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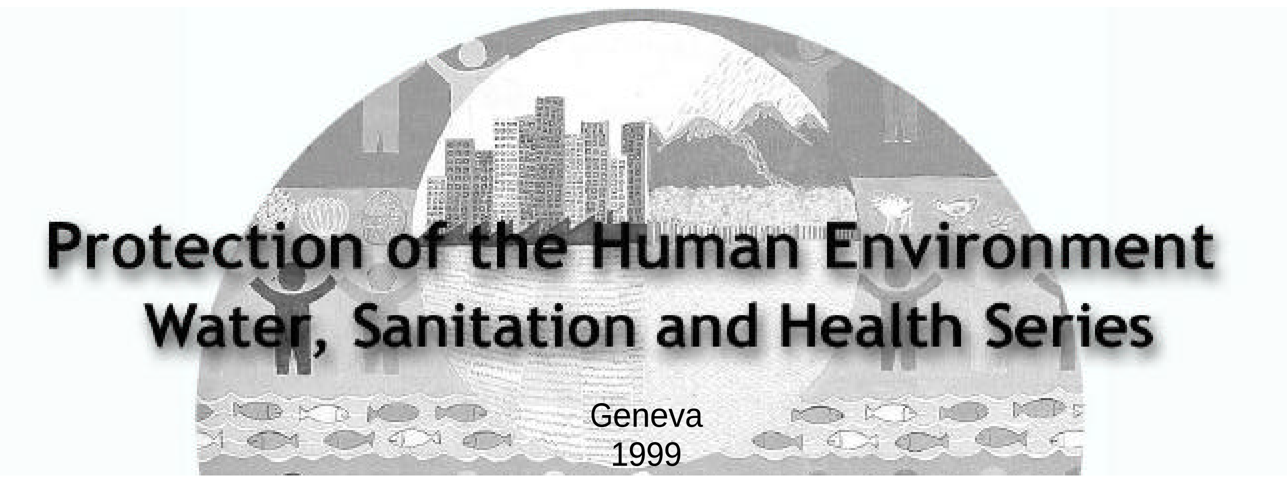
HEALTH-BASED MONITORING OF RECREATIONAL WATERS: THE FEASIBILITY OF A NEW APPROACH (THE 'ANNAPOLIS PROTOCOL')

OUTCOME OF AN EXPERT CONSULTATION, ANNAPOLIS, USA CO-SPONSORED BY USEPA



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FOREWORD

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

In 1994, following discussions between the WHO Regional Office for Europe and WHO Headquarters it was agreed to initiate the development of Guidelines concerning recreational use of the water environment. Guidelines of this type represent primarily a consensus view amongst experts on the risks to health represented by various media and activities; and are based upon critical review of the available evidence. The *Guidelines for Safe Recreational-water Environments*, which result from this process, are being released in two volumes: Volume 1 (WHO, 1998) addresses coastal and freshwaters and Volume 2 (WHO, 1999) swimming pools spas and similar recreational-water environments. Both Volumes are being released as drafts for consultation for a period prior to finalisation.

During the development of Volume 1 of the *Guidelines for Safe Recreational-water Environments* concerns were repeatedly expressed regarding the adequacy and effectiveness of present approaches to monitoring and assessment. In response to these concerns the text *Monitoring Bathing Waters* was developed (Bartram and Rees Eds, 1999 published by E&FN Spon on behalf of WHO, the Commission of the European Communities and USEPA).

Despite evident successes in the protection of public health, present approaches to the regulation of microbiological hazards in recreational waters suffer a series of limitations. During the preparation of *Monitoring Bathing Waters* the United States Environment Protection Agency (USEPA) therefore supported WHO in organising an expert consultation to look into the adequacy and effectiveness of present approaches to monitoring and assessment linked to effective management of microbiological hazards in coastal and freshwater recreational waters. The meeting was implemented in November 1998 in Annapolis, USA. The experts that met there agreed that an improved approach to the regulation of recreational water that better reflected health risk and provided enhanced scope for effective management intervention was necessary and feasible. The output of the meeting was the development of such an approach, which has become known as the *'Annapolis Protocol'*. The 'protocol' is reported here and is also published in Monitoring Bathing Waters. Because this is so different to established practice it includes elements that require substantial testing.

WHO wishes to express its appreciation to the experts that contributed to the meeting and to the development of this report who are listed in Annex 1 to this report. Special thanks are also due to the USEPA for providing financial support for the implementation of the meeting. The USEPA has not undertaken a legal or regulatory review of the outcome of the meeting.

1. Background

1.1 Current regulatory schemes

Recreational water standards have had some successes in driving clean-ups, 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 (Table 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; and
- 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 1 Microbiological Quality of Water Guidelines/Standards 100ml

Country	Shellfish Harvesting		Primary Contact Recreation			Protection of Indigenous Organisms		References
	TC*	**FC****	**TC***	**FC****	**Other**	**TC***	**FC****	
Brazil		100% < 100	80% < 5000[m]	80% < 1000[m]				Brazil Ministerio del Interior (1976)
Colombia			1000	200				Colombia, Ministerio de Salud (1979)
Cuba			1000[a]	200[a] 90% < 400				Cuba, Ministerio de Salud (1986)
EEC,[b] Europe			80% < 500[c] 95% < 10000[d]	80% < 100[c] 95% < 2000[d]	Faecal streptococci 100 [c] Salmonella 0/liter[d] Enteroviruses 0 PFU/liter[d] Enterococci 90% < 100			EEC (1976) CEPPOL (1991)
Ecuador			1000	200				Ecuador, Ministerio de Salud Publica (1987)
France			< 2000	< 500	Faecal streptococci < 100			WHO (1977)
Israel			80% < 1000[g]					Argentina, INCYTH (1984)
Japan	70		1000			1000		Japan, Environmental Agency (1981)
Mexico	70[e] 90% < 230		80% < 1000[f] 100% < 10000[k]			10,000[e] 80% < 10000 100% < 20000		Mexico, SEDUE (1983)
Peru	80% < 1000	80% < 200 100% < 1000	80% < 5000[f]	80% < 1000[f]		80% < 20000	80% < 4000	Peru, Ministerio de Salud (1983)
Poland					*E. coli* < 1000			WHO (1975)
Puerto Rico	70[h] 80% < 230			200[h] 80% < 400				Puerto Rico, JCA (1983)
United States, California	70[e]		80% < 1000 [i,j] 100% < 10000[k]	200 [a,j] 90% < 400[l]				California State Water Resources Board (no date)
United States, USEPA		14[a] 90% < 43			Enterococci 35[a] (marine), 33[a] (fresh) *E. coli* 126[a] (fresh)			USEPA (1986) Dufour and Ballentine (1986)
Former USSR					*E. coli* < 100			WHO (1977)
UNEP/ WHO		80% < 10 100% < 100		50% < 100[n] 90% < 1000[n]				WHO/UNEP (1978)
Uruguay				< 500[n] < 1000[o]				Uruguay, DINAMA (1998)
Venezuela	70[a] 90% < 230	14[a] 90% < 43	90% < 1000 100% < 5000	90% < 200 100% < 400				Venezuela (1978)
Yugoslavia			2000					Argentina, INCYTH (1984)

* Total Coliforms
** Faecal or Thermotolerant Coliforms
a. Logarithmic average for a period of 30 days of at least 5 samples
b. Minimum sampling frequency – fortnightly
c. Guide
d. Mandatory
e. Monthly average
f. At least 5 samples per month
g. Minimum 10 sample per month
h. At least 5 samples taken sequentially from the waters in a given instance
i. Period of 30 days
j. Within a zone bounded by the shoreline and a distance of 1000 feet from the shoreline or the 30 foot depth contour, whichever is further from the shoreline
k. Not a sample taken during the verification period of 48 hours should exceed 10,000/100ml
l. Period of 60 days
m. "Satisfactory" waters, samples obtained in each of the preceding 5 weeks
n. Geometric mean of at least 5 samples
o. Not to be exceeded in at least 5 samples

Source: Adapted from Salas (1998)

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 effectively 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 effectively supports decision-making to protect public health.

1.2 Pathogens

There is a broad spectrum of illnesses that have been associated with swimming in marine and fresh recreational waters. Table 2 is a list of microbes that have been linked to swimming-associated disease outbreaks in the USA between 1985 and 1994.

Table 2 Outbreaks associated with recreational waters in the USA, 1985–1994

Etiological Agent	Number of Cases	Number 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 Gastrointestinal Infections	965	11

Sources: Morbidity and Mortality Weekly Report, 1988, 1990, 1991, 1993; JAWWA, 1996.

Two bacterial pathogens, *E. 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, which might be expected in unmixed small bodies of water containing large numbers of pathogens.

Outbreaks caused by *Leptospira,* Norwalk virus and Adenovirus 3 were more typical in that 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 gastrointestinal infections with an unknown aetiology are more common. Although the cause is unknown, the symptomatology of the illness is frequently similar to that 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 confirm that viruses are candidate organisms for the gastroenteritis observed in epidemiological studies conducted at bathing beaches (Table 3). The data in Table 3 are from acute and convalescent sera obtained from swimmers who suffered from acute gastroenteritis after swimming at a very contaminated beach in Alexandria, Egypt. The sera were obtained from twelve subjects, all of whom were less than 12 years old, on the day after the swimming event and about 15 days later. The sera were tested with Norwalk virus and rotavirus. None of the subjects showed a four-fold increase in titre to rotavirus antigen. However, 33 per cent did show a four-fold 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.

Table 3 Serological response to Norwalk Virus and Rotavirus in individuals with recent swimming associated gastroenteritis

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

The types and numbers of various pathogens in sewage will vary depending on the incidence of disease in the contributing population, and known seasonality in human infections. Hence, numbers will vary greatly across different parts of the world and times of year, but a general indication is given in Table 4.

Table 4 Examples of pathogens and indicator organisms in raw sewage

Pathogen/Indicator[1]	Disease/role	Numbers per Litre
Bacteria		
Campylobacter spp.	Gastroenteritis	37,000
Clostridium perfringens[2]	Indicator	$6x10^5$-$8x10^5$
E. coli	Indicator	10^7-10^8
Salmonella spp.	Gastroenteritis	20-80,000
Shigella	Bacillary dysentery	10-10,000
Viruses		
Polioviruses	Indicator	1800-5,000,000
Rotaviruses	Diarrhoea, vomiting	4000-850,000
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.	Anemia	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, Hepatitis A, etc.
2. From Long & Ashbolt (1994)

Source: Adapted from Yates and Gerba (1998)

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 taken up 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 environmental stresses also can make the task very difficult.

The use of indicator organisms to signal the potential presence of organisms that cause gastrointestinal disease concept has been used successfully for a long time. The faecal indicator bacteria most commonly used today are thermotolerant coliforms, *E. coli* and enterococci or faecal streptococci. However, there are still many questions concerning the effectiveness of the way in which water quality is measured and monitored, and 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 5 provides an overview of possible indicators, describing the strengths and weaknesses of each.

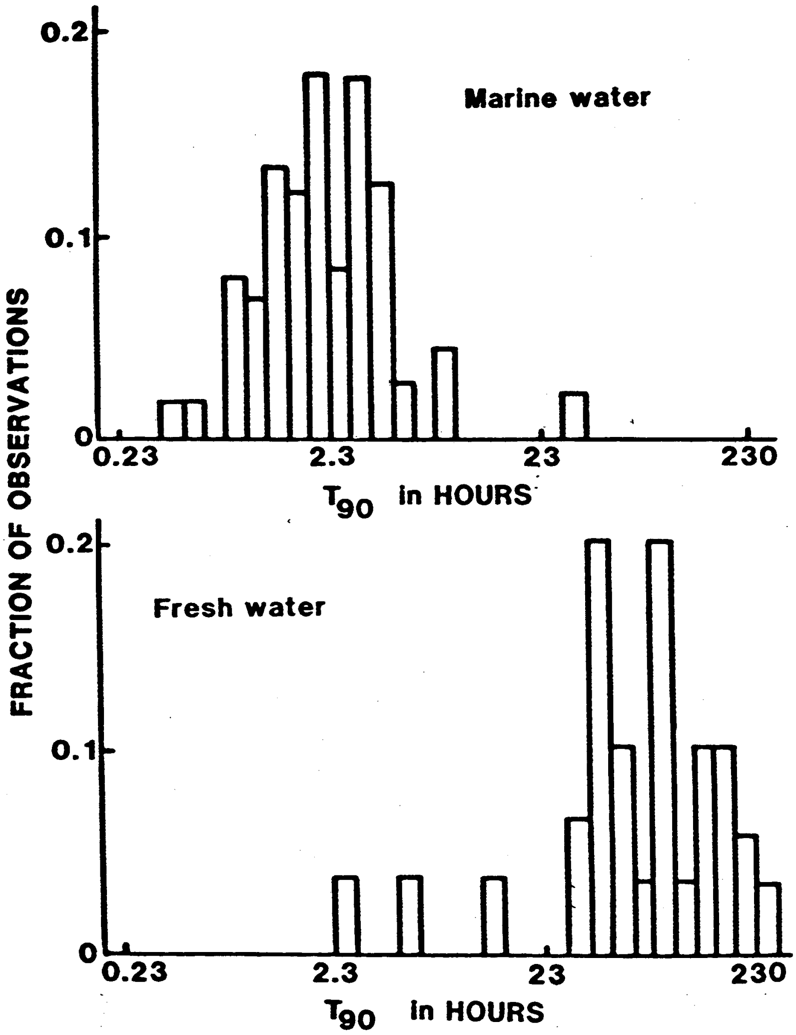
Table 5 Possible sewage contamination indicators and their functions

Indicator/use	Function	
	Pros	**Cons**
Faecal Streptococci /Enterococci	• Marine and potentially freshwater human health indicator. • More persistent in water and sediments than coliforms. • FS may be cheaper than enterococci to assay.	• May not be valid for tropical waters, due to potential growth in soils.
Thermotolerant coliforms	• Indicator of recent faecal contamination.	• Possibly not suitable for 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	• Potentially a freshwater human health indicator. • Indicator of recent faecal contamination. • Potential for typing *E. coli* to aid in sourcing faecal contamination. • Rapid identification possible if define as β-glucuronidase producing bacteria.	• Possibly not suitable for tropical waters due to growth in soils and waters.
Sanitary plastics	• Immediate assessment can be made for each bathing day. • Can be categorized. • Little training of staff required.	• May reflect old sewage contamination and be of little health significance. • Subjective and prone to variable description.
Preceding rainfall (12, 24, 48 or 72h)	• 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 / *Clostridium perfringens*	• Always in sewage impacted waters. • Possibly correlated with enteric viruses and parasitic protozoa. • Inexpensive assay with H_2S production.	• May also come from dog faeces. • May be too conservative an indicator. • Enumeration requires anaerobic culture.
Somatic coliphages	• Standard method well established. • Similar physical behavior 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	• Standard ISO method available. • More persistent than some coliphages. • Host does not grow in environmental waters below 30°C	• Not specific to sewage. • WG49 host may lose plasmid (although F-amp more stable) • Not as persistent in marine waters.
Bacteroides fragilis phages	• Appears to be specific to sewage. • ISO method recently published. • More resistant than other phages in the environment and similar to hardy human enteric viruses.	• Requires anaerobic culture. • Numbers in sewage are lower than other phages, and most humans do not excrete this phage (hence no value for small populations).

Faecal sterols	• Coprostanol largely specific to sewage. • Coprostanol degradation in water similar to die-off of thermotolerant coliforms. • Ratio of 5β/5α stanols > 0.5 is indicative of faecal contamination. i.e. coprostanol/5α-cholestanol > 0.5 indicates human faecal contamination; while C_{29} 5β (24-ethylcoprostanol)/ 5α stanol ratio >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).	• Requires gas chromatographic analysis and is expensive (about $100/sample). • Requires up to 10 L of sample to be filtered through a glass fibre filter (Whatman) to concentrate particulate stanols.
Caffeine	• May be specific to sewage, but unproven to date. • Could be developed into a dip-stick 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 (Animal sourced pathogens)	• Required for potential zoonoses, such as *Cryptosporidium* spp, where faecal indicator bacteria may have died out, or not present.	• Expensive and specialized assay (e.g. Method 1622, USEPA). • Human/animal speciation of serotypes not currently defined.

Die-off in marine and freshwater environments

The differential die-off of indicators in marine and fresh water environments is illustrated for coliforms in Figure 1. The figure, adapted from Chamberlain and Mitchell (1978), shows that in marine waters the mean T_{90} for total coliforms is about 2.2 hours, whereas in fresh waters the mean T_{90} is about 58 hours.



Source: Adapted from Chamberlain and Mitchell (1978)

Figure 1 Survival of coliforms in marine and freshwater environments

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 suggest that enterococci also die-off more rapidly in seawater than in fresh water environments (Table 6).

Table 6 Decay rate estimates for *E. coli* and Enterococci in seawater and freshwater

Die-off Rates (in Days) [1]				References
Freshwater		Sea Water		
E. coli	**Enterococci**	***E. coli***	**Enterococci**	
6.3	34.7			Blitton *et al.*, 1983
2.7	4.2			McFeters and Stuart, 1974
3.1	4.5			Keswick *et al.*, 1982
4.6	3.0	0.8	2.4	Hanes and Fragala, 1967
		0.7	2.6	Omura *et al.*, 1982
3.9[2]	**4.4**	**0.8**	**2.5**	

1. Time required for 90% of the population to die off in days
2. Median values

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. Cioglia and Loddo, (1962) showed that Polio, ECHO and Coxsackie viruses decayed at approximately the same rate in marine and freshwaters (Table 7).

Table 7 Survival of Enteroviruses in seawater and riverwater

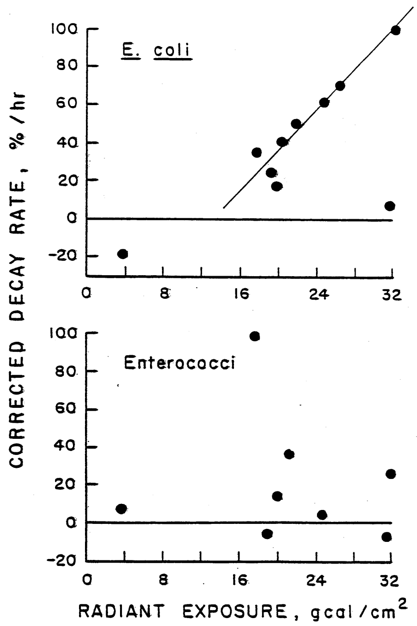
Virus Strain	Die-off Rates (in Days) [1]	
	Sea Water	**River Water**
Polio I	8	15
Polio II	8	8
Polio III	8	8
ECHO 6	15	8
Coxsackie	2	2

1. Maximum number of days required to reduce the virus population by 3 logs
Source: Adapted from Cioglia and Loddo (1962)

If, as appears likely, indicators have different die-off characteristics in marine and freshwater, while viral indicators die-off at similar rates in these environments, then viral pathogens may be present at higher levels in these waters relative to the bacterial indicator numbers. The conclusion may be that higher levels of exposure to viral pathogens may occur in marine waters at similar bacterial indicator levels and this 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 2.



Source: Adapted from Sieracki (1980)

Figure 2 The effect of solar radiation on the die-off of *E. coli* and Enterococci

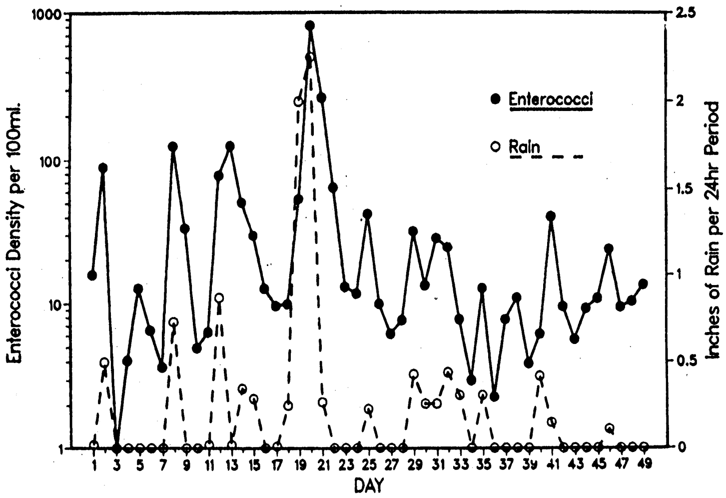
The rate of *E. coli* die-off increases rapidly as solar radiation increases. Conversely, the rate of die-off of enterococci did not increase as the intensity of sunlight increased. Other investigators have observed similar effects of sunlight on indicators. Although human viruses have not been examined under similar experimental conditions, viruses of *E. coli* (coliphages) have been tested and they react 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.

Effects of chlorine

Enterococci and *E. coli* are both sensitive to chlorine, although enterococci are somewhat more resistant to the disinfectant than *E. coli*. For example, to achieve a two-log removal, reported calculated CT values for *E. coli* are in the range of 5 mg min L^{-1}, compared to 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.

Rainfall

Rainfall can have a significant effect on indicator densities in recreational waters. Indicator densities in recreational waters can be increased 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 treatment. In either case, the effect of rainfall on beach water quality can be quite dramatic (Figure 3; R. Calderon, pers. comm.).



Source: Calderon, personal communication

Figure 3 The effect of rainfall on Enterococci densities in bathing beach waters

The effect in this particular setting, a beach on a pond surrounded by forests, was very rapid and usually persisted for one to two days. The highly variable effect rainfall has on water quality can result in frequent beach closings. The important question here is, do high indicator levels that result from animal wastes carried to surface waters by rain water run-off index the same level of risk to swimmers as would exist where the source of the indicators is 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 extra-enteral sources. 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 that the indicator would have any value as a measure of faecal contamination of recreational waters.

The most commonly used indicators for surface water quality, *E. coli*/faecal coliforms and enterococci/faecal streptococci, can readily be detected in the faeces of humans, other warm-blooded animals, and birds (Table 8).

Table 8 Occurrence of Enterococci in faecal samples from humans and other warm-blooded animals

Animal Species	Total No. of Subjects	*E. faecalis* [1]	*E. faecium* [1]
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	-	-

1. percent of total subjects

This list is not exhaustive but it does help 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, then the way in which we currently measure water quality may have to be changed considerably.

1.4 Pollution abatement and water quality

Beaches, especially near urban areas, are often subject to pollution due to sewage and industrial discharges, combined sewer overflows (CSO) and urban runoff. Pollution abatement is therefore a key part of coastal zone management aimed at minimising both 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. Also, combined sewer overflows (CSO) 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.

Treatment

For large urban communities, at least secondary or tertiary sewage treatment plants with disinfection are necessary for on or near shore discharges to protect nearby recreational areas. Public health risks can vary depending on the operation of the plant and effectiveness of disinfection. Smaller communities with lesser population densities usually apply treatment via septic tank systems, latrines, etc. The sub-surface acts as a filter for pathogenic organisms and therefore, such disposal systems result in a very low health risk for recreational areas except in areas with Karst topography where such systems could lead to direct contamination.

The general removal levels of the major pathogen groups by conventional primary, secondary and tertiary sewage treatment are summarised in Table 9.

Table 9 Pathogen removal during sewage treatment

Treatment	Enteric viruses	*Salmonella*	*C. perfringens*[4]	*Giardia*
Raw sewage L^{-1}	100,000-1,000,000	5,000-80,000	100,000	9,000-200,000
[1]Primary treatment				
% removal	50 - 98.3	95.5 - 99.8	30	27 - 64
Nos. remaining L^{-1}	1,700-500,000	160-3,360	70,000	72,000-146,000
[2]Secondary treatment				
% removal	53 - 99.92	98.65 - 99.996	98	
Nos. remaining L^{-1}	80-470,000	3-1,075	2,000	
[3]Tertiary treatment				
% removal	99.983 - 99.9999998	99.99 - 99.9999995	99.9	98.5 to 99.99995
Nos. remaining L^{-1}	0.007-170	0.000004-7	100	0.099-2,951

1. Primary = physical sedimentation.
2. Secondary = primary sedimentation, trickling filter/activated sludge and disinfection.
3. Tertiary = primary sedimentation, trickling filter/activated sludge, disinfection, coagulation-sand filtration and disinfection; note that tertiary does not involve coagulation-sand filtration and second disinfection steps for *C. perfringens*.
4. Source: Long and Ashbolt (1994)

Source: Adapted from Yates and Gerba (1998)

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 PCR (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 nano-filtration methods can also be effective barriers to viruses (over 10^6 removal, Otaki *et al.*, 1998) and other pathogens (Jacangelo *et al.*, 1995; Madireddi *et al.*, 1997). Additionally, revaluation of 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 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 considerably reduce 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 microorganisms may reduce faecal bacterial numbers by 90–99 per cent (Yates and Gerba, 1998). Inactivation of viral and parasitic protozoa is also heavily influenced by temperature. For example, poliovirus type 1 may be inactivated by 99 per cent in five 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 outfalls of sufficient length, diffuser discharge depth and design to ensure a low probability of the sewage plume reaching the designated beach zones. As such, the long outfall is a very low human health risk alternative in that 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 concentrations of organics and nutrients characteristic of 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 further reduce 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 orders-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}).

Pre-treatment with milli-screens with apertures of 1 to 1.5 mm is considered to be the minimum treatment required to remove floatables and thus avoid aesthetic impacts on the designated beach zones. For the same aesthetic considerations, removal of grease and oil should be implemented at the 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 used 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 its inherent pollution potential. Direct injection of sewage to the sub-surface for groundwater recharge is practised in some regions of the world, usually with advanced treatment. Groundwater injection is a no or very low human health risk option for designated beach zones except in Karst topography areas.

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 impacted by the river water quality. The bacterial concentration in river water is determined by faecal pollution from both point and non-point or diffuse sources. Major point sources include sewage effluents, combined sewer overflows (CSOs), industrial effluents and confined animal sources such as feedlots. Non-point sources relate directly to agricultural activity within the watershed, influenced primarily by stock type and density, but a significant contribution is derived from urban surfaces.

Since the transport of microbial contaminants through the watershed to the river and subsequently through the river system to the marine environment is controlled by flow of water, rainfall is a key determinand on concentrations (see Section 1.3). As well as transporting faecal material from the watershed surface to the river, changes in flow are determined by rainfall and the hydrological characteristics of the basin (soils, bedrock, etc) and 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 riverbed. In riverbed sediments, survival times are significantly increased and the bacteria are readily re-suspended when the river flow increases. Combined with increased supply of bacteria from watershed surfaces and some point sources (e.g. CSOs) during rainfall events, all rivers demonstrate a close correlation between flow and bacterial concentration (Figure 4).

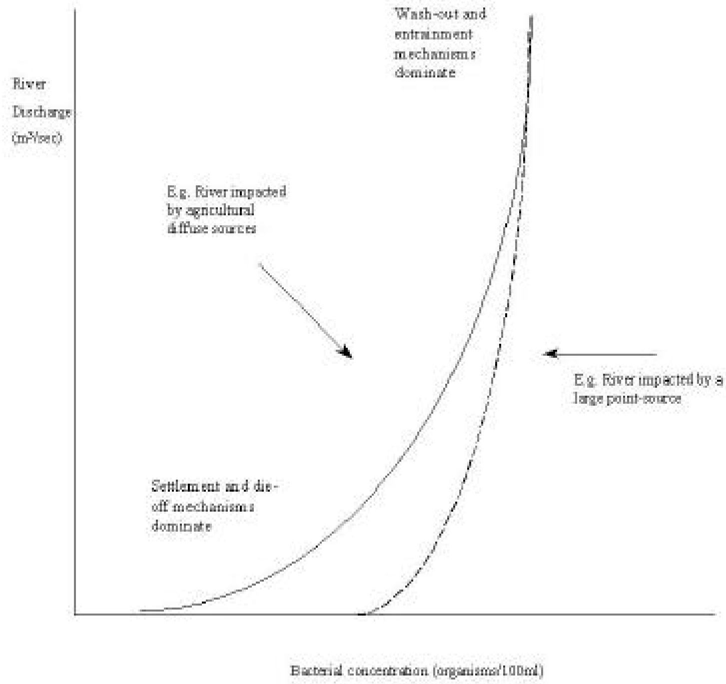


Figure 4 River flow and bacterial concentration

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 bed-sediment (or the catchment surface) store of organisms is exhausted. This phenomenon, however, has only been documented on small streams dominated by diffuse inputs and is less likely 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 (Section 4.1).

2. Alternative approaches to monitoring and assessment programmes

The experts who met in Annapolis in November 1998 agreed that an improved approach to the regulation of recreational water that better reflected health risk and provided 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 was 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. It will be further refined as experience with implementation accumulates. The 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 are also described. Pilot testing of this approach is encouraged.

The proposed approach leads to a classification scheme through which a beach would be assigned to a class (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 to local management actions as well as to 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/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, whilst microbiological quality may vary widely and rapidly even within a small area (section 1). It is of course possible to perform a large number of analyses to obtain an improved 'picture' of the situation 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 to increase the reliability of the overall assessment. This would lead to a series of classes of relative risk as presented schematically in Figure 5.

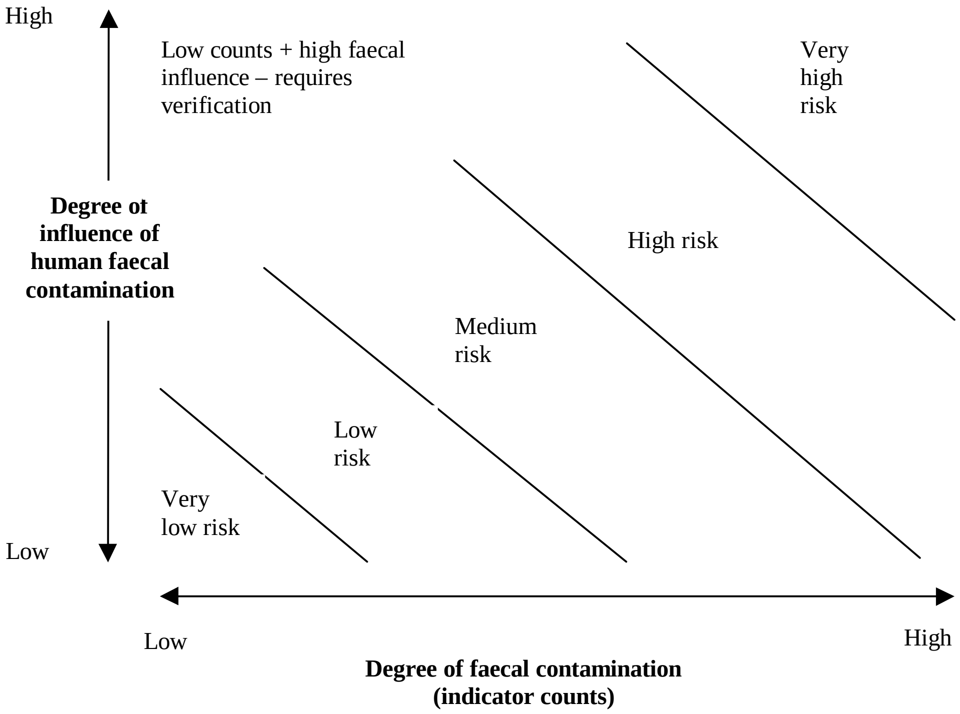


Figure 5 Schematic Representation of Classes of Health Risk

The strengths of such an approach are demonstrated by the case study presented in Box 1.

Box 1: Southern California Case Study

During February 1992, a severe winter storm battered the southern California coastline. Winds, high surf and the deluge of rain took its toll. 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 was leaking into 10 feet of water approximately 90 feet from shore. Water samples were collected directly above the broken pipe and at the adjacent year-round swimming and surfing beach. Coliform concentrations in the samples directly above the pipe break exceeded state standards for recreational contact, but 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 due to the discharge of non-disinfected wastewater. The health officer stated that even though state coliform standards were not exceeded at the beach, "that does not mean viruses that cause gastrointestinal illness, hepatitis or polio aren't present." The health officer's concern stemmed from the fact that activated-sludge treatment alone is only between 90 to 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 health officer. In closing the beach, the health officer took a risk management approach to swimmer health protection, which is to prevent contact with waters known to contain faecal contamination regardless of the density of wastewater "indicator bacteria" measured by the 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 effectively discourage use of areas that are of poor quality, or discourage use at times of increased risk. In addition, if success in discouraging bathing at times of risk can be demonstrated, the classification of a beach might be reasonably upgraded. Since measures to predict and discourage use at times or in areas of elevated risk may be inexpensive, greater cost-benefit and greater possibilities for effective local management intervention are possible.

Figure 6 illustrates the process for assigning a classification to a given beach.

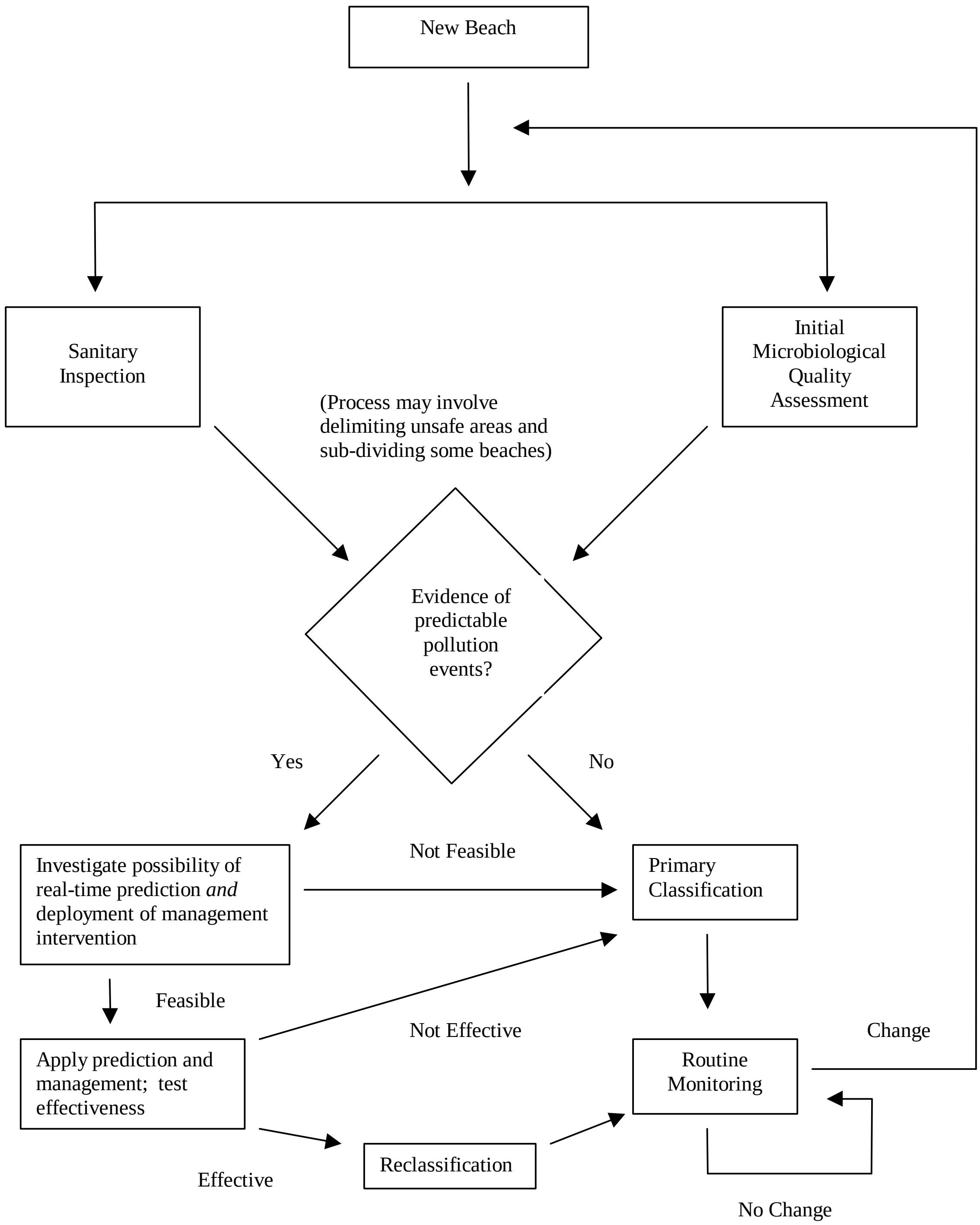


Figure 6: Process followed for a new Beach or Location on entering the Classification Scheme

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); and
- the possibility of "reclassifying" a beach to a higher (better) class if effective management interventions are deployed to reduce human exposure at times or in places of increased risk.

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.

3.1 Sanitary inspection: evaluation of 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 stormwater discharges;
- riverine discharges, where the river is a receiving water for sewage discharges and is either used directly for recreation or discharges near a coastal or lake area used for recreation; and
- bather contamination, including excreta.

All of these sources will lead to the presence of faecal indicators that may be recovered and which may provide a semi-quantitative estimate of health risk as evidenced 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. However, in general, due to the "species barrier", the density of pathogens of public health importance is generally assumed to be less in aggregate in animal excreta than in human excreta and may therefore represent a significantly lower risk to human health. As a result, the use of faecal indicator bacteria alone as an index of risk to human health may significantly over-estimate risks where the indicators derive from sources other than human excreta. Nevertheless, the human health risk associated with pollution of recreational waters from animal excreta is not zero and some pathogens such as *Cryptosporidium* can be transmitted through this route.

The relative risk to human health through direct sewage discharge, riverine discharge contaminated with sewage, and bather contamination has been ranked in the protocol. In doing account is taken 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 is 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.

Sewage discharges

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 beach area; and
- those where discharge is through long sea outfalls, where the sewage is diluted and dispersed and is unlikely to pollute bathing areas.

Whilst the terms 'short' and 'long' are often used, length is generally less important than proper location and effective diffusion which will ensure that pollution is unlikely to reach 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 stormwater run-off and outputs from CSO's are included within the scheme under the category of direct beach outfalls.

The classification is based upon a qualitative assessment of risk of contact/exposure under 'normal' conditions with respect to operation of sewage treatment works, hydro-meteorological and oceanographic conditions.

The potential risk to human health through exposure to sewage can be categorised as shown in Table 10.

Table 10 Risk potential to human health through exposure to sewage

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. 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).
3. Includes combined sewer overflows

The sewage effluent treatments listed in Table 10 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/activated sludge); secondary with disinfection; tertiary (advanced wastewater treatment, including primary sedimentation, trickling filter/activated sludge, and coagulation-sand filtration); tertiary with disinfection; and lagoons (low rate biological treatment). Septic systems are assumed to be equivalent to primary treatment.

Riverine discharges

Riverine discharges are categorised with respect to the sewage effluent load and the degree of dilution (Table 11).

Table 11 Risk potential to human health through 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 includes all the population upstream from the beach to be classified and assumes no instream reduction in 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.

Effluent load is characterised by the total human population in the watershed/catchment above the beach or estuary. The population of relevance is the peak population that 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/discharge during the bathing season. Use of dry weather flow is both 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 in 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 (section 4.1). Episodic input can dominate in areas subject to frequent summer rainstorms such as Northwest Europe.

In practice several discharges into a single river course are likely to occur and where larger discharges are treated to a higher level, then smaller sources (including septic tank discharges) and CSO's may represent the principal source of concern. Pure plug flow is assumed with no 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 varied depending on proximity of the river to the beach.

Bather shedding

While bather shedding is generally of lesser importance than sewage or riverine discharge, pollution is direct and fresh, and therefore potentially of great public health significance. Several studies (section 1.2) have demonstrated accumulation of faecal loads (as indexed 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 for example. The two principal factors of importance are therefore categorised by bather density and degree of dilution (Table 12). Low dilution is assumed to represent no water movement (e.g., lakes, lagoons, coastal embayments). The likelihood of bathers defecating into the water is substantially increased if toilet facilities are not readily available. Under high bather density, the classification should therefore be increased to the next higher class if no sanitary facilities are available at the beach.

Table 12 Risk potential to human health through exposure to sewage from bathers

Bather Shedding	Category
High bather density, high dilution [2]	Low
Low bather density, high dilution	Very low
High bather density, low dilution [1,2]	Medium
Low bather density, low dilution [1]	Low

1. If no water movement
2. Move to next higher category if no sanitary facilities available at beach site

3.2 Microbiological quality assessment

Sewage contamination may be identified by a range of microbial, chemical or visual parameters, as described in Table 5. Each gives a different view of the possible source(s) and thus is appropriately used in a staged approach in 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. These are tabulated for marine and freshwater in Table 13. "Secondary indicators" are described for follow-up analysis to assist in the assessment and management of faecal contamination at beaches.

Table 13 Examples of categories of microbial indicator levels by water source

Water Source	Indicator(s)	Category	95^{th} Percentile
Temperate marine water	Faecal streps Enterococci[3]	A	<10
		B	11-50
		C	51-200
		D	201-1000
		E	>1000
Alternative for tropical marine water [1]	Sulphite reducing Clostridia/ *Clostridium perfringens*	A	<1
		B	1-10
		C	11-50
		D	51-80
		E	>80
Temperate freshwater [2]	Faecal streps Enterococci[3]	A	<10
		B	11-50
		C	51-200
		D	201-1000
		E	>1000
	E. coli	A	<35
		B	36-130
		C	131-500
		D	501-1000
		E	>1000
Optional for tropical freshwater [1]	Sulphite reducing Clostridia *Clostridium perfringe*ns	A	<1
		B	1-10
		C	11-50
		D	51-80
		E	>80

1. Based on preliminary data
2. While studies suggest that there is a differential die-off rate for microbial indicators in marine and freshwaters (see Section 9.1.3), current data are not sufficient to derive separate 95^{th} percentiles for freshwater environments. The above faecal streps/enterococci percentiles are therefore based on data obtained from marine studies, but may be reconsidered when further freshwater studies have been conducted.
3. Source for faecal streps/enterococci 95^{th} Percentile ranges: WHO (1998)

Primary indicators

Minimal non-microbial primary indicators of faecal contamination in marine environments are sanitary plastics/grease. Although a somewhat crude index, they have been used as aesthetic health indicators. Such materials are associated with faecal contamination. In freshwaters, sanitary plastics may also act as non-microbial primary indicators, but grease will not fulfil such a role.

The primary microbial indicators identified are faecal streptococci/enterococci (temperate marine and freshwaters), *E. coli* (temperate freshwaters) and sulphite reducing clostridia/*Clostridium perfringens* (temperate and tropical marine and freshwaters). Table 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 95^{th} 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 sourcing faecal contamination should include sulphite-reducing clostridia/ *Clostridium perfringens* in temperate waters. Consideration must be given to the fact that dog excreta from surface runoff may be a source of these organisms - 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, generally as the level of sewage contamination as referenced by traditional bacterial indicators increases, so does the incidence of swimming related illness. 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 indexed by total and thermotolerant coliforms, *E. coli*, faecal streptococci and enterococci is consistently associated with increased risk to the health of recreators (WHO, 1998).

As noted in section 1, 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. However, this approach 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 end of indicator organism measurements becomes 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, and as such more accurately describes indicator organism densities at a particular location.

The categorisations in Table 13 are based on a minimum of 20 samples of the suggested microbial indicator(s). As the 95^{th} percentile values were derived from limited studies, they are provisional and are meant to serve as a general guideline rather than 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 7 and Box 2 illustrate the steps necessary to assign a primary microbiological categorisation to a given beach.

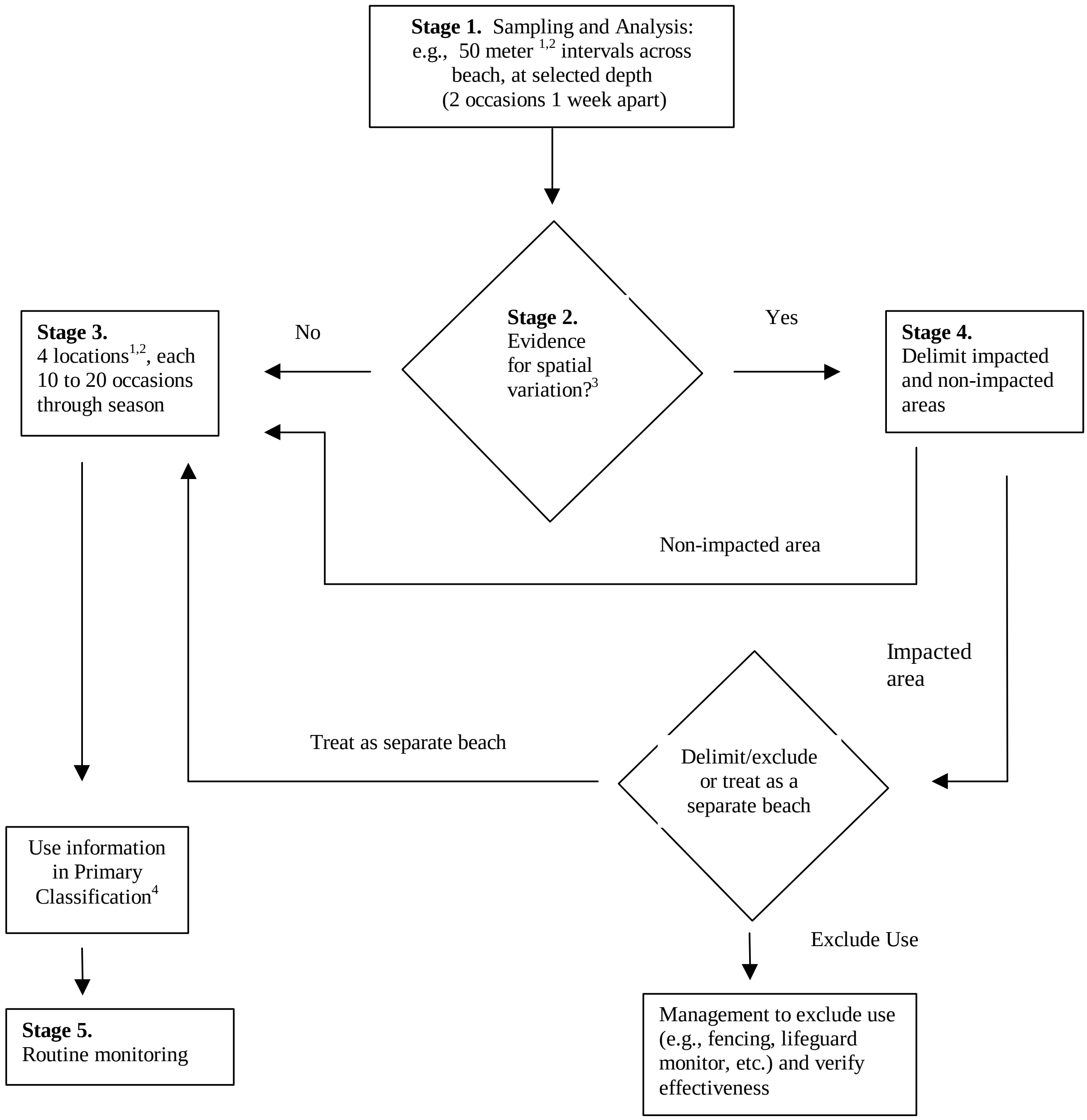


Figure 7: Example sampling protocol for primary microbiological categorization

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1. Less if large historic database.
2. Modified by sanitary inspection.
3. For example, across full band width of microbiological categories.
4. If variation in quality is recognized then reclassification as described in Section 9.4 may be applicable.

Box 2 Example of the practical application of the primary microbiological categorization 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 in Stage 2)

1. Select four evenly distributed sampling locations at no greater than 500 m intervals - if beach is in excess of 2 km, 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 of 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, impacted and non-impacted zones should generally be treated as separate bathing areas and each separately classified.
2. Determine the potential source and extent of the impacted zone.
3. Delimit the non-impacted zone; treat non-impacted zone as in Stage 3 with one of the four identified sample locations at the poorer limit of the impacted zone.
4. For the impacted zone:
   - A monitoring regime for a zone exhibiting spatial variability and likely to be impacted by sewage contamination depends on the extent of the zone.
   - It may be that the impacted 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.* It implies for example fencing combined with general and site warning notices or general and site warning notices plus pro-active individual advice (for instance from life guards) not to use areas. The effectiveness of such management would need to be verified.
   - If the impacted 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 a beach's classification (section 9.5.5).

1. 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.

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 look-up table such as that in Table 14 can be used to determine the primary classification for the beach.

Table 14 Primary classification matrix

		Microbiological Assessment Category (Indicator Counts)				
		A	**B**	**C**	**D**	**E**
Sanitary Inspection Category (Susceptibility to Faecal Influence)	**Very Low**	excellent	excellent	good	good+	fair+
	Low	excellent	good	good	fair	fair+
	Moderate	good*	good	fair	fair	poor
	High	good*	fair*	fair	poor	very poor
	Very High	fair*	fair*	poor*	very poor	very poor

* indicates unexpected result requiring verification

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

4 Reclassification

Microbiological contamination varies widely and rapidly and the risks to human health are principally associated 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*
- the times of contamination can be predicted in some way; *and*
- management interventions can be applied which effectively reduce or prevent exposure at these times, then the beach risk evaluation may reasonably be modified 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. A reclassification should therefore be provisional; 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, due to the conflicts of interest that may arise.

Note that due to the resulting risk assessment it may be appropriate to add an additional dimension 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 are an obvious example. Alternatively this may require an additional 'commentary' element to the classification.

4.1 Simple predictive approaches

It is impossible to predict every type of event that may impact on every beach, as 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 beach's classification to reflect the reduced health risk can be justified.

The issue selected to illustrate this predictive approach is rainfall. To inform the 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, beach inputs must be categorised; primary inputs of concern are combined sewer overflows, riverine and storm drains. Examples of the type of information required for each input are given in Table 15.

Table 15 Discharge sources associated with rainfall

Discharge Source	Background Information	Utility in Prediction
Generic Factors Associated with 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, definition of zone of influence. Catchment and population equivalent loadings 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 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 will 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.	
Combined Sewer Overflows (CSOs)	CSO discharge is 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 typically applied 'rule of thumb' is that effects may become obvious when dry weather flow is exceeded three-fold. 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.	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.

Riverine	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, then rainfall is a likely predictor in a simpler 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., CSO, storm drains, likelihood of sediment resuspension, 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. Resuspension of sediments is a function of extended dry periods and river flow. It is a complex situation with multiple likely sources of contamination, and the health risk is difficult to predict.	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/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 practises will modify the nature and extent of run-off and may vary the threshold.
Storm Drains	Storm drain discharges are associated with localised, generally urban sources. In principle storm drains should not be connected to the sewage system, so therefore should not have a high sewage loading. The likely discharge is generally of low significance to public health, provided that there is no sewage connection. 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, so if they are connected to the sewage system, there will be an increased health risk.	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 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.

A scheme 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 Table 15. While the use of simple predictive approaches requires additional work to plan and implement, they are not highly expensive.

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. While these studies were initially designed 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 of this nature (i.e., "compliance" modelling, budget studies, diffuse source modelling and nearshore modelling) would require tens to hundreds of thousands of US 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 twelve 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 a set of reliable microbiological data covering several years and possibly several locations, as well as a set of variables that have been proven 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 modelling 'success' should be judged on the basis of the explained variance (R^2) of the predictive multivariate model assuming statistical significance. R^2 values of over 60 per cent for a particular beach year have been achieved in previous work. Clearly, 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 multicollinearity) is required in model construction and constant input from a professional statistician in model construction is 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 that the inputs to a bathing water be characterised. It is vital that both low flow and high flow inputs be measured, and quantity and quality measurements are also required.

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 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.

The budget studies will 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 an event will not pose a large risk to public health if 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, and the construction of a diffuse source model of the upstream catchment can offer evidence of the contamination being derived from non-sewage sources. Decisions on remediation strategies can therefore be informed by these studies.

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.

Nearshore 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 nearshore 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, and this requires new scientific information. In addition to these data, elements such as wave height and sedimentary resuspension may be important predictors of microbiological contamination, but are not specifically addressed in current modelling systems.

This approach requires complex finite element modelling, and high level expertise is necessary to successfully use this approach to predict compliance in shallow nearshore waters. However, such approaches can accommodate both constant and episodic inputs to bathing waters, dynamic change in the nearshore waters, and impact under different tidal states and hydro-meteorological conditions.

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, both 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.

Routing monitoring should be undertaken to determine if a beach’s classification status changes over time. If management actions are shown to be effective and a beach can therefore be reclassified, monitoring requirements may be substantially reduced. Examples of classifications and their associated management and monitoring actions are given in Table 16.

Table 16 Examples of classification outcomes and associated management and monitoring actions

Primary classification	Reclassification	Generic statement for public – non verifiable, passive action	Generic management advice - verifiable, active action. Level of action dependent on likely health impact of the event	Monitoring requirements: Sanitary Inspection, Microbiological Quality Assessment
Excellent	-	Excellent beach	N/A	Annual sanitary inspection to ensure no change. Microbiological quality assessment every five years to verify status.
Good	Excellent (defined conditions of contamination)	This beach is of good quality.	No action needed on health grounds. Action may be warranted for local tourist promotion.	Annual sanitary inspection to ensure no change. Microbiological quality assessment every five years to verify status.
Fair	Good (defined area of contamination)	Inform public through advice at beach, tourist locations that bathing at location X is discouraged.	Posting beach - bathing discouraged between specified posts. Restricting access, e.g., not allowing car parking. Discouraging service industries. Fencing area off. Encouraging alternatives via car parks, bus stops and service industries.	Annual sanitary inspection to verify no change. Low-level microbiological quality assessment – four samples on five occasions (equally spaced throughout the bathing season). Abnormal high samples need further verification and additional monitoring and possible review of impacted zone. Annual verification of management intervention effectiveness.
Fair	Good (increased contamination occurs under certain conditions)	Inform public through advice at beach, tourist locations that bathing is discouraged after periods of heavy rainfall.	Posting notice at bathing water. Lifeguards to warn bathers. Closure of car parks and service facilities. Stop tourist buses. Encourage use of alternative beaches, provision of free transport.	Annual sanitary inspection to verify no change. Low-level microbiological quality assessment – four samples on five occasions (equally spaced throughout the bathing season). Abnormal high samples need further verification and additional monitoring and possible review of impacted zone. Annual verification of management intervention effectiveness.
Poor	Good/Fair	This area is of periodic poor quality and bathing is discouraged at certain locations/times.	Active advice similar to that for “Fair” classification.	Annual sanitary inspection to verify no change. Low-level microbiological quality assessment – four samples on five occasions (equally spaced throughout the bathing season). Abnormal high samples need further verification and additional monitoring and possible review of impacted zone. Annual verification of management intervention effectiveness
Very poor	Not affected by local management	This area may be polluted with (nature of pollution) from (define 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 beach. Use poster to inform of alternative locations. Do not allow development of service industries. Make access difficult - 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.

5.1 Direct action on pollution sources

This should be the principal management action as, if successfully undertaken, it will provide 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.

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 impacted zone of the intermittent discharge for a given period following heavy rainfall. Active and verifiable interventions could include posting warnings around the impacted zone following a rainfall event advising bathers not to swim for a period of time. Alongside this, advice could be given as to the location of alternative bathing waters and transportation could be provided to and from those locations. Lifeguards, if present, could reinforce the message. More restrictive measures could be the closure of relevant car parks and service industries (but not sanitary facilities).

5.3 Management interventions on spatial pollution

It is possible for a bathing water to be only partially impacted 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 as outlined in section 5.1 can be undertaken, various options exist for reducing public health exposure. These can range from the passive provision of information to the general public that bathing at the location was not advised, to actively dissuading bathing for instance by not providing public transportation or car parking near the affected area or fencing off the area. As suggested in section 5.2, the policy of dissuasion should be reinforced by information as to alternative bathing areas along with some encouragement, in the form of transport, easier parking or service industries, etc. to entice bathers away from the polluted area.

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 5.1 are not feasible, management actions are needed to reduce the usage 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 service industries. Additionally, information regarding alternative bathing waters of a similar nature but with acceptable water quality needs to be provided.

5.5 Routine monitoring

Under the classification scheme, routine monitoring would always require that an annual sanitary inspection be conducted, 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 undertaken, although the level of monitoring required for a given beach may largely depend upon its classification, as shown in Table 16. 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. Beaches zones greater than two kilometres in length may require additional sampling sites. Further sampling may be necessary if abnormally high samples 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 beach's classification should be revised following a process similar to that described in Figure 6.

6. Evaluation/validation of the proposed approach

A classification scheme of the type proposed would be of value if it accomplishes one or more of the following:

- 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 and thereby reduce 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 the above, both field testing and evaluation of the scientific validity of the approach proposed is required. A limited number of intensive studies would be necessary to test the scientific validity of the approach and in recognition of the importance of this, a protocol has been developed 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 set forth in this chapter.

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 report, which would characterise a bathing water at reasonable cost and within the constraints of limited resources (personnel and equipment/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 insure 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 worldwide. It is desirable that all parties interested in improved monitoring approaches collectively participate 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 all could use (and adhere to) 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.

Microbiological parameters

Two microbial indicators of faecal contamination were selected for this sampling study protocol: faecal streptococci/enterococci and sulphite-reducing *Clostridium/Clostridium perfringens*. The protocol can equally apply to other indicator organisms described in Table 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 as for other environmental testing. There is a large database that describes the precision, accuracy and coefficients of variation for these methods. They were also chosen because they are considered applicable for both marine and fresh water 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 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 and should provide satisfactory data to establish important factors or conditions at a study site which 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 and which will 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 will vary 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 on weekends and holidays). In addition, local events may recur on a routine basis that will affect waters serving a recreational area. The sampling protocol should take account of these factors, so as not to introduce a bias to the data set. Sampling events on sampling days should be on an hourly basis over a 12-hour period, e.g., 7am to 7pm, and these should be at 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 levels 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, this may not be true for short-term evaluations. For such locations this could lead to a gap in the data regarding event contributions to microbial pollutant loading at a beach. If feasible, it is recommended that at least 20 per cent of the study sampling days be during/after rainfall events where there is, or will likely be, local runoff.

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 may likely occur, and also along the designated width of the beach (parallel to the shore line). This then becomes the "box" of water that a single sampling event during a days multiple sampling periods should characterise. This then would comprise a "grid" of sampling locations that would be sampled for each period.

Sampling grid

Spacing of the length of grid samples along the beach should be uniform at 20 meter distances parallel from each other with a minimum of three locations (e.g., minimum of 60 meters total distance). Beaches shorter than this would not be considered valid for incorporation into the sampling validation study.

Sample site distances perpendicular to the shore would be located from the 20 meter grid transects and the locations would be:

1) ankle deep (0.15 meters from grid transect on shore);
2) knee deep (0.5 meters from grid transect); and
3) chest deep (1.3 meters from grid transect).

Samples should be taken at the following depths: ankle depth sample ~ 0.075 meters below the water surface; knee and chest depth samples ~ 0.3 meters below the water surface.

Sand samples, while not an absolute requirement for this sample validation study, are considered desirable data. Sand samples should be taken from the swash zone. Samples should be taken from the top two centimetres of the sand. Enough sand should be taken so that one portion can be used to establish the dry weight and another portion can be used to elute microbial components for quantification.

Sampling/analysis approach

A single, discrete sample would be taken from each location at each period. Each sample will be labelled as to location, day/time taken, and any other distinguishing characteristic needed to identify the sample. Samples would be iced, packaged, and shipped via surface or air transport to the laboratory for processing and analysis. Sample analysis must be initiated within 8-12 hours and all discrete samples would be assayed in triplicate for each dilution (three dilutions - this may be reduced to two if or when the water becomes well characterised for indicator content under various sampling conditions.)

Other test or observational parameters for each sampling period:

Physical/chemical:

- 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).
- percent cloud cover (hourly).
- bather population at each transect point (e.g., photographs) (hourly).
- animal population - presence and number of horses, donkeys, dogs, shore birds (hourly).
- boats anchored or moored within one km of the beach (hourly).
- beach debris and sanitation: sanitary plastics, visible grease balls, algae (daily).
- location of fresh water, storm water, sewage outfall or other intrusion to beach.

- location of bather facilities (showers, lavatories) and relevance of input from these sources (shower runoff, sewage overflow) to beach.

Database requirements

- all raw data should be provided.
- all data would be entered on an EXCEL or LOTUS compatible spreadsheet framework.
- all data entered would be validated for accuracy.
- all data would be duplicated on a separate file for future access.

Analysis (descriptive statistics)

- number of samples taken.
- geometric means - per sample, per replicate, etc.
- standard deviation.
- QA/QC results.
- coefficients of variation, precision, accuracy of methods used by the laboratory.

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