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ADDENDUM TO THE

WHO GUIDELINES FOR SAFE RECREATIONAL WATER ENVIRONMENTS,
VOLUME 1, COASTAL AND FRESH WATERS

LIST OF AGREED UPDATES

Addendum to Guidelines for Safe Recreational Water Environments, Vol 1
World Health Organization – Geneva, Switzerland

CONTENTS

1. INTRODUCTION

This addendum provides updated information based on new scientific evidence to explain matters relating to faecal pollution in the 2003 volume of the Guidelines for Safe Recreational Water Environments, Volume 1, Coastal and Fresh Waters.

The addendum is the product of presentations and discussions that took place at an expert meeting held in January 2009 at World Health Organization headquarters in Geneva. Participants included researchers, regulators, and epidemiologist from seven countries. The meeting was convened to review emerging evidence regarding faecal pollution and human health in connection with recreational bathing waters. Through the course of the meeting it was decided that there was insufficient new evidence nor were there significant advancements in water quality monitoring to warrant a new edition of the Guidelines. Rather, it was decided that updated information based on best available evidence would be better presented in an addendum to the existing Guidelines. Therefore, each participant, according to their expertise, contributed material to this addendum.

The items in the addendum are listed in the numerical order (by page) in which they appear in the Guidelines. Only those references that are not yet listed in the Guidelines are listed in the addendum. For further information regarding the meeting presentations and discussions reference is made to the report of the meeting on WHO Guidelines for Safe Recreational Water Environments Meeting Report, World Health Organization, 14-16 January 2009 - WHO/HSE/WSH/09.07, also available on the Water, Sanitation and Health pages of the WHO web site (www.who.int/water_sanitation_health).

2. UPDATED ITEMS

THROUGHOUT
Replace "Norwalk Virus" with "Norovirus", **except** in Table 4.3.
Likewise on pg. 57, second paragraph, "Norwalk-like viruses" should be changed to Noroviruses.

INTRODUCTION
page xxiiii
Replace paragraph 5 that begins with "Population groups" with:
Some population groups, such as the very young, the elderly, the immunocompromised, and tourists, might be more susceptible to local endemic pathogens and, thus, may be at higher risk to swimming-associated disease. Children are clearly at higher risk because of their swimming behaviour and immature immune systems, while visiting populations may be at higher risk because they have not been previously exposed to local pathogens. Little is known about the risk of disease for the elderly and immunocompromised exposed to recreational waters. Extensive exposure to recreational waters by these higher risk groups should be considered in the development of risk assessments and by managers of water resources.

CHAPTER 4. FAECAL POLLUTION AND WATER QUALITY

Page 54, TABLE 4.1 EXAMPLES OF PATHOGENS AND INDEX ORGANISMS CONCENTRATED IN RAW SEWAGE

Replace second row, "Viruses", with the following:

Viruses		
Adenoviruses	Ocular, respiratory and urinary infections, gastroenteritis	47 600-11 600 000
Enteroviruses	Central nervous system, ocular and respiratory infections	0-3 723
Noroviruses	Gastroenteritis	380-7 100 000
Rotaviruses	Gastroenteritis	400-85 000

Update Footnote (a) with the following references:

Bofill-Mas, et al., 2006; Costán-Longares et al.,2008; Iwai et al., 2009.

Page 55, BOX 4.1 FAECAL STREPTOCOCCI/INTESTINAL ENTEROCOCCI
Replace current Box 4.1 with new Box 4.1 below:

Faecal streptococci and *E. coli* are used to index of faecal pollution in recreational waters. However, they may not be useful in tropical waters due to potential growth in soils/sediments. However, they may not be useful in tropical waters due to potential growth in soils, in fact molecular methods has proved that *E. coli* can become "naturalized" in the environment and do not necessarily indicate recent faecal pollution (Ishii et al., 2007; Ishii and Sadowsky, 2008).

Faecal streptococci is a bacterial group that includes species of different sanitary significance and survival characteristics (Gauci, 1991; Sinton & Donnison, 1994) and species prevalence differs between animal and human faeces (Rutkowski & Sjogren, 1987; Poucher et al., 1991; see Table 9.8 in Bartram & Rees, 2000). The taxonomy of this group has been subject to extensive revision (Ruoff, 1990; Devriese et al., 1993; Janda, 1994; Leclerc et al., 1996) and contains species of two genera—*Enterococcus* and *Streptococcus* (Holt et al., 1993). Although several species of both genera are included under the term enterococci (Leclerc et al., 1996), the species most predominant in the polluted aquatic environments are *Enterococcus faecalis*, *E. faecium* and *E. durans* (Volterra et al., 1986; Sinton & Donnison, 1994; Audicana et al., 1995; Figueras et al., 1998; Borrego et al., 2002). In fresh water *E. faecium* prevails over *E. faecalis* while in seawater occurs the other way around (Figueras et al., 1998).

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

The International Organization for Standardization (ISO) has developed two methods one based on the Membrane Filtration Technique (MF) and the other based on the Most Probable Number (MPN) using a miniaturized 96-well system to enhance precision (Bartram & Rees, 2000—chapter 8). The MF method (ISO 7899-2) employs the classical m-Ent culture media (with 1% sterile solution of TTC incubated for 44 ± 4 h at 36 ±2°C), after which a transplantation of the filter to bile esculin azide agar (incubating for 2 h at 44 ± 0.5 °C) allows for confirming all colonies that appear as dark brown to black as intestinal enterococci. This confirmation step is essential to avoid false positives (Figueras et al., 1996). The MPN (ISO7899-1) enumerates intestinal enterococci on basis to their capacity to growth at 44 ± 0.5°C and of hydrolysing 4-methylumbelliferyl-b-D-glucoside in the presence of thallium acetate, nalidixic acid and 2,3,5-triphenyltetrazolium chloride, in specified liquid medium being the reaction visualized by the emission of fluorescence in 36-72 h. Details are given on the following page.

New approaches to the quantification of faecal indicator organisms in recreational waters are emerging. Molecular methods such as quantitative Polymerase Chain Reaction (qPCR) are being employed in epidemiological studies and showing promise in predicting illness rates in swimmers (Wade et al., 2006; 2008; Ahmed et al., 2008a). Such approaches also have potential utility as a rapid method of water quality

assessment to inform decisions on 'advisory' notices and timely management of health risk at bathing waters. There is an indication of weak but significant correlation (least squares regression R^2 0.46) between intestinal enterococci, enumerated by culture methods (e.g. colony counts from membrane filtration), and genome copy cell equivalents enumerated by qPCR (Haugland et al., 2005). However, it is not recommended that simplistic functional relationships between these parameters are assumed and used to convert between parameter sets because their fate and transport in the environment is very different. It is likely that future epidemiological studies will deploy both culture and molecular methods in parallel and further information on their statistical comparability will emerge in the medium term to underpin a more rigorous comparative evaluation of their public health and management utility.

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

E. coli are bacteria that replaced faecal coliforms as a more specific index of faecal pollution because it is a more specific indicator of faeces from warm blooded animals. It is considered an indicator of recent faecal pollution due to its higher decay rate than intestinal enterococci, both in fresh water and sea water (Table 9.6 in Bartram & Rees, 2000).

Of the two ISO methods, one is based on MF and the other on the MPN (Bartram & Rees, 2000). The MF (ISO 9308-1) allows two alternative procedures the first is the *standard test* and uses lactose TTC agar with Tergitol-7 and requires a probabilistic confirmation of the colonies (at least 10). The second is the *rapid test* that use tryptone soya agar (4-5 h at 36 ± 2°C°C) after which a transplantation of the filter to tryptone bile agar (19-20 h at 44 ± 0.5°C) allows for confirming all the colonies that turn red after the addition of drops of the indole reagent (on their top) as *E. coli*. Transplantation can be avoided if both media are included in the same Petri dish and a programmed incubation is used. This ISO method was designed for drinking water or treated waters and may not be useful for contaminated marine waters or fresh waters with many interfering microbes. The MPN method, ISO 9308-3 (96 wells), enumerates *E. coli* on basis to their capacity to growth at 44 ± 0.5°C in tryptone, salicine triton and of hydrolysing 4-methylumbelliferyl-b-D-glucuronide being the reaction visualized by the emission of fluorescence in 36-72 h. It is worth mentioning that both the MF and MPN ISO methods can provide results for *E. coli* that can be equal or higher than the results obtained for faecal coliforms, because both methods involve less selective conditions that may favour the recovery of stressed *E. coli*. Both MPN ISO methods for intestinal enterococci (ISO7899-1) and *E. coli* had been specifically designed for environmental waters with different degrees of pollution and are not suitable for drinking water because the lower limit of detection is 15 counts per 100 mL. This limit can be reduced to 3.5 counts per 100 mL using 3 microplates per sample, 200 µl per well, 288 wells with a 1:2 dilution (Wiedenmann et al., 2006). Many chromogenic and fluorogenic substrates exist for the specific detection of the same enzymatic activities included in the ISO MPN methods, and various commercial

tests based on these substrates are available (Buckalew et al., 2006; Fricker et al., 2008; Maheux et al., 2008).

While both MF and MPN ISO methods for intestinal enterococci provide quite similar results, this is not the case for *E. coli* where the ISO MPN method can provide higher results (>1 log) than the MF ISO in marine waters with very low levels of faecal pollution, measured by the mean (and range) of intestinal enterococci of 11 (2-36) cfu/100mL. This is due to enzymatic activity from other non-target bacteria (false positives) at low levels of the targeted bacteria (Fiksdal & Tryland, 2008) or even by plant extracts and algae including diatoms (Davies et al., 1994). The MPN methods for intestinal enterococci and *E. coli* had been used for fresh recreational waters in a recent epidemiological study without finding the false positive reactions for *E. coli* mentioned above (Wiedenmann et al., 2006).

Page 56, TABLE 4.2 OUTBREAKS ASSOCIATED WITH RECREATIONAL WATERS IN THE USA, 1985-2006[A]
Replace current table with table below:

TABLE 4.2 OUTBREAKS ASSOCIATED WITH RECREATIONAL WATERS IN THE USA, 1985-2006[a]

Etiological agent	Number of cases	Number of outbreaks
Shigella spp.	1905	25
Escherichia coli 0157:H7	313	15
Leptospira spp.	438	6
Giardia lamblia	76	6
Cryptosporidium parvum	471	8
Norovirus	300	11
Adenovirus 3	595	1
Acute gastrointestinal infections (no agent identified)	2305	42
Naegleria	1	1

[a] Craun et al., 1997; Dziudan et al., 2006; Kramer et al., 1996; Levy et al., 1998; Yoder et al., 2004; 2008.

Page 60, 4.3.2 Risk Assessment
Beginning with the second paragraph that starts "QMRA can be used to…" replace remainder of the section with the following text:

TABLE 4.4 RISK ASSESSMENT PARADIGM FOR ANY HEALTH EFFECT remains as presented.

QMRA can be used to indirectly estimate the risk to human health by predicting infection or illness rates given densities of particular pathogens in recreational waters, assumed rates of ingestion and appropriate dose-response models for the exposed population (US EPA, 2007; Boehm et al., 2009). Application of QMRA to recreational water use is constrained by the current lack of specific water quality data for many pathogens and the fact that pathogen numbers, as opposed to faecal index organisms, vary according to the prevalence of specific pathogens in the contributing population and may exhibit seasonal trends.

These factors suggest a general screening-level risk assessment (SLRA) as the first step to identify where further data collection and quantitative assessment may be most useful. As such, QMRA should be undertaken in an iterative manner to explore where health concerns may arise for the system being modelled. Further, QMRA should be undertaken with stochastic values (using distributions rather than point estimates) to better account for the inherent variability as well as the uncertainty in parameter values. Where estimated risks are consider unacceptable, but uncertainties are high, research to reduce uncertainties is suggested, followed by re-running of the QMRA. Nonetheless, key areas for risk management can still be identified using uncertain QMRA parameter values and initial point estimates (in a screening-level risk assessment).

Caution is required in interpreting the results of a QMRA because the risk of infection or illness from exposure to pathogenic microorganisms is subject to many uncertainties. It should also be recognized that microbial risks are fundamentally different from the risk associated with other contaminants, such as toxic chemicals. Consequently, QMRA has greatest utility to aid in risk management (Section 4.3.3), where relative changes in estimated risks to various scenarios can be explored to provide insight into where management may be most beneficial. Also QMRA can explore risks either below epidemiologically detectable levels or under circumstances that are not suited to epidemiological examination.

Several of the key differences between exposure to pathogens and toxic chemicals are:

- Exposure to pathogens can occur via an environment-to-person pathway, but can also occur due to person-to-person contact (secondary spread);
- whether a person becomes infected or ill after exposure to a pathogen may depend on the person's immune status. This condition implies that exposure events are not independent;
- infectious individuals may be symptomatic or asymptomatic;
- different strains of the same pathogen have a variable ability to cause disease (differing virulence);
- this virulence can evolve and change as the pathogen passes through various infected individuals; and
- pathogens are generally not evenly suspended in water.

Although the differences between exposure to chemical agents and pathogenic microorganisms are widely acknowledged, the conceptual framework for chemical risk assessment (Table 4.4) has been commonly employed for assessing the risk associated with exposure to pathogenic microorganisms. Frameworks have been developed specifically to assess the risks of human infection associated with exposure to pathogenic microorganisms and to account for some of the perceived shortcomings of the chemical risk framework with respect to properties unique to infectious microorganisms. However, to date, these frameworks have not been widely adopted, such as those that attempt to account for prior immunity and secondary spread of infectious agents, so called dynamic models (Soller & Eisenberg, 2008) rather than the more common static (or single pass) models.

In employing the chemical risk framework to carry out a SLRA, representative pathogens for viral, bacterial and parasitic protozoan pathogens (reference pathogens) are used to conservatively characterize each pathogen group. For example, the occurrence of adenovirus, with its associated dose–response curve, may be used as a predictor for all enteric viruses. As such, conservative estimates of exposure to each reference pathogen are initially used to characterize "total" risks from each of the groups of pathogens. The results of the SLRA, often only calculated the end point of infection (not disease), should then indicate an order of magnitude estimate of risk, whether or not further data are required and if risks are likely to be dominated by a single class of pathogen or source (potentially defining options for risk management). It should be emphasized that this SLRA approach presumes that little net error is made by not accounting for either person-to-person transmission of disease or immunity.

Despite the somewhat limited array of microorganisms and exposed sub-populations for which dose–response relationships have been estimated, there is a sufficient array of reference pathogens to at least undertake a SLRA. The range of microbes from which to select relevant reference pathogens for a particular site include: for the enteric viruses, rotavirus, adenovirus, and norovirus; for enteric bacteria, *Salmonella enterica* (various serotypes), *Campylobacter jejuni*, and *E. coli* O157:H7; and for parasitic protozoa, *Cryptosporidium parvum, and Giardia lamblia* (Haas et al., 1999; US EPA, 2005; Teunis et al., 2008). A screening-level QMRA and risk management approach is outlined for a recreational water example in Box 4.2 (adapted from Roser et al., 2006; 2007).

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

The primary advantages of QMRA studies are that the potential advantages and limitations of risk management options may be explored via numerical simulation to examine their potential efficacy, and that risk below epidemiologically detectable levels may be estimated under certain circumstances. The limitations of QMRA studies, as noted earlier, are that limited data are available to carry out these

assessments and, in many cases, the data that are available are highly uncertain and variable. Nevertheless, it may be inferred from several of the available QMRA studies (Sydney and Honolulu) (Mamala Bay Study Commission, 1996; Ashbolt et al., 1997) that they provide supporting evidence for the results of various epidemiological studies.

Thus, QMRA can be a useful tool for screening the risk to public health at recreational water sites and for determining the potential efficacy of management alternatives through the integration of a wide array of disparate data. Finally, QMRA provides credible scientific analysis that can be used in conjunction with or, at times, in lieu of epidemiological investigations to assess risk to human health at recreational water sites.

Page 62, BOX 4.2 SCREENING-LEVEL QMRA APPROACH FOR BATHER RISK
Replace Box 4.2 with the following text:

A freshwater lake that had been closed to swimming due to high faecal indicator levels was evaluated to see if under certain conditions it would be suitable for recreational activity in Sydney, Australia (Roser et al., 2006; 2007). Historic monitoring data and a recent sanitary inspection around the 10.5 hectares lake identified background *E. coli*/enterococci contamination due to waterfowl and periodic contamination due to sewer overflows via the major inflow creek. The concentration of pathogens in waters may be estimated from the mean pathogen densities in raw sewage and their dilution in recreational waters (based on the numbers of index organisms to pathogens; see Table 4.1). As an initial conservative approximation of pathogen numbers in recreational waters, *E. coli* or enterococci may be used as an index for the dilution of sewage-associated bacterial pathogens (e.g., *Salmonella*) and spores of *Clostridium perfringens* or enterococci for the enteric viruses and parasitic protozoa. Alternatively, direct presence/absence measurement of pathogens in 1-10-L volumes of recreational waters may be attempted (Reynolds et al., 1998; Calgua et al., 2008).

An additional important factor in highly transparent water bodies is that of solar inactivation. Modeling solar inactivation is possible based on the work of Davies-Colley et al. (2000) (equation below). This model explains how it is possible to have widely different estimates for the first order reaction constant *k*, and that it can be important to incorporate estimates of solar radiation exposure:

$$k = k_d + k_s S = k_d + k_s \frac{S_0}{KH}$$

Where k_d = dark inactivation coefficient, k_s = sunlight inactivation coefficient and S is the insolation averaged over the water column of depth *H* and light attenuation coefficient *K*, and *So* is insolation incident at the water surface.

In optimised experimental reactors the solar irradiance required for 90% reduction (S_{90}s) are typically 2.5-5 $MJ.m^{-2}$ for bacterial and F-RNA coliphage faecal indictors (e.g. Davies–Colley et al., 2005), and for *Enterococcus faecalis*, *Clostridium perfringens* and the DNA bacteriophage PRD1, 1-2 $MJ.m^{-2}$. For comparison the radiation on sunny summer days in Sydney is about 20-35 $MJ.m^{-2}.d^{-1}$ and seldom drops below 10 $MJ.m^{-2}.d^{-1}$ in summer. Hence the stormwater microbial inflows to the lake could be reduced by over five logs and measured T_{90S} (time for 90% reduction due to sunlight) were 1 to 2 days.

After the general concentrations of pathogens from the three microbial groups have been determined, selected reference pathogens are used for which dose–response data are available (e.g., *Campylobacter jejuni, Salmonella enterica*, *Cryptosporidium parvum*, *Giardia lamblia*, rotavirus and adenoviruses) (WHO, 2004). Note that these specific pathogens may not necessarily be the major etiological agents, but are used as characteristic of the likely pathogens. Risks from viral, bacterial and protozoan pathogens can then be characterized per exposure by applying published dose–response models for infection and illness (Haas et al., 1999).

Hence, using the approach described by Ashbolt et al. (1997) and assuming sewage as the primary source of pathogens (faecal sterols indicated a primary dilution factor of $3x10^{-5}$ [e.g. *ca* 1:30,000]), accidental ingestion of 20-50 mL per swim (Dufour et al., 2006), and bather shedding described by Gerba (2000), the estimated infection risks were generated for five scenarios (Table 4.5).

During dry weather (Scenario 1, Table 4.5) the maximum infection risk for a given pathogen per bathing event was 2.7 x 10^{-6}, supporting the conclusion that at such times primary contact should be acceptable (threshold risk of illness being about 5% for gastroenteritis, Table 4.7). With increasing rainfall input and shorter Lake recovery times (Scenario 2) risks from run-off increased and in the case of *Campylobacter* approached the proposed benchmark (5 x 10^{-3} infection probability.person^{-1}.exposure^{-1}). Scenario 3 illustrated the importance of allowing water quality to recover and the effect on infection risk of not making this allowance. Scenario 4 generated two more risk estimates close to the proposed pathogen benchmark. These high risks arise not only from the input load but also from the use of a more conservative dilution factor. This increase indicates an additional day recovery might be advisable following a very large event (>100 mm). Scenario 5, shedding of pathogens by bathers, appears to be of greatest concern. The risk estimate for rotavirus indicated an infection risk on average of 9.2x10^{-2} person^{-1}.exposure^{-1} despite high-simulated dilution. This high value arose due to the following: the high numbers of viral particles released per bather shedding event (1 event per 1000 bathers (Gerba, 2000) with 1.4 x 10^{4} protozoa per bather per contact day and 1.4 x 10^{7} enteric viruses per bather per contact day); the low dose-response relationship reported in the literature and used in the simulation; and the inclusion of excretion by children in the estimate of the quantity of average faecal matter excreted, which Gerba (2000) estimates is about 100-times higher than for adults.

TABLE 4.5 - RISK ESTIMATES (INFECTION PROBABILITY.PERSON^{-1}.EXPOSURE^{-1}) FOR LAKE SWIMMERS UNDER FIVE DIFFERENT SCENARIOS

Reference Pathogen	Scenario				
	1. Dry Weather	Different amounts of rainfall			5. Bather shedding[b]
		2. Management Trigger Threshold Risk (9.9 mm previous night)	3. Substantial (40 mm) Event followed by three days recovery	4. Large Event (180 mm), Epilimnion Displaced, Five days recovery	
Cryptosporidium	2.5x10^{-6}	1.9x10^{-4}	8.1x10^{-5}	**1.6x10^{-3}**	9.3x10^{-5}
Giardia	3.8x10^{-7}	2.9x10^{-5}	1.2x10^{-5}	2.4x10^{-4}	2.8x10^{-6}
Rotavirus	2.3x10^{-8}	1.8x10^{-6}	7.4x10^{-7}	1.4x10^{-5}	**9.2x10^{-2}**
Enterovirus	6.9x10^{-11}	5.2x10^{-9}	2.2x10^{-9}	4.3x10^{-8}	3.5x10^{-4}
Salmonella	2.3x10^{-11}	1.8x10^{-8}	7.4x10^{-10}	4.6x10^{-9}	-
Campylobacter	2.7x10^{-6}	**2.1x10^{-3}**	8.6x10^{-5}	**5.4x10^{-4}**	-

[a] Infection probabilities close to, or exceeding, the proposed benchmark probability range (0.5-2 x 10^{-3}) are shown in **bold**.

[b] Shedding risks were calculated separately to risks from run-off.

Risk Management Guidance from the QMRA

Because of the variation in run-off event sizes it was seen as essential to estimate Lake recovery rates and using these estimates develop a defensible scheme for estimating when the Lake water quality was likely to have recovered. Within the limits of the method it was estimated that recovery for run-off events involving > 10 mm of rainfall in the previous 24 h would require between 1.5 and 4 days or cumulative global radiation exposure of between 40 and 93 MJ.m^{-2} assuming that solar radiation was the primary inactivation agent.

Hence, the proposed water safety plan for the lake was based on:

- During ‘dry weather’ the small pathogen inputs from contaminated stormwater are unlikely to pose a significant risk to bathers;
- 10 mm of rainfall appears to be a rational trigger for managing access;
- Following substantial rainfall events the probability of infection from pathogens rises to levels of concern, and a no-swim period of a few days (two to three days) should allow for solar radiation driven suitable for primary contact recreation; and
- Bather shedding appears to be the greatest concern during Lake use and hence the need to emphasize good hygiene and potentially limit lake recreator numbers. However, this can be reduced with toilet facilities and education campaigns.

Page 63, 4.3.3 Risk Management
Replace entire sub-section with text below:

To meet health targets ultimately based on a tolerable risk of illness (see section 4.4), achievable objectives need to be established for water quality and associated management. Hazard analysis and critical control point (HACCP) or what has evolved for the water management, water safety plans (WSP) provide an approach. A WSP promotes good operational/management practice and effective quality assurance (QA), similar to that used in the food and beverage industry (Deere et al., 2001) since its codification in 1993 by the Food and Agriculture Organization of the United Nations and WHO *Codex Alimentarius* Commission. WSP for drinking water were developed from the HACCP approach (Davison et al., 2006; Bartram e al. , 2009) and are equally applicable to recreational and reuse water management.

An example WSP outline for recreational waters is described in Table 4.6. This risk management procedure should be approached in an iterative manner, with increasing detail proportional to the scale of the problem and resources available. By design, the WSP addresses principally the needs for information for immediate management action; when applied to recreational water use areas, however, its information outputs are also suitable for use in longer-term classification.

Variation in water quality may occur in response to events (such as rainfall) with predictable outcomes, or the deterioration may be constrained to certain areas or sub-areas of a single recreational water environment. It may be possible to effectively discourage use of areas that are of poor quality or discourage use at times of increased risk. Since measures to predict times and areas of elevated risk and to discourage water contact during these periods may be inexpensive (especially where large point sources are concerned), greater cost effectiveness and improved possibilities for effective local management intervention are possible (see section 4.7).

Page 64, TABLE 4.6 IMPLEMENTATION OF HACCP APPROACH FOR RECREATIONAL WATER MANAGEMENT

Replace entire table with table below and rename table as follows

"TABLE 4.6 WATER SAFETY PLAN OUTLINE FOR RECREATIONAL WATER MANAGEMENT"

Component	Action
System Assessment	
Assemble WSP team	• The team is formed to steer the overall process. Composition of the team should represent all stakeholders and cover all fields of expertise as much as possible. Representatives of health agencies, microbiologist responsible of the analysis, user groups, tourism industry, water and sewage industry, communities, competent authorities, potential polluters, experts in hazard and risk analysis, etc., should all therefore be considered.
Document and describe the system	• Summarize previous data from sanitary surveys, compliance testing, maps specifying sewage inputs, overflow points and stormwater pipes and overflows. • Determine if there are major animal faecal sources within the recreational water catchment. • Reference development applications and appropriate legal requirements. • If no (historic) microbiological/sanitary data are available, collect basic data to fill data gap/deficiency.
Produce and verify pollutant flow charts	• Produce and verify source-to-water flow charts for faecal pollution from source(s) to recreational exposure area(s) for each recreational water catchment. This may require a new sanitary survey. • The series of flow charts should illustrate what happens to water between source(s) and exposure in sufficient detail for potential entry points of different sources of faecal contaminants to be pinpointed and any detected contamination to be traced (WHO, 2009). • This information may best be summarised in conceptual diagrams for normal and potential event conditions to aid in pollutant management (WHO, 2009).
Hazard Identification & Risk Prioritisation	
Identify potential hazards	• Identify human versus different types of animal faecal pollution sources and potential points of entry into recreational waters as either a risk priority or major risk.
Determine existing control measures	• Determine significance of possible exposure risks (based on judgement, quantitative and qualitative risk assessment, as appropriate). • Identify existing control measures to prevent/reduce exposure.
Risk prioritisation	• Identify preventive measures (control points) for all significant risks & prioritise. (See Box 4.2)
Operational Monitoring to Support Risk Management	
Operational monitoring and selection of operational control parameters	• Establish a monitoring regime to give early warning of exceedances beyond operational limits (see Box 4.2, section 4.7.7). Those responsible for the monitoring should be closely involved in developing monitoring and response procedures. Note that monitoring is not limited to water sampling and analysis, but could also include, for example, visual inspection of potential sources of contamination in catchment, flow/overflow gauges, change in river heights, amount of rainfall, wind speed, and direction.
Establish corrective action for deviations	• Identify those points or locations at which management actions can be applied at defined control points to reduce the presence of, or exposure to, hazards to acceptable levels. Examples include municipal sewage

that may occur	discharge points, treatment works operation, combined sewer overflows, illegal connections to combined sewers, etc. • Determine measurable control parameters (e.g. salinity deviation from normal values for seawater) and their operational limits. Ideally, assign target and action limits to pick up trends towards operational limits (e.g., >10–20mm rainfall in previous 24-h period that based on historic data analysis would exceed the beach microbiological criteria or notification of sewer overflow by local agency).
Incidents and emergency responses	• If the corrective action does not bring the system back under control, or if some unforeseen event occurs, it's possible that water quality and safety could become compromised. Under such circumstances a major response is required to prevent potentially significant health impacts. Such broad responses are often terms 'incidents' or 'emergencies'. Under such circumstances signs prohibiting bathing and announcement through load speakers may be the responses as well as fencing and posting signs in the affected area.
Verification and Audit	
Verification	• Obtain objective evidence that the envisaged management actions will ensure that the desired water quality will be obtained or that human recreational exposures will be avoided; e.g. inspect the site for absence of bathers. This would draw from the literature and in-house validation exercises.
Auditing	• Obtain objective data from auditing management actions that the desired water quality or change in human exposure is in fact obtained and that the good operational practices, monitoring and management actions are being complied with at all times.
Supporting Programmes and Management Procedures	
Management	• Prepare and test actions to reduce or prevent exposure in the event of critical limit actions being exceeded. Examples include building an appropriate treatment and/or wastewater disposal system, training personnel, developing an early warning system for bathers, issuing a media release and (ultimately) closing the area for recreational use.
Record keeping	• Ensure that monitoring records are retained in a format that permits external audit and compilation of annual statistics. These should be designed in close liaison with those using the documents and records.

Page 69, BOX 4.4, PERCENTAGE CALCULATION
Replace entire Box 4.4 with content below:

Inappropriate choice of methods for calculation of the 95th percentile or their use on limited data sets is liable to lead to substantial error that can lead to misclassification of recreational waters.

Problems with small numbers
The standard error of any percentile calculation is inversely proportional to the square root of the number of data points included in the calculation and also increases with the variance in the underlying data and the distance of the percentile from the median. This means that any beach classifications made on small numbers of microbiological test results are liable to considerable uncertainty. The following figure illustrates the impact of different sample sizes on the rates of misclassification.

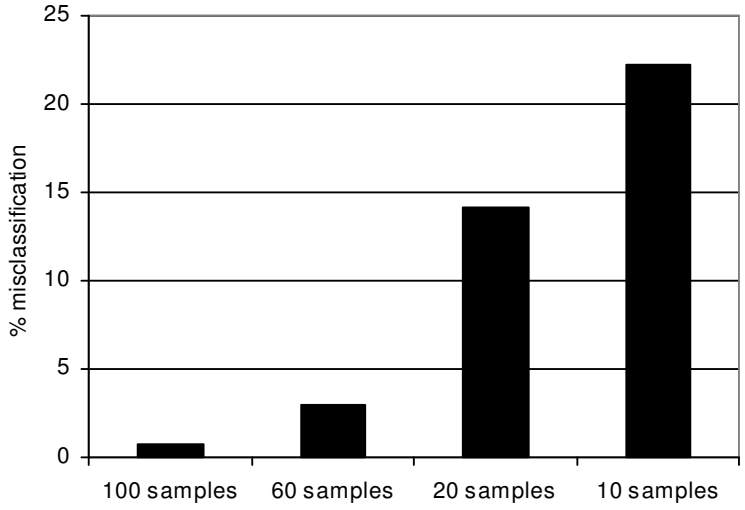

FIGURE 4.4. MISCLASSIFICATION RATES IN BATHING WATERS USING PARAMETRIC 95TH PERCENTILE VALUES ASSUMING A STANDARD DEVIATION OF LOG VALUES OF 0.8 WHERE ACTUAL 95TH PERCENTILE WERE GENERATED TO BE ½ OR 2 TIMES THE STANDARD.

If compliance is estimated from 100 samples, as may be accrued over five bathing seasons with 20 samples per season, the probability of misclassification is less than 1%, but if based on just one season's 20 samples then this would be about 14%. Thus, estimating compliance on too few samples is unlikely to protect public health by allowing too many beaches to pass or protect the interests of beach managers by failing too many good quality beaches. The only exception to this would be very poor water quality beaches where most or all of the first five to ten samples exceed the 95th percentile classification point.

Problems with censored data
Censored data are results that are above or below the limit of detection. This is a common occurrence in bathing water data, which are frequently "left censored" (e.g., data are below the detection limits of the microbiological method). This is especially a problem in good quality bathing water. When censored data is replaced by a single variable such as the limit of detection this leads to a systematic bias in the calculation of the mean and 95th percentile. Furthermore the magnitude and direction of this bias can not be easily predicted (El-Shaarawi & Esterby, 1992) which means that a bathing

area with very low values <10ufc/100mL and that sporadically shows some values as 40 or 60 cfu/100mL may have a 95th percentile above the standard for excellent water quality. A number of adjustments to the parametric calculation of the 95th percentile have been proposed but none of them work well under all circumstances (Hewett & Ganser, 2007).

Problems with distributions other than log normality

The parametric estimation of the 95th percentile is based on the assumption that the data is normally distributed after being log transformed. When this assumption is not correct then there will be errors in the calculated mean and so 95th percentile (Schmoyer et al., 1996). In a study of Irish bathing water quality datasets, the data were not log-normally distributed in 85% of sites where there was disagreement between the parametric and % compliance results (Chawla & Hunter, 2005). It would appear that many bathing water datasets are not log normal, nor is it usually possible to determine the underlying distribution, in part because of censoring within the data or because in excellent bathing areas the low values will prevail.

Recommendations

Bathing water classification cannot be made on small numbers of samples unless the bathing water quality is very poor, otherwise there will be a significant number of misclassifications. At least 60 samples are required and preferably 100 samples.

The parametric calculation of percentiles cannot be recommended for the numerical basis of bathing water classification unless there are no results that are outside the limits of detection of the microbiological method and the dataset is shown to be log normally distributed. Both these conditions are likely to be uncommon. A non-parametric method is more appropriate. The Hazen method has been shown to be the least biased estimator (Hunter 2002, Bartram & Rees, 2000—Chapter 8).

Hazen is a ranking formula where the data are ranked in ascending order. The rth count is then the value of the appropriate percentile where r is given by formula:

$$r = \frac{1}{2} + \frac{Pn}{100}$$

Here *P* is the percentile and *n* the number of values in the dataset (how you practically derive the Pile from this formula an example is required). Because the above formula rarely gives an exact rank, the percentile is calculated by interpolation between the two data-points on either side of the calculated rank. So assume that we have 100 samples for the calculation of the 95th percentile (the 95th percentile = 5th largest out of 100)

$$r = \frac{1}{2} + \frac{5 \times 100}{100}$$

$$r = 5.5$$

So the 95%ile is half way between the 5th and 6th largest value. If the 5th largest value is 115 and the 6th is 111 then the 95th percentile would be 115 - (115-111) x 0.5 = 113.

Now assume only 46 samples;

$$r = \frac{1}{2} + \frac{5 \times 46}{100}$$

$$r = 2.8$$

Hence, the 95th percentile is somewhere between the 2nd and 3rd largest value, but closer to the 3rd largest. If the 2nd largest number is 115 and the 3rd is 111 then the 95th percentile would be 115 - (115-111) x 0.8 = 111.8.

Page 71, Table 4.7. GUIDELINE VALUES FOR MICROBIOLOGICAL QUALITY OF RECREATIONAL WATERS, Continued

Replace current Footnote 7, with text below:

This table may not apply to children, the elderly, or immunocompromised persons because it was developed from studies of young, healthy adults. Presently available data on these special groups (children) does not lend itself to quantifying the degree of protection needed by at higher risk groups and, therefore, no correction factors are applied to the current guidelines.

Add Footnote 8, with text below:

Epidemiological data on fresh waters, obtained using the same methodology as in the seawater study which formed the basis to calculate the attributable risks listed in the table, yielded considerably lower attributable risks at the same concentrations of faecal indicators, while relative risks (2.6 in the fresh water *versus* 2.5 in the seawater study, at values above 50 IE/100 mL) and the threshold of effect (38 IE/100 mL [95th percentile value]) were very similar (Wiedenmann et al., 2004; 2006). This might be partially explained by a higher ratio of pathogens to indicators in seawater. But it might have primarily resulted from a lower susceptibility to infection or disease of the participants in the fresh water study, which could also explain the much lower disease rates in the non-bathers group. As a high susceptibility to infection or disease, dependent on the epidemiological situation present at the time of the study, and this might also occur in freshwater environments (Fewtrell et al., 1992, Prüss, 1998), it is recommended that the information given in the table should also be used to set indicator standard values in freshwaters. The same applies for exposures other than swimming (e.g., high exposure activities such as surfing, dinghy boat sailing, or whitewater canoeing). In addition, it is recommended that the length and frequency of exposure encountered by special interest groups (such as body surfers boat riders, wind surfers, sub-aqua divers, canoeists and dinghy sailors) be taken into account.

Page 71, BOX 4.5 DIFFERENTIAL DIE-OFF OF INDEX BACTERIA AND PATHOGENS IN SEAWATER AND FRESH WATER

Change sentence before Table 4.8 to:

Cioglia & Loddo (1962) showed that poliovirus, echovirus, and coxsackie virus were inactivated at approximately the same rate in marine and fresh waters, but it is important to note that other factors, such as water temperature, are more important than salinity for virus inactivation (Gantzer et al., 1998). Information on survival of viruses in seawater and freshwater is shown in Table 4.8.

Refer to updated Table 4.8 on following page.

Page 71, BOX 4.5 DIFFERENTIAL DIE-OFF OF INDEX BACTERIA AND PATHOGENS IN SEAWATER AND FRESH WATER

Replace Table 4.8 with the table below:

TABLE 4.8. SURVIVAL OF VIRUSES IN SEAWATER AND RIVER WATER

	Die-off rates (in days)	
Virus Strain	**Seawater**	**River water**
Adenovirus 40, 41[a]	1 LTR in 40 days	3.2 LTR in 60 days. 1 LTR in 40 days
Enterovirus	1 LTR in 4 days [b]	1 LTR in 2 days in river water "in situ" [c]
Hepatitis A[d]	2 LTR in 28 days	Estuarine water: 2 LTR in 28 days
Rotavirus[e]	Bovine Rotavirus decay 0.5 LTR per day	Group A virus: 2 LTR > 64 days in tapwater or 10 days in river water ; Group B virus: 3.2 LTR in 60 days

LTR: Log titre reduction

a. Enriquez et al., 1995.

b. Bitton, 1978.

c. O'Brien & Newman, 1977.

d. Sobsey et al., 1988.

e. Loisy et al., 2004; Terrett et al., 1987; Vondefecht et al., 1986.

Page 72, Add after 2nd Paragraph (Section 4.4.4 Guideline values for fresh water)
The faster die-off of index bacteria than certain pathogens, especially viruses (Enriquez et al., 1995) in sea water compared with fresh water (Box 4.5), and the significant differences in swimming-associated GI rates in seawater swimmers and freshwater swimmers at a given level of faecal index organisms observed in certain epidemiological studies as discussed by Dufour (1984; 2007) has prompted national and international authorities to set different standards for seawater and freshwater (e.g., US EPA since 1983, EU since 2006).

Page 72, 3rd full Paragraph (Section 4.4.4 Guideline values for fresh water)
Replace paragraph starting with "Studies using a randomised trial design have been conducted in Germany at fresh water sites. These have…" **with the following text**:

When Wiedenmann et al. (2006) repeated the key studies mentioned in 4.4.3 using a similar randomised controlled trial design in German freshwater bathing sites, and compared their results to those of the eligible studies reviewed by Prüss (1998), they came to the conclusion that the highly variable attributable risks ('excess risks') of gastroenteritis (0.4 to 27.7% in freshwater and 0.5 to 19.5% in seawater studies) were most likely due to differences in the cohort susceptibility as demonstrated, e.g., by Cabelli et al. (1982) for local residents and tourists at Egyptian beaches and in other studies (Prüss 1998, Wiedenmann et al. 2004 and 2006, Dizer et al. 2005, Wiedenmann 2007a). They suggested that in order to account for variable susceptibilities of bathers in different epidemiological studies, a rationale different from the one based solely on the "disease burden" concept might be necessary for relating standards to potential health threats. This rationale might be based on threshold concentrations (No Observable Adverse Effect Level; NOAEL's), above which disease rates start to increase (Wiedenmann, 2007b), or on relative rather than attributable risks.

Page 76, FIGURE 4.4
Change to FIGURE 4.5.
A new figure was added in Box 4.4; therefore the subsequent figure on pg 76 should be changed from Figure 4.4 to Figure 4.5.

Page 82, 4. Animal Inputs
Insert text below after first paragraph on animal inputs:
The emerging technique of Microbial Source Tracking (MST) offers the potential for determining the source of faecal indicator concentrations in a receiving water used for recreational activities. The suite of approaches was recently reviewed in a collection of papers published in Water Research (Vol. 41) and national agencies have produced useful review papers (US EPA, 2005; Edge, 2006) and other valuable review papers can be accessed in the international literature (Ahmed et al., 2008; Field et al., 2003; Field & Samadpour, 2007; Gourmelon et al., 2007; Ishii & Sadowsky, 2008; Isobe et al., 2004; McLaugin et al., 2007; Reischer et al., 2008; Santo Domingo et al., 2007; Stoeckel and Harwood, 2007; Vogel et al., 2007). This suite of methods range from chemical approaches, such as the detection of sterols and fluorescent whitening agents, through microbial techniques based on antibiotic resistance profiling and qPCR assessment of human and ruminant fractions of the bacteriodetes flora. It is prudent to deploy a range of methods to any specific site investigation to gain an understanding of the probable contributors to the faecal loading. Quantitative apportionment of faecal indicators measured in a recreational water to individual contributing species (e.g. human and ruminant) is, at this time, not provided by the suite of MST techniques available.

Page 82, last sentence
Change the text that begins with the sentence "For example, it is not acceptable" to:
The sampling programme should be representative of the range of conditions (dry and wet weather, etc) in the recreational water environment where it is being used. When determining recreational water classification, all routinely collected samples on days when the recreational water area was open to the public should be used. For example, it is not appropriate to resample the bathing water following a high count measured when the beach was open and no 'advisory' notice had been posted and then to use the re-sample result but not the original result (this is not the case where an advisory notice has been posted in which case the sample taken during the period of the posted advisory would be omitted (e.g. discounted) from percentile calculations). On the other hand, reactive samples may be taken following an adverse event or high result from a routine sample. The additional samples may be used to investigate the full impact of the event on the bathing water or to further characterise the area and the impacts of adverse events.

Page 86, BOX 4.6 CASE STUDY, continued
Top of page, part (e), **Replace** "Very high" with "Very high susceptibility to faecal influence"

Part 3 Combined Sanitary and Microbial Water Quality Assessment and Overall Classification

Move: This beach is rated as "poor", to bottom
Change: Sanitary Inspection Category-"Very low", should read "Very high susceptibility to faecal influence"

In table **add** "95th tile" before (intestinal enterococci/100 mL)

Page 93, 4.7.1 Public health advisories and warnings
Insert new box, 4.9, INFORMING THE BATHER WITH IMPROVED ASSESSMENTS OF WATER QUALITY

Predictive models can be used at bathing water areas to derive microbial water quality forecasts (e.g. daily) which can then be made available to the public by beach signage and other methods of disseminating information. These provide bathers and beach users with near-real-time information on likely water quality conditions that are more up to date than the historical results provided by traditional analytical methods. They allow water users to make informed choices on whether or not it is advisable to undertake bathing activities using the predict and protect (precautionary) principle.

These predictive water quality models are usually site specific but can sometimes be developed for adjacent bathing waters if they are affected by common pollution inputs and are predictable by the same environmental factors, which drive water quality. Commonly they work by using input data from the local factors which correlate strongest with factors which cause or affect water quality such as the levels of preceding rainfall (e.g. over previous 12, 24 or 48 hr) from drainage catchments, river or storm water flows, wind direction, turbidity, UV and tidal state.

Working predictive water quality models have been developed and used in a number of countries such in the U.S. for freshwater bathing waters on the Great Lakes (Never & Whitman, 2005; Boehm et al., 2007) and in Scotland for coastal bathing waters (McPhail & Stidson, 2004; 2009). Information of model outputs and water quality predictions are usually posted on the Internet (see References for websites) or by beach advisory notices.

In Scotland, the Scottish Environment Protection Agency (SEPA) issues daily water quality forecasts to the public during the bathing season by a network of electronic variable message signs located at a number of bathing beaches affected by rain events. The predicted water quality conditions are also posted simultaneously on a website and through a telephone text message service.

Electronic messages are switched on through a central national control as to either:
"Good Water Quality is Predicted Today", or
"Bathing Not Advised Today – Risk of Poor Water Quality".
Additional public messages can be provided by alternating text pages.

It is important that the predictive models are validated and checked against real conditions, but once developed they have been shown to be good (correct or precautionary) at predicting water quality. They work particularly well at waters that may be subject to weather-related or other environmental factors that correlate with causing short-term pollution or elevated microbiological events. As further developments are made in information technologies there will be opportunities to extend the methods of disseminating water quality predictions and information to the public by methods that they find useful.

Page 96, Add following new section, after 4.7.1 (and renumber following sections to 4.7.3 and 4.7.4):

4.7.2 ASSESSING AND ACTING ON SINGLE AND/OR HIGH ANALYTICAL RESULTS

Those with responsibility in this area should ensure they are fully appraised of any sanitary survey information for the particular site, any past records of water quality and they have undertaken a recent visual site inspection. There are three principal conditions that might lead beach management agencies to consider posting an advisory notice of likely adverse water quality.

i. *Where climatic conditions, such as high rainfall, produces elevation of faecal indicator bacteria in the recreational waters* (Marsalek & Rochfort, 2004). The microbial source may be agricultural runoff and/or urban surface water. Here, the appropriate management action is to give the public information through signage, ideally, provided through real-time prediction of bathing water quality communicated via electronic means to key communication facilities such as signs at bathing water sites and/or via the internet to tourist information centres and the news media. The water quality levels at which such an advisory might be prudent should be decided in light of local conditions. Examples of predicted faecal indicator bacterial concentrations used to inform decisions on when to deploy advisory signs are presented for Scotland in Box 4.9. In Scotland, limit values from current water quality standards (e.g. 2000 E. coli/100mL and or 200 intestinal enterococci/100mL) are used. These limit values might also be appropriate for locations with no regulator compliance history or data describing water quality.

ii. *Where some rare and/or extreme event causes gross pollution of the bathing water*. Often, the first evidence of such a condition are visual reports of gross pollution indicated by high turbidity and associated sanitary wastes from sewer overflows and/or overflow debris from rivers and drains discharging to the bathing water. Action to protect the public is prudent on observing such conditions, particularly where the visual evidence suggests discharges from the sewerage system or there is telemetric evidence of sewer flooding. Microbiological testing, to confirm such adverse water quality, could provide both confirmation of the high microbial concentrations and a yardstick to ensure a return to more 'normal' water quality for the site affected by the extreme event. However, the protective advisory notice, informing the public of potentially adverse water quality, should be posted on first observation of the extreme event evidence.

iii. *Where weather events do not present a feasible explanation for observed sewer debris at the bathing water but such observations are reporte*d. This may indicate a gross malfunction or leakage of the sewerage system. Here, an advisory notice to inform the public of the new risk should be posted and only removed when the new source of gross pollution has been rectified.

Page 96, 4.8 References
Insert the additional references listed below:

Ahmed W, Stewart J, Gardner T, Powell D (2008a) A real-time polymerase chain reaction assay for quantitative detection of the human-specific enterococci surface protein marker in sewage and environmental waters. *Environmental Microbiology*, 10: 3255-3264.

Ahmed W, Hargreaves M, Goonetilleke A, Katouli M (2008b) Population similarity analysis of indicator bacteria for source prediction of faecal pollution in coastal lake. *Marine Pollution Bulletin*, 56: 1469-1475.

Bartram J, Corrales L, Davison A, Deere D, Drudy D, Gordon B, Howard G, Rinehold A, Stevens M (2009) *Water safety plan manual: step-by-step risk management for drinking-water suppliers*. Geneva, World Health Organization.

Boehm AB, Whitman RL, Nevers MB, Hou D, Weisberg SB (2007) Nowcasting recreational water quality. In: Wymer, L.J. (Ed.), *Statistical Framework for Recreational Water Quality Criteria and Monitoring.*

Boehm A, Nicolas JA, Colford JM, Dunbar LE, Fleming LE, Gold MA, Hansel JA, Hunter PR, Ichida AM, McGee CH.D, Soller JA, Weisberg SB (2009) A sea change ahead for recreational water quality criteria. *Journal of Water and Health*, Doi: 10.2166/wh.2009.122.

Bofill-Mas S, Albiñana-Gimenez N, Clemente-Casares P, Hundesa A, Rodriguez-Manzano J, Allard A, Calvo M, Girones R (2006) Quantification and stability of human adenoviruses and polyomavirus JCPyV in wastewater matrices. *Applied and Environmental Microbiology,*72:7894-6.

Buckalew DW, Hartman LJ, Grimsley GA, Martin AE, Register KM (2006) A long-term study comparing membrane filtration with Colilert® defined substrates in detecting faecal coliforms and *Escherichia coli* in natural waters. *Journal of Environmental Management,* 80:191-197.

Calgua B, Mengewein A, Grunert A, Bofill-Mas S, Clemente-Casares P, Hundesa A, Wyn-Jones AP, Lopez-Pila JM, Girones R (2008) Development and application of a one-step low cost procedure to concentrate viruses from seawater samples. *Journal of Virological Methods,* 153: 79-83.

Chawla R, Hunter PR (2005) Classification of Bathing Water Quality based on the Parametric Calculation of Percentiles is unsound. *Water Research,* 39: 4552-4558.

Costán-Longares A, Mocé-Llivina L, Avellón A, Jofre L, Lucena F (2008) Occurrence and distribution of culturable enteroviruses in wastewater and surface waters on north-eastern Spain. *Journal of Applied Microbiology,* 105:1945-1955.

Davies CM, Apte SC, Peterson SM, Stauber JL (1994) Plant and algal interference in bacterial beta-D-galactosidase and beta-D-glucuronidase assays. *Applied Environmental Microbiology*, 60:3959-3964.

Davies-Colley RJ, Craggs RJ, Park J, Sukias JP, Nagels JW, Stott R (2005) Virus removal in a pilot-scale 'advanced' pond system as indicated by somatic and F-RNA bacteriophages. *Water Science and Technology,* 51:107-110.

Davison A, Deere D, Stevens M, Howard G, Bartram J (2006) *Water Safety Plan Manual*. Geneva, World Health Organization.

Dizer H, Wolf S, Fischer M, López-Pila JM, Röske I, Schmidt R, Szewzyk R, Wiedenmann A (2005) *The EU Bathing Water Directive. Risk assessment and standards.* Bundesgesundheitsblatt Gesundheitsforschung Gesundheitsschutz. 48(5): 607-14. German, English abstract.

Dufour A, Schaub S (2007) The evolution of water quality criteria in the United States, 1922-2003. In Wymer LJ (ed.): *Statistical Framework for Recreational Water Quality Criteria and Monitoring. Statistics in Practice*. John Wiley & Sons Ltd., Chichester, England: 1-12.

Dufour AP, Evans O, Behymer TD, Cantú R. (2006) Water ingestion during swimming activities in a pool: a pilot study. *Journal of Water Health*, 4: 425-430.

Edge TA, Schaefer KA, ed. (2006) *Microbial Source Tracking in Aquatic Ecosystems: The State of the Science and an Assessment of Needs*. National Water Research Institute, Burlington, Ontario. NWRI Scientific Assessment Report Series No. 7 and Linking Water Science to Policy Workshop Series. 23p.

El-Shaarawi, AH, Esterby SR (1992) Replacement of censored observations by a constant: an evaluation. *Water Res*. 26 (6): 835–844.

Enriquez CE, Hurst CJ, Gerba CP (1995) Survival of the enteric adenoviruses 40 and 41 in tap, sea and wastewater. *Water Research*, 29: 2548–2553.

Field, KG, Chern EC, Dick LK, Fuhrman J, Griffith J, Holden PA, La Montagne MG, Le J, Olson B, Simonich MT (2003) A comparative study of culture independent, library-independent genotypic methods of fecal source tracking. *Journal of Water and Health,* 1: 181-194.

Field, KG and Samadpour M (2007) Fecal source tracking, the indicator paradigm, and managing water quality. *Water Research,* 41: 3517 – 3538.

Figueras MJ, Inza I, Polo FL, Feliu MT, Guarro J (1996) A fast method for the confirmation of fecal streptococci from M-enterococcus medium. *Applied Environmental Microbiology*, 62:2177-2178.

Figueras MJ, Inza I, Polo F, Guarro J (1998) Evaluation of the oxolinic acid--esculin--azide medium for the isolation and enumeration of faecal streptococci in a routine monitoring programme for bathing waters. *Canadian Journal of Microbiology,* 44:998-1002.

Fiksdal L, Tryland I (2008) Application of rapid enzyme assay techniques for monitoring of microbial water quality. *Current opinion in Biotechnology*, 19: 289-294.

Fricker CR, DeSarno M, Warden PS, Eldred BJ (2008) False –negative β-D-glucuronidase reactions in membrane lactose glucuronide agar medium used for the simultaneous detection of coliforms and *Escherichia coli* from water. *Letters in Applied Microbiology*, 47: 539-42.

Gourmelon M, Caprais MP, Segura R, Le Mennec C, Lozach S, Piriou JY, Rince A (2007) Evaluation of two library-independent microbial source tracking methods to identify sources of fecal contamination in French estuaries. *Applied Environmental Microbiology,* 73: 4857-4866.

Haugland RA, Siefring SC, Wymer LJ, Brenner KP, Dufour AP (2005) Comparison of Enterococcus measurements in freshwater at two recreational beaches by quantitative polymerase chain reaction and membrane filter culture analysis. *Water Research,* 39(4): 559-568.

Hewett P, Ganser GH (2007) A Comparison of Several Methods for Analyzing Censored Data. *Annals Occupational Hygiene,* 51 (7): 611–632.

Hunter PR (2002) Appropriate calculation of the 95th percentile of microbiological results in determining compliance with bathing water quality standards. *Letters in Applied Microbiology,* 34: 283-286.

Ishii S, Hansen DL, Hicks RE, Sadowsky MJ (2007) Beach sand sediments are temporal sinks and sources of *Escherichia coli* in Lake Superior. *Environmental Science and Technology*, 41: 2203-2209.

Ishii S, Sadowsky MJ (2008) *Escherichia coli* in the environment: implications for water quality and human health. *Microbes and Environment*, 23: 101-108.

Isobe KO, Tarao M, Chiem NH, Mihn LY, Takada H (2004) Effect of environmental factors on the relationship between concentrations of coprostanol and fecal indicator bacteria in tropical (Mekong Delta) and temperate (Tokyo) freshwaters. *Applied and Environmental Microbiology*, 70: 814-821.

Iwai M, Hasegawa S, Obara M, Nakamura K, Horimoto E, Takizawa T, Kurata T, Sogen S, Shiraki K (2009) Continuous existence of norovirus and sapoviruses in raw sewage reveals infection among inhabitants in Toyama, Japan (2006-2008). *Applied Environmental Microbiology*, Doi:10.1128/AEM.01166-08.

Loisy F, Atmar RL, Cohen J, Bosch A, Le Guyader FS (2004) Rotavirus VLP2/6: a new tool for tracking rotavirus in the marine environment. *Research in Microbiology*, 155: 575–578.

Marsalek J, Rochfort Q (2004) Urban wet-weather flows: sources of fecal contamination impacting on recreational waters and threatening drinking-water sources. *Journal of Toxicology and Environmental Health, Part A,* 67: 1765-1777.

Maheux AF, Huppé V, Boissinot M, Picard FJ, Bissonnette L, Bernier JT, Bergeron MG (2008) Analytical limits of four β-glucuronidase and β-galactosidase-based commercial culture methods used to detect *Escherichia coli* and total coliforms. *Journal of Microbiological Methods,* 75: 506-514.

McLaughlin K, Ho Ahn J, Litton RM, Grant SB (2007) Use of salinity mixing models to estimate the contribution of creek water fecal indicator bacteria on an estuarine environment: Newport Bay, California. *Water Research,* 41: 3595-3604.

McPhail CD, Stidson, R (2004) The Scottish Signage Project. *Water and Environment Magazine,* 9(4): 18-19.

McPhail CD, Stidson R (2009) Bathing Water Signage and Predictive Water Quality Models in Scotland. *Aquatic Ecosystem Health and Management Journal,* in press.

Nevers MB, RL Whitman. (2005) Nowcast modeling of *Escherichia coli* concentrations at multiple urban beaches of southern Lake Michigan. *Water Research,* 39:5250-5260.

O'Brien RT and Newman, JS (1977) Inactivation of polioviruses and Coxsackie viruses in surface water. *Applied Environmental Microbiology,* 33: 334-340.

Reischer GH, Haider JM, Sommer R, Stadler H, Keiblinger KM, Hornek R, Zerobin W, Mach RL, Farnleitner AH (2008) Quantitative microbial faecal source tracking with sampling guided by hydrological catchment dynamics. *Environmental Microbiology,* 10: 2598-2608.

Roser DJ, Davies CM, Ashbolt NJ, Morison P (2006) Microbial exposure assessment of an urban recreational lake: a case study of the application of new risk-based guidelines. *Water Science and Technology,* 54: 245-252.

Roser DJ, Ashbolt NJ, Davies CM, Glamore WC, Hawker KM, Miller BM (2007) Application of TMDL and risk assessment principles for pathogen management at an urban recreational lake. In *Watershed Management to Meet Water Quality Standards and TMDLS, 4th Conference Proceedings, March 10-14, 2007 San Antonio, Texas*. Stephenville, Texas: The American Society of Agricultural and Biological Engineers (ASABE), 420-426.

Santo Domingo JW, Bambic DG, Edge TA, Wuertz S (2007) Quo vadis source tracking? Towards a strategic framework for environmental monitoring of fecal pollution. *Water Research,* 41: 3539-3552.

Schmoyer RL, Beauchamp JJ, Brandt CC, Hoffman JR (1996) Difficulties with the Iognormal model in mean estimation and testing. *Environmental and Ecological Statistics* 3: 81-97.

Sobsey MD, Shields PA, Hauchman FS, Davis AL, Rullman VA, Bosch A (1988) Survival and persistence of hepatitis A virus in environmental samples. *In:* Alan R Liss (Ed.), *Viral Hepatitis and Liver Disease,* New York: 121–124.

Soller JA, and Eisenberg JNS (2008) An evaluation of parsimony for microbial risk assessment models. *Environmetics,* 19: 61–78.

Stoeckel DM, Harwood V (2007) Performance, design, and analysis in microbial source tracking studies. *Applied and Environmental Microbiology*, 73(8): 2405–2415.

Terrett LA, Sail LJ, Theil KW, Kohler EM (1987) Physicochemical characterisation of porcine pararotavirus and detection of virus and viral antibodies using cell culture immunofluorescence. *Journal of Clinical Microbiology*, 25: 268–272.

Teunis PF, Moe CL, Liu PE, Miller S, Lindesmith L, Baric RS, Le Pendu J, Calderon RL (2008) Norwalk virus: How infectious is it? *Journal of Medical Virology,* 80: 1468-1476.

US EPA (2007) *Report of the experts scientific workshop on critical research needs for the development of new or revised recreational water quality criteria. June 2007.* Warrenton, Virginia. U.S. Environmental Protection Agency, Office of Water (EPA 823-R-07-006).

US EPA (2005) *Economic Analysis for the Final Long Term 2 Enhanced Surface Water Treatment Rule.* Washington D.C.: U.S. Environmental Protection Agency, (*EPA 815-R-06-001).*

Vogel JR, Stoeckel DM, Lamendella R, Zelt RB, Santo Domingo JW, Walker SR, Oerther DB. (2007) Identifying fecal sources in a selected catchment reach using multiple source-tracking tools. *Journal of Environmental Quality,* 36: 718-729.

Vondefecht SL, Huber AC, Eiden J, Mader LC, Yolken RH (1986) Infectious diarrhoea of infant rats produced by a rotavirus-like agent. *Journal of Medical Virology,* 19: 167–173.

Wade TJ, Calderon RL, Sams E, Beach M, Brenner KP, Williams AH, Dufour AP (2006) Rapidly measured indicators of recreational water quality are predictive of swimming-associated gastrointestional illness. *Environmental Health Perspectives,* 114(1): 24-28.

Wade T, Calderon RL, Brenner KP; Sams E, Beach M, Haugland R, Wymer L, Dufour AP (2008) High sensitivity of children to swimming associated gastrointestinal illness. *Epidemiology*; 19(3): 375-383.

WHO (2004) *Guidelines for Drinking Water Quality, Third Edition, Vol 1.* Geneva, World Health Organization.

Wiedenmann A (2007a) A plausible model to explain concentration-response relationships in randomized controlled trials assessing infectious disease risks from exposure to recreational waters. In Wymer LJ (ed.): *Statistical Framework for Recreational Water Quality Criteria and Monitoring. Statistics in Practice.* Chichester, England: John Wiley & Sons Ltd., 153-177.

Wiedenmann A (2007b) Health-related interpretation of individual microbiological sample results from monitoring programs of recreational waters based on recent epidemiological findings. (German; Engl. abstract), *Umweltmedizin in Forschung und Praxis.* UFP (6) 351-365.

Wiedenmann A, Krüger P, Dietz K, López-Pila JM, Szewzyk R, Botzenhart K (2006) A randomized controlled trial assessing infectious disease risks from bathing in fresh recreational waters in relation to the concentration of Escherichia coli, intestinal enterococci, Clostridium perfringens, and somatic coliphages. *Environmental Health*

Perspectives, 114:228-236.

Wiedenmann A, Krüger P, Gommel S, Eissler A, Hirlinger M, Paul A, Jüngst K, Sieben E, Dietz K (2004) *Epidemiological Determination of Disease Risks from Bathing. Final Report.* UFOPLAN 298 61 503. Umweltbundesamt.

Yoder JS, Blackburn BG, Craun GF, Hill V, Levy DA, Chen N, Lee SH, Calderon RL, Beach MJ (2004) *Surveillance for Waterborne Disease Outbreaks Associated with Recreational Water – United States, 2001 – 2002,* MMWR, Surveillance Summaries, Vol. 53, No. SS-8.

Yoder JS, Hlavsa MC, Craun GF, Hill V, Roberts V, Yu PA, Hicks LA, Alexander NT, Calderon RL, Roy SL, Beach MJ (2008) *Surveillance for Waterborne Disease Outbreaks Associated with Recreational Use and other Aquatic Facility-associated Health events – United States, 2005 – 2006*, MMWR, Surveillance Summaries, Vol. 57, No. SS-9.

Internet resources for Box 4.9

1:http://www.glsc.usgs.gov/main.php?content=research_initiatives_beachhealth&title=Project%20S.A.F.E.0&menu=research_initiatives_projectSAFE

2. http://www.ohionowcast.info/index.asp

3. http://www.sepa.org.uk/water/bathing_waters/bathing_signs.aspx

CHAPTER 10. FAECAL POLLUTION AND WATER QUALITY

Page 195
Change Item 15, Delete "prior" in "The influence of specific events such as rain on the recreational water use areas, especially in relation to the duration of the peak contamination period, should be established and **prior** agreed procedures implemented."