen concentration of 0.2 mg l^{-1} gives an absorbance of about 0.68 units, using a 25 ml test portion and cells of 40 mm pathlength. The main interferences are chloride, SRP, magnesium and manganese (II). Interference problems can be avoided with other Spectrometric methods such as ISO 7890-1 and 7890-2 (ISO, 1986a,b).

Reagents

✓Sulphuric acid ≈ 18 mol l^{-1} = 1.84 g ml^{-1} (Caution, eye protection and protective clothing are necessary).

✓Glacial acetic acid ≈ 17 mol l^{-1} = 1.05 g ml^{-1} (Caution, eye protection and protective clothing are necessary).

✓Alkali solution = 200 g l^{-1}. Dissolve with care 200 g ± 2 g of sodium hydroxide pellets in about 800 ml of water. Add 50 g ± 0.5 g of EDTA-Na_2 and dissolve. Cool to room temperature and make up to 1 litre with water in a measuring cylinder. Store in polyethylene bottle. This reagent is stable indefinitely. (Caution, eye protection and protective clothing are necessary).

✓ Sodium azide solution = 0.5 g l^{-1}. Dissolve with care 0.05 g ± 0.005 g of sodium azide in about 90 ml of water and dilute to 100 ml with water in a measuring cylinder. Store in a glass bottle. This reagent is stable indefinitely. (Caution: this reagent is very toxic if swallowed. Contact between the solid reagent and acid liberates very toxic gas).

✓ Sodium salicylate solution 10 g l^{-1}. Dissolve 1 g ± 0.1 g of sodium salicylate in 100 ml ± 1 ml of water. Store in a glass polyethylene bottle. Prepare this solution freshly on each day of operation.

✓ Nitrate, stock standard solution 1,000 mg l^{-1}. Dissolve 7.215 g ± 0.001 g of potassium nitrate (previously dried at 105 °C for at least 2 h) in about 750 ml of water. Transfer to a 1 litre volumetric flask and make up to one litre mark with water. Store the solution in a glass bottle for not more than two months.

✓ Nitrate, standard solution 100 mg l^{-1}. Pipette 50 ml of the stock standard solution into a 500 ml volumetric flask and make up to the 500 ml mark with water. Store the solution in a glass bottle for not more than one month.

✓ Nitrate, working standard solution 1 mg l^{-1}. Into a 500 ml volumetric flask, pipette 5 ml of standard nitrate solution. Make up to 500 ml mark with water. Prepare the solution freshly on each occasion of use.

Apparatus

Standard laboratory apparatus plus:

✓ Spectrometer, capable of operating at 415 nm with cells of 40 or 50 mm pathlength.

✓ Evaporating dishes, about 50-ml capacity. If the dishes are new or not in regular use, they should be rinsed first with water and taken through the procedure as in the colour development step (see below).

✓ Water bath, boiling, capable of accepting at least six of the evaporating dishes.

✓ Water bath, capable of thermostatic regulation to 25 °C ± 0.5 °C.

Procedure

Warning: This procedure involves the use of concentrated sulphuric acid, acetic acid, sodium hydroxide and sodium azide solutions. Eye protection and protective clothing are essential when using these reagents. They must never be pipetted by mouth.

The maximum test portion volume, which can be used for the determination of nitrate concentrations up to 0.2 mg l^{-1}, is 25 ml. Use smaller test portions as appropriate in order to accommodate higher nitrate concentrations. Because surface water samples contain suspended matter, allow them to settle, centrifuge them or filter them through a washed glass fibre filter before taking the test portion. Neutralise samples with a pH value greater than 8 with acetic acid before taking the test portion.

Carry out a blank test in parallel with the determination, using 5 ml ± 0.05 ml of water instead of the test portion and designate the measured absorbance A_b.

1. Calibration. To prepare the set of calibration solutions add, to a series of clean evaporating dishes, using a burette, 1, 2, 3, 4 and 5 ml respectively of the working nitrate standard solution, corresponding to nitrate amounts of 1, 2, 3, 4 and 5 µg in the respective dishes.

2. Colour development. Add 0.5 ml ± 0.005 ml of sodium azide solution and 0.2 ml ± 0.002 ml of acetic acid. Wait for at least 5 minutes, and then evaporate the mixture to dryness in the boiling water bath. Add 1 ml ± 0.01 ml of sodium salicylate solution, mix well and evaporate the mixture to dryness again. Remove the dish from the water bath and allow the dish to cool to room temperature.

Add 1 ml ± 0.01 ml of sulphuric acid and dissolve the residue in the dish by gentle agitation. Allow the mixture to stand for about 10 minutes. Then add 10 ml ± 0.1 ml of water followed by 10 ml ± 0.1 ml of alkali solution.

Transfer the mixture to a 25-ml volumetric flask but do not make up to the 25 ml mark. Place the flask in the water bath at 25 °C ± 0.5 °C for 10 min ± 2 min. Then remove the flask and make up to the 25 ml mark with water.

3. Spectrometric measurements. Measure the absorbance of the solution at 415 nm in cells of pathlength 40 or 50 mm against distilled water as a reference. Designate the absorbance measured as A_s.

4. Plotting the calibration graph. Subtract the absorbance of the blank solution from the absorbances of each of the calibration solutions and plot a calibration graph.

5. Determination. Pipette the selected test portion of volume V, such that the aliquot contains a mass of nitrate-nitrogen between 1 µg and 5 µg, into a small evaporating dish. Then proceed as in the preceding "Colour development" and "Spectrometric measurements" steps.

6. Correction for test portion absorption. If absorption by the test portion at the analytical wavelength is known, or suspected, to interfere (as may arise with highly coloured samples), carry out the operations given in the preceding "Colour development" and "Spectrometric measurements" steps, on the duplicate test portion but omitting the addition of sodium salicylate solution. Designate the absorbance measured be A_c.

Table 10.7 The effect of other substances on the results obtained with the spectrometric method for nitrate using sulphosalicylic acid

Other substance	Amount of other substance in a 25 ml test portion (µg)	Effect of other substance in a 25 ml test portion	
		m(N) = 0.00 µg (µg N)	m(N) = 5.00 µg (µg N)
Sodium chloride	10,000	+ 0.03	- 0.73
Sodium chloride	2,000	+ 0.01	- 0.16
Sodium sulphate	10,000	+ 0.04	- 0.16
Sodium hydrogen carbonate	10,000	- 0.02	- 0.52
Sodium hydrogen carbonate	2,000	- 0.03	- 0.18
Calcium chloride	5,000	+ 0.23	+ 0.38
Calcium chloride	2,500	+ 0.02	- 0.14
Iron (III) sulphate	20	+ 0.08	- 0.02
Manganese (II) sulphate	20	+ 0.92	+ 0.99
Manganese (II) sulphate	5	+ 0.05	+ 0.13
Zinc sulphate	20	- 0.02	+ 0.07
Copper sulphate	20	+ 0.03	+ 0.19
Ammonium chloride	500	- 0.12	- 0.17

Expression of results

Calculate the absorbance due to nitrate in the test portion, A_t, from the equation:

$$A_t = A_s - A_b$$

or, when a correction for sample absorption has been made, from the equation:

$$A_t = A_s - A_b - A_c$$

In both equations A_s, A_b and A_c refer to the sample, blank and correction absorbances respectively (see relevant sections above). Read off from the calibration graph, the mass of nitrate, *m(N)* in micrograms, corresponding to the absorbance value A_t.

The nitrate content in the sample, in mg l^{-1}, is given by the formula: *m(N)/V* where *V* is the volume of the test portion (in ml). The effect of other substances on this method is provided in Table 10.7.

10.5.3 Spectrometric method for nitrate by reduction of nitrate to nitrite

Nitrates are reduced to nitrites almost quantitatively by amalgamated granulated cadmium. Separate methods for nitrate reduction and for nitrite determination are presented below.

Determination of nitrate: scope, field application and principle

This method is based on the reduction of nitrate ions to nitrite. Because it determines the sum of nitrite and nitrate ions, a separate determination of nitrite must be conducted, and its concentration subtracted from the sum of nitrate and nitrite. At concentrations higher than about 20 µM NO_3-N, calibration factors for a low and high range must be established. The reduction is carried out at a pH of about 8.5. Ammonium chloride buffer is used to control the pH and to complex the liberated cadmium ions (Carlberg, 1972).

Reagents

- Sulphanilamide (SAN) (Caution: possibly hazardous, use fume cupboard, solution must not be discharged to a public sewer). Dissolve 10 g SAN, $NH_2C_6H_4SO_2NH_2$, in a mixture of 100 ml concentrated hydrochloric acid, HCl (36 per cent) and 600 ml bi-distilled water (BdW). After cooling dilute the solution to 1 litre. At room temperature, when stored in glass bottles, the reagent is stable for several months.

- Naphtylamine solution (NED) (Caution: possibly hazardous, use fume cupboard, solution must not be discharged to a public sewer). Dissolve 1 g N-(1-naphthyl)-ethylene-diamine dihydrochloride, $C_{10}H_7NHCH_2NH_22HCl$, in BdW and dilute to 1 litre. The solution should be stored in a tightly closed dark bottle, with 3-4 drops of saturated $HgCl_2$ solution in a refrigerator (Kirkwood, 1992). The solution contains 10 µmoles ml^{-1}.

- Buffer solutions:

25 per cent stock buffer. Dissolve 250 g ammonium chloride, NH_4Cl in BdW and

25 ml concentrated ammonium hydroxide 25 per cent. Dilute to 1 litre.

2.5 per cent work buffer (WB). Dilute 100 ml of stock buffer with BdW to 1 litre.

Wash buffer solution (WbS). Dilute 20 ml of 2.5 per cent WB with BdW to 1 litre.

- Hydrochloric acid 2M. Dilute 165 ml of concentrated commercial HCl (37 per cent) with BdW to 1 litre.

- Mercuric chloride solution 1 per cent (Caution, highly toxic). Dissolve 5 g mercuric chloride, $HgCl_2$, in 500 ml BdW.

- Synthetic seawater (SSW). Dissolve 36 g sodium chloride, NaCl, 12 g magnesium sulphate heptahydrate, $MgSO_47H_2O$, and 0.25 g sodium carbonate, $NaHCO_3$, with BdW and dilute to 1 litre. For analytical purposes this is equivalent to a salinity of 40 psu. For calibration work the SSW may be diluted to the desired salinity.

✔ Nitrate standard stock solution (NO_3-N SSS). Dissolve 0.25278 g potassium nitrate, KNO_3 (molecular weight 101.11) dried at 110 °C to constant weight, in BdW and dilute to 250 ml. Store in a tightly closed dark bottle with 2-3 drops of a saturated $HgCl_2$ solution in a refrigerator (Kirkwood, 1992). The solution contains 10 µmoles ml^{-1}.

✔ Cadmium coarse pulver. Sieve commercially available granulated cadmium and retain and use the fraction between 35 and 40 mesh, i.e. around 0.5 to 0.42 mm. (Caution: Cadmium is a poisonous metal. It should be handled with great care. All operations on the dry metal, particularly the granules, must be carried out in a well-ventilated area, e.g. a fume cupboard. Never inhale the dust. Cadmium must be treated as hazardous waste).

✔ Amalgamated cadmium. The required amount of cadmium metal is about 35 g per reduction column (RC) (Figure 10.6). The sieved granules are rinsed from the oxides by washing with 2M HCl. Then they are washed with plenty of water to eliminate all HCl. All the washed metal is transferred to a round-bottom flask that is filled with 1 per cent $HgCl_2$ solution. The flask is closed with a glass stopper. After this step all contact between air and the metal should be avoided. The flask is rotated for 90 minutes in a horizontal position or shaken with suitable equipment. Finally the flask is opened and the turbid sublimate solution rinsed out with BdW. (Caution: the used $HgCl_2$ solution must not be discharged to a public sewer). When a suitable volume of $HgCl_2$ is collected, 25 ml concentrated HCl is added per litre and then precipitated with hydrogen sulphide, H_2S, or sodium sulphide, Na_2S. The liquid is filtered and the precipitate stored, discarding the clear filtrate.

Apparatus and equipment

✔ Test tubes with glass or plastic stoppers, graduated or marked at 25 ml volume.

✔ Automatic syringe pipettes of 1 ml.

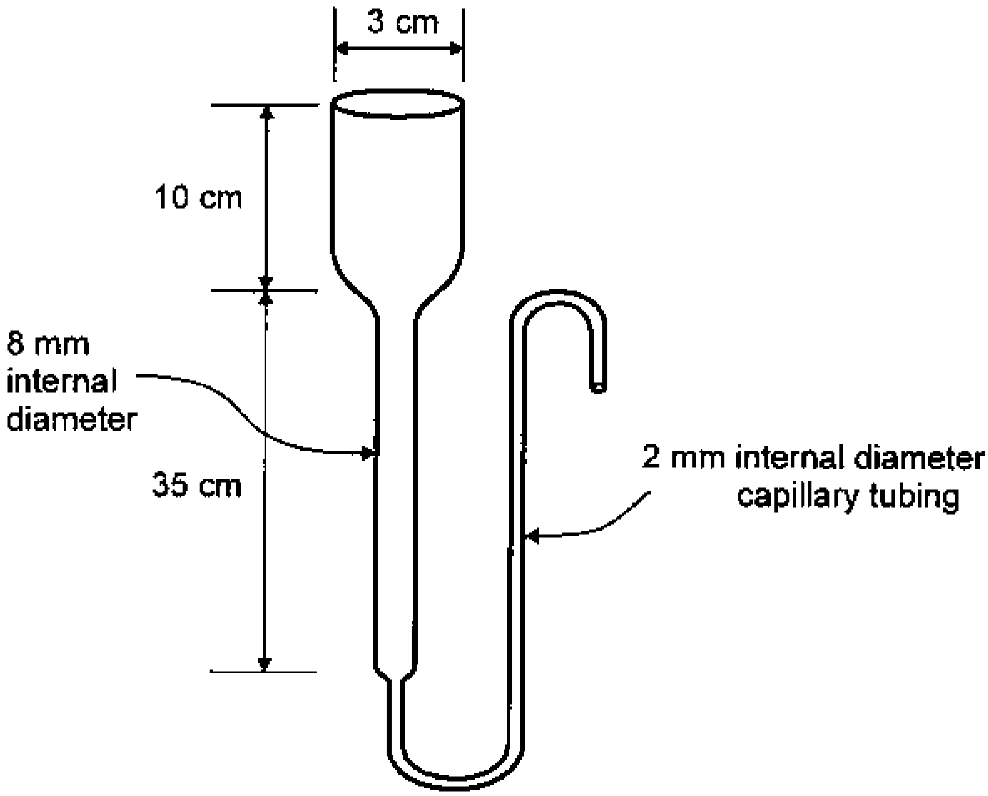
✔ 25 ml automatic pipette.

✔ 500 ml round-bottom flask.

✔ Reduction columns (Figure 10.6).

✔ Spectrometer, with 1, 5 and 10 cm pathlength cells.

Figure 10.6 A reduction column



Preparation of the reduction columns

A small ball of thin copper wire is placed at the bottom of the RC and above the wire a small ball of glass wool. The RC is filled entirely with water. The metal granules are poured into the RC, with a small plastic spoon (making sure that no cavities are formed in the column) and filled to about 1 cm below the reservoir. The amalgamated metal is activated by passing through about 150 ml of Wash buffer Solution (WbS) containing about 100 µM NO_3-N then rinsed thoroughly. The RC is packed with WbS only, before being used for analysis. A newly prepared RC reduces nitrate with an efficiency of 95-100 per cent.

Table 10.8 Preparation of working standard solutions of nitrate by dilution with bi-distilled water or synthetic seawater

Volume of solution D[1] (ml)	Total volume (ml)	Resultant concentration (µM NO_3^--N)
25.0	500	10.00
25.0	1,000	5.00
5.0	1,000	1.00

[1] Solution "D" is prepared by diluting 5 ml of an SSS in 250 ml, giving a solution which contains 0.20 µmoles NO_3^--N per ml

Calibration

There is a significant salinity effect in the calibration for nitrate measurements by manual methods using Hg-Cd reduction columns. Freshly amalgamated RC show a salinity effect of less than 10 per cent while the same RC, after several weeks use, shows a higher discrepancy (up to 30 per cent) when calibration against Working Standard Solution (WSS) made up from BdW and compared with standards in SSW of 35 psu. Working Standard Solution should therefore be made from SSW or the magnitude of the salinity effect should be recorded frequently, where after proper correction should be applied to the data. A series of WSS is prepared from the NO_3-N SSS by dilutions with BdW (or SSW) using volumetric flasks (Table 10.8).

Triplicates of WSS and the blank samples with BdW are analysed as described below. Each RC should be calibrated using blanks and calibration solutions. The linear regression of the absorbances measured in the spectrometer against the concentrations of the WSS (including absorbances of the blank samples, concentration equals 0) gives the calibration factor (cf). Using 5 cm cells, a cf of approximately 4.3 should be obtained.

Analysis

Pour 25 ml of sample into the reservoir. Immediately add 1 ml of WB using an automatic syringe pipette, followed by another 25 ml of the sample. Let this pass through the amalgamated metal. Collect drops in a test tube: the first 25 ml are discarded (use them as washer); the second 25 ml are the nitrite sample. The turbidity reference samples are unnecessary. The RC is now ready to receive the next sample. After every analytical batch, the RC must be flushed with WbS. It should never be left to dry. The concentration of the nitrate in the samples is calculated by multiplying their absorbances (A_s) by the cf:

$$\text{concentration } NO_3\text{-N} = \text{cf} \times A_s \ [\mu M]$$

Control of the reduction efficiency

The reduction efficiency of each RC must be controlled from time to time, preferably for every analytical batch. Duplicates of WSS for nitrite must be analysed, followed by WSS for nitrate of the same concentration:

$$\text{Reduction efficiency}(\%) = \frac{\text{Absorbance of the nitrate WSS} \times 100}{\text{Absorbance of the nitrite WSS}}$$

If the reduction efficiency decreases below 85 per cent, empty the RC, wash the filings quickly with 2M HCl and rinse well with water. Dry the filings, sieve and reamalgamate as described above.

Determination of nitrite: scope, field application and principle

This method is specific for nitrite ions and is applicable to all types of marine waters. It is not appreciably affected by salinity, small changes in reagent concentrations, or by temperature (Grasshoff, 1983). Using 5 cm cells, the detection limit is about 0.02 µM and shows linearity up to about 10 µM. The determination is based on the reaction of nitrite ions with sulphanilamide with the formation of a diazonium compound which, coupled to a second aromatic amine, forms a coloured azo dye.

Reagents

✓ Sulphanilamide (SAN) - as in nitrate determination.

✓ Naphtylamine solution (NED) - as in nitrate determination.

✓ Nitrite standard stock solution (NO_2-N SSS). Dissolve 0.17250 g sodium nitrite, $NaNO_2$ (molecular weight: 69.00), dried at 110 °C to constant weight, in BdW and dilute to 250 ml. Store in a tightly closed dark bottle with 2-3 drops of a saturated $HgCl_2$ solution in a refrigerator (Kirkwood, 1992). The solution contains 10 µM ml^{-1}. (Note: aged solid reagent, even if it is of analytical grade, may contain less than 100 per cent $NaNO_2$ because it is unstable in air, and therefore should not be used for the preparation of SSS).

Apparatus and equipment

✓ Test tubes with glass or plastic stoppers, graduated or marked at 25-ml volume.

✓ Automatic syringe pipettes of 1 ml.

✓ 25 ml automatic pipette.

✓ 500 ml round-bottom flask.

✓ Reduction columns (Figure 10.6).

✓ Spectrometer, with 1, 5 and 10-cm pathlength cells.

Sampling

Nitrite is an intermediate compound in the simplified redox sequence from ammonia to nitrate, and therefore it cannot be preserved properly. Avoid filtration of samples. If samples are slightly turbid or have a visible natural colouration and contain no other disturbing substances, (such as samples taken from nearshore areas), analyse them together with turbidity blanks, but do not filter.

Calibration

Perform calibration in solutions made with BdW. A series of WSS from the NO_2-N SSS is prepared by dilution (Table 10.9). From each of the WSS above, transfer 25-ml triplicates into the tubes. In addition, prepare with BdW one set of triplicates of "blank samples", adding reagents to all tubes as described later. The linear regression of the absorbances against the concentrations of the WSS (including absorbances of the blank samples, concentration equals 0) gives the calibration factor, cf.

Analysis

Transfer 25 ml of sample into the test tubes. If the turbidity has to be determined, transfer 25 ml of sample to a second tube. Pipette 0.5 ml of SAN into the tubes, mix well, and, after 2 minutes but not exceeding 8 minutes, add 0.5 ml of NED into one of the tubes. Stopper and shake. After 8 minutes read at 543 nm in a 5-cm pathlength cell, using the tube without NED as a reference. Colour is stable for two hours.

$$\text{Concentration } NO_2\text{-N} = cf \times A_s \ [\mu M]$$

Table 10.9 Preparation of working standard solutions for nitrite by dilution with bi-distilled water

Volume of solution D[1] (ml)	Total volume (ml)	Resultant concentration ($\mu M\ NO_2^-$-N)
5.0	250	1.00
5.0	500	0.50
1.0	200	0.25
1.0	500	0.10
1.0	1,000	0.05

[1] Solution "D" is prepared by diluting 5.0 ml of a SSS in 1 litre, giving a solution which contains 0.05 µmoles NO_2^--N per ml

10.5.4 Determination of ammonium

Scope, field of application and principle

This method is specific for ammonium and applicable to all kinds of natural waters (Koroleff, 1983b). Ammonium refers to the sum of ammonia and ammonium ions, because the original proportion of each in a water sample is pH dependent. The detection limit is about 0.10 µM (in 5 cm pathlength cells). The Lambert-Beer's law is followed up to about 40 µM. Interferences from amino acids and urea can be neglected. To compensate for the influence of salinity on the developed colour, a correction factor has to be applied (see below).

Reagents

Most of the reagents are caustic and very toxic, therefore mouth pipetting should not be used.

✔ Ammonium-free water (AFW). If the ammonium blank concentrations are higher than 0.3 µM, the water should be treated. Therefore, water that has been passed through an acidic deionisation cation exchange resin should be used. Alternatively, 2 ml of concentrated sulphuric acid (96 per cent) and 1 g of potassium peroxodisulphate, $K_2S_2O_8$, can be added per litre. The solution should be boiled for 10 minutes (without the condenser) to remove ammonium, and then distilled to give a residue of 150 ml. Ammonium-free water should be stored in a tightly sealed plastic container with thick walls.

- Citrate buffer solution. Dissolve 67 g of trisodium citrate dihydrate, $Na_3C_6H_5O_72H_2O$, 34 g boric acid, H_3BO_3, 19 g citric acid dihydrate, $C_6H_8O_72H_2O$, and 30 g sodium hydroxide, NaOH, in AFW and dilute to 1 litre. The solution is stable and should be stored in a well-stoppered glass bottle at room temperature.

- Reagent A, phenol-nitroprusside solution. Dissolve 35 g phenol, C_6H_5OH and 0.4 g of sodium nitroprusside dihydrate, $Na_2Fe(CN)5NO2H_2O$, in AFW and dilute to 1 litre. Store in a tightly closed bottle in a refrigerator. The solution is stable for several months. (Caution, highly toxic and must be treated as hazardous waste).

- Reagent B, hypochlorite solution. Dissolve 4 g of the sodium salt of the dichloro-isocyanuric acid and 15 g NaOH in AFW and dilute up to 1 litre. Store in a tightly closed bottle in a refrigerator. The solution is stable for several months.

- Ammonium Standard Stock solution (NH_3-N SSS). Dissolve 0.13373 g ammonium chloride, NH_4Cl, (molecular weight: 13.49), dried at 110 °C to constant weight, in AFW and dilute to 250 ml. Store in a tightly closed bottle with some drops of chloroform, in a refrigerator. The solution contains 10 µM ml^{-1}.

Table 10.10 Preparation of working standard solutions for ammonia by dilution with ammonium-free water

Volume of solution D[1] (ml)	Total volume (ml)	Resultant concentration (µM NH_3-N)
20.0	250	4.0
5.0	250	1.0
5.0	500	0.5
2.0	500	0.2

[1] Solution "D" is prepared by diluting 5.0 ml of a SSS to 1 litre, giving a solution which contains 0.05 µmoles NH_3-N per ml

Apparatus and equipment

- 25 ml test tubes with ground glass stoppers.
- Automatic syringe pipettes of 1 ml and 2 ml.
- 25 ml automatic pipettes.
- Spectrometer with cells of 1, 5 or 10 cm pathlength.

Analytical procedures

Test tubes should be carefully cleaned according to the following procedure: every tube is filled with about 25 ml water and reagents are added as described later. All ammonium contained in the tubes (dissolved in the water or adhered to the glass walls) will react. Tubes are then rinsed with AFW and stored filled with AFW. The tubes should be kept stoppered when not used. They should not be washed between the different sets of calibrations or analyses, merely rinsed with AFW (Caution: serious contamination from the air can result from smoking).

Calibration

In order to avoid disturbances from variations in pH and salinity in the samples, the calibration can be carried out in two ways. In areas where the salinity variations are small, WSS are diluted with ammonium-free seawater (AFSW) (i.e. surface seawater preferably collected shortly after a plankton bloom) (Table 10.10). For work in estuaries, where the brackish water displays large salinity variations, a calibration in AFW, followed by corrections for the salinity (Table 10.11) of each sample, is preferred. A series of WSS from the NH_3-N SSS is prepared by dilution with AFW or AFSW, using volumetric flasks.

From each of the WSS above, 25 ml triplicates are transferred to the test tubes. In addition, two sets of "blank samples" are prepared, also in triplicate, but with AFW only. To all the tubes, the reagents are added as described below in "Analysis of the samples", but to one of the blank sets a double volume of reagents is added. The blank samples here correct for the absorbance caused by the residual ammonium impurities in the AFW. The second set of blanks, those with double volume of reagents, corrects for the ammonium impurities in the reagents only. The linear regression of absorbances measured against the concentrations of the WSS (including absorbances of the first set of blanks, concentration equals 0) gives the cf. The product of cf and the absorbance of the second sets of blanks will be a constant (K), which is deduced from the results obtained with the samples and may vary from analysis to analysis because of influences such as age of solutions and contaminants from chemicals or air. Using a 5 cm cell, the current cf is approximately 11.

Table 10.11 Salinity correction factors (cf) for ammonia analysis

	Salinity (psu)									
	<8	**11**	**14**	**17**	**20**	**23**	**27**	**30**	**33**	**36**
pH	0.8	10.6	10.5	10.4	10.3	10.2	10.0	9.95	9.90	9.80
cf	1.00	1.01	1.02	1.03	1.04	1.05	1.06	1.07	1.08	1.09

Source: Koroloff, 1983b

Analysis of the samples

1. With the automatic pipette, dispense 25 ml of the sample into the test tubes.

2. With the automatic syringe pipette, add 1.5 ml of citrate buffer and mix (a vortex mixer works very well).

3. Add 0.7 ml of reagent A and using the automatic syringe pipette mix well.

4. Add 0.7 ml of reagent B and with the automatic syringe pipette mix well.

5. Stopper the test tubes, shake and keep in the dark for at least 8 hours until colour has developed. The absorbance will become constant during a maximum of 48 hours.

6. Read at 630 nm in a cell with suitable pathlength. As a reference, AFW is used.

$$\text{Concentration } NH_4^+\text{-N} = cf \times A_s - K\ [\mu M]$$

If the WSS was diluted with AFW and the samples are seawater, the results must be corrected using the correction factors given Table 10.11, depending on the salinity of the sample.

Alternative methods

A manual spectrometric method is given in ISO (1984a) where a blue compound, formed by reaction of ammonium with salycilate and hypo-chlorite ions in the presence of sodium nitrosopentacyanoferrate (III) is analysed at a limit of detection of 0.003-0.008 mg l^{-1}. An automated procedure is given by ISO (1986c) and distillation and titration method in ISO (1984b). Further details are also given in Wetzel and Likens (1990) and APHA/AWWA/WPCF (1995).

10.6 Algal and cyanobacterial identification and quantification

Approaches to the determination of cyanobacterial and algal taxa, numbers and biomass present in a sample are not yet internationally harmonised. The methods used are very variable and can be undertaken at very different levels of sophistication. Rapid and simple methods addressing the composition of a sample at the level of genera (rather than species) are often sufficient for a preliminary assessment of potential hazard and for initial management decisions. Further investigation may be necessary in order to address quantitative questions of whether cyanobacteria are present above a threshold density. More detailed taxonomic resolution and biomass analyses will be required if population development or toxin content is to be predicted. Distinction between these approaches is important because managers must decide how available staff hours can be most effectively invested. In many cases, the priority will be evaluation of a larger number of samples at a lower level of precision. Furthermore, investing time in regular intralaboratory calibrations encompassing these steps is likely to be more effective than investing time in counting protocols that reduce error, e.g. from 20 per cent to 10 per cent, but at quadrupled effort. The choice of methods also requires informed consideration of sources of variability and error at each stage of the monitoring process, from sampling to counting.

10.6.1 Identification

Microscopic examination of a bloom sample is very useful, even when quantification is not being carried out. The information obtained regarding the cyanobacteria detected can provide an instant alert that harmful cyanotoxins may be present. This information can determine the choice of bioassay or analytical technique appropriate for determining toxin levels. Most cyanobacteria can be distinguished readily from other phytoplankton and particles under the microscope at 200-400 times magnification by their morphological features.

Cyanobacterial and algal taxonomy, following the established botanical code, differentiates by genera and species. However, this differentiation is subject to some uncertainty, and organisms classified as belonging to the same species may nonetheless have substantial genetic differences, for example, with respect to microcystin production, and these cannot be differentiated microscopically. The distinction of genera is very important for assessing potential toxicity, but microcystin content varies extremely at the level of genotypes or strains, rather than at the level of species. This is one reason why identification to the taxonomic level of genera (e.g. *Microcystis, Planktothrix, Aphanizomenon* and *Anabaena*) is frequently sufficient. It may be preferable to give only the genus name especially if differentiation between species by microscopy is uncertain on the basis of current taxonomic knowledge, lack of locally available expertise, or lack of characteristic features of the specimens to be identified. This must be emphasised because "good identification practice" has frequently been misunderstood to require determination down to the species level, and this has led to numerous published misidentifications of species. Practitioners in health authorities with some experience in using a microscope can easily learn to recognise the major cyanobacterial genera and some prominent species that occur in the region they are monitoring. Such efforts should not be deterred by the pitfalls of current scientific work in cyanobacterial taxonomy which targets differentiation to the species level.

More precise identification of the dominant organisms down to species level may be useful for a more accurate estimate of toxin content. For example, *Planktothrix agardhii* and *Planktothrix rubescens* have both been shown to contain microcystins, but each species contains different analogues of the toxin with different toxicity.

For establishing cyanobacterial identification in a laboratory, initial consultation and later occasional co-operation with experts on cyanobacterial identification is helpful. Training courses for beginners should focus on the genera and species relevant in the region to be monitored. Experts can assist in deriving an initial list of these taxa and the criteria for their identification. In the course of further monitoring experience, experts should be consulted periodically for quality control and for updating such a list. Helpful publications for determination of genera and species are presented in Box 10.1.

10.6.2 Quantification by direct counting methods

Microscopic enumeration of cyanobacterial cells, filaments or colonies has the advantage of assessing directly the potentially toxic organisms. Little equipment is required other than a microscope. The method may be rather time-consuming, ranging from 10 minutes to 4 hours per sample depending upon the accuracy required and the number of species to be differentiated. Precise and widely accepted counting procedures are time consuming and require a moderate level of expertise, but serve as a basis to assess performance of simplified methods developed to suit the expertise and requirements of sampling programmes tailored to the assessment of toxic cyanobacteria. A summary of methods is provided in Table 10.12. Detailed information on sampling and on counting marine toxic phytoplankton is given in UNESCO (1995, 1996), and for marine cyanobacteria in Falconer (1993).

Sample concentration by sedimentation or centrifugation

Direct counting of preserved cells is typically carried out by Utermöhl's counting techniques (Utermöhl, 1958) using a counting chamber and inverted microscope. Cells from a sample preserved in Lugol's iodine are allowed to sediment onto the glass bottom of the chamber where they can be counted.

Counting chambers and sedimentation tubes are commercially available or can be constructed by the investigator. The most commonly used chambers have a diameter of 2.5 cm and a height of about 0.5-2 cm and thus contain 2-10 ml of sample. These can fit easily on the inverted microscope stage. If larger volumes of water have to be sedimented (for example, when cell density is low) then the height of the tube has to be increased. These extended tubes, however, are too tall to fit on the inverted microscope stage and the light would have to pass through a considerable thickness of liquid before reaching the sedimented specimens. This can be overcome using a tube in two sections, which allows the supernatant to be removed after settling without disturbing the sedimented cells on the bottom glass. The amount of sedimented water required depends on the density of cells, on the counting technique (fields or transects) and on the magnification being used.

Table 10.12 A summary of methods for the quantification of algae

Method	Volume (ml)	Sensitivity (cells per litre)	Preparation time (minutes)
Compound microscope			
Sedgewick-Rafter Cell (counting cell)	1	1,000	15
Palmer-Malloney Cell (counting cell)	0.1	10,000	15
Drops on slide		5,000-10,000	1
Inverted microscope			
Utermöhl (sedimentation chamber)	2-50	20-500[1]	2-24[2]
Epifluorescence microscopy			
Counting on filters (fluorochrome: Calco Flour)	1-100	10-1,000	15

[1] Cells per ml
[2] Hours

Source: UNESCO, 1996

Samples for sedimentation must be equilibrated to room temperature before they are placed in the settling chamber, to prevent air-bubbles from developing. The water sample must be gently inverted several times to ensure even mixing of the particles before being poured into the sedimentation chamber. The chamber must be placed on a horizontal surface to settle making sure that the content is not disturbed or exposed to temperature changes or direct sunshine. Sedimentation times vary depending on the height of the sedimentation tube and the preservative used. Various sedimentation times have been recommended in the literature (Lund *et al.,* 1958). Samples preserved in Lugol's iodine should be allowed at least 3-4 hours per centimetre height of liquid to settle. For samples preserved in neutralised formaldehyde, twice this time is required. Buoyant cyanobacterial cells (e.g. of *Microcystis* spp.) occasionally do not settle unless their gas vesicles are destroyed by applying hydrostatic pressure. Once the samples have settled, phytoplankton density can be determined by counting the organisms on the bottom of the chamber.

If an inverted microscope is not available, and samples with low cyanobacterial density need to be counted, it is possible to concentrate samples sufficiently to enable a drop (of defined volume by using a micropipette) to be counted under a standard microscope. A 100 ml measuring cylinder can be used to sediment the sample, allowing 4 hours per centimetre of sedimentation height. The supernatant can then be carefully abstracted down to the bottom 5 ml. The sample is thus concentrated by a factor of 20. Gentle centrifugation may be applied for further concentration.

Box 10.1 Sources of information for identification of cyanobacteria and algae

Anagnostidis, K. and Komárek, J. 1988 Modern approach to the classification system of cyanophytes. *Archives Hydrobiology* Supplement **80** (Algol. Studies 50-53), 327-472.

Balech, E. 1995 *The genus Alexandrium Halim (Dinoflagellata).* Sherkin Island Marine Station, Ireland.

Bourelly P. 1968 Les algues d'eau douce, T. II. *Les algues d'eau douce. Initiation a la systématique. Tome III. Les algues jaunes et brunes.* Boubée, Paris, 438 pp.

Bourelly P. 1970 Les algues d'eau douce, Tome III. *Les algues d'eau douce. Initiation a la systématique. Tome III. Les algues bleues et rouges. Les Eugléniens, Peridiniens et Cryptomonadines.* Boubée, Paris, 512 pp.

Bourelly, P. 1972 Les algues d'eau douce. T.I. *Les algues d'eau douce. Initiation a la systématique. Tome III. Les algues vertes.* Boubée, Paris, 572 pp.

Carr, N.G. and Whitton, B.A. 1973 *The Biology of the Blue-Green Algae.* Botanical Monographs. 9, Blackwell, Oxford, 676 pp.

Carr, N.G. and Whitton, B.A. 1982 The biology of Cyanobacteria. *Botanical Monographs* **17**, Blackwell, Oxford, 688 pp.

Fay, P. and Vanbaalen, C. 1987 *The Cyanobacteria.* Elsevier, Amsterdam, 543 pp.

Fogg, G.E., Stewart, W.D.P., Fay, P. and Walsby, A.E. 1973 *The Blue-Green Algae.* Academic Press, London, 459 pp.

Hasle, G.R. and Fryxell, G.A. 1995 Taxonomy of Diatoms. In: G.M. Hallegraeff, D.M. Anderson and A.D. Cembella [Eds] *Manual on Harmful Marine Micro-algae.* IOC Manuals and Guides No. 33, UNESCO, United Nations Educational, Scientific and Cultural Organization, Paris, 339-364.

Komárek, J. and Anagnostidis, K. 1985 Modern approach to the classification system of cyanophytes. I. Introduction. *Archives Hydrobiology* Supplement **71** (Algol. Studies 38/39), 291-302.

Komárek, J. and Anagnostidis, K. 1986 Modern approach to the classification system of cyanophytes. *Archives Hydrobiology* Supplement **73** (Algol. Studies 43), 157-226.

Skulberg, O.M., Carmichael, W.W., Codd, G.A. and Skulberg, R. 1994 Taxonomy of toxic cyanophyceae (Cyanobacteria). In: I.R. Falconer [Ed.] *Algal Toxins in Seafood and Drinking*

Water. Academic Press, London, 145-164.

Starmach, K. 1966 *Cyanophyta. Flora slodkowodna Polski 2.* Polska Akademia, Warszawa, 807 pp.

Taylor, F.J.R., Fukuyo, Y. and Larsen, J. 1995 In: G.M. Hallegraeff, D.M. Anderson and A.D. Cembella [Eds] *Manual on Harmful Marine Microalgae.* IOC Manuals and Guides No. 33, UNESCO, United Nations Educational, Scientific and Cultural Organization, Paris, 283-317.

Tomas, C.R. 1993 *Marine Phytoplankton: A Guide to Naked Flagellates and Coccolithophorids.* Academic Press, Inc., New York.

Tomas, C.R. 1996 *Identifying Marine Diatoms and Dinoflagellates.* Academic Press, Inc., New York.

Where sedimentation is not possible, centrifugation (360× g for 15 minutes using 10-20 ml sample) can offer a rapid and convenient method of concentrating a sample (Ballantine, 1953). Centrifugation may be aided by addition of a precipitating agent, such as potassium aluminium sulphate (1 per cent solution) added at 0.05 ml per 10 ml sample. Fixation with Lugol's solution enhances susceptibility to separation by centrifugation. However, buoyant cells may still be difficult to pellet and may require disruption of gas vesicles prior to centrifugation by applying sudden hydrostatic pressure (Walsby, 1994), for example in a well sealed syringe or by banging a cork into the bottle very tightly.

Counting cyanobacteria and algae

When counting cyanobacteria the units to be counted must be defined. The majority of planktonic cyanobacteria are present as filamentous or colonial forms consisting of a large number of cells which are often difficult to distinguish separately. The accuracy of quantitative determination depends on the number of counted objects (e.g. cells or colonies); the relative error is approximately indirectly proportional to the square root of the number of objects counted. The number of colonies, not the number of cells, is decisive for accurate enumeration. However, the number of colonies is often not very high even in water containing a heavy bloom where only several dozen colonies may be present in a 100 ml sample. Both filaments and colonies can differ greatly in the number of cells present, hence results given as number of colonies, for example stating that 1 ml of sample contains an average of 2.43 colonies of *Microcystis aeruginosa,* gives little information on the quantity of cyanobacteria.

Typically unicellular species are counted as cells per millilitre and filamentous species can be counted as the number of filaments, with an average number of cells per filament quoted (often the cells per filament in the first 30 filaments encountered are counted and averaged), or they can be measured as total filament length by estimating the extension of each filament within a counting grid placed in the ocular of the microscope. The latter is more precise if filament length is highly variable. For colonial species, disintegration of the colonies and subsequent counting of the individual cells is preferable to counting colonies and estimating colony size. Colonies sometimes disintegrate after several days when fixed with Lugol's iodine solution. For more stable colonies, disintegration can be achieved by ultrasonication. This often separates cells very effectively and, in cases where colonies do not totally break down into single cells, their size may be reduced sufficiently to allow single cells to be counted. Sometimes, this is not successful and it is necessary to estimate the geometric volume of individual colonies. If colonies are relatively uniform in size, the average number of cells per colony may be determined and then the colonies may be counted. Generally, the use of values for numbers of cells per colony published in the literature is not recommended because the size of colonies varies greatly.

There are several methods for counting organisms. Most approaches aim at counting only a defined part of the sample and calculating back to the volume of the entire sample. The most common methods are: total surface counting, counting in transects and counting in fields. Counting the total chamber bottom may be very time consuming. It is usually only appropriate for very large counting units (cells, colonies, and filaments) at low magnification. Counting cells in transects from one edge of the chamber to the other, passing through the central point of the chamber, is more efficient. Some inverted microscopes are equipped with special oculars that enable the transect width to be adjusted as required. However, in many cases, horizontal or vertical sides of a simple counting grid can be used to indicate the margin of the transect. Back-calculating to 1 ml of sample can be done by measuring the area of the transects and of the chamber bottom, together with the volume of the counting chamber.

Cyanobacteria and algae occurring in randomly selected fields may be counted. When changing the position of the chamber to find the next field, it is preferable to avoid looking through the microscope to ensure random choice of fields. Microscopic field area covered by a counting grid is usually considered as one field. However, if no counting grid is available, the total spherical field can be considered. Back-calculating to 1 ml of sample can be done by measuring the area of the field and of the chamber bottom together with the volume of the counting chamber.

The density of different species in one sample can vary and there can also be several orders of magnitude difference between the sizes of the species, and therefore it is necessary to select the counting method that is adequate for the sample. Total chamber surface counting with low magnification (100×) may be useful for large species whereas transect or field counting with higher (200× to 400×) magnification is used for smaller or unicellular cyanobacteria and algae. Accurate enumeration using transects or fields assumes even distribution of cyanobacteria and algae on the bottom of the chamber surface after sedimentation. Due to the inevitable convection currents in the sedimentation chamber, that are very difficult to avoid, cells very rarely settle evenly on the surface of the bottom glass - they are almost always more dense either in the middle or around the circumference of the chamber. In some cases, density also varies between opposite edges of the bottom glass. The misestimation that arises from uneven distribution can be minimised by transect counting or by taking a fairly even distribution of randomly selected field. Counting four perpendicular diameters can minimise this error. The relation between counting time and accuracy is best if about 100 counting units (cells, colonies, and filaments) are settled in one transect. This may be achieved by diluting or concentrating samples so that the number of units of the important species lies in this range.

Specimens occurring exactly on the margin of the counting area (transect or field) present the common problem of whether to count them or not. For transect counting, those specimens that lie across the left margin are ignored, while those that cross the right margin are included. In field counting, two predetermined sides of the grid are included, the other two are ignored.

There are different recommendations in the published literature concerning how many specimens per species should be counted for reliable results. Mass developments of phytoplankton populations are generally characterised by dominance of 1-3 species. It is unusual for more than six to eight species to contribute to the majority of biomass. Therefore, it is suggested that 400-800 specimens in each sample are counted, leading to a maximum error of the total count of 7-10 per cent. In this situation 10-20 per cent of the error is accounted for by the few dominant species, 20-60 per cent is accounted for by the subdominant species and the rest of the species can be considered as insufficiently counted. If only cyanobacteria are to be counted, and only one or two species are present, counting up to the precision level of 20 per cent (by counting 400 individual units per species) can be accomplished within less than one hour.

Other counting chambers (e.g. Sedgewick-Rafter or haemacytometers) are available for use with a standard microscope. Samples might require prior concentration or dilution. It can also be useful to monitor samples under high magnification with oil-immersion (1,000×) to check the sample for the presence of very small forms that may be overlooked during normal counting.

The use of mechanical or electronic counters for recording cell counts can shorten counting time considerably, especially if only a few species are counted.

Simplifications

One alternative method, which has been found to be useful, is syringe filtration. This method is considerably less time-consuming because it does not depend on lengthy sedimentation times. Water samples (10 ml) are filtered through a membrane filter disc (13 mm) contained in a filtration device. The filter with the captured phytoplankton is dried at room temperature, and then placed on a drop of immersion oil on a microscope slide. A further drop of immersion oil is placed onto the surface of the filter, which makes the filter transparent. The sample is observed under a standard microscope (200× or 400×) without using a micro-cover slip. All cells on the surface of the filter are counted and the number of cells per litre can be calculated.

For optimising the relationship between the time spent and the information gained, various simplifications are possible. No method of enumeration is definitive, and personal creativity as well as understanding of potential pitfalls may compensate for lack of the ideal equipment or time. For each method applied (for improvisations as well as for "benchmark methods") it is crucial to check for reproducibility and comparability of the method established in the laboratory (parallel counts should not deviate by more than 20 per cent). Furthermore, clear statements of the units in which the results are given are of critical, but often unrecognised, importance. Unfortunately, many reported results are unclear about the size of units quoted, i.e. "one colony", or the size of "one filament". Such terminology varies between laboratories and makes it impossible to compare results.

For estimation of error, UNESCO (1996) gives the following equation (see Table 10.12): at a 95 per cent level of confidence, the relative limits of expected concentrations = ± (2 × 100%)/($n^{0.5}$/n). For example, if in a sample volume of 10 ml only 50 cells of species "x" are counted, the result is 5,000 cells per litre. Assuming a deviation of 28 at counting 50, this results in ± 2,800 cells per litre. If the sample was concentrated 10 fold, so that 500 cells were counted, this would result in a higher accuracy of 5,000 ± 900 cells per litre. As a result of the extremely dynamic changes of cyanobacterial density in many water bodies (often amounting to more than 10 fold within a few hours) the precision obtained in counting 50 cells may, in many cases, be quite sufficient for estimating the potential risk involved.

10.6.3 Determination of cyanobacterial biomass

Cell size can vary considerably within species and by a factor of 10 to 100 or more between species; and toxin concentration relates more closely to the amount of dry matter in a sample than to the number of cells. Hence, cell numbers are often not an ideal measure of population size or potential toxicity. This can be overcome by determining biomass. Two approaches are available: estimation from cell counts and average cell volumes, and estimation from chemical analysis of pigment content (chlorophyll *a*).

Cyanobacterial and algal counts and cell volumes

Biovolume can be obtained from cell counts by determining the average cell volume for each species or unit counted, and then multiplying this by the cell number present in the sample to give the total volume of each species. The specific weight of plankton cells is almost 1 mg mm^{-3} and therefore biovolume corresponds quite closely to biomass. Average volumes can be determined by assuming idealised geometric bodies for each species (e.g. spheres for *Microcystis* cells, cylinders for filaments), measuring the relevant geometric dimensions of 10-30 cells (depending upon variability) of each species, and calculating the corresponding mean volume of the respective geometric body.

Simplification for biomass estimates

If the deviation of numbers of dominant species counted in two perpendicular transects is less than 20 per cent between both transects, it is not necessary to count further transects. If the standard deviation of cell dimensions measured on 10 cells is less than 20 per cent, it is not necessary to measure further cells.

If a set of samples from the same water body and only slightly differing sites (e.g. vertical or horizontal profiles) is to be analysed, enumerate all samples, but measure cell dimensions only from one. Check others by visual estimate for deviations of cell dimensions and conduct measurements only if deviations are suspected.

Chlorophyll a *analysis*

The concentration of chlorophyll *a* may be used as a sensitive approximation of algal biomass and as an alternative to counting and measuring biovolumes. Chlorophyll *a* concentrations of mixed phytoplankton populations give an overestimation of the biomass of the cyanobacteria and algae of interest. This degree of overestimation can be assessed by a brief microscopic estimate of the share of cyanobacteria and algae in relation to other phytoplankton biomass. Nevertheless during cyanobacterial mass developments chiefly consisting of one species, chlorophyll *a* may be a good measure of biomass. The method requires relatively simple laboratory equipment and is considerably less time-consuming than microscopic enumeration. A useful analytical protocol is given in ISO (1992). This method involves an extraction procedure with hot ethanol to inactivate chlorophyllase and accelerate the lysis of pigments.

Apparatus

✔ Spectrometer, for use in the visible range up to 800-900 nm, with a resolution of 1 nm, a bandwidth of 2 nm or less, sensitivity less or equal to 0.001 absorbance units and with optical cells of pathlength between 1 cm and 5 cm.

✔ Vacuum filtration device, filter holder with clamp.

✔ Vacuum water pump or electric vacuum pump (in the laboratory).

✔ Glass fibre filters free of organic binder (average pore size 0.7 µm, 47 mm diameter).

✔ Filters for filtration of the extracts (average pore size 0.7 µm, 25 mm diameter), or as an alternative a centrifuge (possibly refrigerated), with an acceleration of 6,000 g and a swinging rotor suitable for extraction tubes.

✔ Extraction vessels, e.g. wide-necked amber glass vials with polytetrafluorethylene (PTFE) lined screw caps, typically of 30 ml to 50 ml capacity and suitable for centrifugation at 6,000 g.

✔ Water bath, adjustable to 75 °C ± 1 °C with a rack for extraction vessels.

Filtration

1. Samples must be shaken before filtration in order to mix thoroughly. Filter a measured volume of the sample (normally between 0.1 and 2 litres, depending on the concentration of algae and cyanobacteria). Pour into the filter cup, drop by drop, recording the volume, to avoid filter clogging.

2. Filter continuously and do not allow the filter to dry during filtration of a single sample. Vacuum pressure during filtration should not exceed 0.5 atmospheres.

3. The vacuum pressure should be reduced just before the filters become dry, in order to leave a thin layer of water and avoid rupture of the algal or cyanobacterial cells.

4. Some analysts recommend adding 0.2 ml of magnesium carbonate suspension (1 per cent (w/v) $MgCO_3$, shaken before use) to the final few millilitres of water in the filter cup. Avoid touching the filter with fingers. It preferable to use forceps. Direct sunshine must also be avoided because chlorophyll degrades rapidly.

5. Filters must either be frozen immediately (see below) or covered with hot ethanol to avoid pigment degradation.

6. Filters must be folded so that the cell layer is protected from rubbing off onto packaging materials. Wrapping folded filters in aluminium foil is a practical solution because it protects the filter and enables labelling on the foil.

Procedure

If extraction is not performed immediately, filters should be placed in individual labelled bags or Petri dishes and stored at -20 °C in darkness. Samples are readily transported in this form.

Extract the filters with a total volume of 10-40 ml (the volume to be used depends on the size of the photometer cuvette) of boiling 90 per cent ethanol (v/v) at 75 °C and leave overnight (24 hours) at +4 °C or in darkness at approximately 20 °C for 24-48 hours. Ethanol containing a denaturant is used successfully in many laboratories. However, denaturants vary and it is prudent to use ethanol without a denaturant or to run comparative analyses to assess the effect of the denaturant. A comparative determination with 90 per cent pure ethanol is recommended.

Homogenisation either by ultrasonication or with a tissue grinder, may be performed to disrupt cells and enhance extraction, after having poured part of the boiling ethanol onto the filter and having used the rest to rinse the apparatus. However, homogenisation is not likely to be essential for extraction of cyanobacteria.

Clarification of the slurry

1. Centrifuge the ethanol and the filter for 15 minutes at 6,000 g. This should result in a clear supernatant.

2. Carefully decant the clear supernatant with a Pasteur pipette into a calibrated flask with stopper. Fill to the mark with ethanol, stopper and mix. This is the extract volume V_e in millilitres.

As an alternative, filter the slurry through a filter (see Apparatus section above) into a calibrated flask with a stopper. Wash the extraction vessel with ethanol and transfer quantitatively into the calibrated flask. Fill to the mark with ethanol, stopper and mix. This is the extract volume V_e.

3. Store the flasks in darkness and proceed promptly to the measurement step.

Measurement

Blank the spectrometer with the same ethanol at each wavelength before reading sample.

1. Transfer the clear extract into the cuvette using a pipette, either a) leaving sufficient volume in the cuvette for the addition of HCl (if it is preferred to proceed by adding HCl

directly into the cuvette) or b) leaving a sufficient volume of extract in the flask for a second measurement after acidification of the extract left in the flask.

2. Record the absorbance at 750 nm (750a) and 665 nm (665a) against a reference cell filled with ethanol. The absorbance at 665 nm should fall between 0.01 and 0.8 units. This may be achieved by choosing a suitable volume of water to be filtered, extractant volume, dilution, or pathlength, etc. To start with, take 0.5 litre of sample, 20 ml of ethanol and a 5 cm cuvette.

3. Proceed with the acidification step, either a) adding HCl directly into the cuvette, or b) acidifying the extract left in the flask. In either case, add 0.01 ml of HCl 3 mol l^{-1} per 10 ml of extract volume and agitate gently for 1 minute.

4. Record absorbance at 750 nm (750b) and 665 nm (665b) after between 5 minutes and 30 minutes.

Calculation and expression of results

1. Calculate absorbance of the extract before acidification: $665_a - 750_a = A_a$

Calculate absorbance of the extract after acidification: $665_b - 750_b = A_b$

2. Calculate chlorophyll a concentration (Chla) in mg m^{-3}

$$\text{Chla} = 29.6 \times (A_a - A_b) \times V_e / V_s \times d$$

where:

V_e is the volume, in millilitres, of the extract
V_s is the volume, in litres, of the filtered water sample
d is the pathlength, in centimetres, of the cuvette
3. Phaeopigment concentration (Phaeo) in mg m^{-3} may be calculated to indicate the portion of inactive cyanobacterial and algal biomass:

$$\text{Phaeo} = 20.8 \times A_b \times V_e / V_s \times d - \text{Chla}$$

Note: The ratio of chlorophyll a to phaeophytin a gives an indication of the effectiveness of sample preservation, as well as of the condition of the cyanobacterial algal population. When samples are concentrated by filtration for the purposes of analysis, the cells die. Consequently, the chlorophyll immediately starts to degrade to phaeopigments. If filters are not extracted rapidly with hot ethanol, or frozen, chlorophyll a concentrations start to reduce. Occasionally, other factors disturb this method, resulting in very low or even negative values for chlorophyll a. If this occurs, the following calculation should be made:

$$\text{Chlorophyl } a + \text{Phaeophytin } a = \frac{12.2(663a) \times V_e}{V_s \times 1} \text{ mg m}^{-1}$$

This should result in a similar value as for the sum of the concentrations of both pigments determined separately, as above.

10.7 Detection of toxins and toxicity

Laboratory methods used to evaluate toxins can vary greatly in their degree of sophistication and the information they provide. Relatively simple, low cost methods can be employed which rapidly evaluate the potential hazard and allow management decisions to be taken. In contrast, highly sophisticated analytical techniques determine precisely the identity and quantity of cyanotoxins. Information obtained from simple rapid screening methods, such as microscopic examination can be used to make an informed decision on the type of bioassay or physicochemical technique that will be adequate. Currently, there is no single method that can be adopted that will provide adequate monitoring for all cyanotoxins in the different sample types that might have to be evaluated. The increasing variety and number of individual cyanotoxins being discovered make the goal of very specific and sensitive analytical methods that would detect all relevant toxins increasingly complex and ultimately unachievable (Yoo *et al.,* 1995).

In conclusion, it is strongly suggested that a monitoring programme of toxin concentrations should not be adopted as a matter of course but only when specifically indicated as discussed in section 10.1. For most recreational sites, monitoring the development of blooms rather than toxins is a more rational approach. A comprehensive review of methods and approaches is given in Lawton *et al.* (1999) and for marine algal and cyanobacterial toxins in UNESCO (1995).

10.8 Elements of good practice

• Monitoring of recreational water-use areas should be sufficient to identify the risk of blooms, taking into account actual or potential accumulation of toxic cyanobacteria and algae.

• Sampling points should be located to represent different water masses (stratified waters, waters coming from river mouths, etc.) in the investigation area and the sources of nutrients (discharges, upwellings, etc.). Possible transport mechanisms of toxic phytoplankton should be considered, possible physical forcings should be identified and sampling schemes arranged accordingly.

• In areas of high risk, sampling for algae should be carried out at least weekly. During development of blooms, sampling should be intensified to daily.

• Monitoring of toxicity (using bioassays, chemical or immunological procedures) is only justified where reason exists to suspect that hazards to human health may be significant. In such cases, long-term information on phytoplankton populations (toxic, harmful and others) should be collected where appropriate.

• Analysis of toxins should only be undertaken where standard, replicable and reliable analyses can be performed.

• Where conditions are such that monitoring is considered essential, temperature, salinity (in marine coastal areas), dissolved oxygen, transparency, presence of surface water stratification, phytoplankton biomass (chlorophyll), surface current circulation (transport of algae) and meteorological patterns (such as seasonal rainfall, storms and special wind regimes) should be considered.

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Chapter 11*: OTHER BIOLOGICAL, PHYSICAL AND CHEMICAL HAZARDS

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In addition to those described in Chapters 3, 7, 8 and 10 there are a number of other diverse biological, physical and chemical hazards that could be encountered in the recreational water environment. Many of these are local in nature and should be addressed in monitoring programmes where they are known or suspected to be locally important. The characteristics of the hazard and the local conditions define the appropriate remedial measures. It is important that standards, monitoring and implementation enable preventative and remedial actions in this time frame that will prevent health effects arising from such hazards. The WHO *Guidelines for Safe Recreational-water Environments* (WHO, 1998) emphasise the importance of identification of circumstances that will support a continuously safe environment for recreation. This includes awareness of biological hazards such as those discussed in this chapter.

The following sections provide a summary of the assessment and control of some biological, physical and chemical hazards encountered in recreational waters. The on-site visit form should be adapted to take account of locally-occurring hazards and any special features of a particular recreational-use area. Inspections should be carried out annually.

During an environmental health assessment, biological risks such as the presence of disease-causing, poisonous or venomous animals or plants, and physical hazards such as extreme water temperatures, should be noted. Effective ways of informing the public and, where possible, protecting them from such hazards should be recommended.

11.1 Biological hazards

11.1.1 Health hazards

Injuries from dangerous aquatic organisms may be sustained in a number of ways, for example accidental encounters with venomous sessile or floating organisms when bathing or treading on stingrays, weeverfish or sea urchins.

Table 11.1 Relative risk to humans posed by selected groups of organisms

Attacks and poisonings by dangerous organisms	Mild discomfort	Requires further medical attention	Requires emergency medical attention
Non-venomous organisms			
Sharks		✓	✓
Barracudas		✓	
Needlefish		✓	✓
Groupers		✓	
Piranhas		✓	
Conger eels		✓	
Moray eels		✓	
Electric fish		✓	
Giant clams		✓	
Seals & sea lions		✓	
Hippopotami		✓	✓
Crocodiles		✓	✓
Venomous invertebrates			
Sponges	✓	✓	
Hydroids	✓	✓	
Portuguese man of war	✓	✓	✓
Jellyfish	✓	✓	
Box-jellyfish		✓	✓
Hard corals	✓	✓	
Sea anemones	✓	✓	
Blue-ringed octopus			✓
Cone shells		✓	✓
Bristle-worms	✓	✓	
Crown of thorns	✓	✓	
Sea urchins (most)	✓		
Flower sea urchin		✓	✓
Venomous vertebrates			
Stingrays		✓	✓
Catfish	✓	✓	

Weeverfish	✓	✓	
Stonefish		✓	✓
Surgeonfish	✓	✓	
Snakes		✓	✓

Source: WHO (1998)

Unnecessary handling, or provocation of venomous organisms during seashore exploration, or invading the territory of animals when swimming may also lead to injury. Table 11.1 lists the relative risk to humans posed by venomous and non-venomous organisms which may be encountered in recreational bathing areas.

11.1.2 Monitoring and assessment

Routine monitoring for biological hazards is justified only where they exist. Monitoring should respond to local conditions; for example where there is a known hazard, such as jellyfish, it is important to identify the source and the nature of the hazard (Raupp *et al.,* 1996).

11.1.3 Remedial measures

Many serious incidents can be avoided through an increase in public education and awareness. Surveillance systems should be in place to provide warnings to the public in areas where it is known that sharks, jellyfish and other hazardous organisms are common (see examples given in Box 11.1) (Fenner, 1998). It is important to identify and to assess the risks from various aquatic organisms in a given region. Where the health outcomes are known to be mild, remedial measures should be based primarily upon raising public awareness and providing information to the public. This may be done in a number of ways (see Chapter 6) and may require only simple messages, such as advice to wear suitable footwear whilst exploring the intertidal area or to avoid handling marine or freshwater organisms. Where the health outcomes are known to be more severe it may be necessary to declare exclusion zones in bathing areas or to restrict bathing where or when appropriate.

11.2 Microbiological hazards

11.2.1 Leptospirosis

Leptospirosis (Weil's disease or haemorrhagic jaundice) is usually characterised by the sudden onset of fever and chills, severe headache, muscular pain, abdominal pain, nausea and conjunctivitis. The causative bacterium is of the genus *Leptospira.* Other symptoms may include aseptic meningitis, conjunctival haemorrhage, rash, jaundice and cough with bloodstained sputum. The organism enters the body either through abraded skin or by contact with mucous membranes. The incubation period is 10-12 days (range 3-30 days) and symptoms persist for approximately one week. Prolonged mental health symptoms may occur after leptospirosis but the relationship is not well documented.

Monitoring and assessment

Water becomes contaminated with leptospires from the urine of infected domestic animals and rodents. Therefore, the common faecal indicator organisms cannot be relied upon as indicators of the presence of the *Leptospira.* The detection of pathogenic leptospires in water is difficult. They are relatively slow growing during enrichment and do not compete well against other more rapid growing organisms. Culture reactions and serology are required to distinguish pathogenic and saprophytic strains. Routine monitoring is not recommended but recreational waters should be examined for leptospires when they are suspected to be the source of an outbreak of leptospirosis. For example, a recent outbreak of leptospirosis following a triathlon race in Springfield, USA, clearly illustrated the need for monitoring, particularly where large numbers of people are at risk.

Box 11.1 Remedial measures to deal with jellyfish in bathing areas in Barcelona, Spain

Jellyfish are commonly found in coastal environments, although their normal environment is approximately 50 miles away from the coastlines in oceanic waters. The factors that govern their presence at the coastline are still unclear. Although dry weather conditions (dry years) are considered to contribute to jellyfish along the coastline, their presence should be considered a natural phenomenon not linked to pollution. Their presence usually indicates a flow of oceanic waters towards the coastline.

In order to alert bathers to the hazards posed by jellyfish an information leaflet has been produced for the public, including illustrations of the most common species, and a code of behaviour with recommendations in the event of encountering jellyfish. It also includes telephone numbers that sailors or fishermen can call if a large number of jellyfish are seen moving in the direction of the coast, so that preventative measures can be undertaken. The Jellyfish Expertise Centre, Institute of Marine Sciences, Barcelona has developed an Internet web page with practical information and recommendations. Jellyfish can be prevented from reaching the beach by removing them from the water using Pelican boats that are used to eliminate undesirable floating pollution on the water. More than 400 jellyfish have been removed in a single day by such means. Preventative measures that have been applied in other regions include the installation of nets and bubble screens. Unfortunately, jellyfish tend to clog nets and to break into pieces that continue to sting. If a jellyfish is detected at the beach, megaphones or loudspeakers alert bathers, safety warning flags can be changed to red to ensure that people stay out of the water. Additionally, the Red Cross and lifeguards can be prepared to deal with bather queries as well as to provide First Aid to those who have been stung. More than 40 stings have been attended in a single day when warnings have been ignored. Several municipalities within Barcelona now have small boats with shallow draughts that can remove jellyfish from bathing areas nearer to the beach.

A jellyfish stranded on the beach should not be handled because it can still sting - its venom-filled pouch remains active. Therefore, it is also important to remove any jellyfish on the beach, although it is necessary to be aware that small broken pieces may still be active while remaining moist. Lightweight protective clothing, such as a lycra suit, or a layer of petroleum jelly spread on unprotected skin, have been shown to protect swimmers against stings. Severe allergic reactions are uncommon unless a person has a history of allergy (atopy or asthma) or has been stung previously or has heart disease. However, cases of toxicity and allergic contact dermatitis and leukocytoclastic vasculitis have been reported following a jellyfish sting.

Source: Maria Figueras, Unitat de Biologia i Microbiologia, Facultat de Medicina i Sciences de la Salut, Sant Llorenc 21 43201 Reus.

Remedial measures, interpretation and reporting

The risk of leptospirosis can be reduced by preventing direct animal access to swimming areas, by treating farm animal wastes prior to discharge and by informing users about the risks of swimming in water that is accessible to domestic and wild animals. Outbreaks of the disease are not common and the risk of leptospirosis associated with swimming areas is low. Outbreaks associated with salt water have never been reported. As a precautionary measure, domestic and wild animals should not have access to swimming areas.

11.2.2 Schistosomes

Among human parasitic diseases, schistosomiasis (sometimes called bilharziasis) ranks second behind malaria in terms of socio-economic impact and health consequences in tropical and subtropical areas. The disease is endemic in 74 developing countries and world-wide some 200 million people are infected.

The major forms of schistosomiasis are caused by five species of water-borne flatworms, or blood flukes, called schistosomes. Humans become infected after contact with water containing the infective stage of the parasite. Intestinal schistosomiasis caused by *Schistosoma mansoni* occurs in the Eastern Mediterranean, Sub Saharan Africa, the Caribbean and South America. Oriental or Asiatic intestinal schistosomiasis, caused by the *S. japonicum* (including *S. mekongi* in the Mekong River basin) group of parasites, is endemic in South East Asia and in the Western Pacific region. Another species of *S. intercalatum* has been reported from 10 African countries. Urinary schistosomiasis, caused by *S. haematobium,* is endemic in Africa and the Eastern Mediterranean. Each of the five species may give rise to acute or chronic disease with widely differing symptoms and clinical signs.

Monitoring and assessment

A computerised global database for schistosomiasis has been established by WHO. The database includes information on epidemiology, control activities, people responsible for control, and water resources for each endemic country.

Schistosomiasis and water are inextricably linked and the high prevalence of schistosomiasis in many parts of the world is closely related to human contact with natural water bodies. Some water contact is occupational and to some extent necessary, but most transmission of schistosomiasis occurs during water contact for domestic and recreational purposes. In endemic areas used by local populations or tourists, monitoring programmes should be implemented. Health education, information and communication are therefore important in a strategy to control morbidity. The objectives of health education are to help people understand that their own behaviour, principally water use practices and indiscriminate urination and defecation, as well as failure to use available screening services or to comply with medical treatment, is a key factor in transmission.

In the short term, where the prevalence of schistosomiasis is high, population-based chemotherapy can reduce the prevalence, severity and morbidity of the disease. Long-term operational and budgetary planning should be made for diagnostic facilities and re-treatment schedules, as well as treatment throughout the health care system and for transmission control. In areas where tourism is important to the economy, molluscides (chemicals that kill the aquatic intermediate host snails) may be applied at specific locations. However, molluscicides are expensive and have impacts on other aquatic life.

11.3 Sun, heat and cold

11.3.1 Health risks

Prolonged recreational use of water can lead to exposure to extreme cold or excessive solar radiation. Staying on the adjacent land area can also lead to enhanced exposure to ultraviolet (UV) radiation, because of the reflection of the sun's rays from the surface of the water. Children are often more at risk because they tend not to use sunglasses and because they spend long periods of time going in and out of the water without a protective sunscreen. Skin cancers and cataracts are important health concerns - the United Nations Environment Programme (UNEP) has estimated that over 2×10^6 non-melanoma skin cancers and 200,000 malignant melanomas occur globally each year. It has been reported by WHO that over half of the world's blind population lose their eyesight because of cataract (WHO, 1993b). It is believed that in about 20 per cent of cataract sufferers disease is triggered by short wave ultraviolet light. Direct exposure to UV radiation has both harmful and beneficial effects on humans.

A number of epidemiological studies have implicated solar radiation as a cause of skin cancer in fair-skinned humans (IARC, 1992) and severe sunburn in children has been shown as a risk factor for malignant melanoma (Katsambas and Nicolaidou, 1996; Weinstock, 1996).

Severe heat stress is also a potential risk for anyone exposed to high temperatures. The most common clinical syndromes are heat cramps, heat exhaustion and heat stroke. Recreational water users commonly expose themselves to prolonged periods of high temperatures, which is often exacerbated if they are undergoing physical exercise.

Exposure to extreme low temperatures, such as often experienced by swimmers in the open sea can also present health risks. When the body temperature falls there is a sense of confusion, a reduction in swimming capability (coupled with an overestimation of swimming capability), a possible loss of consciousness, and death by hypothermic cardiac arrest or drowning.

11.3.2 Remedial measures

Health education campaigns play an important part in the prevention of health effects due to sun, heat and cold exposure. In the UK, for example, 89 per cent of health authorities were found by Sabri and Harvey (1996) to be implementing primary prevention programmes in an attempt to meet the Government's "Health of the Nation" target of stabilising incidence of skin cancer by the year 2005. However, relatively few of these prevention programmes have been evaluated (Melia *et al.,* 1994). Data from Australia have shown that the "Slip! Slop! Slap!" campaign that was initiated in 1980, and its follow-up campaign "Sunsmart" in 1988, were effective in increasing awareness and self-reported sun-protection behaviour (Harvey, 1995). Evaluation of other health education campaigns in various countries have reported improved public knowledge about the dangers of exposure to sunshine but no significant change in sun-protection behaviour (Cameron and McGuire, 1990). Where a corresponding change in behaviour was reported, it was due to the use of sunscreens (Hughes *et al.,* 1993). To avoid the adverse effects of exposure to UV radiation, the correct use of sunscreens and protective clothing should be advocated, taking into consideration the UV index. Currently, however, there is no direct experimental evidence that sunscreens are effective in reducing skin cancer incidence (Sabri and Harvey, 1996). Melia *et al.* (1994) concluded that the benefits of education about sun protection are as yet unproven but, if organised effectively, education is likely in the long term to reduce the risk of most skin cancers and photo-aging. They noted that local initiatives require a multidisciplinary approach to ensure co-operation between general practitioners, dermatologists, pathologists and health promotion officers. Such initiatives should be supported by a national programme promoting sun protection and awareness together with national monitoring of changes in knowledge and behaviour.

Integrated coastal management (ICM) plans may help with remedial measures by displaying information in prominent places at the recreational bathing area, developing campaigns and high profile media events. At the local level, ICM has a large role to play in information dissemination.

11.4 Physical and chemical hazards

11.4.1 Health hazards

Local factors, such as agricultural or, industrial activity, can have a strong influence on the aspects of physical and chemical water quality. Therefore, before standards can be set, it is essential to understand the general characteristics of the water body of interest, together with the effect of local environmental conditions, the processes affecting the concentrations of the physical and chemical variables, and the factors that may modify the toxicity of these variables. It may, therefore, be more appropriate to identify water quality standards on a local basis rather than to adopt national standards for physicochemical aspects of recreational water quality. In determining the likely hazards of physicochemical variables, it is important to evaluate the degree of exposure that recreational users will encounter (the use of wet suits for example, will prolong immersion in cold climates).

11.4.2 Monitoring and assessment

During an inspection for chemical hazards, attention should be paid to the presence of industrial effluent disposal facilities, such as outfalls, sewers and rivers, tributaries, streams or ditches. Adjacent activities and facilities, such as intensive agriculture, electricity generating stations, dredging operations, naval bases, shipyards and terminals, should also be identified and their impact should be assessed. The assessment should also consider the impact of physical characteristics of the local beach and of the meteorological conditions on the dispersion and dilution of contaminants in the recreational-use area. During an inspection it may be necessary to collect representative water samples to confirm the presence of specific chemical contaminants, to establish their magnitude and variability, to identify the source(s) and to evaluate human exposures and health affects. Chemical analyses should only be undertaken where standard, replicable and reliable methods are available. In some cases, simple physical tests, such as pH, turbidity or colour, can be measured on-site and used as surrogates for general chemical contamination. Routine monitoring for specific chemicals should only be considered when the inspection indicates a significant hazard to human health.

In assessing local problems, initial screening for risks associated with ingestion may be undertaken by applying the WHO *Guidelines for Drinking-Water Quality* (WHO, 1993a, 1998), with an appropriate correction factor. Although the Guidelines for Drinking-Water Quality are not, generally, based upon short-term exposures, for the purpose of assessing the risk associated with occasional use of recreational waters, public health authorities can use a reference value of 100 times the value Guideline (for other than acute adverse effects) for an initial screening assessment of recreational water pollution with chemicals with known health effects arising from long-term exposure. It should be emphasised that the exceedence of such a reference value does not necessarily imply a risk, but indicate that public health authorities should evaluate the situation.

If, on consideration, it seems probable that contamination is occurring and recreational users are exposed to significant quantities, chemical analysis will be required to support a quantitative risk assessment. Care should be taken in designing the sampling programme to account, for example, for temporal variations and the effects of water currents. If resources are limited and the situation is complex, samples should be taken first at the point considered to give rise to the worst conditions. Only if this gives cause for concern should there be a need for more extensive sampling.

It is important when evaluating physicochemical hazards that the risks are not overestimated, in relation to risks from other hazards, such as drowning or microbiological contamination, which will be almost invariably much greater.

In areas that are used or proposed for bathing, it is suggested that physico-chemical variables such as pH, salinity, aesthetics, clarity, turbidity, colour, oil and grease, inorganic and organic chemicals are considered. Analytical methods and the minimum sample volumes to be taken for these variables are fully detailed in Bartram and Ballance (1996).

11.5 Elements of good practice

• Monitoring for other locally important hazards is justified only when it is suspected that hazards to human health may be significant. Such occurrences may be highly localised.

• Analyses should only be undertaken where standard, replicable and reliable methods are available for known variables.

• Approaches to the assessment of the significance of locally important hazards depend on the type of hazard and should take account of the magnitude and frequency of the hazards, severity and occurrence of health effects and other local factors.

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Chapter 12*: AESTHETIC ASPECTS

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A clean beach is one of the most important characteristics of a waterside resort sought by visitors (Oldridge, 1992; Morgan *et al.,* 1993). Accumulations of coastal debris raise a number of concerns: risks to marine wildlife, potential human health hazards and threats to the economy of coastal communities especially in tourist areas. In extreme cases people may avoid visiting an area if it is littered with potentially hazardous and unaesthetic items such as sanitary and medical waste. Beach quality can be viewed from two perspectives:

- It is the responsibility of the receiving area to ensure clean beaches and water.

- It is the responsibility of the user to behave in an appropriate manner and to avoid spoiling the beach with litter.

Aesthetics does not deal with a health burden directly but affects well being and health gain. The effects of aesthetic issues on the amenity value of marine and riverine environments have been defined by the World Health Organization (WHO) as: loss of tourist days; resultant damage to leisure/tourism infrastructure; damage to commercial activities dependent on tourism; damage to fishery activities and fishery-dependent activities; and damage to the local, national and international image of a resort (Philipp, 1993). Such effects were experienced in New Jersey, USA in 1987 and Long Island, USA in 1988 where the reporting of medical waste, such as syringes, vials and plastic catheters, along the coastline resulted in an estimated loss of between 37 and 121 million user days at the beach and between US$ 1.3 × 10^9 and US$ 5.4 × 10^9 in tourism-related expenditure (Valle-Levinson and Swanson, 1991).

The robustness of scientific techniques used in litter analysis is of varying quality and methodologies involved for any beach aesthetic programme must be comparable, have a quantitative basis and, more importantly, be easily understood by the end user. The reduction of beach litter for visual, olfactory and health reasons should be a paramount aim for society. Ideally litter should be cut off at source, but in reality this has been found difficult. Fundamental to this aim are universal education programmes.

Box 12.1 Public perception of microbiological quality and aesthetic aspects

A study was carried out in 1987 in the UK to investigate the public perceptions of beach and sea pollution with particular reference to the perception of bathing water quality. Samples were taken from two holiday resorts. On the basis of pre-existing microbiological evidence, the two resorts were chosen so they would have contrasting levels of measured sea pollution.

Interviewers were instructed to select respondents of a wide variety of ages and apparent social classes, recruiting approximately equivalent numbers of men and women on or near the beach in each of the two resorts. The interview schedule was designed to elicit the public's perception of beach and sea pollution, their perception of the quality of bathing water and their reporting of any of a list of symptoms. Respondents were also asked about their, and their children's (if applicable), swimming and other water-related activities. Sampling took place over an eight-week sampling period during the summer months in 1987. All interviews took place on the beach or in the immediate surroundings.

- The microbiological results for Resort 1 indicated higher levels of microbiological contamination than at Resort 2.

- The sea at Resort 1 was more likely to be seen as discoloured, dirty, cloudy, having film, oil or slime than at Resort 2.

- The frequency of reported debris in both the sea and beach was significantly greater at Resort 1.

- There was a higher incidence of discarded food or drink containers reported on the beach than in the sea.

Swimmers and non-swimmers at Resort 1 showed a significantly different percentage of holidaymakers reporting symptoms of illness such as stomach upsets, nausea, diarrhoea or headaches compared with holidaymakers at Resort 2.

Source: University of Surrey, 1987

12.1 Beach litter visual triggers

The presence of clear water does not guarantee that the water is uncontaminated and free from pathogens but the presence of certain items on a beach may however, imply poor microbiological water quality (University of Surrey, 1987) (Box 12.1). Equally, beaches free from litter do not imply that the sanitary quality of the sand is good (Mendes *et al.,* 1997). The general public usually infer that a highly littered beach has poor water quality and it is logical to assume that people prefer to visit a clean beach rather than a dirty beach (Rees and Pond, 1995a). It has been reported by WHO, that *"Good health and well-being require a clean and harmonious environment in which physical, psychological, social and aesthetic factors are all given due importance"* (WHO, 1989 p. 5).

Marine litter is defined as *"solid materials of human origin that are discarded at sea or reach the sea through waterways or domestic or industrial outfall"* (NAS, 1975 p. 104). However, the question remains whether a single item of sanitary waste on a beach necessarily means that a beach is dirty or, alternatively, how many condoms, sanitary towels or metal cans it takes to make a beach aesthetically displeasing. Aesthetics is defined by Collins Concise Dictionary (1995 p. 19) as relating to *"(a) pure beauty rather than to other consideration, (b) relating to good taste".* It relates to personal preferences, which in turn encompass things perceptible by the senses (sight, smell, taste, touch and hearing), gender, socio-economic status, psychological profile, climate, "sense of well being", age, culture, and whether the observer or user is local or a tourist (Dinius, 1981; Williams, 1986; Oldridge, 1992; Morgan *et al.,* 1993; Bonaiuto *et al.,* 1996). Certain aspects of aesthetic pollution have a greater impact on the public than others and it has been suggested that a weighting of importance should be placed on the determinands so that an overall aesthetic index could be established (NRA, 1996).

The perception of the beach user should be taken into account in award schemes (see Chapter 6) of which many exist (Williams and Morgan, 1995). Cognisance of such perception is sadly lacking in all current award schemes (see Chapter 6). Perception by the general public of the beach aesthetic appearance and water quality has become increasingly important (David, 1971; Williams, 1986; House, 1993; NRA, 1996; Williams and Nelson, 1997). The problems of beach litter are being tackled with respect to the physical (Williams and Simmons, 1997a,b) and psychological well-being of the consumer (Williams and Nelson, 1997). Emphasis is being applied increasingly to development of aesthetic health indicators which will aid in the implementation of planning measures to deal with beach health hazards (Philipp *et al.,* 1997). The presence of sewage-related debris (SRD) and medical items tend to evoke stronger feelings of unpleasantness with respect to beach aesthetics than items such as cans or plastic bottles but the tolerance level on a world-wide basis has yet to be quantified. The former items attract media attention because of the potential health risks associated with stepping on syringes, ingesting SRD or other contaminated material (Walker, 1991; Rees and Pond, 1995a). Herring and House (1990) concluded that sewage-derived debris had a greater social impact than any other aesthetic pollution environmental parameter. Williams and Nelson (1997) (Box 12.2) showed that the general public are more affected by mixtures of generic debris categories (e.g. cans, bottles and SRD such as condoms and sanitary towel backings), and it appears that females are more sensitive to beach debris (in particular SRD) than males (which could be due to a higher recognition of these particular items). It has been stated by the UK House of Commons Committee that *"while the risk of infection by serious disease is small, the visible presence of faecal and other offensive materials carried by the sewerage system can mean serious loss of amenity and is therefore an unacceptable form of pollution"* (HCEC, 1990 p. xvii).

Box 12.2 Public perception analysis at Barry Island, South Wales, UK

Public perception to litter was investigated by questionnaire during August 1995 and 1996 at Barry Island beach, South Wales, UK. Results showed that beach users were acutely aware of land-based and marine coastal pollution. A high percentage of beach users (69 per cent) thought the water to be polluted and a large percentage reported a list of litter items as being present on the beach including food packaging (83 per cent) and excrement (27 per cent).

The most prominent items of debris noted on the beach at the end of the day were food packaging, plastic bottles, aluminium cans, excrement and hygiene items. A composition of general litter and sewage-related debris was found to be the more offensive than individual generic items. The most sensitive groups of people to beach litter were females, people in the age range 30-39, and local people when compared with visitors who travelled greater than 10 miles to their destination.

A high degree of concern about the water condition was expressed by the public and a large number, 69 per cent, decided not to enter the water because they believed it to be polluted. Chi-square analysis at the 0.05 level showed females, and also people in the age category 30-40 years old, to be more sensitive to perception of pollution. However, parents still chose to visit the beach for the sand and amenity value without allowing their children into the water. Water quality was the main reason for not swimming (55 per cent), followed by temperature (23 per cent). Floating objects were considered to be the most obtrusive forms of marine debris by 53 per cent of the respondents. Such objects included anything from food packaging and hygiene items to faecal matter. The colour of the water was reported to be unfavourable by 21 per cent of those surveyed, while 14 per cent of respondents commented that the water had a "foul smell" and oil was perceived to be a problem by 12 per cent.

Source: Williams and Nelson, 1997

Dinius (1981) found that water discoloration was a factor that led respondents to make a judgement about the level of pollution of an area. Any visually unpleasant pollutant has the potential to have a negative impact on tourism, whether or not it poses an actual health risk. The aesthetic quality of the Mediterranean has been affected where eutrophication and algal blooms have occurred. There is also evidence of nutrient enrichment in the Baltic Sea, Kattegat, Skagerrak, Dutch Wadden Sea, North Sea and Black Sea (Saliba, 1995). Izmir Bay, Turkey, has been suffering red algal tides and, in 1993, pollution-related illness caused an estimated 10,000 lost working days amongst local swimmers and fishermen using the Bay (Pearce, 1995). Eutrophication has been reported as a problem along virtually every country bordering the Mediterranean. One of the consistently worst affected areas is the northern Adriatic where algae affecting areas of sea water up to 50 km^2 have been reported (Pearce, 1995).

One model for aesthetic standards defines the aesthetic value of recreational waters as (MNWH, 1992):

• Absence of visible materials that may settle to form objectionable deposits; absence of floating debris, oil, scum and other matter.

• Absence of substances producing objectionable colour, odour, taste and turbidity.

• Absence of substances and conditions (or combinations) which produce undesirable aquatic life.

It is imperative that future beach management plans consider the beach users perception of the coastline. Although poor visual appearance of the beach does not necessarily infer danger to health, results of other surveys (see Box 12.1) strongly suggested a link between the presence of certain items of debris and higher bacterial counts in water.

There are a number of human health risks posed by marine debris. Injuries caused by marine debris include entanglement of scuba divers (Cottingham. 1989), cuts caused by broken glass and discarded ring tabs from cans, skin punctures from abandoned syringes and exposure to chemicals from leaking containers washed ashore (Dixon and Dixon, 1981). In addition, munitions and pyrotechnics such as smoke and flame markers have been recovered on beaches (Dixon, 1992). Fishermen and those involved in dredging operations are at particular risk from such items although there are numerous reports of injuries to holidaymakers who have inadvertently picked up such items (Dixon, 1992).

Horsnell (1977) has documented the actual safety hazards arising from individual or small numbers of individual chemical packages. Studies by Dixon (1992) have shown a 63 per cent reduction in dangerous or harmful substances in England and Wales between 1982 and 1992. The reduction was most marked for flammable liquids, oxidising substances and corrosives. Koops (1988) analysed chemical cargoes lost off the Dutch coast that included the gases ethylene oxide and chlorine and the corrosive, sulphur dichloride.

Less obvious health risks are posed by items of SRD and medical waste. Clinical waste represents the potential vectors of infectious diseases such as Hepatitis B and Human Immunodeficiency virus (Walker, 1991). In addition, other visible pollutants, such as discarded food, dead animals, oil, containers and tyres, commonly found along the coastline have been associated with microbiological hazards (Philipp, 1991). Where visible litter is present there are also likely to be high counts of *Escherichia coli* (Philipp, 1991) which are commonly associated with human faecal material. Long-term monitoring of marine debris can therefore become an important part of the process to identify suitable indicators (Pond, 1996).

12.2 Litter survey techniques

There are a number of uses for data gathered by beach survey, including the application of the data to assess the effectiveness of remedial measures; appraisal prior to management programmes; tourism guides (e.g. MCS, 1996); as part of an integrated coastal zone management programme; identification of health hazards and/or particular threats; identification of trends; raising public awareness through public involvement; investigations for identifying the source of the litter, ageing litter and identifying the dynamics of litter in the environment. In all cases the data collected must be of quality suitable for the purpose and, where comparisons are to be made, the data must be standardised. The use of photography as the basis for routine comparisons, training and communication may be important, particularly because the perception of litter is a visual and aesthetic process. Education has a major role to play in respect to the above, both at the formal and informal level.

Litter surveys are conducted to assess types, amounts, distribution and source of litter (Rees and Pond, 1995b) and in turn to assess the effectiveness of legislation. Human health, litter and tourism are intimately connected issues and surveys, such as enumerated below, are needed in order to monitor progress in cleaning up litter (or the lack of it) through time.

Monitoring parameters, sampling stations and sites and sampling frequencies, should all be considered when establishing beach quality monitoring programmes. The problems associated with microbiological sampling of seawater have been well documented (Fleisher, 1985, 1990; Jones *et al.,* 1990; EC, 1995; Rees, 1997) and a number of these factors will apply to aesthetic quality sampling, namely variation between analysts, methods, culture mediums, choice of sampling location, number of transects from which samples are taken, number of sampling points on any stretch of beach, time of sampling (spring to neap tides), frequency of sampling, as well as wind, tide, currents and sunlight. All of these can contribute to inconsistency in results (see Chapters 2 and 9) which raises the question of whether it is possible to take a representative sample. Some of the factors above could certainly apply to beach quality monitoring and it is uncertain whether existing award schemes (see Chapter 6) show realistic representations on which to base any quality assessments.

12.2.1 Survey objectives

Of particular importance is the identification of realistic objectives that must be clearly stated and understood before the survey begins. The objectives of any litter study will define the timing of the sampling period. However, all surveys should encompass varying seasons in order to obtain a representative sample. Baseline studies (to identify the types of material found) are generally carried out over large geographical areas using a low sampling frequency. Assessment studies (to identify density of debris and changes over time) are usually carried out over more intense sampling periods and in smaller geographical areas. Temporal changes, physical characteristics of a beach, tidal patterns and use of the beach can have dramatic influences on the composition of debris found at any one time. It is therefore important that the programme design suits the study aims. Resources may also be a factor determining survey timings. Where sampling is carried out for health reasons it may be desirable to survey throughout the year but the availability of resources may restrict sampling to the bathing season.

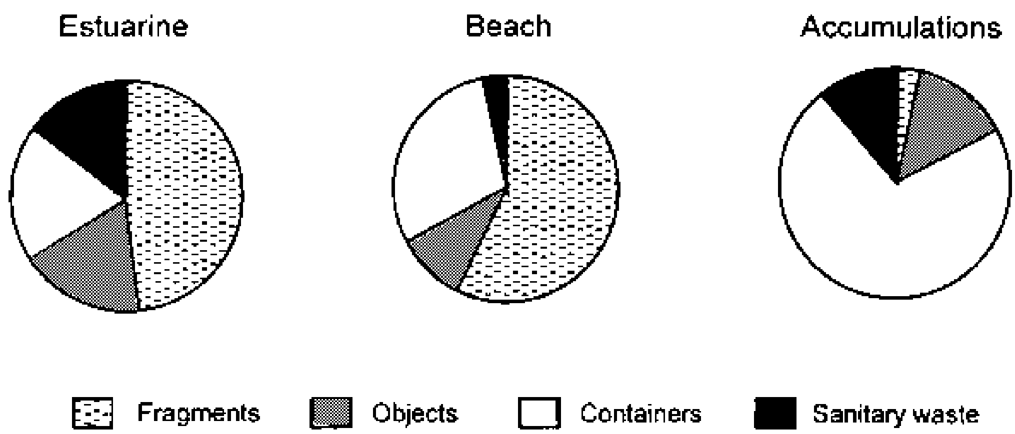
12.2.2 Methods

Surveys can be focused on the beaches, seas or rivers where beach debris is used as an indicator of oceanic, riverine, estuarine or lake conditions. One of the earliest litter surveys to be undertaken was by Garber (1960) and this approach has been used by others (NRA, 1992). The main disadvantage of the method used by Garber (1960) was its subjectiveness. For example, only presence or absence of certain visual characteristics relating to water quality was recorded in section A of the official form whereas in section B scales ranged from "absence" to an amount that was sufficient to be objectionable. No definitions of these categories were given.

Surveys can be on a small scale, such as that by Gilligan *et al.* (1992) in Chatham County, Georgia, USA where four types of site were selected to obtain samples representative of tidal influence; or they can be large scale such as those carried out by the Coastwatch Europe network (Dubsky, 1995) and the Tidy Britain Group (TBG) in the UK (Dixon and Hawksley, 1980).

A number of guides and reviews exist to help survey design (Gilbert, 1987; Ribic *et al.,* 1992; Rees and Pond, 1995b; Earll, 1996). To date, it has been inappropriate to apply a standardised methodology to assess riverine and marine debris, due to the different objectives of the surveys and the diverse nature of coastlines world-wide (Faris and Hart, 1995; Verlander and Mocogni, 1996). Faris and Hart (1995) concluded that monitoring studies can be carried out in a variety of ways provided standardised sampling protocols are established at the beginning and basic requirements are followed. Study objectives will determine the ultimate project design. Studies may be simple enumeration studies, assessing types and litter quantities, or they can be more detailed indicating age and origin of items. They can cover large geographical areas or they can relate to detailed information about specific regions or places (Williams and Simmons, 1997a). The time element, personnel needs and the costs are restricting factors. Details, such as site description, map reference, category definition, wave, wind, current patterns, site topography, physical characteristics of the beach, measurement units, survey frequency and date of survey, all need to be identified and recorded.

Figure 12.1 Plastic litter found at Merthyr Mawr beach, South Wales, UK (After Williams and Simmons, 1997a)



It is important to recognise that undertaking a beach survey can be hazardous. In addition to detailed instructions on how to complete the survey, special attention should be paid to safety aspects. Surveyors should wear appropriate footwear as well as gloves if it is necessary to handle the litter.

Litter can be categorised according to size (Ribic, 1990), composition (Dixon and Dixon, 1983) or weight (YRLMP, 1991). Three main methods of assessing type, amount and distribution of marine debris have been documented:

• Record solid waste generated by ships or pleasure crafts (Dixon and Dixon, 1981).

• Collect or observe litter floating in coastal waters (Cuomo *et al.,* 1988).

• Estimate litter during beach surveys (Rees and Pond, 1995b, 1996; Williams and Simmons, 1997a) (Figures 12.1 and 12.2).

All surveys should be repeated to show temporal changes in amount and type of litter and, hopefully, to determine its source, accumulation rate, standing stock, etc.

Individual items of beach marine debris are usually counted and classified or recorded as presence or absence (Pollard, 1996; Rees and Pond, 1996). The sampling size unit is a function of the survey aims. Examples include:

• The whole beach can be surveyed from splash zone to waters edge (Dubsky, 1995).

• Transects may be used of varying width. The optimum transect width is one which provides a reliable sample.

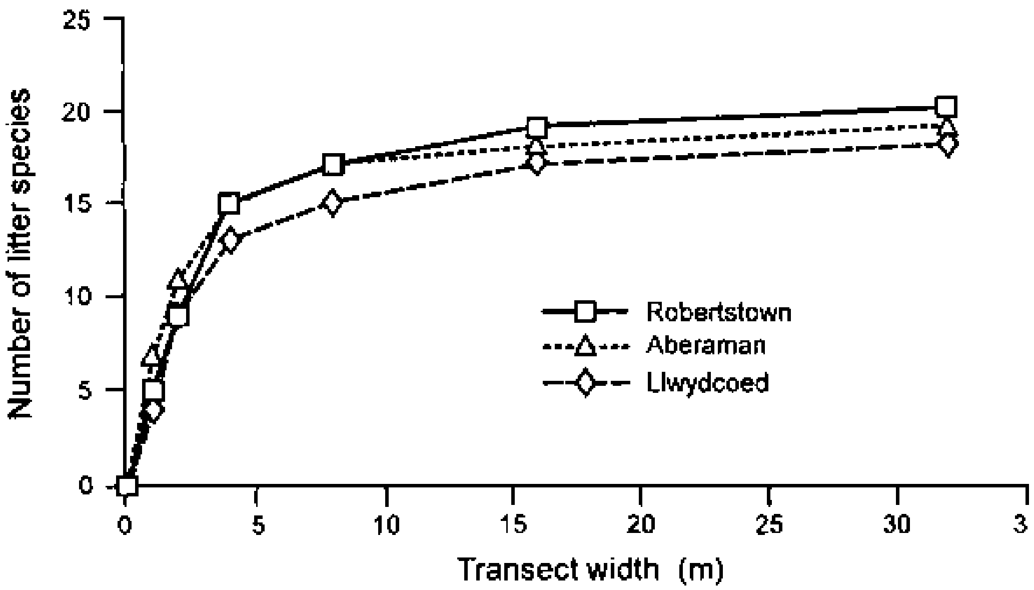
• Transect line quadrats or randomly dispersed quadrats (Dixon and Hawksley, 1980).

• Strand line counts (Williams and Simmons, 1997a).

• Postal surveys (Dixon, 1992).

• The offshore water column can be sampled (Williams *et al.,* 1993).

Figure 12.2 A minimal area curve for beach sites on the River Cynon, South Wales, UK (After Earll *et al*., 1997)



The advantages and disadvantages of various methods of litter surveys are shown in Table 12.1. Survey design and methodological development are considered to be of paramount importance and many statisticians and environmental scientists have provided guidelines to aid formulation of sound survey designs (Gilbert, 1987; Ribic *et al.,* 1992). Common to each approach is an emphasis on formulation of realistic objectives that must be stated and clearly understood before work can progress. No precedent exists regarding the optimum type of data, i.e. qualitative or quantitative. Two approaches to aesthetic surveys are described below: transect surveys and questionnaire surveys.

Transect surveys

Gilbert (1987, p. 7) stated, *"the target population is the set of N population units about which inferences will be made. The sampled population is the set of population units directly available for measurement".* The target population must be limited to litter at sites deemed accessible for sampling purposes, e.g. litter on riverbank sites where both banks can be sampled up to the bank-full position (highest possible water level), beach strandlines, etc. Due to logistical problems of assessing all litter at a site, representative sampling units are needed to provide an accurate portrayal of the whole site. For rivers and beaches, a series of continuous quadrats can be laid starting at the water's edge and finishing at the natural limit of the bank or beach (sites chosen with predominantly natural characteristics). Within each quadrat, litter abundance can be measured in the form of density counts, i.e. the number of individuals of particular litter types within a quadrat. Not every litter type exists on any one particular river or beach.

Table 12.1 The principle advantages and disadvantages of various methods of litter survey

Method	Advantages	Disadvantages
Five strand lines, excluding the vegetation line	Covers a large area of beach where items may accumulate in algae or as mats of debris left as the tide recedes	Can give biased results as the areas between the strand lines are not surveyed and the method only counts the most recent tidal borne debris (1) Only covers surface litter; some litter may be buried (2) It may be difficult to identify strand lines; these vary daily and seasonally (3) Not suitable for beaches with large boulders (4)
Five strand lines, plus the vegetation line	Area covered includes a good cross section; both accumulated and fresh litter is surveyed	As above (1-3)
Top, bottom and vegetation lines	Easy to use; quicker than the above methods	As for the first method (1-3)
Five metre wide strip transect	Covers a large area of beach where items may accumulate in algae or as mats of debris left as the tide recedes	As for the first method (1-3)
One metre wide strip transect	Covers a large area of beach where items may accumulate in algae or as mats of debris left as the tide recedes	As for the first method (1-3)

Random 2×2 quadrats	Fast method of sampling; sampling is not influenced by location of litter and is therefore statistically valid	Results may be variable and depend on the amount of litter present
Random dispersed quadrats	Fast; economic	Possibility of missing litter clumping
Whole beach	Covers all sections of the beach; avoids bias	Time consuming; care needs to be taken not to miss items Best suited to pocket beaches
Postal surveys	Can cover a large geographical area	May be relatively high percentage of non-response

A useful tool adapted to determine whether transect sampling is an appropriate method for river and beach litter assessments (species) and, if found suitable, to assess the optimum transect width size, is classic minimal area analyses (also known as species area curves) derived from the Braun-Blanquet school of phytosociology (Braun-Blanquet 1921, 1932; Cain, 1932, 1935; Gilbertson *et al*., 1985).

Narrow belt transects are more easily studied and enable work to be achieved quicker, but wider transects probably yield more reliable data (Burnham *et al.,* 1980). Therefore, the optimum transect width is one which provides a reliable representation of the litter present, for the minimum amount of work. For example, for investigating riverine litter (Figure 12.2), starting from the site's centre point, a tape is placed up the river bank perpendicular to the river flow. A second tape is then placed parallel to the first, at the smallest possible distance apart (in this case approximately 10 cm). The number of litter types is counted and recorded. The exact initial distance decided upon is unimportant, provided it is small enough to contain only one or two items, because recordings are made in relation to a doubling of transect width and not as a function of the exact width measurement. The transect width is doubled and the number of litter types present counted. The doubling and counting procedure is repeated until the number of litter types at each doubling of the transect width has shown no further increase. The resultant data curve starts to level off at the point that resembles the minimal width necessary to obtain representative samples.

Figure 12.2 shows how three different sites produced similar curves, with the curve gradient indicating the number of litter types found, and the curve beginning to level off after 5 m transect widths. On an objective basis it is very difficult to determine the exact optimum transect width. At a 5 m transect width some 13-15 litter types were identified; but at 15 m width 15-17 types were identified. Detailed field work showed that 20 litter types were present at these sites, i.e. 5 m transect widths covered some 65-75 per cent of the litter present whereas 15 m widths covered some 75-85 per cent. The 5 m transect width has been used in many litter surveys (Dixon and Dixon, 1981; Davies, 1989) but there is no clear scientific justification for this. On applying a pre-specified relative error (Gilbert, 1987), results indicated that any between-site comparisons should only be carried out using litter types known to have a more uniform within-site distribution. Commonly occurring litter types, such as plastic sheeting and sewage-derived articles, could be represented realistically using only three transects. Other litter types needed 64 transects, e.g. packing crates. It is meaningless to compare sites of different litter types, because the within-site variation can be greater than the differences between two sites.

Questionnaire surveys

An alternative approach to transect sampling is to survey the entire beach area. This approach has been followed by the Coastwatch Europe network and has been described in detail by Pond (1996) and Rees and Pond (1995b; 1996). Essentially, the study area is divided into manageable units of 0.5 km in length. The method uses standard questionnaires, translated as necessary (see Rees and Pond, 1996). All surveyor groups are provided with detailed instructions on how to complete the survey, including which items of litter should be included in each category, detailed safety notes and the contact telephone number of a national and local co-ordinator. The survey is conducted over a common time period so that the results can be compared between participating countries and between sites within countries. A six-figure map reference of each unit of coastline is recorded and stored on a database in order to ensure that the same units are surveyed in subsequent surveys.

The survey is conducted as soon as possible after high tide. Surveyors are asked to walk along the intertidal area and to return along the splash zone recording the presence or absence of 17 "general" litter categories, such as sewage-related debris, cans, plastic bottles, etc. and nine "major items of debris", such as household refuse. Quantities of some items of litter are also requested. Surveyors are also asked to record potential threats to the area, to investigate the aesthetic quality of inflows (streams and rivers) and to record other information regarding management aspects of the coastal unit. Once the questionnaire has been completed surveyors return it to a national co-ordinating office for data analysis and report writing.

This approach has a number of advantages: the method is simple to use and can be undertaken by relatively inexperienced surveyors under instruction (see section 12.2.4) and the questionnaire can be adapted to focus on particular areas of interest, for example the Coastwatch Europe network has developed a section of the questionnaire to focus on medical and sanitary waste. It can also be adapted to collect qualitative data. Large areas of coastline can be covered, thus making the sample more representative. The method is also economical and does not require any special equipment or knowledge and can be undertaken in all weather conditions. The main disadvantage is that the method is time-consuming.

12.2.3 Qualitative versus quantitative data analysis

Qualitative data

The problems associated with the techniques used to assess litter and the resulting statistical analyses are comparable with those experienced by ecologists (Ludwig and Goodall, 1978; Ludwig and Reynolds, 1988). Qualitative data can give quick assessments (Hubalek, 1982). Many different pattern types can exist within communities, including spatial dispersion of litter types (species) "within" a site, and relationships "between" sites.

Ludwig and Reynolds (1988) recommended the variance ratio (VR) test of Schluter (1984) to measure association for more than a single pair of litter types. The Null Hypothesis (H_0) is that no association exists between litter types and the expected VR is 1. If an association occurs, then it is either positive (VR < 1), i.e. the pair of litter types occurred together more often than expected if independent; or negative (VR > 1), i.e. the pair of litter types occurred together less often than expected if independent. Litter types may show no association if independent, or when positive and negative associations between litter types cancel out each other.

The chi-square test can detect pair-wise associations of litter types, with H_0 indicating that the litter types were independent. In riverine litter examples quoted by Simmons and Williams (1997) a χ^2_t > 3.84 at the 95 per cent level rejected H_0, and 31 litter pairs were significantly associated. In the context of ecological monitoring, three qualitative (present/absent) binary techniques are common i.e. Ochai, Jaccard and Dice. Jaccard's technique is particularly robust and when using this technique Simmons and Williams (1997) showed that within-site litter transects were generally no more strongly associated than those between-sites. From these qualitative results it appears that within-site litter variations can be as great as between-site variations. If this is the case the representativeness of transects for each site may be questioned because the results from one transect could be dramatically different from another transect, even at the same site. In the Simmons and Williams (1997) study, no strong associations were apparent between sites; the highest index value reached 0.7 with the majority of indices < 0.5. It appears that either significant differences in litter patterns did not exist between sites, or that the sample size was too small to show differences, or that the statistical test was not able to detect the differences.

Several major limitations negate the benefits of collecting by qualitative data alone. A lack of data versatility is a major problem, with few statistical analyses being appropriate. Even the statistical packages available (Ludwig and Reynolds, 1988) require very time-consuming data manipulation to carry out relatively simple calculations. If data are being compiled for several river catchments or marine sites, it is felt that data manipulation problems alone make qualitative analysis an unfavourable proposition.

Quantitative data

Quantitative data makes it amenable to a broader spectrum of analysis methods giving greater versatility. Three basic patterns may be recognised in litter communities, random, clumped and uniform, and the mean, variance and pattern of individuals within a quadrat are quite different between these patterns. Once a pattern has been identified, a test must be proposed concerning the community structure. Initially, it is important to determine if sampling units are discrete (natural) or continuous (arbitrary) because this influences the type of spatial pattern analysis. Based on the continuous nature of sampling units, the quadrat variance method of Ludwig and Goodall (1978) can be undertaken enabling spatial patterns to be observed by sampling a series of continuous quadrats. Data may be collected at all litter sites by a series of 1 m^2 quadrats extending up a river bank or along a beach transect. Quadrat-variance methods are based on examining the changes in the mean and variance of the number of individuals per sampling unit, for a range of sampling units.

Table 12.2 Significantly correlated litter pairs

Correlated litter pairs	Level of significance[1]
Sanitary towel: parity liner	0.000
Panty liner: tampon	0.019
Sanitary towel: tampon	0.007
Plastic sheeting < 30 cm: plastic sheeting 30-60 cm	0.006
Plastic sheeting < 30 cm: plastic sheeting > 60 cm	0.030
Plastic sheeting 30-60 cm: plastic sheeting > 60 cm	0.000

[1] Equivalent to the probability of the correlation arising purely by chance

Source: Simmons and Williams, 1997

Two types of quadrat variance methods can be used: paired-quadrat variance (PQV) and two-term local quadrat variance (TTLQV). The former uses (PQV) changes in quadrat spacing to provide spatial pattern information, whilst the latter (TTLQV) uses changes in quadrat size, through the blocking or combining of adjacent quadrats, to determine pattern intensity and range of densities present. Quantitative analyses can be done using the SPSSx® statistical software package (Norusis, 1983). If results from individual quadrats are combined to produce data representing a 1 m wide belt transect, the data could still be used to indicate whether certain statistical tests would be of future use. Data limitations can be due to small sampling areas and the use of only one transect to represent a site.

The normality of litter data sets should be tested using the Kolmogorov-Smirnov one sample non-parametric test (Miller and Miller, 1988), followed by finding the covariation between litter types, e.g. using the Spearman rank correlation. For example, Simmons and Williams (1997) found correlations between sanitary towels, panty liners and tampon applicators (Table 12.2). Plastic sheeting appeared to be correlated with other plastic sheeting of differing sizes, but not with sewage-derived litter (as indicated in the qualitative litter association analyses described above). This result may highlight one of the problems of using qualitative data in this sort of survey. Associations between plastic sheeting and sewage debris may have been calculated because of their common occurrence at sites. Associations shown by qualitative data may have led to the hypothesis that plastics were introduced to the system from the same sources as the sewage-derived litter, hence their association. However, it appears that although both items are present at the same site, their abundance is not correlated significantly. Major limitations of this type of analysis appear to arise from the number of zeros recorded in the data set; consequently large data sets are needed. A second problem is the realistic interpretation of results. When an expanse of coefficients has been calculated, multivariate methods of pattern recognition, such as cluster and principal component analysis, can and should be used (Derde and Massart, 1983).

Cluster analysis may be used to place similar objects or variables into groups or "clusters", in order to produce a hierarchical tree-like structure known as a dendogram. Dendograms demonstrate graphically, in two dimensions, similarities between variables by the varying distances from the x-axis at which the groups are formed. The closer a group is formed to the x-axis, the stronger the similarities between its constituent parts

(Simmons and Williams, 1997). The cluster analysis approach appears to be a very useful tool for indicating patterns within a data set and reduction of the numbers of zeros recorded is the main key.

Principal Component Analysis (PCA) is an alternative method of pattern recognition that aims to identify principal components that explain correlations among a set of variables (litter types, see Table 12.3). The method condenses information on litter types from many dimensions (N sites) to two or three dimensions that may be more easily interpreted. In addition, it calculates "loadings" to indicate the significance of each of the variables in determining the data structure. The higher the loading, the greater the importance the variable has in determining that component. In Figure 12.3, the first three factors accounted for 20, 16, and 11 per cent of the data variation respectively (Simmons and Williams, 1997).

12.2.4 Volunteers versus "professionals"

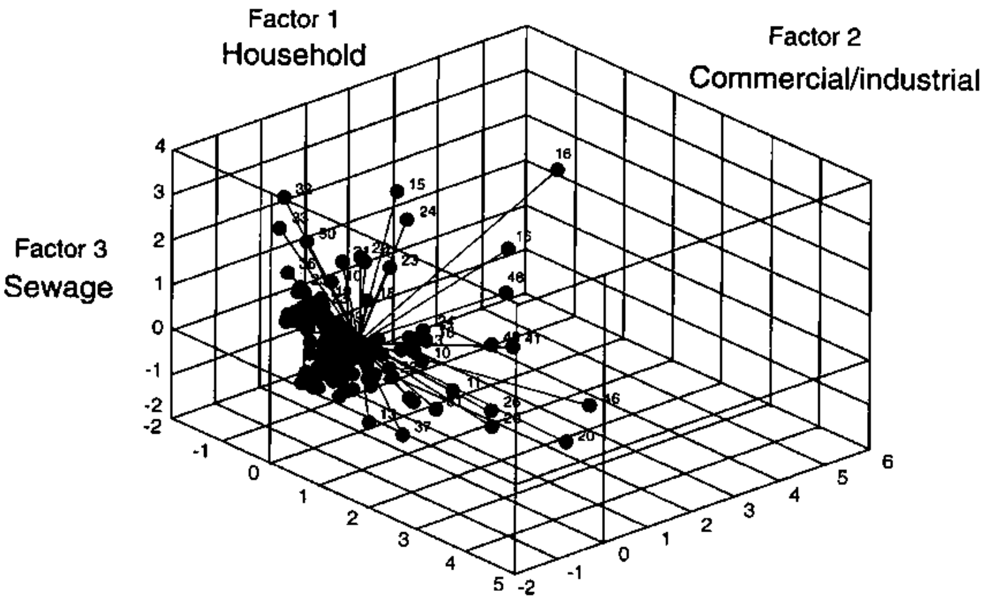
There has been considerable debate about who should conduct litter surveys (Dixon, 1992; Amos *et al.,* 1995; Pollard, 1996; Rees and Pond, 1996). The use of volunteers was discussed at length at the Third International Conference of Marine Debris in 1995 (Paris and Hart, 1995). No conclusions were made, but the Conference recommended that where volunteers are used, clear instructions must be given and good quality assurance procedures must be established. There are both advantages and disadvantages in this approach. The use of volunteers to conduct litter surveys means that a large sample size can be achieved at low cost. However, concerns exist that reporting rates between groups may not be consistent. Trials have shown that volunteers frequently identify litter items incorrectly (Dixon, 1992). This has been investigated recently through the Coastwatch UK programme and it was found that these concerns were largely unfounded (Pond, 1996).

Table 12.3 A litter identification key

Source	Category	Type of litter
Sewage derived	Feminine hygiene	Sanitary towels Panty liners Tampon/applicator
	General	Toilet paper Cotton buds Other/unidentified
Housing materials	Combustible	Fencing Hardboard/wood Other/unidentified
	Non-combustible	Brick/rubble Floor coverings Other/unidentified
Household (large)	Brown goods	Furniture Mattress/foam
	White goods	Other/unidentified
Household (small)	Metal	Cans/tins

Commercial/industrial	Metal	Container drums Sheeting Other/unidentified
	Plastic	Polystyrene Sheeting < 30 cm Sheeting 30-60 cm Sheeting > 60 cm Plastic bags Sweet papers Bottles
	Glass	Bottles Other/unidentified
Transport-associated	Motor vehicles	Cars/parts Motorbikes/parts Other/unidentified
	General	Signs/cones
General	Packaging	Cardboard
	Miscellaneous	Cloth/shoes Rope/fishing line Other/unidentified

Figure 12.3 Principal Component Analysis of litter sites along the River Cynon in summer and winter. Numbers represent litter types described in Table 12.3



12.2.5 Beach quality questionnaires

Beach quality questionnaires should be objective. Several rating systems are based on a limited number of parameters (Table 12.4) and it should be an axiom that ratings must cover physical, human and biological parameters. Nevertheless, many existing systems have been found wanting in this respect. Virtually all the following do not take into consideration the beach user's perception of his or her environment.

The majority of beach quality schemes look at one or only a few of the parameters associated with beach ratings (Table 12.4). Beach aesthetics cannot be rated effectively on one facet alone, e.g. biological parameters. Table 12.4 shows a summary of the scope of a variety of beach awards and rating systems currently in place. Chapter 6 deals with beach award schemes in greater detail.

Table 12.4 An overview of the scope of selected beach awards and rating systems

Component	European Blue Flag	Seaside Award (TBG)	Good Beach Guide (MCS)	NRA (sw region)	Chaverri, 1989	Williams *et al.*, 1993	Beach Quality Rating Scale
Water quality	*	*	*		*	*	*
Education and information	*	*					
Access	*	*			*	*	*
Lifeguards/first aid	*	*				*	*
Litter	*	*			*	*	*
Sanitation	*	*					*
Sewage debris	*	*	*	*	*	*	*
Bathing water safety					*	*	*
Climate					*	*	*
Landscape quality					*	*	*
Beach material					*	*	*
Water temperature					*	*	*
Flora and fauna					*	*	*
Refreshments and facilities	Some	Some				Some	*
Beach regulation (dogs, vehicles, etc.)	*	*				*	*
Weighting of factors							*
Scoring based on preferences priorities of beach users							*
Quantification of most or all factors			*	*		*	
Difference between resort/undeveloped beaches		*					*

TBG Tidy Britain Group
MCS Marine Conservation Society
NRA National Rivers Authority (south-west region)

12.3 Beach cleaning

Increasingly, environmental management systems are being used to assess the routine performance of management approaches to the environment. Sequences of planning, objective setting, implementation, audit and review are becoming commonplace. The audit process for such systems often requires field measurements to assess whether systems are working. Monitoring in terms of "cleanup" is often the response to litter. The cleaning of beaches provides a way of collecting data on the types and quantities of marine debris. However, the primary value of these methods is as public participation exercises and as a way of raising public awareness. The cleaning of beaches cannot solve the problem of marine debris permanently because they do not reduce the quantity of debris at source (Simmons and Williams, 1993). Physical "cleanups" are generally carried out by local authorities (Gilbert, 1996), local voluntary groups or volunteers co-ordinated by national voluntary bodies (Pollard and Parr, 1997). However, cleanups are really only useful at the local level, and they are expensive if undertaken by mechanical means or else they are labour intensive. Conversely, if volunteers are employed the costs are minimal. Site selection for beach cleaning programmes is biased towards areas with easy access, tourist locations and depositing beaches. Where volunteers are used, the collection of litter is the primary task and therefore the recording of the items is likely to be less of a priority (Rees and Pond, 1995a). Amos *et al.* (1995) have shown that volunteers participating in beach cleaning programmes undercounted individual items of debris by 50 per cent.

There are two methods of beach cleaning: mechanical and manual. Mechanical cleaning usually involves motorised equipment using a sieve effect that scoops up sand and retains the litter; therefore it is not selective. Litter retention is a function of the sieve. Most sieve machines are coarse grained allowing small items to pass through. The passage of such vehicles over the beach interferes with the beach ecology and the method is costly (Davidson *et al.,* 1991; Kirby, 1992; Acland, 1994; Llewellyn and Shackley, 1996). In addition this technique cannot be used on pebble beaches. Pressure to clean a beach is intense, especially where authorities wish to promote tourism. The advantages of such mechanical cleanups are that the result is achieved quickly, and large areas can be covered and they can provide an apparently pristine beach for visitors. Mechanical cleanups reduce the need for personal contact thus reducing health risks to individuals.

Manual beach cleaning programmes share many advantages and disadvantages with mechanical cleaning. They can help to raise community awareness of the litter problem and enable the sourcing of the litter (Earll, 1996) from a scientific perspective rather than scooping it up "en mass" for deposition in a landfill site. Manual beach cleanups organised as community events on small areas can ensure that the beach is cleaned of small items missed by mechanical methods (Pollard, 1996).

12.3.1 Economic aspects

Cleaning a country's coastline costs the responsible authorities large amounts of money each year. In the UK, Suffolk District Council estimated that GB£ 50,000 was spent each year on cleaning the grounds around the coast and picking up litter from the foreshore. Authorities in Kent estimated that between GB£ 32,000 and GB£ 48,000 was being spent annually per beach and the direct and indirect costs of dealing with litter on the Kent coast has been estimated at over GB£ 11 million (Gilbert, 1996), which places a strain on the Gross National Product (GNP) of the area. Woodspring District Council reported that GB£ 100,000 was spent on managing litter and sand on just two beaches in the district of Weston Super Mare (Acland, 1994). Nevertheless this expenditure is necessary for tourist beaches.

In 1993, it cost GB£ 937,000 to clean the Bohuslan coast of Sweden (Olin *et al.,* 1994) and more than US$ 1 million was spent in 1988-89 cleaning up the coasts of Santa Monica and Long Beach in California (Kauffman and Brown, 1991). At Studland, Dorset, UK, one million visitors per year along a 6 km stretch of beach results in 12-13 tonnes of litter, collected each week during the summer months at a cost of GB£ 36,000 per annum. Additional costs are incurred when hazardous containers are found and have to be recovered from beaches (Dixon, 1992).

12.3.2 Measurable standards of cleanliness

The public perception of litter is intrinsically linked with standards of cleanliness. A number of issues become pertinent when setting standards or grades of cleanliness and these have been identified by Earll *et al.,* (1997).

- Will the public notice the standards set?

- If the public notice or recognise this material does it matter?

- At what level of littering do these issues become important to the public?

- Are the levels of litter indicative of other pollution, health and environmental hazards?

At present, it appears that very few standards of cleanliness exist regarding beach litter (see Chapter 6). It has been recognised that adequate information is required to support improvements in the cleaning of coastal waters and beaches (Anon, 1972). Coastal authorities, especially in Southern England, responded to the increasing amount of marine litter by extending the cleansing operation beyond the bathing season. The Royal Commission on Environmental Pollution (RCEP, 1984, 1985) noted the substantial costs incurred in beach cleaning operations. In the UK, the Environmental Protection Act (1990) sets standards of cleanliness under the Code of Practice issued under section 89(7) which are considered reasonable to meet. Local Authorities are encouraged to identify as "Category 5 Zone" areas, those beaches in their ownership or control that might reasonably be described as "amenity beaches". For such designated amenity beaches, the minimum standard is that they should be generally clear of all types of litter and refuse between May and September inclusive. This standard applies, not only to items discarded by beach users, but also to items or materials originating from disposal directly to the marine environment. The Code also notes that, in establishing a cleansing standard for beaches, careful consideration should be given to the practical difficulties encountered in collecting and removing litter, and to the damage to sensitive habitats which may result from such operations (DoE, 1991).

Table 12.5 Relative contribution of different sources of the marine debris found in St Brides Bay, Wales, 1997

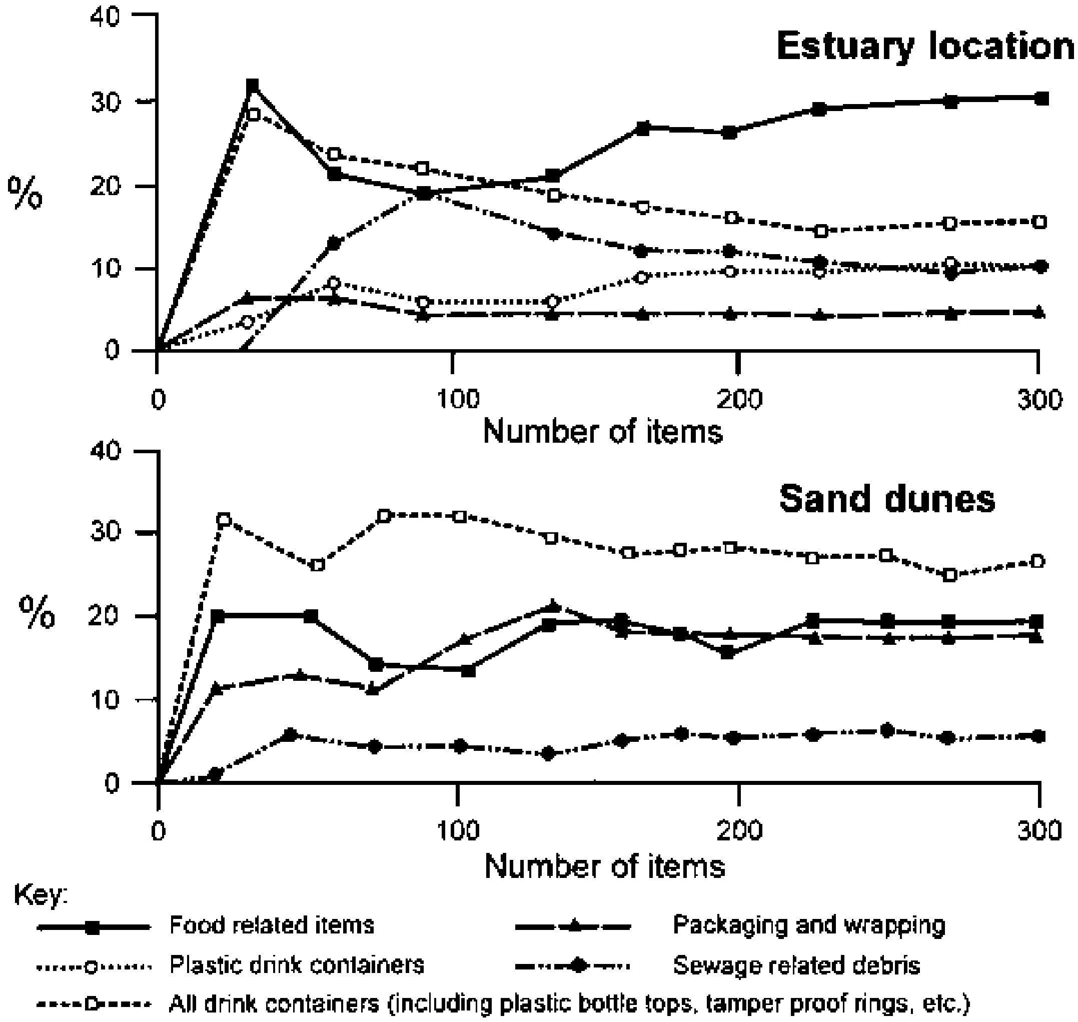
Source	Summer (% of total)	Winter (% of total)	Average over the year (% of total)
Tourism	15.7	1.8	8.8
Shipping	9.1	13.4	11.2
Sewage-related debris	2.9	3.4	3.2
Fishing	8.9	19.5	14.2
Fly tipping	None found	None found	0.0
Medical items	None found	None found	0.0
Non-sourced	63.0	61.7	62.6

12.4 Debris sourcing

An objective of collating and analysing litter is to identify the source because it is only when the source is known that really effective action can be undertaken to remedy the situation. It is essential to have robust quantitative information to enable litter types to be monitored in a systematic manner and to enable assessments and judgements to be made. Sources could be marine (ships), tourists on a beach, fly tipping or a river.

Dixon (1992) concluded that beach litter in the UK consisted mainly of waste generated by ships, sewage discharges and material discarded by the general public, and that discharges of rubbish from ships and other crafts constituted 70 per cent of litter. This is almost the opposite viewpoint to that expressed by Faris and Hart (1995) in the USA, Gabrielides *et al.* (1991) in the Mediterranean, Ross *et al.* (1991) in Canada and Pond and Rees (1994). Litter sourcing seems to be highly site specific and generalisations should be avoided. For example, work carried out at St Brides Bay, Wales in 1997 showed that fly tipping and medical waste were not sources from which litter originated (Table 12.5). Non-sourced litter accounted for 62 per cent of the litter generated. Frost and Cullen (1997) attempted to categorise debris on northern New South Wales beaches by dividing debris into that which has the potential to float and debris that sank. It was then assumed that floatable debris was marine-based and sinking debris was from land-based or *in situ* sources.

Figure 12.4 Cumulative percentage scores for litter: A. Estuary location; B. Sand dunes (After Earll *et al.,* 1997)



In Auckland City, attempts have been made to assess the scale of litter discharged from the City into the coastal marine area (Arnold, 1995). This involved monitoring material trapped by a 19 mm wire net placed on three storm-water catchments representative of each land use type. Comparison of the number of items associated with land-use types showed that industrial areas were the major source (9.69 items per hectare per day) followed by commercial (3.33 items per hectare per day) and residential areas (1.22 items per hectare per day).

An interesting approach for litter is to try to identify the people dropping the litter and to make inferences regarding their life style from the types of litter groups. This could help with direct prevention work. However, the number of items that should be collected in order to characterise life style groups still has to be resolved. One way would be to collect, for example, 200 items (in batches of 40-50 items) and to list them by function rather than by material and to carry out a similar analysis to that shown in Figure 12.2, but by plotting the number of categories against percentage occurrence and/or the number of sampled items against the number of litter categories. The common items that should take priority would show up very clearly and a long "tail" would be shown in the plot (Figure 12.4) (Earll *et al.,* 1997). The above approach could be adopted easily and could be carried out on a routine basis and in a cost-effective way. Photography would be an invaluable aid in this approach.

12.5 Elements of good practice

The following are considered to be the main elements of good practice for monitoring aesthetic parameters.

• Monitoring for specific aesthetic pollution parameters should be undertaken where hazards to human health and well being are suspected.

• Selection of aesthetic pollution parameters for monitoring should take into account local conditions and should consider parameters such as surface accumulation of tar, scums, odours, plastic, macroscopic algae or macrophytes (stranded on the beach and/or accumulated in the water) or cyanobacterial and algal scums, dead animals, sewage-related debris and medical waste.

• Sampling of aesthetic pollution indicators should take into account the perception and requirements of the local and any visiting populations in relation to specific polluting items as well as in relation to the feasibility of their monitoring.

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Chapter 13*: EPIDEMIOLOGY

* *This chapter was prepared by D. Kay and A. Dufour*

Epidemiological data are frequently used to provide a basis for public health decisions and as an aid to the regulatory process. This is certainly true when developing safeguards for recreational waters where hazardous substances or pathogens discharged to coastal and inland waters may pose a serious risk of illness to individuals who use the waters. Epidemiological studies of human populations not only provide evidence that swimming-associated illness is related to environmental exposure, but also can establish an exposure-response gradient which is essential for developing risk-based regulations. Epidemiology has played a significant role in providing information that characterises risks associated with exposure to faeces contaminated recreational waters. The use of epidemiological studies to define risk associated with swimming in contaminated waters has been criticised because the approach used to collect the data is not experimental in nature. This perception is unlikely to change, given the highly variable environments where recreational exposures take place. Although the variables may be difficult to control, it is possible to carry out credible studies by following certain standard practices that are given below. This Chapter discusses the place of epidemiological investigations in providing information to support recreational water management and the scientific basis of "health-based" regulation.

13.1 Methods employed in recreational water studies

Epidemiology is the scientific study of disease patterns in time and space. Epidemiological investigations can provide strong evidence linking disease incidence and environmental or other exposures. However, this statistical inference does not provide absolute proof of a direct cause and effect, although the combination of strong statistical association with biological plausibility offers strong evidence of causality. Epidemiological methods can quantify the probability that observed relationships occurred by "chance" factors. The methods employed can range from the study of recorded outbreaks of illness (i.e. seeking to infer the causes of morbidity from existing patterns of recorded individuals who are ill (and called "cases") and possible "controls" who are not ill), through to carefully designed studies in which volunteers are exposed to a hazard (such as faecally contaminated bathing water) and then followed up for a suitable period to investigate the incidence of illness. The type of study employed is dependent on:

- The objectives of the study, i.e. the required use of the data to be acquired.

• The nature of the exposure and illness under study.

• Available epidemiological and biostatistical expertise, together with economic constraints.

It is vital that these three elements are considered at the outset of any investigation. The first element, "objective(s) of the study", is perhaps the most important aspect because the available types of study discussed below each provide data with distinct potential uses. It is vital that the data produced by the more rudimentary epidemiological designs are not over-interpreted and that their limitations are understood clearly by the scientific and policy communities.

There have been a many epidemiological investigations for the health impacts of exposure to faecally polluted recreational waters reported in the scientific literature since 1953 (Table 13.1). These investigations fall into three main design categories described in the following subsections.

13.1.1 Retrospective case-control studies

Retrospective case-control studies are used to determine whether a particular personal characteristic or environmental factor is related to disease occurrence. Cases refer to persons who have a specific illness or disease. Controls, who do not have the illness or disease, are selected. The selection may seek to "match" for variables such as age and sex, or an unmatched design can be employed in which possible confounders are controlled during the analysis phase. Cases and controls are queried to determine if their exposure to environmental hazards have been similar. For example, cases of typhoid fever and their matched controls may be questioned about their past activity with respect to swimming events. The results of questioning may, for example, show that swimming activity is more likely to have occurred with typhoid cases than with controls, indicating a potential association between swimming and the disease. This type of study is most useful in disease outbreaks where a retrospective case-control study may be conducted to determine if certain activities or exposures were related to the disease or illness under investigation. This approach also may be useful in establishing the relationship between serious illness, such as hepatitis, and exposure to bathing waters. The advantage of conducting retrospective case-control studies is that they are not very costly and are reasonably easy to carry out. Their disadvantage is that, while the linkage between disease and exposure can be determined, it is seldom possible to determine the magnitude of the exposure.

Table 13.1 A summary of epidemiological studies

Country	Water body	Indicator	Symptom(s)	Reference
Australia	Sea	Faecal coliform Faecal streptococci	E/ENT/R	Corbett *et al*., 1993
	Sea	Faecal coliform Faecal streptococci *C. perfringens*	GI/R/O	Harrington *et al*., 1993
Canada	Fresh	Total staphylococci	R/GI	Lightfoot, 1989
	Fresh	Total staphylococci Faecal coliform Faecal streptococci	R/GI	Seyfried *et al.,* 1985a,b
Egypt	Sea	Enterococci *E. Coli*	GI	El Sharkawi and Hassan 1982
France	Fresh	Total conforms Faecal coliforms Faecal streptococci *Aeromonas spp.* *P. aeruginosa*	All + S S GI S S	Ferley *et al*., 1989
	Sea	Faecal streptococci Total coliforms Faecal coliforms	E/S/GI	Foulon *et al*., 1983
Hong Kong	Sea	*E. coli* *Klebsiella* Faecal streptococci *Enterococci* *Staphylococci* *P. aeruginosa* *Candida albicans* Total fungi	GI+S GI+S GI+S GI+S ENT GI+S O E+O	Cheung *et al*., 1990
Israel	Sea	*S. aureus* Enterococci *E. coli* Total staphylococci *P. aeruginosa*	GI GI GI	Fattal *et al.,* 1991
	Sea	Enterococci *E. coli* *S. aureus* *P. aeruginosa*	GI	Fattal *et al.,* 1986
Netherlands	Fresh	*P. aeruginosa*	ENT	Asperen *et al.,* 1995
South Africa	Sea	Faecal coliform *E. coli* Faecal streptococci Total staphylococci	GI/R/S	Schirnding *et al.,* 1993
	Sea	Faecal coliform *E. coli* Faecal streptococci Total staphylococci	GI/R/S	Schirnding *et al.,* 1992
Spain	Sea	Faecal streptococci	S/E/ENT/GI	Mujeriego *et al.,* 1982

UK	Sea	Faecal streptococci Faecal coliform	E/S/ENT/R	Fleisher *et al*., 1996
	Sea	Faecal streptococci	GI	Kay *et al*., 1994
	Fresh	Total coliforms Faecal coliforms Faecal streptococci Total staphylococci *P. aeruginosa*	E/S/ENT/R	Fewtrell *et al*., 1993
	Sea	Faecal streptococci	GI	Fleisher *et al*., 1993
	Fresh	Total coliforms Faecal coliforms Faecal streptococci Total staphylococci *P. aeruginosa* Enterovirus	E/S/ENT/R	Fewtrell *et al*., 1992
	Sea	Total coliforms Faecal coliforms Faecal streptococci Salmonella Enterovirus	ENT/GI/S/O	Alexander and Heaven, 1991
	Sea	Total coliform Faecal coliform Faecal streptococci	E/GI/ENT/R	Balarajan *et al*., 1991
	Sea	Total coliforms Faecal coliforms Faecal streptococci	E/S/ENT/R	Jones *et al*., 1991
	Sea	NR	GI	Brown *et al*., 1987
	Sea	Total coliform	O	PHLS, 1958
USA	Fresh	Enterococci *E. coli*	GI	Dufour, 1984
	Sea	Enterococci	GI	Cabelli, 1983
	Both	Total coliform	ENT/GI/R	Stevenson, 1953

E Eye symptoms
S Skin complaints
GI Gastro-intestinal symptoms
ENT Ear nose and throat symptoms
R Respiratory illness
O Other
NR Not reported

13.1.2 Prospective cohort study

The second type of investigation is a prospective cohort study. In this study, individuals are recruited immediately before or, more commonly, after participation in some form of recreational water exposure. A control group is similarly recruited and both cohorts are followed up for a period of time. The exposure status of the bathers and non-bathers is self-selected and not randomised in this type of study. During the follow-up period, data are acquired on the symptoms experienced by the two cohorts using questionnaire interviews, either in person or by means of telephone inquiry. The quality of the recreational water environment is defined through environmental sampling on the day of exposure. The exposure data are often combined to produce a "daily mean" value for the full group of bathers using a particular water on any one day. Many days of exposure are required to define adequately the relationship between "exposure day" water quality and disease. Thus, data on "exposure" are available which can be related to "illness" outcome through an exposure-response curve predicting illness from indicator bacterial concentration. However, this approach will not provide a unique exposure measure (i.e. microbial indicator concentration) for each exposed individual and may lead to systematic misclassification bias. In addition, indicator organism counts are an indirect, and very often very inadequate, estimate of exposure to pathogens.

13.1.3 Randomised trials

A third approach, the randomised trial, is also a "prospective" study design but it differs from the cohort study outlined above in several respects. First, volunteers are recruited at the outset of the experiment. This group is generally interviewed prior to exposure, given medical examinations and then randomised into bather and non-bather groups. Both groups report to a test beach on a predetermined day. Typically, the bathers undertake a brief period of water exposure whilst the non-bathers remain on the beach. In well conducted studies, supervisors monitor each group and note the time and place of exposure undertaken by each bather. Both groups may also be given similar food, in the form of a packed lunch, and may undertake an identical short interview on the study day. Typically, volunteers report for a post-exposure interview and medical examination a week after exposure and complete a further postal questionnaire to examine any illnesses with longer incubation periods. During the exposure period, samples should be acquired from the bathing area in sufficient number to characterise fully water quality at the time and place of exposure for each bather, so that exposure can be assigned to each individual based on the time and place of exposure.

13.2 Major studies

13.2.1 UK Public Health Laboratory Service retrospective studies

The most widely quoted example of a retrospective case-control study was completed by the UK Public Health Laboratory Service between 1953 and 1958 (PHLS, 1959). This study was designed to identify any link between bathing in sewage-polluted waters and cases of poliomyelitis or enteric fever (paratyphoid). All cases of these two notifiable illnesses reported in seaside District Council areas were used in the study. Controls (matched for age and sex and selected, where possible, from the same street) were also identified. The availability of water quality records for the beaches used by the identified "cases" was also investigated, but microbiological information was rarely available for the appropriate times and locations of the exposure event(s). All cases and controls were interviewed by local medical staff to determine their bathing history. The authors concluded that there was no evidence linking the incidence of poliomyelitis and a history of sea bathing.

Role of retrospective case control studies

Although the retrospective design is not useful for developing exposure-response relationships, it is appropriate for linking illnesses to environmental exposures.

13.2.2 US Environmental Protection Agency prospective studies

The United States Environmental Protection Agency (US EPA) conducted a series of retrospective studies in the mid-1970s (Cabelli, 1983). At the time, these studies were the most extensive and carefully conducted prospective epidemiological investigations ever attempted. The main elements of the design are described briefly below.

Trials were conducted on Saturdays and Sundays, i.e. weekend-only bathers and non-bathing beachgoers were recruited in the hope of avoiding the multiple exposure problem. Demographic data were acquired during the initial beach interview and during a subsequent telephone interview. Bathing activity was recorded during the initial beach interview; bathers were defined as those who had experienced full head immersion, thus risking ingestion of water via the nasal and oral orifices. Wet hair at the time of recruitment was used as an indicator of head immersion and defined the exposure status of the participant.

The recruitment of study participants targeted family groups that included non-bathing controls. A letter was posted 1-2 days after the beach contact to remind all participating families that they should be recording all illness. At 8-10 days after recruitment the respondents for each family group were contacted by telephone to record any symptoms that had developed since the day they were at the beach.

Only gastro-intestinal (GI) symptoms were considered to be related to both swimming and pollution and the water quality indicators chosen. A subgroup of highly credible gastro-intestinal (HCGI) symptoms was defined as vomiting, diarrhoea accompanied by fever or which was disabling, or nausea and/or stomach ache accompanied by fever.

A range of bacterial indicators was used to characterise water quality at different locations, namely 11 indicators at New York City beaches, 5 at Lake Pontchartrain and 2 in Boston. Water quality was measured at times of maximum swimming activity, with 3-4 samples collected from 2-3 sites at chest depth, 12 inches below the surface of the water. This sampling design was used to characterise water quality for each test day. The geometric mean of these samples was used as the exposure estimate for all bathers on that day.

A rate difference between the bather and non-bather groups was used to quantify the swimming-associated morbidity rate for any particular symptom. Each trial day was associated with a specific water quality and an attack rate. However, individual trial days were not used as the raw data of subsequent analyses because the non-bathing control group for each day was of insufficient size. Analyses, therefore, was performed on water quality data gathered by summer and beach, or data from beach trial days that formed natural clusters of similar indicator densities.

Bivariate log-linear least squares regression was used to define the relationship between the seasonal swimming associated rate for GI symptoms (i.e. the rate difference between bathers and non-bathers) and water quality (as $\log_{10}$ geometric mean faecal indicator concentration). Cabelli (1983) reported statistically significant relationships between enterococcus density and swimming associated GI symptoms (GI r^2 = 74%; HCGI r^2 = 52% i.e. 52 per cent of the variance in HCGI symptom reporting was explained by the predictor variable enterococcus density). These relationships have been used to quantify the risk implied by the previous (NTAC, 1968) standards leading to the formulation of the most recent US Federal standard systems (US EPA, 1986).

Role of non-randomised prospective approach

Precise measurement of water quality at the time of exposure for each swimmer is not possible. The advantages of the non-randomised prospective approach include:

- Selection of participants after voluntary swimming activity allows a broad range of swimmers to be studied.

- Swimming-associated illness is expressed in terms of exposed populations rather than the probability of individual infection. Under some circumstances this approach may be meaningful from a public health perspective.

13.2.3 UK prospective randomised trial studies

The randomised trial involved recruitment of healthy adult volunteers at seaside towns with adjacent beaches that had traditionally passed EC Imperative Standards (Jones *et al.,* 1991; Kay *et al.,* 1994). After initial interviews and medical checks, volunteers reported to the specified bathing location on the trial day where they were randomised into bather and non-bather groups.

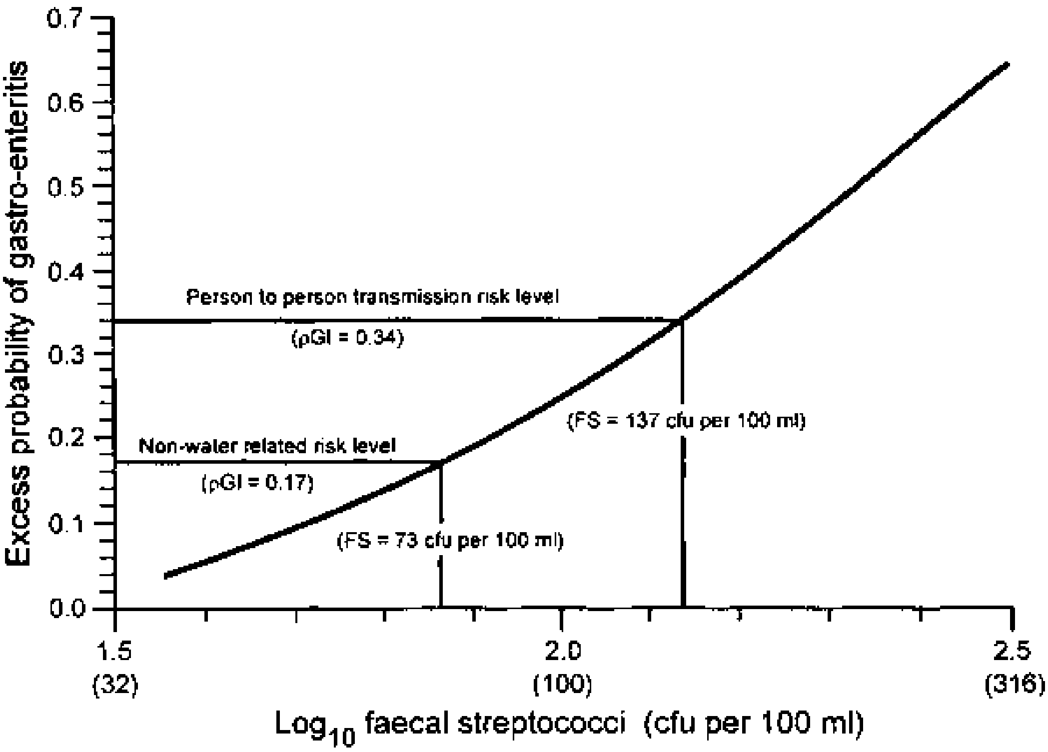
Bathers entered the water at specified locations where intensive water quality monitoring was taking place under the supervision of a marshall who recorded their activities. All bathers immersed their heads on three occasions. On exiting the water bathers were asked if they had swallowed water. The locations and times of exposure were known for each bather and, thus, a more precise estimate of "exposure" (i.e. indicator bacterial concentration) could be assigned to each bather (Fleisher *et al.,* 1993; Kay *et al.,* 1994). A control group of non-bathers came to the beach and had a picnic of identical type to that provided for all volunteers. One week after exposure all volunteers returned for further interviews and medical examinations and later they completed a final postal questionnaire, three weeks after exposure.

Detailed water quality measurements were completed at marked locations that defined "swim zones". Samples were collected synchronously at locations 20 m apart every 30 minutes and at three depths (i.e. surf zone, 1 m depth and at chest depth, 1.3-1.4 m). In the case of the latter two sampling depths, samples were collected at approximately 30 cm below the surface of the water. Five bacterial indicators were enumerated. Faecal coliforms and faecal streptococci were analysed using triplicate filtrations for each of three dilutions (i.e. nine plates per bacterial enumeration) to narrow the confidence limits on enumeration of total coliforms. Total staphylococci and *Pseudomonas aeruginosa* were also enumerated.

The initial analysis of the data from the UK randomised trial experiments centred on the links between water quality and gastro-enteritis (see Fleisher *et al.,* 1993 and Kay *et al.,* 1994). The data were analysed for relationships between water quality, as indexed by any of the five bacterial indicators measured at any of the three depths (i.e. 15 potential predictor variables) and gastro-enteritis. Only faecal streptococci, measured at chest depth, provided a statistically significant relationship between water quality and the risk of gastro-enteritis. This result was replicated at three of the four sites examined and at the fourth site concentrations of faecal streptococci were generally below the threshold level at which an effect was observed at the other three locations. No site specific differences were observed in terms of the exposure response curve. The relationship between faecal streptococci concentrations in recreational waters and the excess probability of gastro-enteritis in the exposed population is shown in Figure 13.1. This trend was not apparent with any other bacterial indicator enumerated and the volunteers reporting symptoms (or the research team) could not have known the concentrations of faecal streptococci to which each bather was exposed.

Multiple logistic regression analyses also allowed for the assessment of the effects of concomitant factors (i.e. other predictors of GI symptoms) on the relationship between water quality and illness. The analysis showed that other factors were significant predictors of GI illness. These included non-water-related risk (NWR) factors such as certain food types (see Table 13.2) and person-to-person transmission (PPT) from sick household members. These factors were independent of, and did not confound, the relationship between water quality and gastro-enteritis and can therefore provide markers against which the risk of GI illness from sewage contaminated sea water can be measured (see Figure 13.1). For example, exposure to NWR related risk factors in Table 13.2 represents a risk equivalent to a faecal streptococci concentration of approximately 70 organisms per 100 ml, whereas sharing a household with a person exhibiting GI symptoms represents an equivalent risk to a single exposure to recreational water containing approximately 140 faecal streptococcci per 100 ml.

Figure 13.1 The dose response curve linking faecal streptococci with excess probability of gastro-enteritis



This UK study, therefore, provides two sources of information. It presents an exposure-response relationship that defines the risk attributable to different levels of faecal streptococci exposure and, it quantifies the risk of other commonly experienced risks in society. As a result it provides scope for a system of risk-based standards.

These trials have also examined the relationships between non-enteric illnesses and exposure to sea water. Significant exposure-response relationships have been reported between acute febrile respiratory illnesses and faecal streptococci concentrations and between ear infections and faecal coliform concentrations. In addition, eye ailments were elevated in the bather group but unrelated to faecal indicator concentrations in the water (Fleisher *et al.,* 1996). Standard systems, to date, have centred on gastro-enteritis as the main outcome. However, as more evidence mounts on these non-enteric illnesses it may be prudent to consider their inclusion into future standard systems incorporating the concept of total disease burden through the use of Disability Adjusted Life Years (DALYs) or other means of cross-comparison between illnesses.

Table 13.2 Non-water-related risk factors for gastro-enteritis

Age (grouped by 10-year intervals)
Gender
History of migraine headaches
History of stress or anxiety
Frequency of diarrhoea (often, sometimes, rarely or never)
Current use of prescription drugs
Illnesses within 4 weeks prior to the trial day lasting more than 24 hours
Use of prescription drugs within 4 weeks prior to the trial day
Consumption of any of the following foods in the period from 3 days prior to 7 days after the trial day: mayonnaise, purchased sandwiches, chicken, eggs, hamburgers, hot dogs, raw milk, cold meat pies or seafood
Illness in the household within 3 weeks after the trial day
Alcohol consumption within the 7-day period after the trial
Frequency of usual alcohol consumption
Taking of laxatives within 4 weeks of the trial day
Taking of other stomach remedies within 4 weeks of the trial day
Additional bathing within 3 days prior and 3 weeks after the trial day[1]

[1] This was included in order to control for possible confounding due to multiple exposures among bathers and exposure among non-bathers prior to or after the trial day

Limitations of the randomised study protocol

The scope of UK randomised trial protocol is limited and should not be over-interpreted. The limitations include the fact that the studies were conducted in north European marine waters with a high tidal range where all waters commonly passed EU Imperative coliform criteria and the US EPA enterococci criteria. It may not be as applicable, however, in the standards design process for Mediterranean bathing waters where solar radiation is more intense, turbidity is lower and tidal activity is limited. Similar comments could be made concerning the application of these results to freshwater environments.

Furthermore, the results apply only to healthy adult volunteers, and may not be applicable directly to infants or chronically sick people. This is of particular relevance in the consideration of sampling depth. Adult chest depth predicted gastro-enteritis in adult bathers, but the UK operational sampling depth of 1 m (or less) might be more appropriate for child bathers. Another limitation is that the results may not be applicable to special interest groups such as surfers, sail-boarders and other high exposure activities that may involve prolonged contact with the water (often at some distance from the beach).

Role of the randomised prospective design

Randomisation of the exposed (bather) and non-exposed (non-bather) groups removes the problem of self-selection bias that is always potentially present where the exposure status is self-selected. More precise definition of water quality to which each bather was exposed (ideally through measurements taken at the place and time of exposure) provides better definition of the exposure for each bather. The multiple interviews facilitate data acquisition on a broad range of potential risk factors for the illness outcomes and allow accurate case definitions to be made.

13.3 Choice of study design

The primary criteria to be considered in the choice of an appropriate epidemiological study protocol are the objectives of the study and the validity of the findings, both of which determine the use to which the data acquired can be put. A secondary consideration is the scientific capacity and resource availability of the society in which the study is to be conducted. If, for example, the primary objective is to define an exposure-response relationship with maximum precision, then the randomised trial provides the most appropriate protocol. Its suitability derives from its tight control and relatively precise measurement of exposure (i.e. the water quality experienced by each bather) and the extensive data acquired on NWR risk factors. However, there are circumstances, even in affluent developed nations, where the implementation of the randomised trial is not feasible. For example, epidemiological investigations conducted to investigate the health implications of water sports activities, such as white water canoeing, would find a randomised design almost impossible to apply. It would clearly be inappropriate (and irresponsible) to expect a cohort of volunteers recruited from the general public to participate in a potentially dangerous activity for which they would not be skilled. Furthermore, randomisation of existing water sports participants would imply that the non-exposed group would agree willingly not to participate in their sport for a given period. In such circumstances an improved prospective cohort design (Cabelli, 1983) is the most appropriate. The application of this protocol to special interest groups of water sports enthusiasts in marine and fresh waters has been reported by Fewtrell *et al.* (1992, 1993, 1994).

Other circumstances may limit the application of the randomised design; for example, ethical constraints can preclude the inclusion of young people in the volunteer group. This was true for the UK studies but it was not found to be a problem in the randomised trial pilot investigation conducted in the Netherlands during 1996 (van Asperen *et al.,* 1997). The Netherlands randomised trial involved a volunteer group of children with no ethical constraints and studied their exposure to fresh recreational waters. Clearly, the identification and resolution of any potential ethical problems are an important preliminary step in the application of a randomised design.

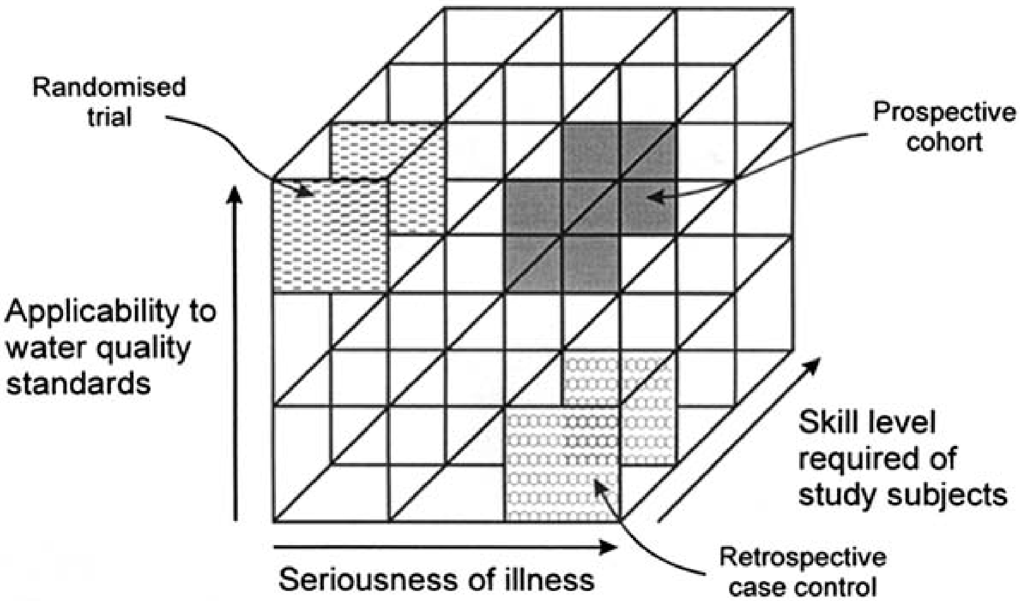
Where a non-randomised prospective protocol design is applied, i.e. the activity status of the participants is not determined by the researcher, a number of points should be noted. The measure of exposure (i.e. water quality) should, as far as possible, be attributed to small groups of exposed individuals. In effect, daily mean values applied to large groups exposed on one day will mask variability in indicator concentration and underestimate the numbers exposed to high indicator concentration. If the misclassification is random, it will tend to underestimate the slope of the dose-response curve through systematic misclassification bias. Attributing an exposure level to as small a group as possible requires intensive spatial and temporal water quality sampling.

Similarly, such studies should seek to acquire extensive data on NWR risk (and other potential confounding) factors. This might imply recruitment and follow-up interviews of considerable length with well-trained specialist health professionals conducting the data acquisition exercise. As with all epidemiological investigations, tight statistical control and early inputs to protocol design are essential to ensure that the information derived from the data acquired is maximised.

Through appropriate logistic regression analysis, the non-randomised prospective design can produce exposure-response relationships with information on NWR risk factors. However, such relationships should be treated with caution in standards design because of the possibility of misclassification bias and the potential underestimation of effect. Thus, if data on special interest groups or watersports activities are required, then a non-randomised prospective design may be appropriate, provided that any exposure-response relationships are treated with caution in their application to standards design of the type outlined in the *Guidelines for Safe Recreational Water Environments: Coastal and Fresh Waters* (WHO, 1998).

The retrospective case control design is clearly the only possible outbreak investigation tool and it does provide a means for establishing a link between specific pathogens and water exposure. It can provide guidance on maximum acceptable concentrations of pollution in recreational waters, provided that exposure data are available, but it is not designed to produce a credible dose-response curve of the type required in health-based standards design.

Figure 13.2 Choice of epidemiological protocols in recreational water epidemiology



In summary, the choice of epidemiological protocol design will be driven by the study objectives: i.e. the requirements of risk managers for precise exposure-response relationships, as well as logistical and ethical constraints on project implementation and the resources available. Figure 13.2 illustrates these choices in three dimensions as a guide to appropriate choice of epidemiological protocols in recreational water epidemiology.

13.4 Elements of good practice

• The design of any epidemiological component is critical because it affects every aspect of the recreational water study. It should address why the study is being done, i.e. what is the objective, and how it will be conducted. For example, it should address whether the research will use a case-control, cohort or cross-sectional approach to collect the data. It should also consider how the data will be analysed. These elements should be thoroughly described in the description of the design of the study.

• Health outcomes and exposure should be clearly defined. The endpoint result of exposure to microbiological hazards, as well as the exposure itself are key factors in describing the results of epidemiological studies. The endpoint might be self-reported symptomatology, indicative of exposure to a potentially broad spectrum of pathogens or it may be more specific, as with the isolation of an etiological agent or the reactivity of subject sera to known antigens. Efforts should be made to make the response to exposure endpoint as specific as possible.

• The population to be studied should be well defined in terms of the participating individuals. This will include demographic information, the means of selecting the population sample and the nature of exclusions, e.g. pregnant women or individuals being treated with steroids or immunosuppressive agents.

• The numerical size of exposed and non-exposed groups is another critical factor that must be considered in the conduct of epidemiological studies. The sizes of these groups are governed by the frequency of occurrence of the health effect under study. Illnesses or infections that occur at higher frequencies require smaller groups. The size of the required populations is also affected by the magnitude of the differences in the frequency of illness or infections between exposed and non-exposed groups. The smaller the differences to be detected between exposed and non-exposed groups the larger the number of subjects required in each group. Expert advice should be sought with regard to population size before conducting an epidemiological study.

• The approaches for collecting exposure and health effects data should be described in detail. This includes the use of questionnaires and other sources of health data, as well as methods used for collecting exposure data, such as microbiological analytical methods for enumerating microorganisms in water.

• Data analysis should include the steps taken to control selection, misclassification and confounding bias. The statistical evaluation procedures should be fully described.

• All of the measures taken to ensure the quality of the data should be described including the technical qualifications of all scientists participating in the study.

• The study plan should be submitted to a Human Investigations Committee, or its equivalent, to ensure that any regulatory limitations regarding human studies will be met, especially confidentiality restrictions and informed consent procedures.

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DIRECTIVE 2006/7/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL

of 15 February 2006

concerning the management of bathing water quality and repealing Directive 76/160/EEC

THE EUROPEAN PARLIAMENT AND THE COUNCIL OF THE EUROPEAN UNION,

Having regard to the Treaty establishing the European Community, and in particular Article 175(1) thereof,

Having regard to the proposal from the Commission [1],

Having regard to the opinion of the European Economic and Social Committee [2],

Having regard to the opinion of the Committee of the Regions [3],

Acting in accordance with the procedure laid down in Article 251 of the Treaty [4], in the light of the joint text approved by the Conciliation Committee on 8 December 2005,

Whereas:

(1) Building on the Commission's Communication on sustainable development, the European Council has singled out objectives as general guidance for future development in priority areas such as natural resources and public health.

(2) Water is a scarce natural resource, the quality of which should be protected, defended, managed and treated as such. Surface waters in particular are renewable resources with a limited capacity to recover from adverse impacts from human activities.

(3) Community policy on the environment should aim at a high level of protection, and contribute to pursuing the objectives of preserving, protecting and improving the quality of the environment and of protecting human health.

(4) In December 2000, the Commission adopted a Communication to the European Parliament and the Council on the development of a new bathing water policy and initiated a large-scale consultation of all interested and involved parties. The main outcome of this consultation was general support for the development of a new Directive based on the latest scientific evidence and paying particular attention to wider public participation.

(5) Decision No 1600/2002/EC of the European Parliament and of the Council of 22 July 2002 laying down the Sixth Community Environment Action Programme [5] contains a commitment to ensuring a high level of protection of bathing water, including by revising Council Directive 76/160/EEC of 8 December 1975 concerning the quality of bathing water [6].

(6) Pursuant to the Treaty, in preparing policy on the environment the Community is, *inter alia,* to take account of available scientific and technical data. This Directive should use scientific evidence in implementing the most reliable indicator parameters for predicting microbiological health risk and to achieve a high level of protection. Further epidemiological studies should be undertaken urgently concerning the health risks associated with bathing, particularly in fresh water.

(7) In order to increase efficiency and wise use of resources, this Directive needs to be closely coordinated with other Community legislation on water, such as Council Directives 91/271/EEC of 21 May 1991 concerning urban waste-water treatment [7], 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources [8] and Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy [9].

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(1) OJ C 45 E, 25.2.2003, p. 127.

(2) OJ C 220, 16.9.2003, p. 39.

(3) OJ C 244, 10.10.2003, p. 31.

(4) Opinion of the European Parliament of 21 October 2003 (OJ C 82 E, 1.4.2004, p. 115). Council Common Position of 20 December 2004 (OJ C 111 E, 11.5.2005, p. 1) and Position of the European Parliament of 10 May 2005 (not yet published in the Official Journal). European Parliament Legislative Resolution of 18 January 2006 (not yet published in the Official Journal) and Council Decision of 20 December 2005.

(5) OJ L 242, 10.9.2002, p. 1.

(6) OJ L 31, 5.2.1976, p. 1. Directive as last amended by Regulation (EC) No 807/2003 (OJ L 122, 16.5.2003, p. 36).

(7) OJ L 135, 30.5.1991, p. 40. Directive as last amended by Regulation (EC) No 1882/2003 of the European Parliament and of the Council (OJ L 284, 31.10.2003, p. 1).

(8) OJ L 375, 31.12.1991, p. 1. Directive as amended by Regulation (EC) No 1882/2003.

(9) OJ L 327, 22.12.2000, p. 1. Directive as amended by Decision No 2455/2001/EC (OJ L 331, 15.12.2001, p. 1).

(8) Appropriate information on planned measures and progress on implementation should be disseminated to stakeholders. The public should receive appropriate and timely information on the results of the monitoring of bathing water quality and risk management measures in order to prevent health hazards, especially in the context of predictable short-term pollution or abnormal situations. New technology that allows the public to be informed in an efficient and comparable way on bathing waters across the Community should be applied.

(9) For the purpose of monitoring, harmonised methods and practices of analysis need to be applied. Observation and quality assessment over an extended period are necessary in order to achieve a realistic bathing water classification.

(10) Compliance should be a matter of appropriate management measures and quality assurance, not merely of measuring and calculation. A system of bathing water profiles is therefore appropriate to provide a better understanding of risks as a basis for management measures. In parallel, particular attention should be attached to adherence to quality standards and coherent transition from Directive 76/160/EEC.

(11) On 17 February 2005 the Community ratified the UNECE Convention on Access to Information, Public Participation in Decision-Making and Access to Justice in Environmental Matters (the Århus Convention). It is therefore appropriate for this Directive to include provisions on public access to information and to provide for public participation in its implementation to supplement Directive 2003/4/EC of the European Parliament and of the Council of 28 January 2003 on public access to environmental information ([1]) and Directive 2003/35/EC of the European Parliament and of the Council of 26 May 2003 providing for public participation in respect of the drawing up of certain plans and programmes relating to the environment ([2]).

(12) Since the objectives of this Directive, namely the attainment by the Member States, on the basis of common standards, of a good bathing water quality and a high level of protection throughout the Community, cannot be sufficiently achieved by the Member States and can be better achieved at Community level, the Community may adopt measures, in accordance with the principle of subsidiarity as set out in Article 5 of the Treaty. In accordance with the principle of proportionality, as set out in that Article, this Directive does not go beyond what is necessary in order to achieve those objectives.

(13) The measures necessary for the implementation of this Directive should be adopted in accordance with Council Decision 1999/468/EC of 28 June 1999 laying down the procedures for the exercise of implementing powers conferred on the Commission ([3]).

(14) The continued importance of a Community bathing water policy is evident each bathing season as it protects the public from accidental and chronic pollution discharged in or near Community bathing areas. The overall quality of bathing waters has improved considerably since Directive 76/160/EEC came into force. However, that Directive reflects the state of knowledge and experience of the early 1970. Patterns of bathing water use have since changed, as has the state of scientific and technical knowledge. Therefore, that Directive should be repealed,

HAVE ADOPTED THIS DIRECTIVE:

CHAPTER I

GENERAL PROVISIONS

Article 1

Purpose and scope

1. This Directive lays down provisions for:

(a) the monitoring and classification of bathing water quality;

(b) the management of bathing water quality; and

(c) the provision of information to the public on bathing water quality.

2. The purpose of this Directive is to preserve, protect and improve the quality of the environment and to protect human health by complementing Directive 2000/60/EC.

3. This Directive shall apply to any element of surface water where the competent authority expects a large number of people to bathe and has not imposed a permanent bathing prohibition, or issued permanent advice against bathing (hereinafter bathing water). It shall not apply to:

(a) swimming pools and spa pools;

(b) confined waters subject to treatment or used for therapeutic purposes;

([1]) OJ L 41, 14.2.2003, p. 26.

([2]) OJ L 156, 25.6.2003, p. 17.

([3]) OJ L 184, 17.7.1999, p. 23.

(c) artificially created confined waters separated from surface water and groundwater.

Article 2

Definitions

For the purposes of this Directive, the following definitions shall apply:

1. the terms 'surface water', 'groundwater', 'inland water', 'transitional waters', 'coastal water' and 'river basin' have the same meaning as in Directive 2000/60/EC;

2. 'competent authority' means the authority or authorities that a Member State has designated to ensure compliance with the requirements of this Directive or any other authority or body to which that role has been delegated;

3. 'permanent' means, in relation to a bathing prohibition or advice against bathing, lasting for at least one whole bathing season;

4. 'large number' means, in relation to bathers, a number that the competent authority considers to be large having regard, in particular, to past trends or to any infrastructure or facilities provided, or other measures taken, to promote bathing;

5. 'pollution' means the presence of microbiological contamination or other organisms or waste affecting bathing water quality and presenting a risk to bathers' health as referred to in Articles 8 and 9 and Annex I, column A;

6. 'bathing season' means the period during which large numbers of bathers can be expected.

7. 'management measures' means the following measures undertaken with respect to bathing water:

   (a) establishing and maintaining a bathing water profile;

   (b) establishing a monitoring calendar;

   (c) monitoring bathing water;

   (d) assessing bathing water quality;

   (e) classifying bathing water;

   (f) identifying and assessing causes of pollution that might affect bathing waters and impair bathers' health;

   (g) giving information to the public;

   (h) taking action to prevent bathers' exposure to pollution;

   (i) taking action to reduce the risk of pollution;

8. 'short-term pollution' means microbiological contamination as referred to in Annex I, column A, that has clearly identifiable causes, is not normally expected to affect bathing water quality for more than approximately 72 hours after the bathing water quality is first affected and for which the competent authority has established procedures to predict and deal with as set out in Annex II;

9. 'abnormal situation' means an event or combination of events impacting on bathing water quality at the location concerned and not expected to occur on average more than once every four years;

10. 'set of bathing water quality data' means data obtained in accordance with Article 3;

11. 'bathing water quality assessment' means the process of evaluating bathing water quality, using the assessment method defined in Annex II;

12. 'cyanobacterial proliferation' means an accumulation of cyanobacteria in the form of a bloom, mat or scum;

13. the term 'public concerned' has the same meaning as in Council Directive 85/337/EEC of 27 June 1985 on the assessment of the effects of certain public and private projects on the environment [1].

CHAPTER II

QUALITY AND MANAGEMENT OF BATHING WATER

Article 3

Monitoring

1. Member States shall annually identify all bathing waters and define the length of the bathing season. They shall do so for the first time before the start of the first bathing season after 24 March 2008.

2. Member States shall ensure that monitoring of the parameters set out in Annex I, column A, takes place in accordance with Annex IV.

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(1) OJ L 175, 5.7.1985, p. 40. Directive as last amended by Directive 2003/35/EC of the European Parliament and of the Council (OJ L 156, 25.6.2003, p. 17).

3. The monitoring point shall be the location within the bathing water where:

(a) most bathers are expected; or

(b) the greatest risk of pollution is expected, according to the bathing water profile.

4. A monitoring calendar for each bathing water shall be established before the start of each bathing season and for the first time before the start of the third full bathing season after the entry into force of this Directive. Monitoring shall take place no later than four days after the date specified in the monitoring calendar.

5. Member States may introduce monitoring of the parameters set out in Annex I, column A, during the first full bathing season following the entry into force of this Directive. In that case, monitoring shall take place with the frequency specified in Annex IV. The results of such monitoring may be used to build up the sets of bathing water quality data referred to in Article 4. As soon as Member States introduce monitoring under this Directive, monitoring of the parameters set out in the Annex to Directive 76/160/EEC may cease.

6. Samples taken during short-term pollution may be disregarded. They shall be replaced by samples taken in accordance with Annex IV.

7. During abnormal situations, the monitoring calendar referred to in paragraph 4 may be suspended. It shall be resumed as soon as possible after the end of the abnormal situation. New samples shall be taken as soon as possible after the end of the abnormal situation to replace samples that are missing due to the abnormal situation.

8. Member States shall report any suspension of the monitoring calendar to the Commission, giving the reasons for the suspension. They shall provide such reports on the occasion of the next annual report provided for in Article 13 at the latest.

9. Member States shall ensure that the analysis of bathing water quality takes place in accordance with the reference methods specified in Annex I and the rules set out in Annex V. However, Member States may permit the use of other methods or rules if they can demonstrate that the results obtained are equivalent to those obtained using the methods specified in Annex I and the rules set out in Annex V. Member States that permit the use of such equivalent methods or rules shall provide the Commission with all relevant information about the methods or rules used and their equivalence.

Article 4

Bathing water quality assessment

1. Member States shall ensure that sets of bathing water quality data are compiled through the monitoring of the parameters set out in Annex I, column A.

2. Bathing water quality assessments shall be carried out:

(a) in relation to each bathing water;

(b) after the end of each bathing season;

(c) on the basis of the set of bathing water quality data compiled in relation to that bathing season and the three preceding bathing seasons; and

(d) in accordance with the procedure set out in Annex II.

However, a Member State may decide to carry out bathing water quality assessments on the basis of the set of bathing water quality data compiled in relation to the preceding three bathing seasons only. If it so decides, it shall notify the Commission beforehand. It shall also notify the Commission if it subsequently decides to revert to carrying out assessments on the basis of four bathing seasons. Member States may not change the applicable assessment period more than once every five years.

3. Sets of bathing water data used to carry out bathing water quality assessments shall always comprise at least 16 samples or, in the special circumstances referred to in Annex IV, paragraph 2, 12 samples.

4. However, provided that either:

— the requirement of paragraph 3 is satisfied, or

— the set of bathing water data used to carry out the assessment comprises at least eight samples, in the case of bathing waters with a bathing season not exceeding eight weeks,

a bathing water quality assessment may be carried out on the basis of a set of bathing water quality data relating to fewer than four bathing seasons if:

(a) the bathing water is newly identified;

(b) any changes have occurred that are likely to affect the classification of the bathing water in accordance with Article 5, in which case the assessment shall be carried out on the basis of a set of bathing water quality data consisting solely of the results for samples collected since the changes occurred; or

(c) the bathing water had already been assessed in accordance with Directive 76/160/EEC, in which case equivalent data gathered under that Directive shall be used and, for this purpose, parameters 2 and 3 of the Annex to Directive 76/160/EEC shall be deemed to be equivalent to parameters 2 and 1 of column A of Annex I to this Directive.

5. Member States may subdivide or group together existing bathing waters in the light of bathing water quality assessments. They may group existing bathing waters together only if these waters:

(a) are contiguous;

(b) received similar assessments for the preceding four years in accordance with paragraphs 2, 3 and 4(c); and

(c) have bathing water profiles all of which identify common risk factors or the absence thereof.

Article 5

Classification and quality status of bathing waters

1. As a result of the bathing water quality assessment carried out in accordance with Article 4, Member States shall, in accordance with the criteria set out in Annex II, classify bathing water as:

(a) 'poor';

(b) 'sufficient';

(c) 'good'; or

(d) 'excellent'.

2. The first classification according to the requirements of this Directive shall be completed by the end of the 2015 bathing season.

3. Member States shall ensure that, by the end of the 2015 bathing season, all bathing waters are at least 'sufficient'. They shall take such realistic and proportionate measures as they consider appropriate with a view to increasing the number of bathing waters classified as 'excellent' or 'good'.

4. However, notwithstanding the general requirement of paragraph 3, bathing waters may temporarily be classified as 'poor' and still remain in compliance with this Directive. In such cases, Member States shall ensure that the following conditions are satisfied:

(a) in respect of each bathing water classified as 'poor', the following measures shall be taken with effect from the bathing season that follows its classification:

(i) adequate management measures, including a bathing prohibition or advice against bathing, with a view to preventing bathers' exposure to pollution;

(ii) identification of the causes and reasons for the failure to achieve 'sufficient' quality status;

(iii) adequate measures to prevent, reduce or eliminate the causes of pollution; and

(iv) in accordance with Article 12, alerting the public by a clear and simple warning sign and informing them of the causes of the pollution and measures taken, on the basis of the bathing water profile.

(b) If a bathing water is classified as 'poor' for five consecutive years, a permanent bathing prohibition or permanent advice against bathing shall be introduced. However, a Member State may introduce a permanent bathing prohibition or permanent advice against bathing before the end of the five-year period if it considers that the achievement of 'sufficient' quality would be infeasible or disproportionately expensive.

Article 6

Bathing water profiles

1. Member States shall ensure that bathing water profiles are established in accordance with Annex III. Each bathing water profile may cover a single bathing water or more than one contiguous bathing waters. Bathing water profiles shall be established for the first time by 24 March 2011.

2. Bathing water profiles shall be reviewed and updated as provided for in Annex III.

3. When establishing, reviewing and updating bathing water profiles, adequate use shall be made of data obtained from monitoring and assessments carried out pursuant to Directive 2000/60/EC that are relevant for this Directive.

Article 7

Management measures in exceptional circumstances

Member States shall ensure that timely and adequate management measures are taken when they are aware of unexpected situations that have, or could reasonably be expected to have, an adverse impact on bathing water quality and on bathers' health. Such measures shall include information to the public and, if necessary, a temporary bathing prohibition.

Article 8

Cyanobacterial risks

1. When the bathing water profile indicates a potential for cyanobacterial proliferation, appropriate monitoring shall be carried out to enable timely identification of health risks.

2. When cyanobacterial proliferation occurs and a health risk has been identified or presumed, adequate management measures shall be taken immediately to prevent exposure, including information to the public.

Article 9

Other parameters

1. When the bathing water profile indicates a tendency for proliferation of macro-algae and/or marine phytoplankton, investigations shall be undertaken to determine their acceptability and health risks and adequate management measures shall be taken, including information to the public.

2. Bathing waters shall be inspected visually for pollution such as tarry residues, glass, plastic, rubber or any other waste. When such pollution is found, adequate management measures shall be taken, including, if necessary, information to the public.

Article 10

Cooperation on transboundary waters

Wherever a river basin gives rise to transboundary impacts on bathing water quality, the Member States involved shall cooperate as appropriate in implementing this Directive, including through the appropriate exchange of information and joint action to control those impacts.

CHAPTER III

EXCHANGE OF INFORMATION

Article 11

Public participation

Member States shall encourage public participation in the implementation of this Directive and shall ensure the provision of opportunities for the public concerned:

— to find out how to participate, and

— to formulate suggestions, remarks or complaints.

This shall relate, in particular, to the establishment, review and updating of lists of bathing waters in accordance with Article 3(1). Competent authorities shall take due account of any information obtained.

Article 12

Information to the public

1. Member States shall ensure that the following information is actively disseminated and promptly made available during the bathing season in an easily accessible place in the near vicinity of each bathing water:

(a) the current bathing water classification and any bathing prohibition or advice against bathing referred to in this Article by means of a clear and simple sign or symbol;

(b) a general description of the bathing water, in non-technical language, based on the bathing water profile established in accordance with Annex III;

(c) in the case of bathing waters subject to short-term pollution:

— notification that the bathing water is subject to short-term pollution,

— an indication of the number of days on which bathing was prohibited or advised against during the preceding bathing season because of such pollution, and

— a warning whenever such pollution is predicted or present,

(d) information on the nature and expected duration of abnormal situations during such events;

(e) whenever bathing is prohibited or advised against, a notice advising the public and giving reasons;

(f) whenever a permanent bathing prohibition or permanent advice against bathing is introduced, the fact that the area concerned is no longer a bathing water and the reasons for its declassification; and

(g) an indication of sources of more complete information in accordance with paragraph 2.

2. Member States shall use appropriate media and technologies, including the Internet, to disseminate actively and promptly the information concerning bathing waters referred to in paragraph 1 and also the following information in several languages, when appropriate:

(a) a list of bathing waters;

(b) the classification of each bathing water over the last three years and its bathing water profile, including the results of monitoring carried out in accordance with this Directive since the last classification;

(c) in the case of bathing waters classified as being 'poor', information on the causes of pollution and measures taken with a view to preventing bathers' exposure to pollution and to tackle its causes as referred to in Article 5(4); and

(d) in the case of bathing waters subject to short-term pollution, general information on:

— conditions likely to lead to short-term pollution,

— the likelihood of such pollution and its likely duration,

— the causes of the pollution and measures taken with a view to preventing bathers' exposure to pollution and to tackle its causes.

The list referred to in point (a) shall be available each year before the start of the bathing season. The results of the monitoring referred to in point (b) shall be made available on the Internet upon completion of the analysis.

3. The information referred to in paragraphs 1 and 2 shall be disseminated as soon as it is available and with effect from the start of the fifth bathing season after 24 March 2008.

4. Member States and the Commission shall, wherever possible, provide information to the public using geo-referenced technology and present it in a clear and coherent manner, in particular through the use of signs and symbols.

Article 13

Reports

1. Member States shall provide the Commission with the results of the monitoring and with the bathing water quality assessment for each bathing water, as well as with a description of significant management measures taken. Member States shall provide this information annually by 31 December in relation to the preceding bathing season. They shall begin providing it once the first bathing water quality assessment has been carried out in accordance with Article 4.

2. Member States shall notify the Commission annually before the start of the bathing season of all waters identified as bathing waters, including the reason for any change compared to the preceding year. They shall do so for the first time before the start of the first bathing season after 24 March 2008.

3. When monitoring of bathing water has started under this Directive, annual reporting to the Commission in accordance with paragraph 1 shall continue to take place pursuant to Directive 76/160/EEC until a first assessment can be made under this Directive. During that period, parameter 1 of the Annex to Directive 76/160/EEC shall not be taken into account in the annual report, and parameters 2 and 3 of the Annex to Directive 76/160/EEC shall be assumed to be equivalent to parameters 2 and 1 of column A of Annex I to this Directive.

4. The Commission shall publish an annual summary report on bathing water quality in the Community, including bathing water classifications, conformity with this Directive and significant management measures undertaken. The Commission shall publish this report by 30 April every year, including via the Internet. When establishing the report the Commission shall, wherever possible, make best use of data collection, assessment and presentation systems under related Community legislation, in particular Directive 2000/60/EC.

CHAPTER IV

FINAL PROVISIONS

Article 14

Report and review

1. The Commission shall, by 2008, submit a report to the European Parliament and to the Council. The report shall have particular regard to:

(a) the results of an appropriate European epidemiological study conducted by the Commission in collaboration with Member States;

(b) other scientific, analytical and epidemiological developments relevant to the parameters for bathing water quality, including in relation to viruses; and

(c) World Health Organisation recommendations.

2. Member States shall, by the end of 2014, submit written observations to the Commission on that report including on the need for any further research or assessments which may be required to assist the Commission in its review of this Directive under paragraph 3.

3. In the light of the report, the Member States' written observations and an extended impact assessment and bearing in mind experience gained from implementing this Directive, the Commission shall, no later than 2020, review this Directive with particular regard to the parameters for bathing water quality, including whether it would be appropriate to phase out the 'sufficient' classification or modify the applicable standards, and shall present if necessary appropriate legislative proposals in accordance with Article 251 of the Treaty.

Article 15

Technical adaptations and implementing measures

1. It shall be decided in accordance with the procedure referred to in Article 16(2):

(a) to specify the EN/ISO standard on the equivalence of microbiological methods for the purposes of Article 3(9);

(b) to lay down detailed rules for the implementation of Articles 8(1), 12(1)(a) and 12(4);

(c) to adapt the methods of analysis for the parameters set out in Annex I in the light of scientific and technical progress;

(d) to adapt Annex V in the light of scientific and technical progress;

(e) to lay down guidelines for a common method for the assessment of single samples.

2. The Commission shall present a draft of the measures to be taken in accordance with paragraph 1(b) with respect to Article 12(1)(a) by 24 March 2010. Before doing so, it shall consult representatives of Member States, regional and local authorities, relevant tourist and consumer organisations and other interested parties. After the adoption of relevant rules, it shall publicise them via the Internet.

Article 16

Committee procedure

1. The Commission shall be assisted by a committee.

2. Where reference is made to this paragraph, Articles 5 and 7 of Decision 1999/468/EC shall apply, having regard to the provisions of Article 8 thereof.

The period laid down in Article 5(6) of Decision 1999/468/EC shall be set at three months.

3. The Committee shall adopt its rules of procedure.

Article 17

Repeal

1. Directive 76/160/EEC is hereby repealed with effect from 31 December 2014. Subject to paragraph 2, this repeal shall be without prejudice to Member States' obligations concerning the time limits for transposition and application set out in the repealed Directive.

2. As soon as a Member State has taken all necessary legal, administrative and practical measures to comply with this Directive, this Directive will be applicable, replacing Directive 76/160/EEC.

3. References to the repealed Directive 76/160/EEC shall be construed as being made to this Directive.

Article 18

Implementation

1. Member States shall bring into force the laws, regulations and administrative provisions necessary to comply with this Directive by 24 March 2008. They shall forthwith inform the Commission thereof.

When Member States adopt these measures, they shall contain a reference to this Directive or shall be accompanied by such a reference on the occasion of their official publication. The methods of making such reference shall be laid down by Member States.

2. Member States shall communicate to the Commission the texts of the main provisions of national law that they adopt in the field covered by this Directive.

Article 19

Entry into force

This Directive shall enter into force on the 20th day following its publication in the *Official Journal of the European Union*.

Article 20

Addressees

This Directive is addressed to Member States.

Done at Strasbourg, 15 February 2006.

For the European Parliament	*For the Council*
The President	*The President*
J. BORRELL FONTELLES	H. WINKLER

ANNEX I

For inland waters

	A	B	C	D	E
	Parameter	Excellent quality	Good quality	Sufficient	Reference methods of analysis
1	Intestinal enterococci (cfu/100 ml)	200 (*)	400 (*)	330 (**)	ISO 7899-1 or ISO 7899-2
2	Escherichia coli (cfu/100 ml)	500 (*)	1 000 (*)	900 (**)	ISO 9308-3 or ISO 9308-1

(*) Based upon a 95-percentile evaluation. See Annex II.
(**) Based upon a 90-percentile evaluation. See Annex II.

For coastal waters and transitional waters

	A	B	C	D	E
	Parameter	Excellent quality	Good quality	Sufficient	Reference methods of analysis
1	Intestinal enterococci (cfu/100 ml)	100 (*)	200 (*)	185 (**)	ISO 7899-1 or ISO 7899-2
2	Escherichia coli (cfu/100 ml)	250 (*)	500 (*)	500 (**)	ISO 9308-3 or ISO 9308-1

(*) Based upon a 95-percentile evaluation. See Annex II.
(**) Based upon a 90-percentile evaluation. See Annex II.

ANNEX II

Bathing water assessment and classification

1. Poor quality

Bathing waters are to be classified as 'poor' if, in the set of bathing water quality data for the last assessment period [a], the percentile values [b] for microbiological enumerations are worse [c] than the 'sufficient' values set out in Annex I, column D.

2. Sufficient quality

Bathing waters are to be classified as 'sufficient':

1. if, in the set of bathing water quality data for the last assessment period, the percentile values for microbiological enumerations are equal to or better [d] than the 'sufficient' values set out in Annex I, column D; and

2. if the bathing water is subject to short-term pollution, on condition that:

   (i) adequate management measures are being taken, including surveillance, early warning systems and monitoring, with a view to preventing bathers' exposure by means of a warning or, where necessary, a bathing prohibition;

   (ii) adequate management measures are being taken to prevent, reduce or eliminate the causes of pollution; and

   (iii) the number of samples disregarded in accordance with Article 3(6) because of short-term pollution during the last assessment period represented no more than 15 % of the total number of samples provided for in the monitoring calendars established for that period, or no more than one sample per bathing season, whichever is the greater.

3. Good quality

Bathing waters are to be classified as 'good':

1. if, in the set of bathing water quality data for the last assessment period, the percentile values for microbiological enumerations are equal to or better [d] than the 'good quality' values set out in Annex I, column C; and

2. if the bathing water is subject to short-term pollution, on condition that:

   (i) adequate management measures are being taken, including surveillance, early warning systems and monitoring, with a view to preventing bathers' exposure, by means of a warning or, where necessary, a bathing prohibition;

   (ii) adequate management measures are being taken to prevent, reduce or eliminate the causes of pollution; and

   (iii) the number of samples disregarded in accordance with Article 3(6) because of short-term pollution during the last assessment period represented no more than 15 % of the total number of samples provided for in the monitoring calendars established for that period, or no more than one sample per bathing season, whichever is the greater.

4. **Excellent quality**

Bathing waters are to be classified as 'excellent':

1. if, in the set of bathing water quality data for the last assessment period, the percentile values for microbiological enumerations are equal to or better than the 'excellent quality' values set out in Annex I, column B; and

2. if the bathing water is subject to short-term pollution, on condition that:

   (i) adequate management measures are being taken, including surveillance, early warning systems and monitoring, with a view to preventing bathers' exposure, by means of a warning or, where necessary, a bathing prohibition;

   (ii) adequate management measures are being taken to prevent, reduce or eliminate the causes of pollution; and

   (iii) the number of samples disregarded in accordance with Article 3(6) because of short-term pollution during the last assessment period represented no more than 15 % of the total number of samples provided for in the monitoring calendars established for that period, or no more than one sample per bathing season, whichever is the greater.

NOTES

(a) 'Last assessment period' means the last four bathing seasons or, when applicable, the period specified in Article 4(2) or (4).

(b) Based upon percentile evaluation of the $\log_{10}$ normal probability density function of microbiological data acquired from the particular bathing water, the percentile value is derived as follows:

   (i) Take the $\log_{10}$ value of all bacterial enumerations in the data sequence to be evaluated. (If a zero value is obtained, take the $\log_{10}$ value of the minimum detection limit of the analytical method used instead.)

   (ii) Calculate the arithmetic mean of the $\log_{10}$ values (μ).

   (iii) Calculate the standard deviation of the $\log_{10}$ values (σ).

   The upper 90-percentile point of the data probability density function is derived from the following equation: upper 90-percentile = antilog ($\mu + 1{,}282\ \sigma$).

   The upper 95-percentile point of the data probability density function is derived from the following equation: upper 95-percentile = antilog ($\mu + 1{,}65\ \sigma$).

(c) 'Worse' means with higher concentration values expressed in cfu/100 ml.

(d) 'Better' means with lower concentration values expressed in cfu/100 ml.

ANNEX III

The bathing water profile

1. The bathing water profile referred to in Article 6 is to consist of:

   (a) a description of the physical, geographical and hydrological characteristics of the bathing water, and of other surface waters in the catchment area of the bathing water concerned, that could be a source of pollution, which are relevant to the purpose of this Directive and as provided for in Directive 2000/60/EC;

   (b) an identification and assessment of causes of pollution that might affect bathing waters and impair bathers' health;

   (c) an assessment of the potential for proliferation of cyanobacteria;

   (d) an assessment of the potential for proliferation of macro-algae and/or phytoplankton;

   (e) if the assessment under point (b) shows that there is a risk of short-term pollution, the following information:

   — the anticipated nature, frequency and duration of expected short-term pollution,

   — details of any remaining causes of pollution, including management measures taken and the time schedule for their elimination,

   — management measures taken during short-term pollution and the identity and contact details of bodies responsible for taking such action,

   (f) the location of the monitoring point.

2. In the case of bathing waters classified as 'good', 'sufficient' or 'poor', the bathing water profile is to be reviewed regularly to assess whether any of the aspects listed in paragraph 1 have changed. If necessary, it is to be updated. The frequency and scope of reviews is to be determined on the basis of the nature and severity of the pollution. However, they are to comply with at least the provisions and to take place with at least the frequency specified in the following table.

Bathing water classification	'Good'	'Sufficient'	'Poor'
Reviews are to take place at least every	four years	three years	two years
Aspects to be reviewed (points of paragraph 1)	(a) to (f)	(a) to (f)	(a) to (f)

In the case of bathing waters previously classified as 'excellent', the bathing water profiles need be reviewed and, if necessary, updated only if the classification changes to 'good', 'sufficient' or 'poor'. The review is to cover all aspects mentioned in paragraph 1.

3. In the event of significant construction works or significant changes in the infrastructure in or in the vicinity of the bathing water, the bathing water profile is to be updated before the start of the next bathing season.

4. The information referred to in paragraph 1(a) and (b) is to be provided on a detailed map whenever practicable.

5. Other relevant information may be attached or included if the competent authority considers it appropriate.

ANNEX IV

Bathing water monitoring

1. One sample is to be taken shortly before the start of each bathing season. Taking account of this extra sample and subject to paragraph 2, no fewer than four samples are to be taken and analysed per bathing season.

2. However, only three samples need be taken and analysed per bathing season in the case of a bathing water that either:

   (a) has a bathing season not exceeding eight weeks; or

   (b) is situated in a region subject to special geographical constraints.

3. Sampling dates are to be distributed throughout the bathing season, with the interval between sampling dates never exceeding one month.

4. In the event of short-term pollution, one additional sample is to be taken to confirm that the incident has ended. This sample is not to be part of the set of bathing water quality data. If necessary to replace a disregarded sample, an additional sample is to be taken seven days after the end of the short-term pollution.

ANNEX V

Rules on the handling of samples for microbiological analyses

1. Sampling point

Where possible, samples are to be taken 30 centimetres below the water's surface and in water that is at least one metre deep.

2. Sterilisation of sample bottles

Sample bottles are:

— to undergo sterilisation in an autoclave for at least 15 minutes at 121 °C, or

— to undergo dry sterilisation at between 160 °C and 170 °C for at least one hour, or

— to be irradiated sample containers obtained directly from manufacturer.

3. Sampling

The volume of the sampling bottle/container is to depend on the quantity of water needed for each parameter to be tested. The minimum content is generally 250 ml.

Sample containers are to be of transparent and non-coloured material (glass, polyethene or polypropylene).

In order to prevent accidental contamination of the sample, the sampler is to employ an aseptic technique to maintain the sterility of the sample bottles. There is no further need for sterile equipment (such as sterile surgical gloves or tongs or sample pole) if this is done properly.

The sample is to be clearly identified in indelible ink on the sample and on the sampling form.

4. Storage and transport of samples before analysis

Water samples are to be protected at all stages of transport from exposure to light, in particular direct sunlight.

The sample is to be conserved at a temperature of around 4 °C, in a cool box or refrigerator (depending on climate) until arrival at the laboratory. If the transport to the laboratory is likely to take more than four hours, then transport in a refrigerator is required.

The time between sampling and analysis is to be kept as short as possible. It is recommended that samples be analysed on the same working day. If this is not possible for practical reasons, then the samples shall be processed within no more than 24 hours. In the meantime, they shall be stored in the dark and at a temperature of 4 °C ± 3 °C.