

Guidelines for drinking-water quality

Fourth edition
incorporating
the first, second and
third addenda



World Health
Organization

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Preface

Access to safe drinking-water is essential to health, a basic human right and a component of effective policy for health protection.

The importance of water, sanitation and hygiene for health and development has been reflected in the outcomes of a series of international policy forums. This includes the adoption of the Sustainable Development Goals by countries, in 2015, which include a target and indicator on safe drinking-water. Further, the United Nations (UN) General Assembly declared in 2010 that safe and clean drinking-water and sanitation is a human right, essential to the full enjoyment of life and all other human rights. These commitments build on a long history of support including the UN General Assembly adopting the Millennium Development Goals in 2000 and declaring the period 2005–2015 as the International Decade for Action, “Water for Life”.

Access to safe drinking-water is important as a health and development issue at national, regional and local levels. In some regions, it has been shown that investments in water supply and sanitation can yield a net economic benefit, because the reductions in adverse health effects and health-care costs outweigh the costs of undertaking the interventions. This is true for investments ranging from major water supply infrastructure through to water treatment in the home. Experience has also shown that interventions in improving access to safe water favour the poor in particular, whether in rural or urban areas, and can be an effective part of poverty alleviation strategies.

The World Health Organization (WHO) published four editions of the *Guidelines for drinking-water quality* (in 1983–1984, 1993–1997, 2004 and 2011), as successors to the previous WHO *International standards for drinking water*, which were published in 1958, 1963 and 1971. Since 1995, the Guidelines have been updated through a process of rolling revision, whereby a limited number of sections within each edition are updated as feasible, including in response to new evidence, uncertainty about best practice or requests from stakeholders. New editions of the Guidelines usually introduce major new recommendations and are published following comprehensive review.

Leading the process of the development of the fourth edition was the Water, Sanitation, Hygiene and Health Unit within WHO headquarters. The Chemical Safety Unit and the Risk Assessment and Management Unit provided input on chemical

hazards, and the Radiation Programme provided input on radiological hazards. All six WHO regional offices participated in the process, in consultation with Member States.

This version of the Guidelines, *Guidelines for drinking-water quality: fourth edition incorporating the first, second and third addenda*,¹ supersedes previous editions of the Guidelines, including the fourth edition incorporating the first and second addenda, published in 2022; the fourth edition incorporating the first addendum, published in 2017; the fourth edition, published in 2011; and the previous International Standards. These Guidelines integrate into the fourth edition the updates of 2022 and subsequent updates, as listed below.

The primary goal of the Guidelines is to protect public health associated with drinking-water quality. The overall objectives of the Guidelines are to:

- provide an authoritative basis for the effective consideration of public health in setting national or regional drinking-water policies and actions;
- provide a comprehensive preventive risk management framework for health protection, from catchment to consumer, that covers policy formulation and standard setting, risk-based management approaches and surveillance;
- emphasize achievable practices and the formulation of sound regulations that are applicable to low-income, middle-income and industrialized countries alike;
- summarize the health implications associated with contaminants in drinking-water, and the role of risk assessment and risk management in disease prevention and control;
- summarize effective options for drinking-water management; and
- provide guidance on hazard identification and risk assessment.

The fourth edition of the Guidelines, including its updates, further develops concepts, approaches and information introduced in previous editions, such as the comprehensive preventive risk management approach for ensuring drinking-water quality that was introduced in the third edition. It considers:

- drinking-water safety, including minimum procedures and specific guideline values, and how these are intended to be used;
- approaches used in deriving the Guidelines, including guideline values;
- microbial hazards, which continue to be the primary concern in both developing and developed countries. Experience has shown the value of a systematic approach to securing microbial safety. This edition builds on the preventive principles introduced in the third edition on ensuring the microbial safety of drinking-water through a multiple-barrier approach, highlighting the importance of source water protection;
- climate change, which results in changing water temperature and rainfall patterns, severe and prolonged drought or increased flooding, and its implications for water quality and water scarcity, recognizing the importance of managing these impacts as part of water management strategies;

¹ Up to 2017, the Guidelines incorporating the addenda were accompanied by separate addenda publications that detailed the updates made to the Guidelines. Separate addenda are no longer published, but will be referenced in the naming of the Guidelines between editions up to the fifth edition.

- chemical contaminants in drinking-water, including information on chemicals not considered previously (e.g. pesticides used for vector control in drinking-water); revisions of existing chemical fact sheets, taking into account new scientific information; and reduced coverage in the Guidelines in cases where new information suggests a lesser priority;
- key chemicals responsible for large-scale health effects through drinking-water exposure (e.g. arsenic, fluoride, lead, nitrate, selenium and uranium), with the Guidelines providing guidance on identifying local priorities and on management;
- the important roles of many different stakeholders in ensuring drinking-water safety; this edition furthers the discussion introduced in the third edition of the roles and responsibilities of key stakeholders in ensuring drinking-water safety; and
- guidance for a broader range of supplies including rainwater harvesting and other non-piped supplies or dual-piped systems.

The Guidelines are accompanied by a series of supporting publications. These include internationally peer-reviewed risk assessments for specific chemicals (see the list of [chapter 12](#) background documents in [Annex 2](#)) and other publications explaining the scientific basis of the development of the Guidelines and providing guidance on good practice in their implementation (see [Annex 1](#)). The WHO *Guidelines for drinking-water quality: small water supplies* (published in 2024; see [Annex 1](#))—a companion publication to these Guidelines—provides guidance on applying the Guidelines recommendations to small water supplies in particular. The related publication *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* (also published in 2024; see [Annex 1](#)) provides resources to support implementation of WHO recommendations on risk management and surveillance. Additional supporting publications that have informed the update of the Guidelines, including the updates to the fourth edition, are referenced throughout.

Key updates to the Guidelines in 2017 were:

- new guidance on microbial risk assessment, aggregating multiple barriers for overall water treatment performance and microbial detection methods ([chapter 7](#));
- new or updated fact sheets for barium; bentazone; chlorine dioxide, chlorate and chlorite; dichlorvos; dicofol; diquat; MCPA; nitrate and nitrite; and perchlorate, with corresponding updates to guideline values or health-based values ([chapter 12](#)); and
- additional guidance on risk management considerations and monitoring of lead ([chapter 12](#)).²

² See *Guidelines for drinking-water quality, fourth edition: first addendum* (WHO, 2017) for the detailed list of changes made to the fourth edition of the Guidelines.

Key updates to the Guidelines in 2022 were:

- clarification that manganese can be a concern in some areas because of the potential extent of exposure at concentrations of human health significance, considering the updated WHO guideline value ([section 2.5.3](#));
- updated information on the adequacy of water supply ([section 5.3](#));
- updated information on climate change, emergencies, and water used in food production and processing ([chapter 6](#));
- a new section on reuse of wastewater ([chapter 6](#));
- an explanation of reference values, which are a new type of value included in the Guidelines ([section 8.2](#));
- additional guidance on assessing chemical mixtures ([section 8.2.8](#));
- updated guidance on management of radionuclides, including interpretation and application of the WHO screening values and guidance levels, and management of radon ([chapter 9](#));
- updated information on cyanobacteria, including considerations for use of an alert level framework ([sections 8.5.1, 10.1 and 11.5](#));
- new or updated fact sheets for anatoxin-a variants, asbestos, bentazone, chromium, cylindrospermopsins, iodine, manganese, microcystins, nickel, organotins, saxitoxins, silver, tetrachloroethene and trichloroethene. These either reaffirm or update the guideline values and health-based values and, in some instances, establish reference values ([chapter 12](#)). Corresponding updates have been made to the chemical summary tables ([chapter 8 and Annex 3](#)), aesthetic considerations for manganese ([section 10.2](#)), factors influencing leaching of nickel in nickel-containing pipes and fittings ([Annex 5](#)), and analytical achievability and treatment performance tables for cyanobacteria, cyanotoxins and manganese ([Annexes 4 and 5](#)); and
- updated references in chapters 1–3 and in the above-mentioned sections, WHO web links throughout the Guidelines and web links in [Annex 1](#).

As with earlier updates to the fourth edition, the latest revisions were identified by the Guideline Development Group with its working groups on chemicals, microbial aspects and protection and control. Most of these updates draw on separate supporting resources to the Guidelines published by WHO after the 2022 version. The revised information on pesticides used for vector control is based on consultations with the WHO teams responsible for the prequalification programme and the development of guidance on larval source management.

Key updates included in the current version of the Guidelines are:

- emphasis that in addition to community-managed small water supplies, small water supplies more broadly need explicit consideration in policies and regulations and provision of support;
- the clarified role of sanitary inspections to support risk assessment and management, including water safety plans (WSPs) ([chapter 4](#));
- updated guidance on WSPs ([chapter 4](#)) including emphasizing the role of governments and based on additional experience in WSP development and implementation;

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- new or updated matrices to support prioritization of action based on microbial water quality test results and WSP audit scores or sanitary inspection scores ([section 5.5.2](#));
- updated information on microbial pathogens including summary information ([section 7.1](#), particularly [Tables 7.1](#) and [7.2](#), and [section 7.2.2](#)), highlighting priority waterborne pathogens ([section 7.1.4](#)), new or updated fact sheets ([sections 11.1 to 11.4](#)), and considerations for buildings ([section 6.10](#)) and health-care facilities ([section 6.11](#));
- updated information on pesticides used for vector control in drinking-water to reflect WHO's latest guidance ([section 8.6](#)) with deletion of pesticide-specific fact sheets (previously section 12.2);
- updated references in the above-mentioned sections including the *Guidelines for drinking-water quality: small water supplies*; and
- added references to WHO resources to aid implementation of Guidelines recommendations including *Lead in drinking-water: health risks, monitoring and corrective actions*, *Management of cyanobacteria in drinking-water supplies: information for regulators and water suppliers, 2nd edition* and *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)).

The Guidelines are addressed primarily to water and health regulators, policy-makers and their advisors, to assist in the development of national policies and regulations. The Guidelines and associated documents are also used by many others as a source of information on water quality and health, and on effective management approaches.

The Guidelines are recognized as representing the position of the UN system on issues of drinking-water quality and health by UN-Water, the body that coordinates among the over 30 UN agencies and programmes concerned with water issues.

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Abbreviations used in text

2,4-D	2,4-dichlorophenoxyacetic acid
2,4-DB	2,4-dichlorophenoxybutyric acid
2,4-DP	dichlorprop
2,4,5-T	2,4,5-trichlorophenoxyacetic acid
2,4,5-TP	2,4,5-trichlorophenoxy propionic acid; fenoprop
AAS	atomic absorption spectrometry
Absor	absorptiometry
ADI	acceptable daily intake
AES	atomic emission spectrometry
AIDS	acquired immunodeficiency syndrome
AK	acanthamoebic keratitis
AMPA	aminomethylphosphonic acid
ARfD	acute reference dose
ATX	anatoxin
BDCM	bromodichloromethane
BMD	benchmark dose
BMDL	lower confidence limit on the benchmark dose
BMDL _x	lower 95% confidence limit on the benchmark dose for an x% response
BTEX	benzene, toluene, ethylbenzene and xylenes
Bti	<i>Bacillus thuringiensis israelensis</i>
bw	body weight
CAS	Chemical Abstracts Service
Col	colorimetry
CPVC	chlorinated polyvinyl chloride
CSAF	chemical-specific adjustment factor
Ct	product of disinfectant concentration and contact time
CYN	cylindrospermopsin

DAEC	diffusely adherent <i>E. coli</i>
DALY	disability-adjusted life year
DBCM	dibromochloromethane
DBCP	1,2-dibromo-3-chloropropane
DBP	disinfection by-product
DCA	dichloroacetic acid
DCB	dichlorobenzene
DCP	dichloropropane
DDT	dichlorodiphenyltrichloroethane
DEHA	di(2-ethylhexyl)adipate
DEHP	di(2-ethylhexyl)phthalate
DNA	deoxyribonucleic acid
DPD	<i>N,N</i> -diethyl-1,4-phenylenediamine sulfate
EAAS	electrothermal atomic absorption spectrometry
EAEC	enteroaggregative <i>E. coli</i>
ECD	electron capture detector
EDTA	ethylenediaminetetraacetic acid; edetic acid
EHEC	enterohaemorrhagic <i>E. coli</i>
EIEC	enteroinvasive <i>E. coli</i>
ELISA	enzyme-linked immunosorbent assay
EPEC	enteropathogenic <i>E. coli</i>
ETEC	enterotoxigenic <i>E. coli</i>
F ₀	parental generation
F ₁	first filial generation
FAAS	flame atomic absorption spectrometry
FAO	Food and Agriculture Organization of the United Nations
FD	fluorescence detector
FID	flame ionization detector
FPD	flame photodiode detector
GAC	granular activated carbon
GAE	granulomatous amoebic encephalitis
GC	gas chromatography
GL	guidance level (used for radionuclides in drinking-water)
GV	guideline value
HAA	haloacetic acid
HAdV	human adenovirus
HAiV	human aichivirus
HAstV	human astrovirus

ABBREVIATIONS USED IN TEXT

HAV	hepatitis A virus
HCB	hexachlorobenzene
HCBD	hexachlorobutadiene
HCH	hexachlorocyclohexane
HEV	hepatitis E virus
HIV	human immunodeficiency virus
HPC	heterotrophic plate count
HPeV	human parechovirus
HPLC	high-performance liquid chromatography
IARC	International Agency for Research on Cancer
IC	ion chromatography
ICP	inductively coupled plasma
ICRP	International Commission on Radiological Protection
ID ₅₀	estimated dose associated with 50% infection rate
IDC	individual dose criterion
IPCS	International Programme on Chemical Safety
IQ	intelligence quotient
ISO	International Organization for Standardization
JECFA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	Joint FAO/WHO Meeting on Pesticide Residues
LC	liquid chromatography
LED	light-emitting diode
LOAEL	lowest-observed-adverse-effect level
LRV	log ₁₀ reduction value
MC	microcystin
MCB	monochlorobenzene
MCPA	4-(2-methyl-4-chlorophenoxy)acetic acid
MCPB	2,4-MCPB; 4-(4-chloro-o-tolyloxy)butyric acid; 4-(4-chloro-2-methylphenoxy)butanoic acid
MCP	2(2-methyl-chlorophenoxy) propionic acid; mecoprop
MDL	method detection limit
MDR	multidrug resistant
MERS-CoV	Middle East respiratory syndrome coronavirus
MMT	methylcyclopentadienyl manganese tricarbonyl
MS	mass spectrometry
MS/MS	tandem mass spectrometry
MTBE	methyl <i>tertiary</i> -butyl ether
MX	3-chloro-4-dichloromethyl-5-hydroxy-2(5H)-furanone

NDMA	<i>N</i> -nitrosodimethylamine
NOAEL	no-observed-adverse-effect level
NOEL	no-observed-effect level
NTA	nitrilotriacetic acid
NTM	non-tuberculous <i>Mycobacterium</i>
NTS	non-typhoidal <i>Salmonella</i>
NTP	National Toxicology Program (USA)
NTU	nephelometric turbidity unit
PAC	powdered activated carbon
PAH	polynuclear aromatic hydrocarbon
PCE	tetrachloroethene
PCP	pentachlorophenol
PCR	polymerase chain reaction
PD	photoionization detector
PDA	photodiode array
PMTDI	provisional maximum tolerable daily intake
PPA	protein phosphatase assay
PT	purge and trap
PTDI	provisional tolerable daily intake
PTMI	provisional tolerable monthly intake
PTWI	provisional tolerable weekly intake
PVC	polyvinyl chloride
QMRA	quantitative microbial risk assessment
RNA	ribonucleic acid
SARS-CoV	severe acute respiratory syndrome coronavirus
SI	Système international d'unités (International System of Units)
SODIS	solar water disinfection
STX	saxitoxin
sp.	species (singular)
spp.	species (plural)
subsp.	subspecies (singular)
TBA	terbuthylazine
TCB	trichlorobenzene
TCE	trichloroethene
TCID ₅₀	estimated dose associated with 50% infection rate of tissue culture cells
TCU	true colour unit

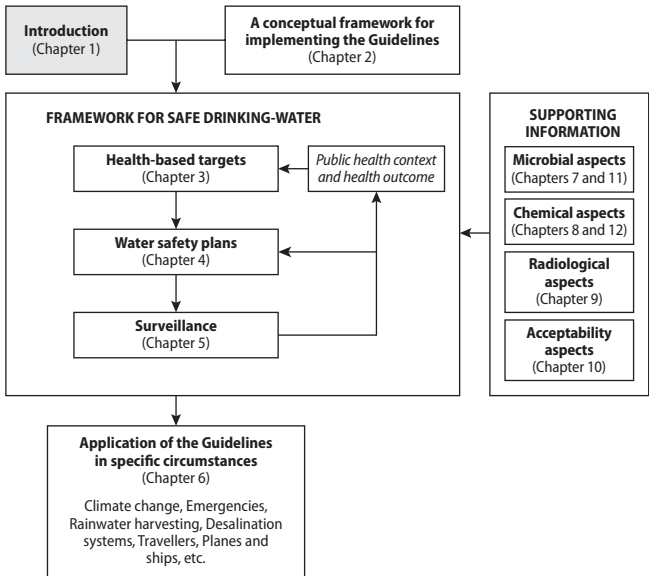
ABBREVIATIONS USED IN TEXT

TD ₀₅	tumorigenic dose ₀₅ , the dose associated with a 5% excess incidence of tumours in experimental animal studies
TDI	tolerable daily intake
TDS	total dissolved solids
THM	trihalomethane
TID	thermal ionization detector; total indicative dose
UF	uncertainty factor
UN	United Nations
UNICEF	United Nations Children's Fund
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USA	United States of America
UV	ultraviolet
UVPAD	ultraviolet photodiode array detector
WHO	World Health Organization
WHOPES	World Health Organization Pesticide Evaluation Scheme
WSP	water safety plan
YLD	years of healthy life lost in states of less than full health (i.e. years lived with a disability)
YLL	years of life lost by premature mortality

1

Introduction

The primary purpose of the *Guidelines for drinking-water quality* is the protection of public health. The Guidelines provide the recommendations of the World Health Organization (WHO) for managing the risk from hazards that may compromise the safety of drinking-water. The recommendations should be considered in the context of managing the risk from other sources of exposure to these hazards, such as waste, air, food and consumer products.



1.1 General considerations and principles

Water is essential to sustain life, and a satisfactory (adequate, safe and accessible) supply must be available to all. Improving access to safe drinking-water can result in tangible benefits to health. Every effort should be made to achieve drinking-water that is as safe as practicable.

Safe drinking-water, as defined by the Guidelines, does not represent any significant risk to health over a lifetime of consumption, including different sensitivities that may occur between life stages. Those at greatest risk of waterborne disease are infants and young children, people who are debilitated and the elderly, especially when living

under unsanitary conditions. Those who are generally at risk of waterborne illness may need to take additional steps to protect themselves against exposure to waterborne pathogens, such as boiling their drinking-water. Safe drinking-water is required for all usual domestic purposes,

Diseases related to contamination of drinking-water constitute a major burden on human health. Interventions to improve the quality of drinking-water provide significant benefits to health.

including drinking, food preparation and personal hygiene. The Guidelines are applicable to packaged water and ice intended for human consumption. However, water of higher quality may be required for some special purposes, such as renal dialysis and cleaning of contact lenses, or for certain purposes in food production and pharmaceutical use. The Guidelines may not be suitable for the protection of aquatic life or for some industries.

The Guidelines are intended to support the development and implementation of risk management strategies that will ensure the safety of drinking-water supplies through the control of hazardous constituents of water. These strategies may include national or regional standards developed from the scientific basis provided in the Guidelines. The Guidelines describe reasonable minimum requirements of safe practice to protect the health of consumers and derive numerical “guideline values” for constituents of water or indicators of water quality. When defining mandatory limits, it is preferable to consider the Guidelines in the context of local or national environmental, social, economic and cultural conditions. The Guidelines should also be part of an overall health protection strategy that includes sanitation and other strategies, such as managing food contamination. This strategy would also normally be incorporated into a legislative and regulatory framework that adapts the Guidelines to address local requirements and circumstances (see also [section 2.6](#)).

The main reason for not promoting the adoption of international standards for drinking-water quality is the advantage provided by the use of a risk–benefit approach (qualitative or quantitative) in the establishment of national standards and regulations. Further, the Guidelines are best used to promote an integrated preventive management framework for safety applied from catchment to consumer. The Guidelines provide a scientific point of departure for national authorities to develop drinking-water regulations and standards appropriate for the national situation. In developing standards and regulations, care should be taken to ensure that scarce resources are not unnecessarily diverted to the development of standards and the monitoring of substances of relatively minor importance to public health. The approach followed in these Guidelines is intended to lead to national standards and regulations that can be readily implemented and enforced and are protective of public health.

The nature and form of drinking-water standards may vary among countries and regions. There is no single approach that is universally applicable. It is essential in the development and implementation of standards that the current or planned legislation relating to water, health and local government is taken into account and that the capacity of regulators in the country is assessed. Approaches that may work in one country or region will not necessarily transfer to other countries or regions. It is essential that each country review its needs and capacities in developing a regulatory framework.

The judgement of safety—or what is an acceptable level of risk in particular circumstances—is a matter in which society as a whole has a role to play. The final judgement as to whether the benefit resulting from the adoption of any of the Guidelines or guideline values as national or local standards justifies the cost is for each country to decide.

Although the Guidelines describe a quality of water that is acceptable for life-long consumption, the establishment of these Guidelines, including guideline values, should not be regarded as implying that the quality of drinking-water may be degraded to the recommended level. Indeed, a continuous effort should be made to maintain drinking-water quality at the highest possible level.

An important concept in the allocation of resources to improving drinking-water safety is that of incremental improvement towards long-term health-based targets. Priorities set to remedy the most urgent problems (e.g. protection from pathogens; see [section 1.1.2](#)) may be linked to long-term targets of further water quality improvements (e.g. improvements in the acceptability of drinking-water in terms of its taste, odour and appearance; see [section 1.1.6](#)).

An important concept in the allocation of resources to improving drinking-water safety is that of incremental improvement towards long-term water quality targets.

1.1.1 Framework for safe drinking-water

The basic and essential requirements to ensure the safety of drinking-water are a “framework” for safe drinking-water, comprising health-based targets established by a competent health authority, adequate and properly managed systems (adequate infrastructure, proper monitoring and effective planning and management) and a system of independent surveillance.

A holistic approach to the risk assessment and risk management of a drinking-water supply increases confidence in the safety of the drinking-water. This approach entails systematic assessment of risks throughout a drinking-water supply—from the catchment and its source water through to the consumer—and identification of the ways in which these risks can be managed, including methods to ensure that control measures are working effectively. It incorporates strategies to deal with day-to-day management of water quality, including upsets and failures. In this respect, climate change—in

In Stockholm, in 1999, it was agreed that future guidelines for drinking-water, wastewater and recreational water¹ should integrate assessment of risk, risk management options and exposure control elements within a single framework with embedded quality targets (see the supporting document *Water quality—Guidelines, standards and health*; [Annex 1](#)). Following this approach, the assessment of risk is not a goal in its own right, but rather a basis for decision-making. The framework for safe drinking-water and the recommended approach for regulations, policies and programmes are based on this overall framework, known as the Stockholm Framework (see [chapter 2](#)).

¹ See WHO (2006) and WHO (2021), respectively, for *Guidelines for the safe use of wastewater, excreta and greywater in agriculture and aquaculture, Volumes 1–4* and *Guidelines on recreational water quality*. WHO advice on the safe management of excreta, which is a primary source of contamination of drinking-water, is covered in the WHO *Guidelines on sanitation and health* (WHO, 2018).

the form of increased and more severe periods of drought or more intense rain-fall events leading to flooding—can have an impact on both the quality and the quantity of water and will require planning and management to minimize adverse impacts on drinking-water supplies. Climate change also needs to be considered in the light of demographic change, such as the continuing growth of cities, which itself brings significant challenges for drinking-water supply.

In support of the framework for safe drinking-water, the Guidelines provide a range of supporting information, including microbial aspects ([chapters 7 and 11](#)), chemical aspects ([chapters 8 and 12](#)), radiological aspects ([chapter 9](#)) and acceptability aspects ([chapter 10](#)). [Figure 1.1](#) provides an overview of the interrelationships among the individual chapters of the Guidelines in ensuring drinking-water safety.

The Guidelines are applicable to all piped and non-piped drinking-water supplies, including those serving large metropolitan areas, small towns and communities and individual households. The Guidelines are also applicable to a range of specific circumstances ([chapter 6](#)), including buildings, travellers and conveyances (aeroplanes and ships).

1.1.2 Microbial aspects

Securing the microbial safety of drinking-water supplies is based on the use of multiple barriers, from catchment to consumer, to prevent the contamination of drinking-water or to reduce contamination to levels not injurious to health. Safety is increased if multiple barriers are in place, including protection of water resources, proper selection and operation of a series of treatment steps and management of distribution systems (piped or otherwise) to maintain and protect treated water quality. The preferred strategy is a management approach that places the primary emphasis on preventing or reducing the entry of pathogens into water sources and reducing reliance on treatment processes for removal of pathogens.

In general terms, the greatest microbial risks are associated with ingestion of water that is contaminated with faeces from humans or animals (including birds). Faeces can be a source of pathogenic bacteria, viruses, protozoa and helminths.

Faecally derived pathogens are the principal concerns in setting health-based targets for microbial safety. Microbial water quality often varies rapidly and over a wide range. Short-term peaks in pathogen concentration may increase disease risks considerably and may trigger outbreaks of water-borne disease. Furthermore, by the time microbial contamination is detected, many people may have been exposed. For these reasons, reliance cannot be placed solely on end-product testing, even when frequent, to determine the microbial safety of drinking-water.

The potential health consequences of microbial contamination are such that its control must always be of paramount importance and must never be compromised.

Particular attention should be directed to a water safety framework and implementing comprehensive water safety plans (WSPs) to consistently ensure drinking-water safety and thereby protect public health (see [chapter 4](#)). Failure to ensure drinking-water safety may expose the community to the risk of outbreaks of intes-

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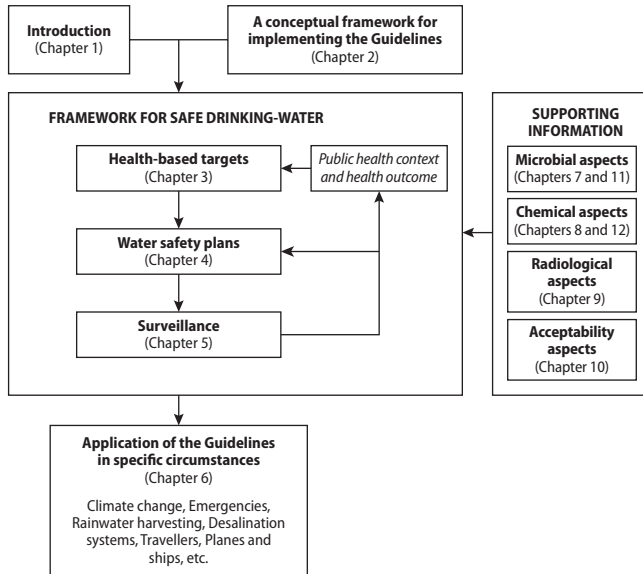


Figure 1.1 Interrelationships among the individual chapters of the *Guidelines for drinking-water quality* in ensuring drinking-water safety

tinal and other infectious diseases. Outbreaks of waterborne disease are particularly to be avoided because of their capacity to result in the simultaneous infection of a large number of persons and potentially a high proportion of the community.

In addition to faecally borne pathogens, other microbial hazards, such as guinea worm (*Dracunculus medinensis*), toxic cyanobacteria and *Legionella*, may be of public health importance under specific circumstances.

Although water can be a very significant source of infectious organisms, many of the diseases that may be waterborne may also be transmitted by other routes, including person-to-person contact, food intake and droplets and aerosols. Depending on the circumstances and in the absence of waterborne outbreaks, these routes may be more important than waterborne transmission.

Microbial aspects of water quality are considered in more detail in [chapter 7](#), with fact sheets on specific microorganisms provided in [chapter 11](#).

1.1.3 Disinfection

Disinfection is of unquestionable importance in the supply of safe drinking-water. The destruction of pathogenic microorganisms is essential and very commonly involves the use of reactive chemical agents such as chlorine.

Disinfection is an effective barrier to many pathogens (especially bacteria) during drinking-water treatment and should be used for surface waters and for groundwater subject to faecal contamination. Residual disinfection is used to provide a partial safeguard against low-level contamination and growth within the distribution system.

Chemical disinfection of a drinking-water supply that is faecally contaminated will reduce the overall risk of disease but may not necessarily render the supply safe. For

example, chlorine disinfection of drinking-water has limitations against the protozoan pathogens—in particular *Cryptosporidium*. Disinfection efficacy may also be unsatisfactory against pathogens within flocs or particles, which protect them from the action of disinfectants. High levels of turbidity can protect microorganisms from the effects of disinfection, stimulate the growth of bacteria and give rise to a significant chlorine demand. It is essential that an overall management strategy is implemented in which multiple barriers, including source water protection and appropriate treatment processes, as well as protection during storage and distribution, are used in conjunction with disinfection to prevent or remove microbial contamination.

The use of chemical disinfectants in water treatment usually results in the formation of chemical by-products. However, the risks to health from these by-products are extremely small in comparison with the risks associated with inadequate disinfection, and it is important that disinfection efficacy not be compromised in attempting to control such by-products.

Disinfection should not be compromised in attempting to control disinfection by-products.

Some disinfectants, such as chlorine, can be easily monitored and controlled as a drinking-water disinfectant, and frequent monitoring is recommended wherever chlorination is practised.

Disinfection of drinking-water is considered in more detail in [chapter 7](#) and [Annex 5](#), with fact sheets on specific disinfectants and disinfection by-products provided in [chapter 12](#).

1.1.4 Chemical aspects

The health concerns associated with chemical constituents of drinking-water differ from those associated with microbial contamination and arise primarily from the ability of chemical constituents to cause adverse health effects after prolonged periods of exposure. There are few chemical constituents of water that can lead to health problems resulting from a single exposure, except through massive accidental contamination of a drinking-water supply. Moreover, experience shows that in many, but not all, such incidents, the water becomes undrinkable owing to unacceptable taste, odour and appearance.

The great majority of evident water-related health problems are the result of microbial (bacterial, viral, protozoan or other biological) contamination. Nevertheless, an appreciable number of serious health concerns may occur as a result of the chemical contamination of drinking-water.

In situations where short-term exposure is not likely to lead to health impairment, it is often most effective to concentrate the available resources for remedial action on finding and eliminating the source of contamination, rather than on installing expensive drinking-water treatment for the removal of the chemical constituent.

There are many chemicals that may occur in drinking-water; however, only a few are of immediate health concern in any given circumstance. The priority given to both

monitoring and remedial action for chemical contaminants in drinking-water should be managed to ensure that scarce resources are not unnecessarily directed towards those of little or no health concern (see the supporting documents *Chemical safety of drinking-water* and *Developing drinking-water quality regulations and standards*; [Annex 1](#)).

There are few chemicals for which the contribution from drinking-water to overall intake is an important factor in preventing disease. One example is the effect of fluoride in drinking-water in protecting against dental caries. The Guidelines do not attempt to define minimum desirable concentrations for chemicals in drinking-water.

Guideline values are derived for many chemical constituents of drinking-water. A guideline value normally represents the concentration of a constituent that does not result in any significant risk to health over a lifetime of consumption. A number of provisional guideline values have been established based on the practical level of treatment performance or analytical achievability. In these cases, the guideline value is higher than the calculated health-based value.

The chemical aspects of drinking-water quality are considered in more detail in [chapter 8](#), with fact sheets on specific chemical contaminants provided in [chapter 12](#).

1.1.5 Radiological aspects

The health risks associated with the presence of naturally occurring radionuclides in drinking-water should also be taken into consideration, although the contribution of drinking-water to total exposure to radionuclides is very small under normal circumstances.

Formal guideline values are not set for individual radionuclides in drinking-water. Rather, the approach used is based on screening drinking-water for gross alpha and gross beta radiation activity. Although finding levels of activity above screening values does not indicate any immediate risk to health, it should trigger further investigation to determine the radionuclides responsible and the possible risks, taking local circumstances into account.

The guidance levels for radionuclides recommended in these Guidelines do not apply to drinking-water supplies contaminated during emergencies arising from accidental releases of radioactive substances to the environment.

Radiological aspects of drinking-water quality are considered in more detail in [chapter 9](#).

1.1.6 Acceptability aspects: taste, odour and appearance

Water should be free of tastes and odours that would be objectionable to the majority of consumers.

In assessing the quality of drinking-water, consumers rely principally upon their senses. Microbial, chemical and physical constituents of water may affect the appearance, odour or taste of the water, and the consumer will evaluate the quality and acceptability of the water on the basis of these criteria. Although these constituents may have no direct health effects, water that is highly turbid, is highly coloured or has an objectionable taste or odour may be regarded by consumers as unsafe and rejected.

In extreme cases, consumers may avoid aesthetically unacceptable but otherwise safe drinking-water in favour of more pleasant but potentially unsafe sources. It is therefore wise to be aware of consumer perceptions and to take into account both health-related guideline values and aesthetic criteria when assessing drinking-water supplies and developing regulations and standards.

Changes in the normal appearance, taste or odour of a drinking-water supply may signal changes in the quality of the raw water source or deficiencies in the treatment process and should be investigated.

Acceptability aspects of drinking-water quality are considered in more detail in [chapter 10](#).

1.2 Roles and responsibilities in drinking-water safety management

Preventive management is the preferred approach to ensuring drinking-water safety and should take account of the characteristics of the drinking-water supply from catchment and source to its use by consumers. As many aspects of drinking-water

A preventive integrated management approach with collaboration from all relevant agencies is the preferred approach to ensuring drinking-water safety.

quality management are often outside the direct responsibility of the water supplier, it is essential that a collaborative multiagency approach be adopted to ensure that agencies with responsibility for specific areas within the water cycle are involved in the management of water quality. One example is where catchments and source waters are beyond the drinking-water supplier's jurisdiction. Consultation with other authorities will generally be necessary for other elements of drinking-water quality management, such as monitoring and reporting requirements, emergency response plans and communication strategies.

Major stakeholders that could affect or be affected by decisions or activities of the drinking-water supplier should be encouraged to coordinate their planning and management activities where appropriate. These could include, for example, health and resource management agencies, consumers, industry and plumbers. Appropriate mechanisms and documentation should be established for stakeholder commitment and involvement.

1.2.1 Surveillance and quality control

In order to protect public health, a dual-role approach, differentiating the roles and responsibilities of service providers from those of an authority responsible for independent oversight protective of public health ("drinking-water supply surveillance"), has proven to be effective.

Organizational arrangements for the maintenance and improvement of drinking-water supply services should therefore take into account the vital and complementary roles of the agency responsible for surveillance and of the water supplier. The two functions of surveillance

Drinking-water suppliers are responsible at all times for the quality and safety of the water that they produce.

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and quality control are best performed by separate and independent entities because of the conflict of interest that arises when the two are combined. In this:

- national agencies provide a framework of targets, standards and legislation to enable and require suppliers to meet defined obligations;
- agencies involved in supplying water for consumption by any means should be required to ensure and verify that the systems they administer are capable of delivering safe water and that they routinely achieve this;
- a surveillance agency is responsible for independent (external) surveillance through periodic audit of all aspects of safety and/or verification testing.

In practice, there may not always be a clear division of responsibilities between the surveillance and drinking-water supply agencies. In some cases, the range of professional, governmental, nongovernmental and private institutions may be wider and more complex than that discussed above. Whatever the existing framework, it is important that clear strategies and structures be developed for implementing WSPs, quality control and surveillance, collating and summarizing data, reporting and disseminating the findings and taking remedial action. Clear lines of accountability and communication are essential.

Surveillance is an investigative activity undertaken to identify and evaluate potential health risks associated with drinking-water. Surveillance contributes to the protection of public health by promoting improvement of the quality, quantity, accessibility, coverage (i.e. populations with reliable access), affordability and continuity of drinking-water supplies (termed “service indicators”). The surveillance authority must have the authority to determine whether a water supplier is fulfilling its obligations.

Surveillance of drinking-water quality can be defined as “the continuous and vigilant public health assessment and review of the safety and acceptability of drinking-water supplies” (WHO, 1976).

In most countries, the agency responsible for the surveillance of drinking-water supply services is the ministry of health (or public health) and its regional or departmental offices. In some countries, it may be an environmental protection agency; in others, the environmental health departments of local government may have some responsibility.

Surveillance requires a systematic programme of surveys, which may include auditing, analysis, sanitary inspection and institutional and community aspects. It should cover the whole of the drinking-water system, including sources and activities in the catchment, transmission infrastructure, treatment plants, storage reservoirs and distribution systems (whether piped or unpiped).

Ensuring timely action to prevent problems and ensure the correction of faults should be one aim of a surveillance programme. There may at times be a need for penalties to encourage and ensure compliance. The surveillance agency must therefore be supported by strong and enforceable legislation. However, it is important that the agency develops a positive and supportive relationship with suppliers, with the application of penalties used as a last resort.

The surveillance agency should be empowered by law to compel water suppliers to recommend the boiling of water or other measures when microbial contamination that could threaten public health is detected.

1.2.2 Public health authorities

In order to effectively support the protection of public health, a national entity with responsibility for public health will normally act in four areas:

- 1) *surveillance of health status and trends*, including outbreak detection and investigation, generally directly but in some instances through a decentralized body;
- 2) directly establishing drinking-water *norms and standards*. National public health authorities often have the primary responsibility for setting norms on drinking-water supply, which may include the setting of water quality targets, performance and safety targets and directly specified requirements (e.g. treatment). Normative activity is not restricted to water quality but also includes, for example, regulation and approval of materials and chemicals used in the production and distribution of drinking-water (see [section 8.5.4](#)) and establishing minimum standards in areas such as domestic plumbing (see [section 1.2.10](#)). Nor is it a static activity, because as changes occur in drinking-water supply practice, in technologies and in materials available (e.g. in plumbing materials and treatment processes), so health priorities and responses to them will also change;
- 3) representing health concerns in *wider policy development*, especially health policy and integrated water resource management (see [section 1.2.4](#)). Health concerns will often suggest a supportive role towards resource allocation to those concerned with drinking-water supply extension and improvement, will often involve lobbying for the primary requirement to satisfy drinking-water needs above other priorities and may imply involvement in conflict resolution;
- 4) *direct action*, generally through subsidiary bodies (e.g. regional and local environmental health administrations) or by providing guidance to other local entities (e.g. local government) in surveillance of drinking-water supplies. These roles vary widely according to national and local structures and responsibilities and frequently include a supportive role to community suppliers and households, where local authorities often intervene directly.

Public health surveillance (i.e. surveillance of health status and trends) contributes to verifying drinking-water safety. It takes into consideration disease in the entire population, which may be exposed to pathogenic microorganisms from a range of sources, not only drinking-water. National public health authorities may also undertake or direct research to evaluate the role of water as a risk factor in disease, through case-control, cohort or intervention studies, for example. Public health surveillance teams typically operate at national, regional and local levels, as well as in cities and rural health centres. Routine surveillance includes:

- ongoing monitoring of reportable diseases, many of which can be caused by waterborne pathogens;
- outbreak detection;

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- long-term trend analysis;
- geographic and demographic analysis;
- feedback to water authorities.

Public health surveillance can be enhanced in a variety of ways to identify possible waterborne outbreaks in response to suspicion about unusual disease incidence or following deterioration of water quality. Epidemiological investigations include:

- outbreak investigations;
- intervention studies to evaluate intervention options;
- case-control or cohort studies to evaluate the role of water as a risk factor in disease.

However, public health surveillance cannot be relied upon to provide information in a timely manner to enable short-term operational response to control waterborne disease. Limitations include:

- outbreaks of non-reportable disease;
- time delay between exposure and illness;
- time delay between illness and reporting;
- low level of reporting;
- difficulties in identifying causative pathogens and sources.

The public health authority operates reactively, as well as proactively, against the background of overall public health policy and in interaction with all stakeholders. In accounting for public health context, priority will normally be afforded to disadvantaged groups. This will generally entail balancing drinking-water safety management and improvement with the need to ensure access to reliable supplies of safe drinking-water in adequate quantities.

In order to develop an understanding of the national drinking-water situation, the national public health authority should periodically produce reports outlining the state of national water quality and highlighting public health concerns and priorities in the context of overall public health priorities. This implies the need for effective exchange of information between local, regional and national agencies.

National health authorities should lead or participate in the formulation and implementation of policy to ensure access to some form of reliable, safe drinking-water supply. Where this has not been achieved, appropriate tools and education should be made available to implement individual or household-level treatment and safe storage.

1.2.3 Local authorities

Local environmental health authorities often play an important role in managing water resources and drinking-water supplies. This may include catchment inspection and authorization of activities in the catchment that may have an impact on source water quality. It can also include verifying and auditing (surveillance) of the management of formal drinking-water supplies. Local environmental health authorities may also give specific guidance for the design and implementation of community and household drinking-water systems and correcting deficiencies, and they may also be responsible for surveillance of water supplies managed by communities and

households. They have an important role to play in educating consumers where household water treatment is necessary.

Support provided by local authorities generally includes education programmes about drinking-water supply and water quality. Such programmes should normally include:

- water hygiene awareness raising;
- basic technical training and technology transfer in drinking-water supply and management;
- consideration of and approaches to overcoming sociocultural barriers to acceptance of water quality interventions;
- motivation, mobilization and social marketing activities;
- a system of continued support, follow-up and dissemination of the water quality programme to achieve and maintain sustainability.

These programmes can be administered at the community level by local health authorities or other entities, such as nongovernmental organizations and the private sector. If the programme arises from other entities, the involvement of the local health authority in the development and implementation of the water quality education and training programme is strongly encouraged.

Behaviour change approaches for sanitation and hygiene to be implemented by local authorities are summarized in Table 5.1 of the WHO *Guidelines on sanitation and health* (WHO, 2018).

1.2.4 Catchment protection

Catchment protection is an integral aspect of the proactive management of drinking-water quality. Prevention of microbial and chemical contamination of source water is the first barrier against drinking-water contamination of public health concern.

Natural and human use factors, along with catchment management will influence water quality downstream and in aquifers. This will have an impact on the treatment steps required to ensure safe water, and preventive action may be preferable to upgrading treatment.

The characteristics of the local catchment and influence of land use on water quality should be assessed as part of catchment management and more broadly, as part of WSPs (see [section 1.2.5](#)). This assessment is not normally undertaken by health authorities or drinking-water supply agencies alone and should take into consideration:

- land cover modification;
- extraction activities;
- construction/modification of waterways;
- application of fertilizers, herbicides, pesticides and other chemicals;
- livestock density and application of manure;
- road construction, maintenance and use;
- various forms of recreation;
- urban or rural residential development, with particular attention to excreta disposal, sanitation, landfill and waste disposal;

- other potentially polluting human activities, such as industry, mining and military sites.

Catchment management may be the responsibility of catchment management agencies and/or other entities controlling or affecting water resources, such as industrial, agricultural, navigation and flood control entities.

The extent to which the responsibilities of health or drinking-water supply agencies include catchment management varies greatly between countries and communities. Regardless of government structures and sector responsibilities, it is important that health authorities liaise and collaborate with sectors managing the water resource and regulating land use in the catchment.

Establishing close collaboration between the public health authority, water supplier and catchment management agency assists recognition of the health hazards potentially occurring in the system. It is also important for ensuring that the protection of drinking-water sources is considered in decisions for land use or regulations to control contamination of water resources. Depending on the setting, this may include involvement of further sectors, such as agriculture, traffic, tourism or urban development.

To ensure the adequate protection of drinking-water sources through catchment management, national authorities will normally interact with other sectors in formulating broader national policy for integrated water resource management. Regional and local structures for implementing the policy will be set up, and national authorities will guide regional and local authorities by providing tools.

Regional environmental or public health authorities have an important task in participating in the preparation of integrated water resource management plans to ensure the best available drinking-water source quality. For further information, see the supporting documents *Protecting groundwater for health* and *Protecting surface water for health* (Annex 1).

1.2.5 Drinking-water suppliers

Drinking-water supplies vary from very large urban systems servicing large populations with tens of millions of people to small supplies providing water to very small populations (see [section 1.2.6](#) for more information on small water supplies).

Drinking-water suppliers are responsible for quality assurance and quality control (see [section 1.2.1](#)). Their key responsibilities are to prepare and implement WSPs (for more information, see [chapter 4](#)).

Although water supply planning should address the catchment that may impact source water quality, in many cases, the water supplier is not responsible for its management. The roles of the water supplier with respect to catchments are to participate in interagency water resource management activities, and as part of water safety planning, to understand the risks arising from potentially contaminating activities and incidents and to use this information in assessing risks to the drinking-water supply and developing and applying appropriate management. Although drinking-water suppliers may not undertake catchment surveys and pollution risk assessment alone, their role is to recognize the need for them and to initiate multiagency collaboration—for example, with health and environmental authorities.

Experience has shown that an association of stakeholders in drinking-water supply (e.g. operators, managers and specialist groups such as small water suppliers, scientists, sociologists, legislators and politicians) can provide a valuable non-threatening forum for the interchange of ideas.

1.2.6 Small water supplies (including community-managed supplies)

Small water supplies including community-managed systems are common worldwide in developed and developing countries. Small water supplies may be piped or non-piped, and they may be managed by households, communities or professional entities (e.g. municipalities, private operators, public utilities and other formalized entities). Although small water supplies are diverse, these supplies are often associated with operational, managerial, technical and resourcing challenges. For example, there may be reliance on often untrained and sometimes unpaid community members in the administration and operation of these types of supplies. Drinking-water supplies in periurban areas—the communities surrounding major towns and cities—in developing countries may also have the characteristics of small water supplies. Given these common challenges, small water supplies, including community-managed supplies, require explicit consideration in policies and regulations, and require the provision of tailored support.

Effective and sustainable programmes for community-managed drinking-water systems require the active support and involvement of local communities. These communities should be involved at all stages of such programmes, including initial surveys; decisions on siting of wells, siting of intakes or establishing protection zones; monitoring and surveillance of drinking-water supplies; reporting faults, carrying out maintenance and taking remedial action; and supportive actions, including sanitation and hygiene practices.

A community may already be highly organized and taking action on health or drinking-water supply issues. Alternatively, it may lack a well-developed drinking-water system; some sectors of the community, such as women, may be poorly represented; and there may be disagreements or factional conflicts. In these situations, achieving community participation will take more time and effort to bring people together, resolve differences, agree on common aims and take action. Visits, possibly over several years, will often be needed to provide support and encouragement and to ensure that the structures created for safe drinking-water supply continue to operate. This may involve setting up hygiene and health educational programmes to ensure that the community:

- is aware of the importance of drinking-water quality and its relationship with health and of the need for safe drinking-water in sufficient quantities for domestic use for drinking, cooking and hygiene;
- recognizes the importance of surveillance and the need for a community response;
- understands and is prepared to play its role in the surveillance process;
- has the necessary skills to perform that role;
- is aware of requirements for the protection of drinking-water supplies from pollution.

For further information, see *Guidelines for drinking-water quality: small water supplies* (WHO, 2024); the supporting documents *Water safety planning for small community water supplies* and *Compendium of drinking-water systems and technologies from source to consumer* ([Annex 1](#)); Simpson-Hébert, Sawyer & Clarke (1996); Sawyer, Simpson-Hébert & Wood (1998); and Brikké (2000).

1.2.7 Water vendors

Vendors selling water to households or at collection points are common in many parts of the world where scarcity of water or faults in or lack of infrastructure limits access to suitable quantities of drinking-water. Water vendors use a range of modes of transport to carry drinking-water for sale directly to the consumer, including tanker trucks and wheelbarrows or trolleys. In the context of these Guidelines, water vending does not include bottled or packaged water (which is considered in [section 6.15](#)) or water sold through vending machines.

There are a number of health concerns associated with water supplied to consumers by water vendors. These include access to adequate volumes and concern regarding inadequate treatment or transport in inappropriate containers, which can result in contamination.

More detailed information on treatment of vended water, undertaking a risk assessment of vended water supplies, operational monitoring of control measures, management plans and independent surveillance is included in [section 6.3](#).

1.2.8 Individual consumers

Everyone consumes water from one source or another, and consumers often play important roles in the collection, treatment and storage of water. Consumer actions may help to ensure the safety of the water they consume and may also contribute to improvement or contamination of the water consumed by others. Consumers have the responsibility for ensuring that their actions do not have an adverse impact on water quality. Installation and maintenance of household plumbing systems should be undertaken preferably by qualified and authorized plumbers (see [section 1.2.10](#)) or other persons with appropriate expertise to ensure that cross-connections or backflow events do not result in contamination of local water supplies.

In most countries, there are populations whose water is derived from household sources, such as private wells and rainwater. In households using non-piped water supplies, appropriate efforts are needed to ensure safe collection, storage and perhaps treatment of their drinking-water. In some circumstances, households and individuals may wish to treat water in the home to increase their confidence in its safety. This would be relevant where community supplies are absent or where community supplies are known to be contaminated or causing waterborne disease (see [chapter 7](#)). Public health surveillance or other local authorities may provide guidance to support households and individual consumers in ensuring the safety of their drinking-water. Such guidance is best provided in the context of a community education and training programme.

1.2.9 Certification agencies

Certification is used to verify that devices and materials used in the drinking-water supply meet a given level of quality and safety. Certification is a process in which an independent organization validates the claims of the manufacturers against a formal standard or criterion or provides an independent assessment of possible risks of contamination from a material or process. The certification agency may be responsible for seeking data from manufacturers, generating test results, conducting inspections and audits and possibly making recommendations on product performance.

Certification has been applied to technologies used at household and community levels, such as hand pumps; materials used by water supplies, such as treatment chemicals; and devices used in the household for collection, treatment and storage.

Certification of products or processes involved in the collection, treatment, storage and distribution of water can be overseen by government agencies or private organizations. Certification procedures will depend on the standards against which the products are certified, certification criteria and the party that performs the certification.

Certification can also be applied to the implementation of WSPs. This can take the form of an independent organization or party undertaking audits to verify that plans have been properly designed, are being implemented correctly and are effective.

National, local government or private (third-party auditing) certification programmes have a number of possible objectives:

- certification of products to ensure that their use does not threaten the safety of the user or the general public, such as by causing contamination of drinking-water with toxic substances, substances that could affect consumer acceptability or substances that support the growth of microorganisms;
- product testing, to avoid retesting at local levels or prior to each procurement;
- ensuring uniform quality and condition of products;
- certification and accreditation of analytical and other testing laboratories;
- control of materials and chemicals used for the treatment of drinking-water, including the performance of devices for household use;
- ensuring that WSPs are effective.

An important step in any certification procedure is the establishment of standards, which must form the basis of assessment of the products. These standards should also—as far as possible—contain the criteria for approval. In procedures for certification on technical aspects, these standards are generally developed in cooperation with the manufacturers, the certifying agency and the consumers. The national public health authorities should have responsibility for developing the parts of the approval process or criteria relating directly to public health. For further information on the control of materials and chemicals used for the treatment of drinking-water, see [section 8.5.4](#).

1.2.10 Plumbing

Significant adverse health effects have been associated with inadequate plumbing systems within public and private buildings arising from poor design, incorrect installation, alterations and inadequate maintenance.

Numerous factors influence the quality of water within a building's piped distribution system and may result in microbial or chemical contamination of drinking-water. Outbreaks of gastrointestinal disease can occur through faecal contamination of drinking-water within buildings arising from deficiencies in roof storage tanks and cross-connections with wastewater pipes, for example. Poorly designed plumbing systems can cause stagnation of water and provide a suitable environment for the proliferation of opportunistic pathogens including *Legionella*. Plumbing materials, pipes, fittings and coatings can result in elevated heavy metal (e.g. lead) concentrations in drinking-water, and inappropriate materials can be conducive to bacterial growth. Potential adverse health effects may not be confined to the individual building. Exposure of other consumers to contaminants is possible through contamination of the local public distribution system, beyond the particular building, through cross-contamination of drinking-water and backflow.

The delivery of water that complies with relevant standards within buildings generally relies on a plumbing system that is not directly managed by the water supplier. Reliance is therefore placed on proper installation of plumbing and, for larger buildings, on building-specific WSPs (see [section 6.10](#)).

To ensure the safety of drinking-water supplies within the building system, plumbing practices must prevent the introduction of hazards to health. This can be achieved by ensuring that:

- pipes carrying either water or wastes are watertight, durable, of smooth and unobstructed interior and protected against anticipated stresses;
- cross-connections between the drinking-water supply and the wastewater removal systems do not occur;
- roof storage systems are intact and not subject to intrusion of microbial or chemical contaminants;
- hot and cold water systems are designed to minimize the proliferation of *Legionella* (see also [sections 6.10, 6.11 and 11.1](#));
- appropriate protection is in place to prevent backflow;
- the system design of multistorey buildings minimizes pressure fluctuations;
- waste is discharged without contaminating drinking-water;
- plumbing systems function efficiently.

It is important that plumbers are appropriately qualified, have the competence to undertake necessary servicing of plumbing systems to ensure compliance with local regulations and use only materials approved as safe for use with drinking-water.

Design of the plumbing systems of new buildings should normally be approved prior to construction and be inspected by an appropriate regulatory body during construction and prior to commissioning of the buildings.

For more information on the essential roles of proper drinking-water system and waste system plumbing in public health, see the supporting document *Health aspects of plumbing* ([Annex 1](#)).

1.3 Supporting resources to the Guidelines

1.3.1 Published documents

These Guidelines are accompanied by separate texts that provide background information substantiating the derivation of the Guidelines and providing guidance on good practice towards their effective implementation. These are available as published texts and for download from the WHO website. Reference details are provided in [Annex 1](#).

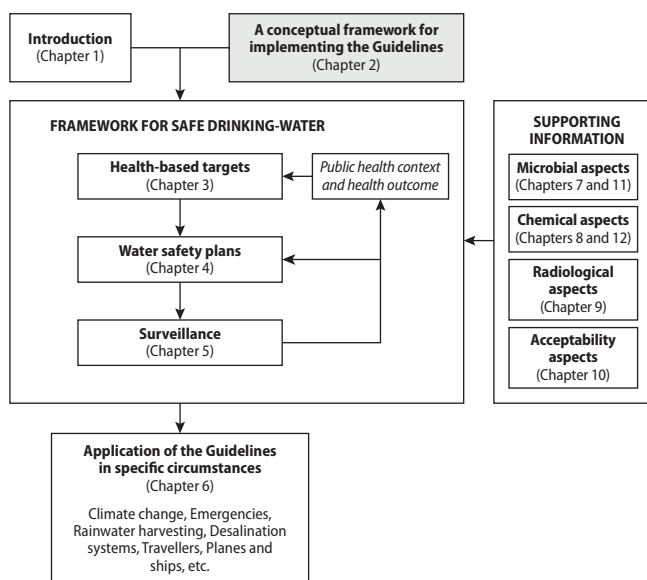
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A conceptual framework for implementing the Guidelines

The basic and essential requirement to ensure the safety of drinking-water is the implementation of a “framework for safe drinking-water” based on the Guidelines. This framework provides a preventive, risk-based approach to managing water quality. It would be composed of health-based targets established by a competent health authority using the Guidelines as a start-

ing point, adequate and properly managed systems (adequate infrastructure, proper monitoring and effective planning and management) and a system of independent surveillance. Such a framework would normally be enshrined in national standards, regulations, or guidelines, in conjunction with relevant policies and programmes (see [sections 2.6](#) and [2.7](#)). Resultant regulations and policies should be appropriate to local circumstances, taking into consideration environmental, social, economic and cultural issues and priority setting.

The framework for safe drinking-water is a preventive management approach comprising three key components:



- 1) health-based targets based on an evaluation of health risks ([section 2.1](#) and [chapter 3](#));
- 2) water safety plans (WSPs), which identify and manage priority risks throughout the entire water supply ([section 2.2](#) and [chapter 4](#));
- 3) a system of independent surveillance that verifies that the above are operating properly ([section 2.3](#) and [chapter 5](#)).

Verification to determine whether the performance of the drinking-water supply is in compliance with the health-based targets and whether the WSP itself is effective may be undertaken by the supplier, surveillance agencies or a combination of the two (see [section 4.3.2](#)).

2.1 Health-based targets

Health-based targets are an essential component of the drinking-water safety framework. They should be established by a high-level authority responsible for health in consultation with others, including water suppliers and affected communities. They should take account of the overall public health situation and contribution of drinking-water quality to disease due to waterborne microbes and chemicals, as a part of overall water and health policy. They must also take account of the importance of ensuring access to water for all consumers.

Health-based targets provide the basis for the application of the Guidelines to all types of drinking-water suppliers. Some constituents of drinking-water may cause adverse health effects from single exposures (e.g. pathogenic microorganisms) or long-term exposures (e.g. many chemicals). Because of the range of constituents in water, their mode of action and the nature of fluctuations in their concentrations, there are four principal types of health-based targets used as a basis for identifying safety requirements:

- 1) *Health outcome targets*: Where waterborne disease contributes to a measurable and significant burden, reducing exposure through drinking-water has the potential to appreciably reduce the risks and incidence of disease. In such circumstances, it is possible to establish a health-based target in terms of a quantifiable reduction in the overall level of disease. This is most applicable where adverse effects follow shortly after exposure, where such effects are readily and reliably monitored and where changes in exposure can also be readily and reliably monitored. This type of health outcome target is primarily applicable to some microbial hazards in developing countries and chemical hazards with clearly defined health effects largely attributable to water (e.g. fluoride, nitrate/nitrite and arsenic). In other circumstances, health outcome targets may be the basis for evaluation of results through quantitative risk assessment models. In these cases, health outcomes are estimated based on information concerning high-dose exposure and dose-response relationships. The results may be employed directly as a basis for the specification of water quality targets or provide the basis for development of the other types of health-based targets. Health outcome targets based on information on the impact of tested interventions on the health of real populations are ideal,

but rarely available. More common are health outcome targets based on defined levels of tolerable risk, either absolute or fractions of total disease burden, usually based on toxicological studies in experimental animals and occasionally based on epidemiological evidence.

- 2) *Water quality targets*: Water quality targets are established for individual drinking-water constituents that represent a health risk from long-term exposure and where fluctuations in concentration are small. They are typically expressed as guideline values (concentrations) of the substances or chemicals of concern.
- 3) *Performance targets*: Performance targets are employed for constituents where short-term exposure represents a public health risk or where large fluctuations in numbers or concentration can occur over short periods with significant health implications. These are typically technology based and expressed in terms of required reductions of the substance of concern or effectiveness in preventing contamination.
- 4) *Specified technology targets*: National regulatory agencies may establish other recommendations for specific actions for smaller municipal, community and household drinking-water supplies. Such targets may identify specific permissible devices or processes for given situations and/or for generic drinking-water system types.

It is important that health-based targets are realistic under local operating conditions and are set to protect and improve public health. Health-based targets underpin the development of WSPs, provide information with which to evaluate the adequacy of existing installations and assist in identifying the level and type of inspection and analytical verifications that are appropriate.

Most countries apply several types of targets for different types of supplies and different contaminants. In order to ensure that they are relevant and supportive, representative scenarios should be developed, including description of assumptions, management options, control measures and indicator systems for performance tracking and verification, where appropriate. These should be supported by general guidance addressing the identification of national, regional or local priorities and progressive implementation, thereby helping to ensure that best use is made of limited resources.

Health-based targets are considered in more detail in [chapter 3](#).

For guidance on how to prioritize constituents based on greatest risk to public health, the reader should refer to [section 2.5](#) and the supporting document *Chemical safety of drinking-water* ([Annex 1](#)).

2.2 WSPs

Water safety planning is a comprehensive risk assessment and management approach that encompasses all steps in the water supply from catchment to consumer. The key components of a WSP are:

- *preparation*: assembling a multidisciplinary team with the expertise and authority to lead WSP development and implementation ([section 4.1](#));

- *system assessment*: identifying risks within the drinking-water supply (from source through treatment to the point of consumption) and determining where improvements are needed ([section 4.2](#));
- *monitoring*: ensuring control measures in the drinking-water supply are operating as intended and that the WSP is effective ([section 4.3](#));
- *management and communication*: describing actions to be taken in normal operation, incident conditions and emergency situations ([section 4.4](#));
- *review and improvement*: ensuring the WSP remains effective by regularly assessing and updating it in response to operational experiences, as well as system changes and audit findings, thereby supporting continuous improvement ([section 4.5](#)).

The elements of a WSP build on the multiple-barrier principle, the principles of hazard analysis and critical control points and other systematic management approaches.

Many drinking-water supplies provide adequate safe drinking-water in the absence of formalized WSPs. Major benefits of developing and implementing a WSP for these supplies include the systematic and detailed assessment and prioritization of hazards, strengthened operational procedures, more targeted and effective monitoring, improved documentation and better targeting of capital investments. In addition, a WSP provides for an organized and structured system to minimize the chance of failure through oversight or lapse of management and for contingency plans to respond to system failures or unforeseen events that may have an impact on water quality, such as increasing severe droughts, heavy rainfall or flood events.

WSPs are applicable to large utilities with piped distribution systems, to piped and non-piped community supplies. The complexity of a WSP should vary with the circumstances. The development of formal WSPs at the household level may not always be practical, but promotion of sanitary inspection with simple good practice is important. WSPs are considered in more detail in [chapter 4](#). This chapter also includes information on the role of sanitary inspections.

2.3 Surveillance

Surveillance agencies are responsible for an independent (external) and periodic review of all aspects of quality and public health safety and should have the power to investigate and to compel action to respond to and rectify incidents of contamination-caused outbreaks of waterborne disease or other threats to public health. The act of surveillance includes identifying potential drinking-water contamination and waterborne illness events and promoting and supporting improvement of the quality, quantity, accessibility, coverage, affordability and continuity of drinking-water supplies.

Surveillance of drinking-water requires a systematic programme of assessment and data collection that should include auditing of WSPs or sanitary inspection, water quality testing, analysis of water quality test results, and institutional and community aspects. It should cover the whole of the drinking-water system, including sources and activities in the catchment, transmission infrastructure, whether piped or unpiped, treatment plants, storage reservoirs and distribution systems.

Surveillance findings should inform incremental improvement of drinking-water supplies, and prioritizing improvements in systems presenting the greatest overall risk to public health is important. Grading schemes can support prioritization by helping to distinguish between lower- and higher-risk sites for the relative safety of drinking-water supplies (see [chapter 4](#)). Such schemes will typically take account of both analytical findings and sanitary inspection or WSP audit through approaches such as those presented in [section 5.5.2](#). This is particularly important for small water supplies where the frequency of testing is low and exclusive reliance on analytical results is particularly inappropriate.

The role of surveillance is discussed in [section 1.2.1](#) and [chapter 5](#).

2.4 Verification of drinking-water quality

Drinking-water safety is secured by application of a WSP, which includes monitoring the efficiency of control measures using appropriately selected determinants. In addition to this operational monitoring, a final verification of quality is required.

Verification is the use of methods, procedures or tests in addition to those used in operational monitoring to determine whether the performance of the drinking-water supply is in compliance with the stated objectives outlined by the health-based targets and whether the WSP needs modification or revalidation.

Verification of drinking-water may be undertaken by the supplier, surveillance agencies or a combination of the two (see [section 4.3.2](#)). Although verification is most commonly carried out by the surveillance agency, a utility-led verification programme can provide an additional level of confidence, supplementing regulations that specify monitoring parameters and frequencies.

2.4.1 Microbial water quality

For microbial water quality, verification is likely to be based on the analysis of faecal indicator microorganisms, with the organism of choice being *Escherichia coli* or, alternatively, thermotolerant coliforms (see [sections 4.3.2, 7.4](#) and [11.6](#)). Monitoring of specific pathogens may be included on very limited occasions to verify that an outbreak was waterborne or that a WSP has been effective. *Escherichia coli* provides conclusive evidence of recent faecal pollution and should not be present in drinking-water. Under certain circumstances, additional indicators, such as bacteriophages or bacterial spores, may be used.

However, water quality can vary rapidly, and all systems are at risk of occasional failure. For example, rainfall can greatly increase the levels of microbial contamination in source waters, and waterborne outbreaks often occur following rainfall. Results of analytical testing must be interpreted taking this into account.

2.4.2 Chemical water quality

Assessment of the adequacy of the chemical quality of drinking-water relies on comparison of the results of water quality analysis with guideline values. These Guidelines provide guideline values for many more chemical contaminants than will actually affect any particular water supply, so judicious choices for monitoring and surveillance should be made prior to initiating an analytical chemical assessment.

For additives (i.e. chemicals deriving primarily from materials and chemicals used in the production and distribution of drinking-water), emphasis is placed on the direct control of the quality of these commercial products. In controlling drinking-water additives, testing procedures typically assess whether the product meets the specifications (see [section 8.5.4](#)).

As indicated in [chapter 1](#), most chemicals are of concern only following long-term exposure; however, some hazardous chemicals that occur in drinking-water are of concern because of effects arising from sequences of exposures over a short period. Where the concentration of the chemical of interest (e.g. nitrate/nitrite, which is associated with methaemoglobinaemia in bottle-fed infants) varies widely, even a series of analytical results may fail to fully identify and describe the public health risk. In controlling such hazards, attention must be given to both knowledge of causal factors such as fertilizer use in agriculture and trends in detected concentrations, as these will indicate whether a significant problem may arise in the future. Other hazards may arise intermittently, often associated with seasonal activity or seasonal conditions. One example is the occurrence of blooms of toxic cyanobacteria in surface water.

A *guideline value* represents the concentration of a constituent that does not exceed tolerable risk to the health of the consumer over a lifetime of consumption. Guideline values for some chemical contaminants (e.g. nitrate) are set to be protective for susceptible subpopulations. These guideline values are also protective of the general population over a lifetime.

It is important that recommended guideline values are scientifically justified, practical and feasible to implement as well as protective of public health. Guideline values are not normally set at concentrations lower than the detection limits achievable under routine laboratory operating conditions. Moreover, some guideline values are established taking into account available techniques for controlling, removing or reducing the concentration of the contaminant to the desired level. In some instances, therefore, provisional guideline values have been set for contaminants for which calculated health-based values are not practically achievable.

2.5 Identifying priority concerns

These Guidelines cover a large number of potential constituents in drinking-water in order to meet the varied needs of countries worldwide. Generally, however, only a few constituents will be of public health concern under any given circumstances. It is essential that the national regulatory agency and local water authorities identify and respond to the constituents of relevance to the local circumstances. This will ensure that efforts and investments can be directed to those constituents that have the greatest risk or public health significance.

Health-based targets are established for potentially hazardous water constituents and provide a basis for assessing drinking-water quality. Different parameters may require different priorities for management to improve and protect public health. In general, the priorities, in decreasing order, are to:

- ensure an adequate supply of microbially safe water and maintain acceptability to discourage consumers from using potentially less microbially safe water;

- manage key chemical hazards known to cause adverse health effects;
- address other chemical hazards, particularly those that affect the acceptability of drinking-water in terms of its taste, odour and appearance;
- apply appropriate technologies to reduce contaminant concentrations in the source to below the guideline or regulated values.

The two key features in choosing hazards for which setting a standard is desirable on health grounds are the health impacts (severity) associated with the substance and the probability of significant occurrence (exposure). Combined, these elements determine the risk associated with a particular hazard. For

Many microbial and chemical constituents of drinking-water can potentially cause adverse human health effects. The detection of these constituents in both raw water and water delivered to consumers is often slow, complex and costly, which limits early warning capability and affordability. Reliance on water quality determination alone is insufficient to protect public health. As it is neither physically nor economically feasible to test for all drinking-water quality parameters, the use of monitoring effort and resources should be carefully planned and directed at significant or key characteristics.

microbial hazards, the setting of targets will be influenced by occurrence and concentrations in source waters and the relative contribution of waterborne organisms to disease. For chemical hazards, the factors to be considered are the severity of health effects and the frequency of exposure of the population in combination with the concentration to which they will be exposed. The probability of health effects clearly depends on the toxicity and the concentration, but it also depends on the period of exposure. For most chemicals, health impacts are associated with long-term exposure. Hence, in the event that exposure is occasional, the risk of an adverse health effect is likely to be low, unless the concentration is extremely high. The substances of highest priority will therefore be those that occur widely, are present in drinking-water sources or drinking-water all or most of the time and are present at concentrations that are of health concern.

Guidance on determining which chemicals are of importance in a particular situation is given in the supporting document *Chemical safety of drinking-water* ([Annex 1](#)).

Although WHO does not set formal guideline values for substances on the basis of consumer acceptability (i.e. substances that affect the appearance, taste or odour of drinking-water), it is not uncommon for standards to be set for substances and parameters that relate to consumer acceptability. Although exceeding such a standard is not a direct issue for health, it may be of great significance for consumer confidence and may lead consumers to obtain their water from an alternative, less safe source. Such standards are usually based on local considerations of acceptability.

Priority setting should be undertaken on the basis of a systematic assessment based on collaborative effort among all relevant agencies and may be applied at national and system-specific levels. At the national level, priorities need to be set in order to identify the relevant hazards, based on an assessment of risk—i.e. severity and exposure. At the level of individual water supplies, it may be necessary to also prioritize constituents for effective system management. These processes may require the input

of a broad range of stakeholders, including health, water resources, drinking-water supply, environment, agriculture and geological services/mining authorities, to establish a mechanism for sharing information and reaching consensus on drinking-water quality issues.

2.5.1 Undertaking a drinking-water quality assessment

In order to determine which constituents are, indeed, of concern, it will be necessary to undertake a drinking-water quality assessment. It is important to identify what types of drinking-water systems are in place in the country (e.g. piped water supplies, non-piped water supplies, vended water) and the quality of drinking-water sources and supplies.

Additional information that should be considered in the assessment includes catchment type (protected, unprotected), geology, topography, sanitary conditions including wastewater discharges, agricultural land use, industrial activities, records of previous monitoring, future projections (e.g. climate, population growth) and local and community knowledge. Information from sanitary inspections and/or WSPs should be considered. The wider the range of data sources used, the more useful the results of the process will be.

In many situations, authorities or consumers may have already identified a number of drinking-water quality problems, particularly where they cause obvious health effects or acceptability problems. These existing problems would normally be assigned a high priority.

Drinking-water supplies that represent the greatest risks to public health should be identified, with resources allocated accordingly.

2.5.2 Assessing microbial priorities

The most common and widespread health risk associated with drinking-water is microbial contamination, the consequences of which mean that its control must always be of paramount importance. Priority needs to be given to improving and developing the drinking-water supplies that represent the greatest public health risk.

The most common and widespread health risk associated with drinking-water is microbial contamination, the consequences of which mean that its control must always be of paramount importance.

Health-based targets for microbial contaminants are discussed in [section 3.2](#), and a comprehensive consideration of microbial aspects of drinking-water quality is contained in [chapter 7](#).

2.5.3 Assessing chemical priorities

Not all of the chemicals with guideline values will be present in all water supplies or, indeed, all countries. If they do exist, they may not be found at levels of concern. Conversely, some chemicals without guideline values, or not addressed in the Guidelines may nevertheless be of legitimate local concern under special circumstances.

Risk management strategies (as reflected in national standards and monitoring activities) and commitment of resources should give priority to those chemicals that pose a risk to human health or to those with significant impacts on the acceptability of water.

Only a few chemicals have been shown to cause widespread health effects in humans as a consequence of exposure through drinking-water when they are present in excessive quantities. These include fluoride, arsenic and nitrate. Human health effects associated with lead (from domestic plumbing) have also been demonstrated in some areas, and there is concern because of the potential extent of exposure to manganese, selenium and uranium in some areas at concentrations of human health significance. Manganese and iron are of widespread significance also because of their effects on acceptability through discoloration. These constituents should be taken into consideration as part of any priority-setting process. In some cases, assessment will indicate that no risk of significant exposure exists at the national, regional or system level.

Drinking-water may be only a minor contributor to the overall exposure to a particular chemical, and in some circumstances controlling the levels in drinking-water, at potentially considerable expense, may have little impact on overall exposure. Drinking-water risk management strategies should therefore be considered in conjunction with other potential sources of human exposure.

The process of “short-listing” chemicals of concern may initially be a simple classification of high and low risk to identify broad issues. This may be refined using data from more detailed assessments and analysis and may take into consideration rare events, variability and uncertainty.

Guidance on how to undertake prioritization of chemicals in drinking-water is provided in the supporting documents *Chemical safety of drinking-water* and *Developing drinking-water quality regulations and standards* ([Annex 1](#)). These deal with issues including:

- the probability of exposure (including the period of exposure) of the consumer to the chemical;
- the concentration of the chemical that is likely to give rise to health effects (see also [section 8.5](#));
- the evidence of health effects or exposure arising through drinking-water, as opposed to other sources, and relative ease of control of the different sources of exposure.

Additional information on the hazards and risks of many chemicals not included in these Guidelines is available from several sources, including WHO Environmental Health Criteria monographs and Concise International Chemical Assessment Documents, reports by the Joint Food and Agriculture Organization of the United Nations (FAO)/WHO Meeting on Pesticide Residues and the Joint FAO/WHO Expert Committee on Food Additives and information from competent national authorities. These information sources have been peer reviewed and provide readily accessible information on toxicology, hazards and risks of many less common contaminants. They can help water suppliers and health officials to decide upon the significance (if any) of a detected chemical and on the response that might be appropriate.

2.6 Developing drinking-water quality standards

Health-based targets, including numeric guideline values and other targets described in the *Guidelines for drinking-water quality*, are not intended to be mandatory limits, but are provided as the scientific point of departure for development of national or regional numerical drinking-water quality standards. No single approach is universally applicable, and the nature and form of drinking-water standards may vary among countries and regions.

In developing national drinking-water standards based on these Guidelines, it will be necessary to take account of a variety of environmental, social, cultural, economic, dietary and other conditions affecting potential exposure. This may lead to national standards that differ appreciably from these Guidelines, both in scope as well as in risk targets. A programme based on modest but realistic goals—including fewer water quality parameters of priority health concern at attainable levels consistent with providing a reasonable degree of public health protection in terms of reduction of disease or disease risk within the population—may achieve more than an overambitious one, especially if targets are upgraded periodically.

To ensure that standards are acceptable to consumers, communities served, together with the major water users, should be involved in the standards-setting process. Public health agencies may be closer to the community than those responsible for its drinking-water supply. At a local level, they also interact with other sectors (e.g. education), and their combined action is essential to ensure active community involvement. Further guidance on considerations for developing drinking-water quality standards and drinking-water regulations (see [section 2.7](#)), including adapting guideline values (see [section 2.6.1](#)) and periodic revision of standards (see [section 2.6.2](#)), is provided in the supporting document *Developing drinking-water quality regulations and standards* ([Annex 1](#)).

2.6.1 Adapting guideline values to locally relevant standards

In order to account for variations in exposure from different sources (e.g. water, food) in different parts of the world, the proportion of the tolerable daily intake allocated to drinking-water in setting guideline values for many chemicals will vary. Where relevant exposure data are available, authorities are encouraged to develop context-specific guideline values that are tailored to local circumstances and conditions. For example, in areas where the intake of a particular contaminant in drinking-water is known to be much greater than that from other sources (e.g. air and food), it may be appropriate to allocate a greater proportion of the tolerable daily intake to drinking-water to derive a guideline value more suited to the local conditions.

Daily water intake can vary significantly in different parts of the world, seasonally and particularly where consumers are involved in manual labour in hot climates. Local adjustments to the daily water consumption value may be needed in setting local standards, as in the case of fluoride, for example.

Volatile substances in water may be released into the air during showering and through a range of other household activities. Under such circumstances, inhalation may become a significant route of exposure. Where such exposure is shown to be important for a particular substance (i.e. high volatility, low ventilation rates and high

rates of showering/bathing), it may be appropriate to adjust the guideline value. For those substances that are particularly volatile, such as chloroform, the correction factor would be approximately equivalent to a doubling of exposure. For further details, the reader should refer to [section 8.2.9](#).

2.6.2 Periodic review and revision of standards

As knowledge increases, there may be changes to specific guideline values or consideration of new hazards for the safety of drinking-water. There may also be changes in the technology of drinking-water treatment and analytical methods for contaminants. Changes in the country context affecting public health risks, as well as changes in capacity for management and monitoring parameters, should also be considered. National or subnational standards must therefore be subjected to periodic review and should be structured in such a way that changes can be made readily. Changes may need to be made to modify standards, remove parameters or add new parameters, but no changes should be made without proper justification through risk assessment and prioritization of resources for protecting public health. Where changes are justified, it is important that they are communicated to all stakeholders.

2.7 Drinking-water regulations and supporting policies and programmes

The incorporation of a preventive risk management and prioritization approach to drinking-water quality regulations, policies and programmes will:

- ensure that regulations support the prioritization of drinking-water quality parameters to be tested, instead of making mandatory the testing of every parameter in these Guidelines;
- ensure implementation of appropriate sanitation measures at community and household levels and encourage action to prevent or mitigate contamination at source;
- identify drinking-water supplies that represent the greatest risks to public health and thus determine the appropriate allocation of resources.

2.7.1 Regulations

The alignment of national drinking-water quality regulations with the principles outlined in these Guidelines will ensure that:

- there is an explicit link between drinking-water quality regulations and the protection of public health;
- regulations are designed to ensure safe drinking-water from source to consumer, using multiple barriers;
- regulations are based on good practices that have been proven to be appropriate and effective over time;
- a variety of tools are in place to build and ensure compliance with regulations, including education and training programmes, incentives to encourage good practices and penalties, if enforcement is required;

- regulations are appropriate and realistic within national, subnational and local contexts, including specific provisions or approaches for certain contexts or types of supplies, such as small drinking-water supplies;
- stakeholder roles and responsibilities, including how they should work together, are clearly defined;
- “what, when and how” information is shared between stakeholders—including consumers—and required action is clearly defined for normal operations and in response to incidents or emergencies;
- regulations are adaptable to reflect changes in contexts, understanding and technological innovation and are periodically reviewed and updated;
- regulations are supported by appropriate policies and programmes.

The aim of drinking-water quality regulations should be to ensure that the consumer has access to sustainable, sufficient and safe drinking-water. Enabling legislation should provide broad powers and scope to related regulations and include public health protection objectives, such as the prevention of waterborne disease and the provision of an adequate supply of drinking-water. Drinking-water regulations should focus on improvements to the provision and safety of drinking-water through a variety of requirements, tools and compliance strategies. Although sanctions are needed within regulations, the principal aim is not to shut down deficient water supplies.

Drinking-water quality regulations are not the only mechanism by which public health can be protected. Other regulatory mechanisms include those related to source water protection, infrastructure, water treatment and delivery, surveillance and response to potential contamination and waterborne illness events.

Drinking-water quality regulations may also provide for interim standards, permitted deviations and exemptions as part of a national or regional policy, rather than as a result of local initiatives. This may take the form of temporary exemptions for certain communities or areas for defined periods of time. Short-term and medium-term targets should be set so that the most significant risks to human health are managed first. Regulatory frameworks should support long-term progressive improvements.

2.7.2 Supporting policies and programmes

Developing and promulgating regulations alone will not ensure that public health is protected. Regulations must be supported by adequate policies and programmes. This includes ensuring that regulatory authorities, such as enforcement agencies, have sufficient resources to fulfil their responsibilities and that the appropriate policy and programme supports are in place to assist those required to comply with regulations. In other words, the appropriate supports need to be in place so that those being regulated and those who are responsible for regulating are not destined to fail.

Implementation or modification of policies and programmes to provide safe drinking-water should not be delayed because of a lack of appropriate regulation. Even where drinking-water regulations do not yet exist, it may be possible to encourage, and even enforce, the supply of safe drinking-water through, for example, educational efforts or commercial, contractual arrangements between consumer and supplier (e.g. based on civil law).

2. A CONCEPTUAL FRAMEWORK FOR IMPLEMENTING THE GUIDELINES

In countries where universal access to safe drinking-water at an acceptable level of service has not been achieved, policies should refer to expressed targets for increases in sustainable access to safe drinking-water. Such policy statements should be consistent with achievement of the United Nations Sustainable Development Goals (<https://sdgs.un.org/goals/>) as part of the 2030 Agenda for Sustainable Development and should take into account the recognition of the human right to water by the United Nations General Assembly (UNGA, 2010a, b) and criteria for adequate water supply outlined in General Comment 15 on the Right to Water of the United Nations Committee on Economic, Social and Cultural Rights (<http://umn.edu/humanrts/gencomm/escgencom15.htm>) and associated documents.

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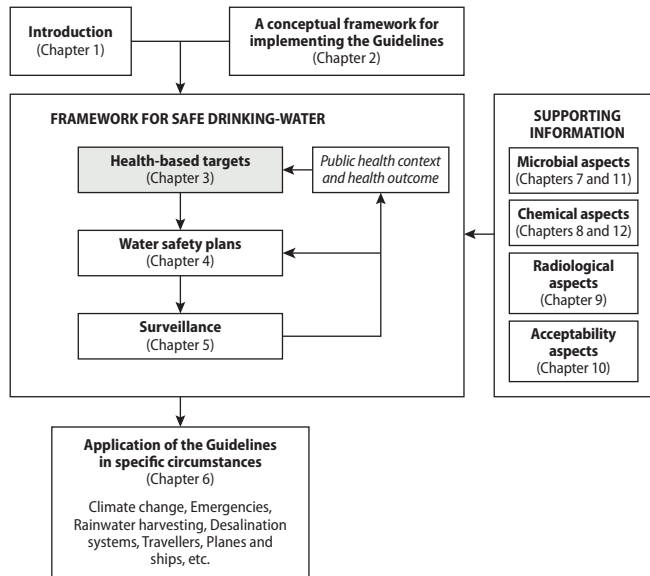
Health-based targets

Health-based targets are measurable health, water quality or performance objectives that are established based on a judgement of safety and on risk assessments of waterborne hazards. These Guidelines describe four distinct types of health-based targets, applicable to all types of hazards and water supplies:

- 1) health outcome targets (e.g. tolerable burdens of disease);
- 2) water quality targets (e.g. guideline values for chemical hazards);
- 3) performance targets (e.g. log reductions of specific pathogens);
- 4) specified technology targets (e.g. application of defined treatment processes).

These targets are common components of existing drinking-water guidelines or standards that are used to protect and improve drinking-water quality and, consequently, human health. They provide benchmarks for water suppliers and regulators to confirm the adequacy of existing systems or the need for improvement. They underpin the development of water safety plans and verification of successful implementation. Where

Health-based targets can be used to support incremental improvement by marking out milestones to guide progress towards water safety and public health goals.



required, health-based targets can be used to support incremental improvement by marking out milestones to guide progress towards water safety and public health goals. This normally requires periodic review and updating of priorities and targets. In turn, norms and standards should also be periodically updated (see [section 2.6.2](#)).

Health-based targets should assist in determining specific interventions appropriate to delivering safe drinking-water, including control measures such as source protection and treatment processes.

3.1 Setting health-based targets

The use of health-based targets is applicable in countries at all levels of development. To ensure effective health protection and improvement, targets need to be realistic, measurable, based on scientific data and relevant to local conditions (including economic, environmental, social and cultural conditions) and financial, technical and institutional resources. Health-based targets should be part of an overall public health policy, taking into account public health status and trends and the contribution of drinking-water to the transmission of infectious disease and to overall exposure to hazardous chemicals both in individual settings and within overall health management.

Although water can be a source of microbial, chemical or radiological hazards, it is by no means the only source. In setting targets, consideration needs to be given to other sources, including food, air, person-to-person contact and consumer products, as well as poor sanitation and personal hygiene. Where the overall burden of disease from multiple exposure routes is very high, there is limited value in setting strict targets for drinking-water. For example, there is limited value in establishing a strict target for a chemical hazard if drinking-water provides only a small proportion of the total exposure to that chemical. The cost of meeting such targets could unnecessarily divert funding from other, more pressing health interventions and is not consistent with the public health objective of reducing overall levels of risk from all sources of exposure to environmental hazards (Prüss-Ustün et al., 2016; WHO, 2019).

It is also important to take account of the impact of the proposed intervention on overall rates of disease. For some pathogens and their associated diseases, interventions in water quality may be ineffective and may therefore not be justified. This may be the case where other routes of exposure dominate. For others, long experience has shown the effectiveness of improving drinking-water supply and quality management in the control of waterborne diseases such as typhoid and dysentery.

Meeting health-based targets should be viewed in the context of broader public health policy, including initiatives to improve sanitation, waste disposal, personal hygiene and public education on ways to reduce both personal exposure to hazards and impacts of personal activity on water resources. Improved public health, reduced carriage of pathogens and reduced human impacts on water

The judgement of safety—or what is a tolerable burden of disease in particular circumstances—is a matter in which society as a whole has a role to play. The final judgement as to whether the benefit resulting from the adoption of any of the health-based targets justifies the cost is for each country to decide.

Table 3.1 Benefits of health-based targets

Target development stage	Benefit
Formulation	Provides insight into the health of the population Reveals gaps in knowledge Supports priority setting Increases the transparency of health policy Promotes consistency among national health programmes Stimulates debate
Implementation	Inspires and motivates collaborating authorities to take action Improves commitment Fosters accountability Guides the rational allocation of resources
Evaluation	Supplies established milestones for incremental improvements Provides opportunity to take action to correct deficiencies and/or deviations Identifies data needs and discrepancies

resources all contribute to drinking-water safety (Howard et al., 2002). Public health prioritization would normally indicate that the major contributors to disease should be dealt with preferentially, taking account of the costs and impacts of potential interventions. However, this does not mean ignoring lesser targets if they can be easily achieved for little cost, as long as this does not divert attention from major targets.

An important concept in the allocation of resources to improving drinking-water safety is the possibility of establishing less stringent transitional targets supported by sound risk management systems in order to encourage incremental improvements of the quality of drinking-water. In this regard, health-based targets can be used as the basis for supporting and measuring incremental progress in water quality improvement. Improvements can relate to progression through increasingly tighter targets or evolution through target types that more precisely reflect the health protection goals (e.g. from specified technology targets to performance targets).

The processes of formulating, implementing, communicating and evaluating health-based targets provide benefits to the overall preventive management of drinking-water quality. These benefits are outlined in [Table 3.1](#).

3.2 Disability-adjusted life years, tolerable disease burden and reference level of risk

At a national level, decisions about risk acceptance and tolerable burdens of disease are complex and need to take account of the probability and severity of impact in addition to the environmental, social, cultural, economic and political dimensions that play important roles in decision-making. Negotiations are an important part of these processes, and the outcome may very well be unique in each situation. Notwithstanding the complexity of these decisions, definitions of tolerable burdens of disease and reference levels of risk are required to provide a baseline for the development of health-based targets and as a point of departure for decisions in specific situations.

Descriptions of tolerable burdens of disease relating to water are typically expressed in terms of specific health outcomes such as maximum frequencies of diarrhoeal disease or cancer incidence. However, these descriptions do not consider the severity of the outcomes. The various hazards that may be present in water are associated with very diverse health outcomes with different impacts ranging from mild diarrhoea to potentially severe outcomes such as typhoid, cancer or skeletal fluorosis.

A common “metric” is needed that can be used to quantify and compare the burden of disease associated with different water-related hazards, taking into account varying probabilities, severities and duration of effects. Such a metric should be applicable regardless of the type of hazard (microbial, chemical or radiological) to enable the use of a consistent approach for each hazard. The metric used in these Guidelines is the disability-adjusted life year, or DALY (Box 3.1). The World Health Organization has used DALYs quite extensively to evaluate public health priorities and to assess the disease burden associated with environmental exposures, particularly for microbial hazards.

A key advantage of using the DALY is its aggregation of different impacts on the quality and quantity of life and its focus on actual outcomes rather than potential risks; hence, it supports rational public health priority setting. DALYs can be used to define tolerable burden of disease and the related reference level of risk.

“Tolerable burden of disease” represents an upper limit of the burden of health effects associated with waterborne disease that is established by national policy-makers. “Reference level of risk” is an equivalent term used in the context of quantitative risk assessments.

In these Guidelines, the tolerable burden of disease is defined as an upper limit of 10^{-6} DALY per person per year. This upper-limit DALY is approximately equivalent to a 10^{-5} excess lifetime risk of cancer (i.e. 1 excess case of cancer per 100 000 people ingesting drinking-water at the water quality target daily over a 70-year period), which is the risk level used in these Guidelines to determine guideline values for genotoxic carcinogens.

Expressing health-based targets for chemical hazards in DALYs has the advantage of enabling comparisons with microbial risks. However, use of the DALY approach for chemicals has been limited in practice due to gaps in knowledge.

The 10^{-6} DALY tolerable burden of disease target may not be achievable or realistic in some locations and circumstances in the near term. Where the overall burden of disease by multiple exposure routes (water, food, air, direct personal contact, etc.) is very high, setting a 10^{-6} DALY per person per year level of disease burden from waterborne exposure alone will have little impact on the overall disease burden. Setting a less stringent level of acceptable risk, such as 10^{-5} or 10^{-4} DALY per person per year, from waterborne exposure may be more realistic, yet still consistent with the goals of providing high-quality, safer water.

3.3 Types of health-based targets

The nature and typical application of health-based targets are presented in Table 3.2. Health-based targets differ considerably with respect to the amount of resources

Box 3.1 Disability-adjusted life years

The various hazards that can be present in water can have very different health outcomes. Some outcomes are mild (e.g. diarrhoea), whereas others can be severe (e.g. cholera, haemolytic uraemic syndrome associated with *Escherichia coli* O157 or cancer). Some are acute (e.g. diarrhoea), whereas others are delayed (e.g. infectious hepatitis or cancer). Some especially relate to certain age ranges and groups (e.g. skeletal fluorosis in older adults often arises from long-term exposure to high levels of fluoride in childhood; infection with hepatitis E virus has a very high mortality rate among pregnant women). In addition, any one hazard may cause multiple effects (e.g. gastroenteritis, Guillain-Barré syndrome, reactive arthritis and mortality associated with *Campylobacter*).

In order to support public health priority setting, a common metric is required that can be applied to all types of hazard and takes into account different health outcomes, including probabilities, severities and duration of effects. The disability-adjusted life year (DALY) provides this metric.

The basic principle of the DALY is to weight each health impact in terms of severity within the range of 0 for good health to 1 for death. The weighting is then multiplied by duration of the effect and the number of people affected. In the case of death, duration is regarded as the years lost in relation to normal life expectancy. Using this approach, a mild diarrhoea with a severity weighting of 0.1 and lasting for 7 days results in a DALY of 0.002, whereas death resulting in a loss of 30 years of life equates to a DALY of 30.

Hence, $DALY = YLL \text{ (years of life lost)} + YLD \text{ (years lived with a disability or illness)}$. In this context, *disability* refers to a condition that detracts from good health.

For example, infection with rotavirus (in developed countries) causes:

- mild diarrhoea (severity rating of 0.1) lasting 7 days in 97.5% of cases;
- severe diarrhoea (severity rating of 0.23) lasting 7 days in 2.5% of cases;
- rare deaths of very young children in 0.015% of cases.

The DALY per case can then be calculated as follows:

$$\begin{aligned} DALY &= (0.1 \times 7/365 \times 0.975) + (0.23 \times 7/365 \times 0.025) + (1 \times 70 \times 0.00015) \\ &= 0.0019 + 0.0001 + 0.0105 \\ &= 0.0125 \end{aligned}$$

Infection with *Cryptosporidium* can cause watery diarrhoea (severity weighting of 0.067) lasting for 7 days with extremely rare deaths in 0.0001% of cases. This equates to a DALY per case of 0.0015.

Further information on the use of DALYs in establishing health-based targets is included in the supporting document *Quantifying public health risk in the WHO Guidelines for drinking-water quality* (Annex 1).

needed for their development and implementation and in relation to the precision with which the public health benefits of risk management actions can be defined. The most precise are health outcome targets, which underpin the derivation of the remaining targets, as shown in [Figure 3.1](#). Each target type is based on those above it in [Table 3.2](#), and assumptions with default values are introduced in moving down between target types. The targets towards the top of the table require greater scientific and technical inputs and are therefore more precisely related to the level of health protection. Target types at the bottom of [Table 3.2](#) require the least interpretation by practitioners in implementation, but depend on a number of assumptions (e.g. establishing specified technology targets in the absence of sufficient source water quality data to apply performance targets for microbial pathogens). Efforts should be made to collect additional information when critical data for applying the next stage of target setting may not be available. This incremental improvement will ensure that the health-based targets will be as pertinent as possible to local circumstances.

Table 3.2 Nature and application of health-based targets

Type of target	Nature of target	Typical applications	Notes
Health outcome	Defined tolerable burden of disease	High-level policy target set at national level, used to inform derivation of performance, water quality and specified technology targets	These Guidelines define a tolerable burden of disease of 10^{-6} DALY per person per year
	No adverse effect or negligible risk	Chemical or radiological hazards	Derived from international chemical or radionuclide risk assessments
Water quality	Guideline values	Chemical hazards	Based on individual chemical risk assessments
		Microbial water quality targets are not normally applied	<i>Escherichia coli</i> is used as an indicator of faecal contamination and to verify water quality
		Radiological water quality targets are not normally applied	Radiological screening levels are applied
Performance	Specified removal of hazards	Microbial hazards (expressed as log reductions)	Specific targets set by water supplier based on quantitative microbial risk assessment and health outcome targets or generic targets set at national level
		Chemical hazards (expressed as percentage removal)	Specific targets set by water supplier based on chemical guideline values or generic targets set at national level
Specified technology	Defined technologies	Control of microbial and chemical hazards	Set at national level; based on assessments of source water quality, frequently underpinned by established or validated performance of the specified technology (e.g. requirement of filtration for surface water)

When establishing health-based targets, care should be taken to account for short-term events and fluctuations in water quality along with “steady-state” conditions. This is particularly important when developing performance and specified technology targets. Short-term water quality can significantly deteriorate, for example, following heavy rain and during maintenance. Catastrophic events can result in periods of very degraded source water quality and greatly decreased efficiency in many processes, or even system failure, greatly increasing the likelihood of a disease outbreak. Events like these provide additional justification for the long-established “multiple-barrier principle” in water safety.

For chemical hazards, health-based targets most commonly take the form of water quality targets, using the guideline values outlined in [section 8.5](#). Performance targets expressed as percentage removals or specified technology targets can also be applied to chemical hazards.

3. HEALTH-BASED TARGETS

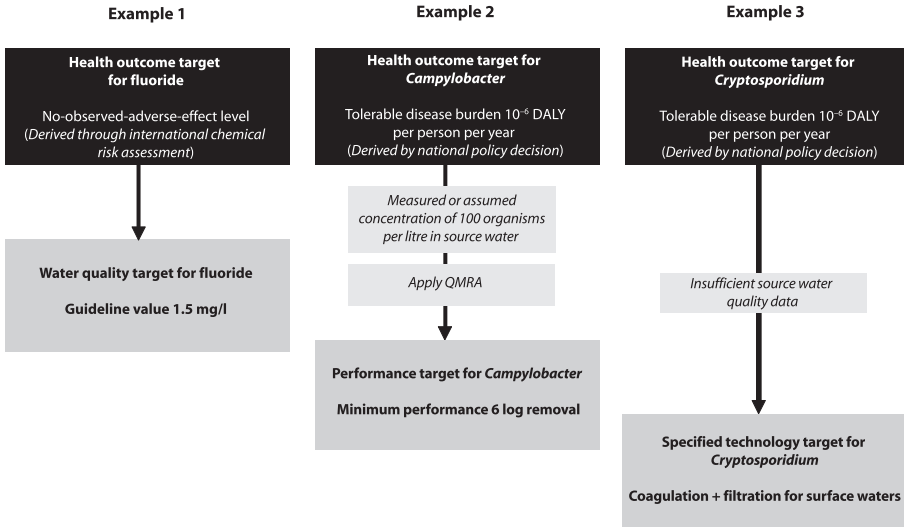


Figure 3.1 Examples of how to set health-based targets for various hazards

For microbial hazards, health-based targets usually take the form of performance or specified technology targets. The choice of target will be influenced by the number of data available on source water quality, with performance targets requiring more information. Water quality targets are typically not developed for pathogens, because monitoring finished drinking-water for pathogens is not considered a feasible or cost-effective option. Concentrations of pathogens equivalent to a health outcome target of 10^{-6} DALY per person per year are typically less than 1 organism per 10^4 – 10^5 litres. Therefore, it is more feasible and cost-effective to monitor for indicator organisms such as *E. coli*.

In practice, risks to public health from drinking-water are often attributable to a single hazard at a time; therefore, in deriving targets, the reference level of risk is applied independently to each hazard.

3.3.1 Health outcome targets

The most direct descriptions of drinking-water safety are health outcome targets, such as upper limits on frequencies of diarrhoeal disease or cancer incidence. These upper limits represent tolerable burdens of disease and are typically set at the national level. They underpin the derivation of water quality, performance and specified technology targets (Figure 3.1). These Guidelines define a tolerable burden of disease of 10^{-6} DALY per person per year. For threshold chemicals, the health outcome target is based on no-observed-adverse-effect levels (see section 8.2).

Health outcome targets must be translated into water quality, performance or specified technology targets in order to be actioned by the water supplier as part of the water safety plan (WSP).

3.3.2 Water quality targets

Water quality targets are the most common form of health-based target applied to chemicals that may be found in drinking-water. The guideline values for individual chemicals described in [section 8.5](#) provide water quality targets that can be used to verify that WSPs have been effective in managing risks from chemicals in drinking-water.

Guideline values are established on the basis of international risk assessments of the health effects associated with exposure to the chemical in water. In developing national drinking-water standards (or health-based targets) based on these guideline values, it will be necessary to take into consideration a variety of environmental, social, cultural, economic, dietary and other conditions affecting potential exposure, as well as the default assumptions that are used to derive the guideline values. Exposure from chemicals in drinking-water is typically minor in comparison with that from other sources (e.g. food, consumer products and air), with a few important exceptions (e.g. arsenic and fluoride). This may lead to national targets that differ appreciably from the guideline values. In some cases, it may be appropriate to take action to prevent exposure to a chemical from sources other than drinking-water (e.g. lead from soldered cans and from paint).

One example is that of the health-based target for fluoride in drinking-water. A guideline value of 1.5 mg/l is recommended in [Table A3.3](#) of [Annex 3](#), with a comment that “Volume of water consumed and intake from other sources should be considered when setting national standards”. Thus, in a country with a warm climate year-round and where piped water is the preferred source of drinking-water, authorities may select a health-based target for fluoride that is lower than this guideline value, as water consumption is expected to be higher. On a similar note, the health-based target should be reviewed in terms of its impact on the most vulnerable section of the population.

Where water treatment processes have been put in place to remove or reduce specific chemicals (see [section 8.4](#) and [Annex 5](#)), water quality targets should be used to determine appropriate treatment requirements.

It is important that water quality targets are established only for those chemicals that, following rigorous assessment, have been determined to be of health concern or of concern for the acceptability of the drinking-water to consumers. There is little value in undertaking measurements for chemicals that are unlikely to be in the system, that will be present only at concentrations much lower than the guideline value or that have no human health effects or effects on drinking-water acceptability. One example is that of radionuclides in drinking-water, which may be present in such minute quantities that their contribution to the overall health risks from drinking-water will be negligible. Analysis of individual radionuclides requires sophisticated and expensive procedures; hence, in such cases, measurements of gross alpha and gross beta activities may be adopted as the screening tests for the presence of radionuclides in drinking-water, as discussed in [section 9.3](#).

Water quality targets are also used in the certification process for chemicals that occur in water as a result of treatment processes or from materials in contact with water. In such applications, assumptions are made in order to derive standards for materials and chemicals that can be employed in their certification. Generally, allow-

ance must be made for the incremental increase over levels found in water sources. For some materials (e.g. domestic plumbing), assumptions must also account for the relatively high release of some substances for a short period following installation.

Escherichia coli remains an important indicator of faecal contamination for verification of water quality, but measurements of *E. coli* do not represent a risk-based water quality target. The use of *E. coli* as an indicator organism is discussed in more detail in [chapter 7](#).

Further guidance on considerations for establishing water quality targets as part of drinking-water quality standards is included in the supporting document *Developing drinking-water quality regulations and standards* ([Annex 1](#)).

3.3.3 Performance targets

Although performance targets can be applied to chemical hazards, the most common application is for control of microbial hazards in piped supplies. Performance targets assist in the selection and use of control measures that are capable of preventing pathogens from breaching the barriers of source protection, treatment and distribution systems or preventing growth within the distribution system.

Performance targets define requirements in relation to source water quality. Ideally, this should be based on system-specific data; more commonly, however, targets will be specified in relation to broad categories of source water quality and type (see [section 7.2](#)). The derivation of performance targets requires the integration of factors such as tolerable disease burden (acceptable risk), including severity of disease outcomes, and, for pathogens, quantitative microbial risk assessment (see [section 7.2](#)). There are insufficient data, and it is not realistic, to derive performance targets for all potentially waterborne pathogens. The practical approach is to derive targets for reference pathogens representing groups of pathogens (e.g. bacteria, viruses and protozoa). Selection of reference pathogens should take into account variations in susceptibility to treatment as well as local conditions, including prevalence of waterborne transmission and source water characteristics.

The most common application of performance targets is in identifying appropriate combinations of treatment processes to reduce pathogen concentrations in source water to a level that will meet health outcome targets and hence be safe. This is normally expressed in terms of log reductions. Selection of processes requires evidence that they will meet required performance targets (i.e. validation; see [section 4.2.7](#)). Examples of treatment processes and pathogen reductions are given in [section 7.3](#).

Performance targets can be applied to catchment controls that are aimed at reducing pathogen concentrations through preventive measures and to measures to prevent ingress of contamination through distribution systems. Performance targets are also important in certification of point-of-use devices and specified technologies used for drinking-water treatment. Certification of devices is discussed elsewhere (see [section 1.2.9](#)).

Performance targets can be applied to chemical hazards. In comparison with targets for microbial hazards, they are typically applied to specific chemicals, with performance measured in terms of percentage reduction (see [section 8.4](#)).

3.3.4 Specified technology targets

Specified technology targets typically take the form of recommendations concerning technologies applicable in certain circumstances (e.g. filtration and disinfection of surface water). Selection of technologies is usually based on qualitative assessments of source water type and quality (e.g. impacted surface water, protected groundwater). Specified technology targets are most frequently applied to small community supplies and to devices used at the household level. They can be applied to both microbial and chemical hazards.

Smaller municipal and community drinking-water suppliers often have limited resources and ability to develop individual system assessments and health-based targets. National regulatory agencies may therefore directly specify technology requirements or approved options. These may include, for example:

- specific and approved treatment processes in relation to source types and characteristics;
- providing guidance on requirements for protection of well heads;
- requirements for protection of drinking-water quality in distribution systems.

It is important to review specified targets on a regular basis to ensure that they are kept up to date in terms of the prevailing scientific knowledge about the technology and its application.

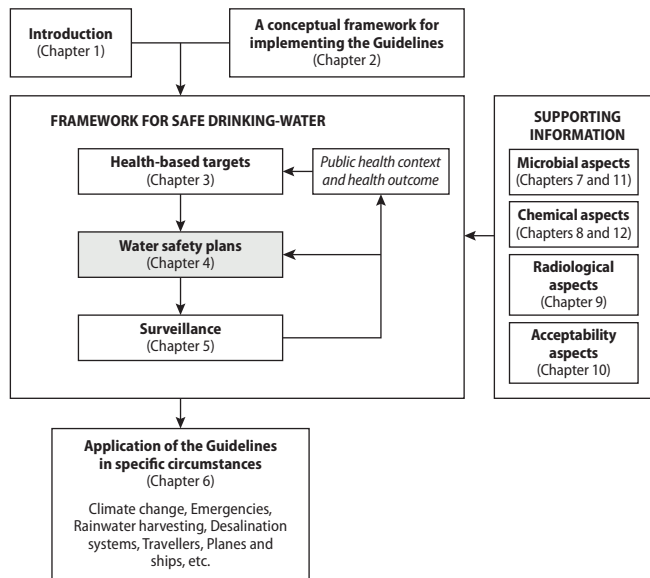
4

Water safety plans

The most effective means of consistently ensuring the safety of a drinking-water supply is through the use of a comprehensive risk assessment and risk management approach that encompasses all steps in the water supply from catchment to consumer. In these Guidelines, such approaches are termed water safety plans (WSPs). The WSP approach has been developed to organize and

systematize a long history of management practices applied to drinking-water and to ensure the applicability of these practices to the management of drinking-water quality. WSPs represent an evolution of the concept of sanitary inspections and vulnerability assessments that include and encompass the whole of the water supply system and its operation. The WSP approach draws on many of the principles and concepts from other risk management approaches, in particular the multiple-barrier approach and hazard analysis and critical control points (as used in the food industry).

This chapter focuses on the key principles of WSPs for different water supply types; it is not a comprehensive guide to their application in practice. Practical information on how to develop and implement a WSP is available in the supporting documents *Water safety plan manual* and *Water safety planning for small community water supplies* ([Annex 1](#)). A guide to equitable water safety planning ([Annex 1](#)) provides information to ensure improvements through WSPs benefit all users. *Climate resilient*



water safety plans ([Annex 1](#)) supports more resilient water supplies by helping water suppliers to better manage the risks posed by climate change to water quality and quantity (see also [section 6.1](#)).

The effective and sustained implementation of water safety planning at scale requires coordinated engagement of all stakeholders involved in the safe management of drinking-water supplies. Governments play a critical role in establishing an enabling environment through the development of policy instruments (e.g. national or subnational drinking-water policies, legislation, regulations and standards; [section 2.7](#)) that promote or support WSP adoption. These efforts should be complemented by the provision of support, such as WSP implementation guidelines and training programmes. A risk-based drinking-water quality surveillance programme should also be in place to monitor and strengthen WSP implementation, for example, through support visits and WSP auditing ([chapter 5](#)). For information on establishing a national WSP programme, refer to *Think big, start small and scale-up* ([Annex 1](#)).

WSPs vary in complexity, as appropriate for the situation. In many cases, they will be quite simple, focusing on the key hazards and hazardous events identified for the specific drinking-water supply system. The wide range of examples of control measures given in the following text does not imply that all of these are appropriate in all cases.

A WSP has five key components, which are guided by health-based targets (see [chapter 3](#)) and overseen through drinking-water quality surveillance (see [chapter 5](#)). They are as follows.

- 1) *Preparation*: establishing a team to lead the development and ongoing implementation of the WSP, and identifying potential stakeholders who can contribute to and support water safety planning.
- 2) *System assessment*: determining whether the drinking-water supply chain as a whole can consistently supply safe drinking-water and highlighting areas where improvements are needed to reduce health risks. The system assessment identifies hazards and hazardous events and assesses the related likelihood and severity, as well as existing control measures, in order to identify priority risks and determine where additional control is required. This stage can also include the assessment of design criteria of new systems.
- 3) *Monitoring*: determining whether control measures are operating as intended, and that the WSP is effective. For each control measure identified, an appropriate means of operational monitoring should be defined to ensure that any deviation from required performance is rapidly detected and rectified in a timely manner. WSP verification includes compliance monitoring to ensure that the health-based targets are being achieved, in addition to WSP auditing and determining user satisfaction.
- 4) *Management and communication*: describing actions to be taken during normal operation, incident conditions, or emergency situations. This stage also involves implementing targeted supporting programmes to facilitate effective water safety planning.
- 5) *Review and improvement*: ensuring that the WSP is up to date and reflects lessons learned from WSP operational experiences. This involves conducting regular

WSP reviews to ensure the WSP is kept current, as well as reviewing the WSP after a significant incident, near miss or emergency, and updating as needed.

The primary objectives of a WSP in ensuring good drinking-water supply practice are the prevention or minimization of contamination of source waters, the reduction or removal of contamination through treatment processes and the prevention of contamination during storage, distribution and handling of drinking-water. These objectives are equally applicable to large piped drinking-water supplies and small water supplies (including community-managed supplies); see [section 1.2.6](#).

WSPs are a powerful tool for the drinking-water supplier to manage the supply safely. Key benefits of effective WSP implementation include:

- better public health protection;
- decreased incidence of diarrhoeal disease;
- demonstration of “due diligence”;
- improved compliance;
- rationalizing and strengthening existing operational procedures, leading to gains in efficiency, improvement of performance and quicker response to incidents, and reduced operational costs;
- better targeted and justification for long-term capital investments based on risk assessment, supporting leveraging of funding;
- improved management of existing staff knowledge and identification of critical gaps in skills for staff;
- improved stakeholder communication and relationships, including supporting increased consumer confidence in the safety of the water supply.

WSPs are typically developed for individual drinking-water systems. Where a defined entity is responsible for a drinking-water supply, its responsibility should include the preparation and implementation of a WSP. In contexts where WSPs are mandated in regulations, the WSP should be reviewed and approved by the authority responsible for protection of public health (or another agency designated by the authority) to help ensure that it will deliver water of a quality consistent with the defined targets (see also [section 5.1](#)).

Many water suppliers may face practical challenges in initiating, developing and implementing a WSP. These include mistaken perceptions that the approach is inflexible; that developing a WSP always requires external expertise; that WSPs supersede, rather than build on, existing good practices; and that WSPs are necessarily complicated and are not appropriate for small water supplies.

Although WSP implementation demands a certain minimum adherence to the steps involved ([Figure 4.1](#)), it is a flexible approach that should rely on the water supplier’s existing practices and fit the way that a supplier is organized.

The WSP is a vital step in identifying the hazards, hazardous events and risks associated with the water supply. Particularly where the water supplier does not manage the catchment, treatment is critical to ensure that it is operating at its optimum at all times, as this is often the key barrier that prevents hazards from reaching drinking-water. It must be recognized that even if other hazards are identified in the

WSP STAGE	PURPOSE
Preparation <i>Section 4.1</i>	Establish a team to lead the development and ongoing implementation of the WSP.
System assessment <i>Section 4.2</i>	Conduct a comprehensive assessment of the entire water supply system to identify potential hazards, evaluate existing control measures, and prioritize risks. Implement a progressive improvement plan to ensure the consistent delivery of safe drinking-water.
Monitoring <i>Section 4.3</i>	Implement a defined operational monitoring plan that ensures the water supply's control measures are operating as intended, and verify that the WSP is working effectively and meeting health-based targets.
Management and communication <i>Section 4.4</i>	Provide documented procedures to be followed during normal or incident conditions, or emergency situations. Implement programmes to support the effective implementation of water safety planning.
Review and improvement <i>Section 4.5</i>	Ensure that the WSP is up to date and reflects operational experiences through periodic review, and review the WSP following significant incidents and emergencies.

Figure 4.1 Overview of the stages of water safety planning

Source: Adapted from WHO (2023)

catchment, remediation may take time, and this should not be a reason for delaying the start of WSP preparation and implementation. Similarly, initiating the process of ensuring that the distribution system is intact and managed appropriately is a vital step that is under the control of the water supplier.

Many of the procedures inherent in the WSP, such as documenting the system and ensuring that standard operating procedures are established for each of the treatment processes and the operation of the distribution system, are simply normal good practice in a drinking-water supply. The WSP should therefore build on and improve existing practice.

WSPs should also not be seen as a competing initiative to existing programmes already being undertaken. For example, existing programmes for operator training and competency, consumer education on water conservation, stakeholder outreach, etc. can all be integrated into the WSP framework, given their supporting contribution to overall WSP objectives.

WSPs embraces a progressive (or “stepwise”) approach to incrementally improving the safety of drinking-water supplies. For example, it will not be possible in many cases to fully implement all the identified improvements within a WSP at once, but the mapping of the system, the identification of the hazards and hazardous events, and the assessment of the risks will provide a framework for prioritizing actions and will identify the requirements for continuing improvement as resources become available.

Box 4.1 WSPs for small water supplies, including the role of sanitary inspection

Small water supplies (including those managed by communities) remain a significant challenge for many countries, partly because human, technical and financial resources are limited. Given these common challenges, small water supplies require explicit consideration in policies and regulations. This includes adopting regulatory approaches that promote water safety planning. The introduction of WSPs helps to identify simple and cost-effective steps that can be taken to protect and improve such supplies. To establish a foundation for WSPs, it is important that health authorities emphasize the importance of safe drinking-water to the local community and that regulatory approaches promote a shift towards professionalized operation and management of small water supplies. Further, the competent national or regional authority should act as a source of information and guidance on appropriate management of small water supplies. Approaches to verification in these circumstances will depend on the capacity of local authorities and communities and should be defined in national policy.

For small water supplies, basic WSPs (e.g. utilizing generic WSP “templates”, which may in some cases be developed by a statutory body or accredited third-party organization) to more comprehensive WSPs may be considered depending on local factors, including management capacity, resources and population served. In these settings, guidance on household water treatment, storage, handling and use may also be required. WSPs dealing with user practices should be linked to a hygiene education programme and advice to households in maintaining water safety.

Sanitary inspection is a valuable tool to support and a starting point for water safety planning for small water supplies. A sanitary inspection is a simple on-site evaluation, traditionally performed using a checklist, to determine if priority risk factors are present that can lead to contamination of the water supply. Risk factors identified should be managed through remedial actions and ongoing operations and management activities to keep water supplies safe.

Sanitary inspections support WSPs by aiding in hazard identification. Sanitary inspections can also support monitoring activities. When sanitary inspections are routinely carried out and combined with planning and action for improvement, ongoing maintenance and monitoring, they can also serve as a simplified risk management approach where a WSP is not feasible (e.g. for an individual household supply), or as an interim approach while WSP capacity is strengthened (e.g. for a community-managed supply).

For additional guidance on water safety planning in the context of small water supplies, refer to the *Guidelines for drinking-water quality: small water supplies* and *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)).

See *Water safety plan manual* ([Annex 1](#)) for more information on the practical application of progressive improvement principles.

Refer to [Box 4.1](#) for further information on WSP considerations for small water supplies. Information is also included on the role of sanitary inspections.

4.1 Preparation

The first stage in developing a WSP is to form a multidisciplinary team of experts with a thorough understanding of the drinking-water system involved. The team should be led by the drinking-water supplier and have sufficient expertise in drinking-water abstraction, treatment, distribution storage, and user practices. Typically, such a team would include individuals involved in each stage of the supply of drinking-water and in many cases representatives from a wider group of stakeholders with collective responsibility for the water supply system from catchment to consumer. Teams could include engineers, catchment and water managers, water quality specialists, environmental or public health professionals, operational staff and representatives of consumers or from the community. In most settings, the team will include

members from external agencies. Therefore, conducting a stakeholder mapping exercise is a valuable undertaking, which can help to determine which stakeholders should be directly involved in the process, and those that should be kept abreast of the process and its outcomes. The team should seek ad hoc expertise as needed during key stages of WSP development or review, for example, when strengthening climate change and equity considerations during WSP development or during WSP review. For small water supplies, additional external expertise is likely to be useful in addition to operational personnel.

4.2 System assessment

Effective management of the drinking-water supply requires a comprehensive understanding of the system, the range and magnitude of hazards and hazardous events that may affect the system and the ability of existing processes and infrastructure to manage actual or potential risks. It also requires an assessment of capabilities to meet targets. When a new system or an upgrade of an existing system is being planned, the first step in developing a WSP is the collection and evaluation of all available relevant information and consideration of what risks may arise during delivery of water to the consumer.

Assessment of the drinking-water system supports subsequent steps in the WSP in which effective strategies for control of hazards and hazardous events are planned and implemented.

The assessment and evaluation of a drinking-water system are enhanced through an accurate system description, including a flow diagram. The system description should provide an overview of the drinking-water system, including characterization of the source, identification of potential pollution sources in the catchment, measures for catchment protection, treatment processes, storage and mechanisms for distribution (including piped and non-piped systems), and consumer practices (e.g. storage, treatment and handling, if applicable). It is essential that the description and the flow diagram of the drinking-water system are conceptually accurate. If the description is not complete, up to date or correct, it is possible to overlook potential hazards and hazardous events that may be significant. To ensure accuracy, the system description should be validated by visually checking against features observed on the ground.

Effective risk management requires the identification of potential hazards and hazardous events and an assessment of the level of risk presented by each. In this context:

- a *hazard* is a contaminant or condition that may adversely affect the supply of safe drinking-water (e.g. microbial, chemical, radiological, acceptability or quantity-related hazards);
- a *hazardous event* is an event that results in a hazard being introduced to, or inadequately removed from, the water supply;
- *risk* is the product of the likelihood of occurrence of a hazardous event and its severity (or consequences).

Data on the occurrence of hazards in source waters and in drinking-water combined with information concerning the effectiveness of existing controls enable an assessment of whether health-based targets can be achieved with the existing infra-

structure. They also assist in identifying catchment management measures, treatment processes and distribution system operating conditions that would reasonably be expected to achieve those health-based targets if improvements are required.

It may often be more efficient to invest in preventive processes within the catchment than to invest in major treatment infrastructure to manage a hazard.

To ensure the accuracy of the assessment, including an overall estimate of risk, it is essential that all elements of the drinking-water system (catchment, treatment, distribution/storage and consumer practices) are considered and that interactions among these elements are taken into consideration.

4.2.1 New systems

When new drinking-water supply sources are being investigated or developed, it is prudent to undertake a wide range of analyses to establish overall safety and adequacy of water sources, and to determine potential sources of contamination of the drinking-water supply source. These analyses would normally include an assessment of water quality data (including undertaking additional sampling if necessary), hydrological analysis, geological assessment and land use inventories to determine potential hazards, and assess the adequacy of source water quantity (see [section 2.5](#) for more information).

When designing new systems, all water quality factors should be taken into account in selecting technologies for abstraction, treatment, distribution and storage of new sources. Variations in the turbidity and other parameters of raw surface waters can be considerable, and allowance must be made for this. Treatment plants should be designed to take account of variations known or expected to occur with significant frequency rather than for average water quality; otherwise, for example, filters may rapidly become blocked or sedimentation tanks overloaded. The chemical aggressiveness of some groundwaters may affect the integrity of borehole casings and pumps, leading to unacceptably high levels of iron in the supply, eventual breakdown and expensive repair work. In such cases, the quality, availability and acceptability of drinking-water may be reduced and public health endangered. Distribution and storage systems should be designed to ensure materials used do not contaminate the water supply.

For an overview of drinking-water systems and technologies and to inform selection, see the supporting document *Compendium of drinking-water systems and technologies from source to consumer* ([Annex 1](#)).

4.2.2 Collecting and evaluating available data

Areas that should be taken into consideration as part of the assessment of the drinking-water system include all real or potential hazards and hazardous events associated with each step in the drinking-water system that could result in contamination or interruption of supply. In most cases, consultation with public health and other sectors, including land and water users and all those who regulate activities in the catchment, will be required for the analysis of catchments. A structured approach is important to ensure that significant issues are not overlooked and that areas of greatest risk are identified.

The overall assessment of the drinking-water system should take into consideration any historical water quality data that may assist in understanding source water characteristics and drinking-water system performance both over time and following specific events (e.g. heavy rainfall). For examples of information to consider in assessing components of the drinking-water system, see the supporting document *Water safety plan manual*, in particular Modules 2 and 3 ([Annex 1](#)).

Prioritizing hazards for control

Once potential hazards and their sources have been identified, the risk associated with each hazard and hazardous event should be compared so that priorities for risk management can be established and documented. Although there are numerous hazards and hazardous events that can compromise the drinking-water supply, not all of them will require the same degree of attention.

The risk associated with each hazard and hazardous event may be described by identifying the likelihood of occurrence (e.g. certain, possible, rare) and evaluating the severity of consequences if the hazard occurred (e.g. insignificant, major, catastrophic). The aim should be to distinguish between important and less important hazards and hazardous events. The approach used typically involves a semiquantitative matrix.

Simple scoring matrices often apply technical information from guidelines, scientific literature and industry practice with well-informed “expert” judgement based on knowledge and experience of WSP team members, supported by peer review or benchmarking. Scoring should be specific for each drinking-water system, as each system is unique. Where WSP templates are developed for technologies used by small drinking-water systems, the scoring will be specific to the technology rather than the individual drinking-water system.

By using risk ranking, control measures can be prioritized in relation to their significance. A variety of semiquantitative and qualitative approaches to ranking risk can be applied, and Module 4 of the supporting document *Water safety plan manual* ([Annex 1](#)) provides further guidance. An example of a semiquantitative approach is given in [Table 4.1](#). Application of this matrix relies to a significant extent on expert opinion to make judgements on the public health risk posed by hazards or hazardous events.

Table 4.1 Example of a simple scoring matrix for ranking risks

Likelihood	Severity of consequences				
	Insignificant	Minor	Moderate	Major	Catastrophic
Almost certain	5	10	15	20	25
Very likely	4	8	12	16	20
Likely	3	6	9	12	15
Unlikely	2	4	6	8	10
Very unlikely	1	2	3	4	5

Risk score	< 6	6–9	10–15	> 15
Risk level	Low	Medium	High	Very high

An example of descriptors that can be used to rate the likelihood of occurrence and severity of consequences is given in [Table 4.2](#). A “cut-off” point should be determined, above which all risks will require immediate attention. There is little value in expending large amounts of effort to consider very low risks.

Table 4.2 Examples of definitions of likelihood and severity categories that can be used in risk scoring

Item	Rating	Definition
<i>Likelihood categories</i>		
Almost certain	5	Has occurred in the past and is expected to happen again
Very likely	4	Has occurred in the past and has the potential to happen again
Likely	3	Is possible and under certain circumstances could happen
Unlikely	2	Is possible and cannot be ruled out completely
Very unlikely	1	Has not occurred in the past, and it is highly improbable that it will happen in the future
<i>Severity categories</i>		
Catastrophic	5	Potential illness or death
Major	4	Potential long-term health effects
Moderate	3	Long-term or widespread non-compliance, quantity or acceptability issue (not health related)
Minor	2	Short-term or localized non-compliance, quantity or acceptability issue (not health related)
Insignificant	1	Negligible impact on water quality, acceptability or quantity

Control measures

The assessment and planning of control measures should ensure that health-based targets will be met and should be based on hazard identification and risk assessment. The level of control applied to a hazard and hazardous event should be proportional to the associated risk ranking. Assessment of control measures involves:

- identifying existing control measures for each significant hazard or hazardous event from catchment to consumer;
- validating whether the control measures are effective in reducing risk to acceptable levels;
- if improvement is required, evaluating strengthened or additional control measures that could be applied.

Identification and implementation of control measures should be based on the multiple-barrier principle. The strength of this approach is that a failure of one barrier may be compensated by effective operation of the remaining barriers, thus

minimizing the likelihood of contaminants passing through the entire system and being present in sufficient amounts to cause harm to consumers. Many control measures may contribute to control more than one hazard, whereas some hazards may require more than one control measure for effective control. Examples of control measures are provided in the following sections.

A control measure is an activity or process to prevent, eliminate or reduce the risk of a hazardous event to an acceptable level. These measures are applied collectively to ensure that drinking-water consistently meets health-based targets.

All control measures are important and should be afforded ongoing attention. They should be subject to operational monitoring and control, with the means of monitoring and frequency of data collection based on the nature of the control measure and the rapidity with which change may occur (see [section 4.3](#)).

4.2.3 Catchment protection

Effective catchment management has many benefits. By decreasing the contamination of the source water, the amount of treatment required can be reduced. This may reduce the production of treatment by-products and minimize operational costs, and also reduce associated carbon emissions.

Hazard identification

Understanding the reasons for variations in raw water quality and quantity is important, as it will influence the requirements for treatment, treatment efficiency and the resulting health risk associated with the finished drinking-water, as well as establishing the needs for alternative water sources to ensure adequate supply into the future.

In general, raw water quality is influenced by both natural and human use factors. Important natural factors include wildlife, climate, topography, geology, hydrology and vegetation. Human use factors include pollution from point sources (e.g. wastewater discharges) and non-point sources (e.g. surface runoff). For example, discharges of municipal wastewater can be a major source of pathogens; urban runoff and livestock can contribute substantial microbial load; body contact recreation can be a source of faecal contamination; and agricultural runoff, including agrochemicals and manure, can lead to increased challenges to treatment.

Similarly, water quantity is influenced by natural fluctuations in weather patterns, longer-term climate change and human use factors (e.g. extraction for irrigation or industrial uses). Reduced water quantity can also influence water quality, for example, saline intrusion in coastal aquifers experiencing overextraction or low water levels in reservoirs mobilizing contaminants present in sediments.

Whether water is drawn from surface or underground sources, it is important that the characteristics of the local catchment or aquifer are understood and that the scenarios that could lead to water pollution or water scarcity are identified and managed. The extent to which potentially polluting activities in the catchment can be reduced may appear to be limited by competition for water and pressure for increased development in the catchment. However, introducing good practices in land use and in containment of hazards is often possible without substantially restricting activities,

and collaboration between stakeholders may be a powerful tool to reduce pollution without reducing beneficial development.

Catchment protection provides the first barrier in protection of drinking-water quality, and can help to ensure adequate water quantity, which could be affected by activities that include illegal dam construction or diversions, or overextraction. Where catchment management is beyond the jurisdiction of the drinking-water supplier, the planning and implementation of control measures will require coordination with other agencies. These may include planning authorities, sanitation authorities, catchment boards, environmental and water resource regulators, road authorities, emergency services and agricultural, industrial and other commercial entities whose activities have an impact on water quality and quantity. It may not be possible to apply all aspects of catchment protection initially; nevertheless, priority should be given to catchment management. This will contribute to a sense of ownership and joint responsibility for drinking-water resources through multistakeholder bodies that assess pollution risks and develop plans for improving management practices for reducing these risks.

Groundwater from deep and confined aquifers is usually microbially safe and chemically stable in the absence of direct contamination; however, shallow or unconfined aquifers can be subject to contamination from discharges or seepages associated with agricultural practices (e.g. pathogens, nitrates and pesticides), on-site sanitation and sewerage (e.g. pathogens and nitrates) and industrial wastes. For guidance and examples of hazards and hazardous situations that should be taken into consideration as part of a hazard analysis and risk assessment, see the supporting document *Water safety plan manual*, in particular Module 3, and the supporting documents *Protecting groundwater for health* and *Protecting surface water for health* ([Annex 1](#)). Further information on the impacts of climate change on water sources is presented in *Climate resilient water safety plans* ([Annex 1](#)) and [section 6.1](#).

Control measures

Control measures in the catchment include those that are under the direct responsibility of the water supplier, as well as measures not under their direct control such as regulations to protect water resources (land use planning and watershed management) from potentially polluting activities. This could be supported by promoting awareness in the community of the impact of human activity on water quality.

Where a number of water sources are available, there may be flexibility in the selection of water for treatment and supply. It may be possible to avoid taking water from rivers and streams when water quality is poor (e.g. following heavy rainfall) in order to reduce risk and prevent potential problems in subsequent treatment processes.

Retention of water in reservoirs can reduce the number of faecal microorganisms through settling and inactivation, including solar (ultraviolet; UV) disinfection, but also provides opportunities for the introduction of contamination. Most pathogenic microorganisms of faecal origin (enteric pathogens) do not survive indefinitely in the environment. Substantial die-off of enteric bacteria will occur over a period of weeks. Enteric viruses and protozoa will often survive for longer periods (weeks to months) but are often removed by settling and antagonism from indigenous microbes. Retention also allows suspended material to settle, which makes subsequent disinfection more effective and reduces the formation of disinfection by-products

(DBPs). However, reservoir sediments can be nutrient rich, and when sediments are mobilized (e.g. from changing flow patterns, redox conditions or reservoir levels), this may promote cyanobacterial growth, which, under certain conditions, may lead to the release of chemical compounds that affect the acceptability of the water and/or harmful toxins. Historic sediments in polluted catchments may also contain chemicals that can be mobilized during drought or low-oxygen conditions, posing additional water quality risks. In these instances, appropriate source protection, selective abstraction and downstream treatment control measures should be considered, among other controls.

Control measures for groundwater sources should include protecting the aquifer and the local area around the abstraction point from contamination and ensuring the physical integrity of the spring source, well or bore (surface sealed, casing intact, etc.); further information can be found in the supporting document *Protecting groundwater for health* ([Annex 1](#)).

For examples of control measures to consider for surface water and catchments, see the supporting document *Protecting surface water for health* ([Annex 1](#)). Further information on the use of indicator organisms in catchment characterization is also available in chapter 4 of the supporting document *Assessing microbial safety of drinking water* ([Annex 1](#)).

4.2.4 Treatment

After source water protection, the next barriers to contamination of the drinking-water system are those of water treatment processes, including disinfection and physical removal of hazards.

Hazard identification

Hazards may be introduced during treatment, or hazardous events may allow contaminants to pass through treatment in significant concentrations. For example, constituents of drinking-water can be introduced through the treatment process, including chemical additives used in the treatment process or products in contact with drinking-water. Sporadic high turbidity in source water can overwhelm treatment processes, allowing enteric pathogens into treated water and the distribution system. Similarly, suboptimal filtration following filter backwashing can lead to the introduction of pathogens into the distribution system.

For guidance on hazards and hazardous events that can have an impact on the performance of drinking-water treatment, see the supporting document *Water safety plan manual*, in particular Module 3 ([Annex 1](#)).

Control measures

Control measures may include pretreatment, coagulation, flocculation, sedimentation, filtration and disinfection.

Pretreatment includes processes such as roughing filters, microstrainers, off-stream storage and bankside filtration. Pretreatment options may be compatible with a variety of treatment processes ranging in complexity from simple disinfection to membrane processes. Pretreatment can reduce or stabilize the microbial, natural organic matter and particulate load.

Coagulation, flocculation, sedimentation (or flotation) and filtration remove particles, including microorganisms (bacteria, viruses and protozoa). It is important that processes are optimized and controlled to achieve consistent and reliable performance. Chemical coagulation is the most important step in determining the removal efficiency of coagulation, flocculation and clarification processes. It also directly affects the removal efficiency of granular media filtration units and has indirect impacts on the efficiency of the disinfection process. While it is unlikely that the coagulation process itself introduces any new microbial hazards to finished water, a failure or inefficiency in the coagulation process could result in an increased microbial load entering drinking-water distribution.

Various filtration processes are used in drinking-water treatment, including granular, slow sand, precoat and membrane (microfiltration, ultrafiltration, nanofiltration and reverse osmosis) filtration. With proper design and operation, filtration can act as a consistent and effective barrier for pathogenic microorganisms and may in some cases be the only treatment barrier (e.g. for removing *Cryptosporidium* oocysts by direct filtration when chlorine is used as the sole disinfectant).

Application of an adequate concentration of disinfectant is an essential element for most treatment systems to achieve the necessary level of microbial risk reduction. Taking account of the level of microbial inactivation required for the more resistant microbial pathogens through the application of the *Ct* concept (product of disinfectant concentration and contact time) for a particular pH and temperature ensures that other, more sensitive microbes are also effectively controlled. Where disinfection is used, measures to minimize DBP formation should be taken into consideration.

The most commonly used disinfection process is chlorination. Ozonation, UV irradiation, chloramination and application of chlorine dioxide are also used. These methods are generally very effective in killing bacteria and viruses, although adenovirus is less sensitive to UV light. Protozoa are the least sensitive to chemical disinfectants. For example, *Cryptosporidium* is highly resistant to chlorine; however, *Cryptosporidium* is sensitive to UV light. For effective removal or inactivation of protozoal cysts and oocysts, filtration with the aid of coagulation and flocculation (to reduce particles and turbidity) followed by disinfection (by one or a combination of disinfectants) is the most practical method.

Storage of water after disinfection and before supply to consumers can improve disinfection by increasing disinfectant contact times. This can be particularly important for more resistant microorganisms, such as *Giardia*.

Further information can also be found in [chapter 11](#) for individual pathogens and the supporting document *Water treatment and pathogen control* ([Annex 1](#)).

4.2.5 Piped distribution systems

Water treatment should be optimized to prevent microbial growth, corrosion of pipe materials and the formation of deposits.

Maintaining good water quality in the distribution system will depend on the design and operation of the system and on maintenance and survey procedures to prevent contamination and to prevent and remove the accumulation of internal deposits.

Hazard identification

The protection of the distribution system is essential for providing safe drinking-water. Because of the nature of the distribution system, which may include many kilometres of pipe, storage tanks, interconnections with industrial users and the potential for tampering and vandalism, opportunities for microbial and chemical contamination exist. In addition to contamination concerns, water quantity hazards may arise from leaks or bursts, which could occur from, for example, physical damage, ageing infrastructure or illegal connections. For guidance on hazards and hazardous events in piped distribution systems, see the supporting document *Water safety plan manual*, in particular Module 3 ([Annex 1](#)). Further information can also be found in the supporting document *Water safety in distribution systems* ([Annex 1](#)).

When contamination by enteric pathogens or hazardous chemicals occurs within the distribution system, it is likely that consumers will be exposed to the pathogens or chemicals. In the case of pathogen ingress, even where disinfectant residuals are employed to limit microbial occurrence, they may be inadequate to overcome the contamination or may be ineffective against some or all of the pathogen types introduced. As a result, pathogens may occur in concentrations that could lead to infection and illness.

Where water is supplied intermittently, the resulting low water pressure will allow the ingress of contaminated water into the system through breaks, cracks, joints and pinholes. Intermittent supplies are not desirable but are very common in many countries and are frequently associated with contamination. The control of water quality in intermittent supplies represents a significant challenge, as the risks of infiltration and backflow increase considerably. The risks may be elevated seasonally as soil moisture conditions increase the likelihood of a pressure gradient developing from the soil to the pipe. Where hazards enter the pipes in an intermittent supply, the charging of the system when supply is restored may increase risks to consumers, as a concentrated “slug” of contaminated water can be expected to flow through the system. Where household storage is used to overcome intermittent supply, localized use of disinfectants to reduce microbial proliferation may be warranted.

Drinking-water entering the distribution system may contain free-living amoebae and environmental strains of various heterotrophic bacterial and fungal species. Under favourable conditions, amoebae and heterotrophs, including strains of *Klebsiella* and *Pseudomonas* may colonize distribution systems and form biofilms. There is limited to no evidence to implicate the occurrence of most microorganisms from biofilms (an exception is *Legionella*, which can colonize water systems in buildings) with adverse health effects in the general population through drinking-water (see [section 7.1](#), including [Tables 7.1](#) and [7.2](#) for more information).

Water temperatures and nutrient concentrations are not generally elevated enough within the distribution system to support the growth of *E. coli* (or enteric pathogenic bacteria) in biofilms. Thus, the presence of *E. coli* should be considered as evidence of recent faecal contamination.

Natural disasters, including flood, drought and earth tremors, may significantly affect piped water distribution systems.

Control measures

Water entering the distribution system must be safe and ideally should also be biologically stable. The distribution system itself must provide a secure barrier to contamination as the water is transported to the user.

Water distribution systems should be fully enclosed, and storage reservoirs and tanks should be securely roofed with external drainage to prevent contamination. A number of strategies can be adopted to maintain the quality of water within the distribution system, including use of backflow prevention devices, maintaining positive pressure throughout the system and implementation of effective maintenance procedures such as flushing programmes.

Other control measures to protect against external contamination and microbial growth include maintaining disinfectant residuals and avoiding low flows and stagnation in storage tanks, loops and dead-end sections. Pipe replacement or relining programmes should be implemented to remove degraded and corroded pipework. It is also important that appropriate security measures be put in place to prevent unauthorized access to or interference with the drinking-water system infrastructure. For additional guidance on distribution system control measures, see Module 4 in the supporting document *Water safety plan manual* ([Annex 1](#)). Further information is also available in the supporting documents *Water safety in distribution systems* and *Safe piped water* ([Annex 1](#)).

4.2.6 Small water supplies (including community-managed supplies)

Hazard identification

For small water supplies, including those managed by communities, identification of hazards and hazardous events would ideally be performed on a case-by-case basis. In some settings, however, it may be necessary to make general assumptions of hazards and hazardous events that are relevant for technologies or system types and that may be defined at a national or regional level. In this context, sanitary inspection forms, which include priority risk factors that can contaminate a water supply, can be useful (see [Box 4.1](#) for more information).

For additional examples of hazards and hazardous events potentially associated with various small water supplies, see *Water safety planning for small community water supplies* and *A field guide to improving small drinking-water supplies: water safety planning for rural communities*.

Control measures

The control measures put in place ideally depend on the characteristics of the individual drinking-water supply, including source water type and associated catchment. In some settings however, particularly where general assumptions of hazards and hazardous events are made, standard measures may be applied for each technology or system type.

In many cases, contamination of groundwater supplies can be controlled by a combination of simple measures. In the absence of fractures or fissures, which may allow rapid transport of hazards to the source, groundwater in confined or deep aquifers will generally be free of pathogenic microorganisms. Boreholes and wells should

have an impermeable apron, and should be sealed or encased to a reasonable depth to stop the ingress of surface water.

Rainwater harvesting systems, particularly those involving storage in above-ground tanks, can be a relatively safe supply of water (see [section 6.2](#)). The principal sources of contamination are birds, small mammals and debris collected on roofs. The impact of these sources can be minimized by simple measures: guttering should be cleared regularly, overhanging branches should be kept to a minimum (because they can be a source of debris and can increase access to roof catchment areas by birds and small mammals) and inlet pipes to tanks should include a filter box (e.g. to remove leaf litter). First-flush diverters, which prevent the first flush of rainwater (e.g. 20–25 litres) from entering tanks, are recommended, as this water is typically of poorer quality due to the build-up of contaminants on the roof between rain events. If first-flush diverters are not available, a detachable downpipe can be used manually to provide the same result.

In general, surface waters and shallow groundwater under the direct influence of surface water will usually require treatment (e.g. filtration and disinfection) to ensure microbial safety. The first barrier is based on minimizing contamination from human waste, livestock and other hazards at the source.

The greater the protection of the water source, the less the reliance on treatment or disinfection.

Water should be protected during storage and delivery to consumers by ensuring that the distribution and storage systems are enclosed. This applies to all small water supplies, including vendor-supplied water (see [section 6.3](#)). Where household water, collection, transport, treatment and storage is applied, respective consideration also needs to be given (see the supporting document *Managing water in the home*; [Annex 1](#) and [section 7.3.2](#)). This guidance should be used in conjunction with hygiene education programmes to provide households with skills to manage their water hygiene.

For control of chemical hazards, reliance may be placed primarily on initial screening of sources and on ensuring the quality and performance of treatment chemicals, materials and devices available for this use, including water storage systems.

Refer to *Water safety planning for small community water supplies, A field guide to improving small drinking-water supplies: water safety planning for rural communities* and *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)) for examples of control measures for small water supplies.

4.2.7 Validation of control measures

For the WSP to be relied on for anticipating and managing the hazards and hazardous events for which it was set in place, it should be supported by evidence. Validation is concerned with obtaining evidence on the performance of control measures. Depending on the type of control, validation can be done by site inspection, using existing data (e.g. water supplier historical records, from industry bodies or manufacturers) and scientific literature, technical standards, benchmarking with other authorities or targeted monitoring programmes to demonstrate performance under normal and exceptional

circumstances. In some contexts (e.g. lower resource or capacity settings), the WSP team may adopt a judgement-based control measure validation approach initially. As capacities, experience and resources are gained, the WSP team should move towards a more rigorous data-based approach as appropriate.

Validation of treatment processes is required to show that the treatment processes can operate as required and achieve required levels of hazard reduction. In the case of microbial hazards, these required levels commonly take the form of performance targets based on the use of reference pathogens (see [section 7.2](#)). Validation can be undertaken during pilot stage

Validation is an investigative activity to obtain evidence that the control measure can effectively control the corresponding hazardous event. It is typically an intensive activity when a system is initially constructed or rehabilitated. It provides information on reliably achievable water quality in preference to assumed values and also to define the operational criteria required to ensure that the control measure contributes to effective control of hazards.

studies or during initial implementation of a new or modified water treatment system. It is also a useful tool in the optimization of existing treatment processes.

The first stage of validation is to consider data and information that already exist. This stage will inform the testing requirements. It is important that data used in validation are relevant for system-specific conditions, as variations in water composition and quality, for example, may have a large impact on the efficacy of control measures.

Validation is not used for day-to-day management of drinking-water supplies; as a result, microbial parameters that may be inappropriate for operational monitoring can be used, and the lag time for return of results and additional costs from pathogen measurements can often be tolerated. Parameters should be chosen to reflect the microorganisms being targeted by treatment (see [section 7.2](#)). Increasingly, indicator parameters are being used in validation. For example, coliphage can be used to assess the effectiveness of virus removal by filtration processes or to measure the effectiveness of disinfection processes, whereas *Clostridium perfringens* can be used to measure the effectiveness of the removal of protozoa by filtration processes.

Validation should not be confused with routine operational monitoring, which is designed to show that validated control measures continue to work effectively (see [section 4.2](#)). The validation process often leads to improvements in operating performance through the identification of the most effective and robust operating modes. Additional benefits of the validation process may include identification of more suitable operational monitoring parameters for unit performance. Validation studies can also enhance the resilience of the water supplies by assessing treatment performance across a range of future scenarios, such as coagulation and flocculation effectiveness during varying cyanobacterial bloom intensities or raw water turbidity shifts following precipitation events.

4.2.8 Upgrade and improvement

The assessment of the drinking-water system may indicate that existing practices and control measures may not ensure drinking-water safety. In some instances, all that may be needed is to review, document and formalize these practices and address any areas

where improvements are required; in others, major management or infrastructure changes may be needed. The assessment of the system should be used as a basis to develop a plan to address identified needs for full implementation of a WSP.

Improvement of the drinking-water system may encompass a wide range of issues, such as:

- capital works;
- training;
- enhanced operational procedures;
- community consultation programmes;
- research and development;
- developing incident protocols;
- communication and reporting.

Improvement plans can include shorter-term (e.g. 1 year) or longer-term programmes. Shorter-term improvements might include, for example, improvements to community consultation and the development of community awareness programmes. Longer-term capital works projects could include covering of water storages or enhanced coagulation and filtration.

Implementation of improvement plans may have significant budgetary implications and therefore may require detailed analysis and careful prioritization in accord with the outcomes of risk assessment. Implementation of plans should be monitored to confirm that improvements have been made and are effective. Control measures often require considerable expenditure, and decisions about water quality improvements cannot be made in isolation from other aspects of drinking-water supply that compete for limited financial resources. Priorities will need to be established, and improvements may need to be phased in over a period of time.

4.3 Monitoring

4.3.1 *Operational monitoring and maintaining control*

Operational monitoring is a planned and routine set of activities used to determine that control measures continue to work effectively, and helps to ensure that timely corrective action is taken when predefined limits are not met.

Selecting operational monitoring parameters

Operational monitoring can include measurement of parameters or observational activities. The parameters selected for operational monitoring should reflect the effectiveness of each control measure, provide a timely indication of performance, be readily measured and provide the opportunity for an appropriate response. Examples include measurable variables, such as free chlorine residuals, pH and turbidity, or observable factors, such as the integrity of vermin-proof screens.

Operational monitoring assesses the performance of control measures at appropriate time intervals. The intervals may vary widely—for example, from online control of residual chlorine to quarterly checking of the integrity of the plinth surrounding a well.

Enteric pathogens or indicator organisms are often of limited use for operational monitoring, because the time taken to process and analyse water samples does not allow operational adjustments to be made prior to supply.

A range of parameters can be used in operational monitoring (see also the supporting documents *Water safety in distribution systems*, *Safe piped water* and *Water quality and health – review of turbidity*; [Annex 1](#)).

- For source waters, these include turbidity, UV absorbency, cyanobacterial growth, flow and retention time, colour, conductivity, and integrity of protective (e.g. fences) or abstraction infrastructures (e.g. well seals). Particular attention should be given during severe weather events including floods (see the supporting documents *Protecting groundwater for health* and *Protecting surface water for health*; [Annex 1](#)).
- For treatment, parameters may include disinfectant concentration and contact time, UV intensity, pH, light absorbency, membrane integrity, turbidity and colour (see the supporting document *Water treatment and pathogen control*; [Annex 1](#)).
- In piped distribution systems, operational monitoring parameters may include the following:
 - *Free chlorine residual monitoring* provides a rapid indication of problems that will direct measurement of microbial parameters. A sudden disappearance of an otherwise stable residual can indicate ingress of contamination. Alternatively, difficulties in maintaining residuals at points in a distribution system or a gradual disappearance of residual may indicate that the water or pipework has a high oxidant demand due to growth of bacteria.
 - *Oxidation–reduction potential* (or redox potential) measurement can also be used in the operational monitoring of disinfection efficacy. It is possible to define a minimum level of oxidation–reduction potential necessary to ensure effective disinfection. This value has to be determined on a case-by-case basis; universal values cannot be recommended. Further research and evaluation of oxidation–reduction potential as an operational monitoring technique are highly desirable.
 - *Heterotrophic bacteria* present in a supply can be a useful indicator of changes, such as increased microbial growth potential, increased biofilm activity, extended retention times or stagnation and a breakdown of integrity of the system. The numbers of heterotrophic bacteria present in a supply may reflect the presence of large contact surfaces within the treatment system, such as in-line filters, and may not be a direct indicator of the condition within the distribution system (see the supporting document *Heterotrophic plate counts and drinking-water safety*; [Annex 1](#)).
 - *Pressure measurement and turbidity* are also useful operational monitoring parameters in piped distribution systems.

For guidance on the management of distribution systems, including operation and maintenance), and the development of monitoring programmes, see the supporting documents *Water safety in distribution systems* and *Safe piped water* ([Annex 1](#)).

Many of the parameters noted above are applicable to small water supplies. Minimally, turbidity, and if chlorinated, pH and free chlorine residual should be monitored

Table 4.3 Examples of operational monitoring parameters that can be used to monitor control measures

Operational parameter	Raw water	Coagulation	Sedimentation	Filtration	Disinfection	Distribution system
pH		✓	✓		✓	✓
Turbidity (or particle count)	✓	✓	✓	✓	✓	✓
Dissolved oxygen	✓					
Stream/river flow	✓					
Rainfall	✓					
Colour	✓					
Conductivity (total dissolved solids)	✓					
Organic carbon	✓		✓			
Chemical dosage		✓			✓	
Flow rate		✓	✓	✓	✓	
Net charge		✓				
Streaming current value		✓				
Headloss				✓		
Ct (disinfectant concentration × contact time)					✓	
Disinfectant residual					✓	✓
Oxidation–reduction potential					✓	
DBPs					✓	✓
Heterotrophic bacteria					✓	✓
Hydraulic pressure						✓

and can be measured with little training and low costs. Regular observations should also be conducted, for example, of the integrity of physical infrastructure such as fences/barriers and storage tank covers. These should be complemented by regular physical assessments of the water, especially after heavy rains, to monitor whether any obvious changes in water quality occur (e.g. changes in colour, odour, taste or turbidity). For further information, refer to *Water safety planning for small community water supplies* and *A field guide to improving small drinking-water supplies: water safety planning for rural communities* ([Annex 1](#)). The *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)), which includes basic guidance for conducting operations and maintenance, can also support operational monitoring for a range of small water supply scenarios.

Examples of operational monitoring parameters are provided in [Table 4.3](#).

Establishing operational and critical limits

Control measures need to have defined limits for operational acceptability—termed operational limits—that can be applied to operational monitoring parameters. Operational limits should be defined for parameters applying to each control measure. If monitoring shows that an operational limit has been exceeded, then predetermined corrective actions (see [section 4.4](#)) need to be applied. The detection of the deviation and implementation of corrective action should be possible in a time frame adequate to maintain performance and water safety.

For some control measures, a second series of “critical limits” may also be defined, outside of which confidence in water safety would be lost. Deviations from critical limits will usually require urgent action, including immediate notification of the appropriate health authority.

Operational and critical limits can be upper limits, lower limits, a range or an “envelope” of performance measures.

4.3.2 Verification

Verification provides a check on the overall performance of the drinking-water supply chain and the safety of drinking-water being supplied to consumers.

For microbial verification, testing is typically for faecal indicator bacteria in treated water and water in distribution. For verification of chemical safety, testing for chemicals of concern may be at the end of treatment, in distribution or at the point of consumption (depending on whether the concentrations are likely to change in distribution). Trihalomethanes and haloacetic acids are the most common DBPs and occur at among the highest concentrations in drinking-water. Under many circumstances, they can serve as a suitable measure that will reflect the concentration of a wide range of related chlorinated DBPs.

In addition to operational monitoring of the performance of the individual components of a drinking-water system, it is necessary to undertake *verification* for reassurance that the system as a whole is operating safely. Verification may be undertaken by the supplier, by an independent authority or by a combination of these, depending on the administrative regime in a given country. It typically includes testing for faecal indicator organisms and hazardous chemicals, auditing that WSPs are being implemented as intended and are working effectively, and monitoring user satisfaction. In the case of small water supplies, sanitary inspections can be considered as an alternative to auditing where WSPs are not in place, or between WSP audit events.

Frequencies of sampling should reflect the need to balance the benefits and costs of obtaining more information. Sampling frequencies are usually based on the population served or on the volume of water supplied, to reflect the increased population risk. Frequency of testing for individual characteristics will also depend on variability. Sampling and analysis are required most frequently for microbial and less often for chemical constituents. This is because even brief episodes of microbial contamination can lead directly to illness in consumers, whereas episodes of chemical contamination that would constitute an acute health concern, in the absence of a specific event (e.g. chemical overdosing at a treatment plant), are rare. Sampling frequencies for water leaving treatment depend on the quality of the water source and the type of treatment.

Plans should be developed to respond to results that do not meet water quality targets. These should include investigation of the cause of non-compliance and, where necessary, corrective action, such as boil water advisories. Repeated failure to meet targets should lead to review of the WSP and development of improvement plans.

Verification also includes WSP auditing to ensure that WSPs are in place, consistently applied, and achieving their water safety objectives (see below). User satisfaction monitoring also supports verification by providing a mechanism to capture users' experiences and perceptions on drinking-water safety, helping to identify service gaps and build trust in safe drinking-water management. More information on this aspect of verification can be found in *Water safety plan manual* ([Annex 1](#)).

Microbial water quality

Verification of the microbial quality of drinking-water typically includes testing for *Escherichia coli* as an indicator of faecal pollution. In practice, testing for thermotolerant coliform bacteria can be an acceptable alternative in many circumstances. Although *E. coli* is useful, it has limitations. Enteric viruses and protozoa are more resistant to disinfection; consequently, the absence of *E. coli* will not necessarily indicate freedom from these organisms. Under certain circumstances, the inclusion of more resistant indicators, such as bacteriophages and/or bacterial spores, should be considered (see [section 7.4](#)).

Verification of the microbial quality of water in supply must be designed to ensure the best possible chance of detecting contamination. Sampling should therefore account for potential variations of water quality in distribution. This will normally mean taking account of locations and of times of increased likelihood of contamination.

Faecal contamination will not be distributed evenly throughout a piped distribution system. In systems where water quality is good, this significantly reduces the probability of detecting faecal indicator bacteria in the relatively few samples collected.

The chances of detecting contamination in systems reporting predominantly negative results for faecal indicator bacteria can be increased by using more frequent presence/absence testing. Presence/absence testing can be simpler, faster and less expensive than quantitative methods. Comparative studies of the presence/absence and quantitative methods demonstrate that the presence/absence methods can maximize the detection of faecal indicator bacteria. However, presence/absence testing is appropriate only in a system where the majority of tests for indicator organisms provide negative results.

The more frequently the water is examined for faecal indicator organisms, the more likely it is that contamination will be detected. Frequent examination by a simple method is more valuable than less frequent examination by a complex test or series of tests.

The nature and likelihood of contamination can vary seasonally, with rainfall and with other local conditions. Sampling should normally be random but should be increased at times of epidemics, flooding or emergency operations or following interruptions of supply or repair work.

Recommended minimum sample numbers for verification of the microbial quality of drinking-water are shown in [Table 4.4](#).

Table 4.4 Recommended minimum frequency of faecal indicator testing in piped distribution systems based on population size^{a, b}

Population	Total number of samples
< 5000	Monthly sampling
5000–100 000	Monthly sampling per 5000 population
> 100 000–500 000	Monthly sampling per 10 000 population plus an additional 12 samples per month
> 500 000	Monthly sampling per 50 000 population plus an additional 50 samples per month

^a Parameters such as free chlorine residual, turbidity and pH should be tested more frequently as part of operational and verification monitoring.

^b For more guidance on small water supplies and non-piped systems, refer to *Guidelines for drinking-water quality: small water supplies* (WHO, 2024). See also [section 5.2](#).

Chemical water quality

Issues that need to be addressed in developing chemical verification include the availability of appropriate analytical facilities, the cost of analyses, the possible deterioration of samples, the stability of the contaminant, the likely occurrence of the contaminant in various supplies, the most suitable point for monitoring and the frequency of sampling.

For a given chemical, the location and frequency of sampling will be determined by its principal sources (see [chapter 8](#)) and variability in its concentration. Substances that do not change significantly in concentration over time require less frequent sampling than those that might vary significantly.

In many cases, water quality analysis can be once per year, or even less, particularly for drinking-water sourced from stable groundwaters, where the concentrations of naturally occurring substances of concern will vary very slowly over time. Concentrations of naturally occurring substances are likely to be more variable in surface waters, and therefore drinking-water sourced from surface water may require a greater number of samples, depending on the contaminant and its importance.

Sampling locations will depend on the water quality characteristic being examined. Sampling at the treatment plant or at the head of the distribution system may be sufficient for constituents whose concentrations do not change during delivery. However, for those constituents whose concentrations can change during distribution, sampling should be undertaken following consideration of the behaviour or source of the specific substance. Samples should include points near the extremities of the distribution system and taps connected directly to the mains in houses and large multioccupancy buildings. Lead, for example, should be sampled at consumers' taps, as the source of lead is usually service connections or plumbing in buildings.

For further information, see the supporting documents *Chemical safety of drinking-water*, *Developing drinking-water quality regulations and standards* and *Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)).

Source waters

Verification testing of source waters is particularly important where there is no water treatment. It will also be useful following failure of the treatment process or as part of an investigation of a waterborne disease outbreak. The frequency of testing will depend on the reason for carrying out the sampling. Testing frequency may be:

- on a regular basis (the frequency of verification testing will depend on several factors, including the size of the population supplied, seasonal variation, parameter stability, reliability of the quality of the drinking-water, degree of treatment and the outcomes of the catchment risk assessment (see [section 4.2.3](#)));
- on an occasional basis (e.g. random or during visits to community-managed drinking-water supplies);
- increased following degradation of source water quality resulting from predictable incidents, emergencies or unplanned events considered likely to increase the potential for a breakthrough in contamination (e.g. following a flood, drought, wildfire or upstream spills).

Prior to commissioning a new drinking-water supply, a wider range of analyses should be carried out, including parameters identified as potentially being present from a review of data from similar supplies or from a risk assessment of the source.

Piped distribution systems

The choice of sampling points will be dependent on the individual water supply. The nature of the public health risk posed by pathogens and the contamination potential throughout distribution systems mean that collection of samples for microbial analysis (and associated parameters, such as free chlorine residual, pH and turbidity) will typically be done frequently and from dispersed sampling sites. Careful consideration of sampling points and frequency is required for chemical constituents that arise from piping and plumbing materials and that are not controlled through their direct regulation and for constituents whose concentrations change in distribution, such as trihalomethanes. The use of stratified random sampling in distribution systems has proven to be effective.

Small water supplies (including community-managed supplies)

With small water supplies frequently having more limited capacity, skills and resources than larger supplies, there may be an increased need for the surveillance authority to undertake verification activities. See [section 5.2.2](#) for adapting surveillance activities within such settings.

With respect to verification testing, the parameters recommended for the minimum monitoring of community supplies are those that best establish the hygienic state of the water and thus the risk of waterborne disease. The essential parameters of water quality are *E. coli*—thermotolerant (faecal) coliforms are accepted as suitable substitutes—and chlorine residual (if chlorination is practised). These should be supplemented, where appropriate, by measurement of pH (if chlorination is practised) and turbidity.

These parameters may be measured on site using relatively unsophisticated testing equipment, and improved and relatively low cost systems continue to be developed. On-site testing is essential for the determination of turbidity and chlorine residual, which change rapidly during transport and storage; it is also important for the other parameters where laboratory support is lacking or where transportation problems would render conventional sampling and analysis impractical.

Other health-related parameters of local significance should also be measured. Comprehensive analysis of the chemical quality of all sources is recommended prior to commissioning as a minimum, and periodically thereafter at a suitable frequency considering the outcomes of a risk assessment and other context-specific factors. In situations where resources for chemical water quality monitoring are especially limited, a cross-sectional survey approach may be considered to support decision-making. The overall approach to control of chemical contamination is outlined in [chapter 8](#). See also sections 3.3.3–3.3.8 of the *Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)) for additional considerations for verification monitoring of small water supplies.

Quality assurance and quality control

Appropriate quality assurance and analytical quality control procedures should be implemented for all activities linked to the production of drinking-water quality data. These procedures will ensure that the data are fit for purpose—in other words, that the results produced are of adequate accuracy. Fit for purpose, or adequate accuracy, will be defined in the water quality monitoring programme, which will include a statement about accuracy and precision of the data. Because of the wide range of substances, methods, equipment and accuracy requirements likely to be involved in the monitoring of drinking-water, many detailed, practical aspects of analytical quality control are concerned. These are beyond the scope of this publication.

Guidance on the design and implementation of a quality assurance programme for water quality laboratories is provided in [chapter 9](#) of *Water quality monitoring: a practical guide to the design and implementation of freshwater quality studies and monitoring programmes* (Bartram & Ballance, 1996). Ideally, laboratories should be accredited to a recognized standard for demonstrating technical competence and quality assurance such as ISO/IEC 17025:2017, *General requirements for the competence of testing and calibration laboratories* published by the International Organization for Standardization (ISO).

Guidance on sampling is given in the ISO standards listed in [Table 4.5](#).

WSP auditing

In addition to testing of water quality, verification should include audits of WSPs to demonstrate that the plans have been properly designed, are being implemented correctly and are effective.

Audits can be undertaken as part of internal or external reviews and may form part of surveillance by independent authorities. Auditing can have both an assessment and a compliance-checking function. Further information can be found in the supporting document *A practical guide to auditing water safety plans* ([Annex 1](#)).

Table 4.5 ISO standards for water quality giving guidance on sampling^a

ISO standard no.	Title (water quality)
5667-1:2023	Sampling—Part 1: Guidance on the design of sampling programmes and sampling techniques
5667-3:2024	Sampling—Part 3: Preservation and handling of water samples
5667-4:2016	Sampling—Part 4: Guidance on sampling from lakes, natural and man-made
5667-5:2006	Sampling—Part 5: Guidance on sampling of drinking water and water from treatment works and piped distribution systems
5667-6:2014	Sampling—Part 6: Guidance on sampling of rivers and streams
5667-11:2009	Sampling—Part 11: Guidance on sampling of groundwaters
5667-13:2011	Sampling—Part 13: Guidance on sampling of sludges
5667-14:2014	Sampling—Part 14: Guidance on quality assurance and quality control of environmental water sampling and handling
5667-16:2017	Sampling—Part 16: Guidance on biotesting of samples
5667-17:2008	Sampling—Part 17: Guidance on sampling of bulk suspended solids
5667-20:2008	Sampling—Part 20: Guidance on the use of sampling data for decision making—Compliance with thresholds and classification systems
5667-21:2010	Sampling—Part 21: Guidance on sampling of drinking water distributed by tankers or means other than distribution pipes
5667-23:2011	Sampling—Part 23: Guidance on passive sampling in surface waters
TS 13530:2009	Guidance on analytical quality control for chemical and physicochemical water analysis
17381:2003	Selection and application of ready-to-use test kit methods in water analysis

^a ISO has also established quality management standards relating to drinking-water supply, including ISO 24510:2024, Activities relating to drinking water and wastewater services—Guidelines for the assessment and for the improvement of the service to users; and ISO 24512:2024, Activities relating to drinking water and wastewater services—Guidelines for the management of drinking water utilities and for the assessment of drinking water services.

In the case of small water supplies, sanitary inspections can be considered as an alternative to auditing where WSPs are not in place, or between WSP audit events. This activity would normally be carried out by the surveillance agency (see [section 5.1.2](#) for more information), ideally complementing risk assessment and management carried out by the small water supply service provider (see [Box 4.1](#) for more information).

4.4 Management and communication

4.4.1 Management procedures

Many management procedures describe actions to be taken to maintain optimal operation under normal operating conditions (often referred to as “standard operating procedures”). These will include both responses to normal variations in operational monitoring parameters and responses when operational monitoring parameters reach critical limits. All activities, including standard operating procedures applied during

normal conditions, and planned responses to incidents and emergencies, should be documented and implemented in practice.

Incident response

A significant deviation in operational monitoring where a critical limit is exceeded (or in verification) is often referred to as an

“incident”. An incident is any situation that represents some degree of loss in system control that could compromise the drinking-water supply, or have the potential to escalate to an emergency. As part of a WSP, management procedures should be defined for response to incidents and emergencies, including unforeseen events.

Incident response plans can have a range of alert levels. These can be minor early warning, necessitating no more than additional investigation, through to those that may escalate and trigger an emergency. Emergencies are likely to require the resources of organizations beyond the drinking-water supplier, particularly the public health authorities.

Incident response plans typically include defined responsibilities and contacts, trigger indicators with alert levels, specified response actions, locations of procedures and equipment (including backups), and supporting logistic, technical and reference information.

The plan may need to be followed at very short notice, so standby rosters, effective communication systems and up-to-date training and documentation are required.

Staff should be trained in response procedures to ensure that they can manage incidents or emergencies effectively. Incident and emergency response plans should be periodically reviewed and practised. This improves preparedness and provides opportunities to improve the effectiveness of plans before an emergency occurs.

The preparation of clear procedures, definition of accountability and provision of equipment for the sampling and storing of water in the event of an incident can be valuable for follow-up epidemiological or other investigations, and the sampling and storage of water from early on during a suspected incident should be part of the response plan.

Emergency response

Water suppliers should develop plans to be invoked in the event of an emergency. These plans should consider potential natural disasters (e.g. earthquakes, floods, damage to electrical equipment by lightning strikes), accidents (e.g. spills in the watershed, interruptions in electricity supply), damage to treatment plant and distribution system and human actions (e.g. strikes, sabotage or cyber-attacks). Emergency plans should clearly specify responsibilities for coordinating measures to be taken, a communication plan to alert and inform users of the drinking-water supply and plans for providing and distributing emergency supplies of drinking-water.

Plans should be developed in consultation with relevant regulatory authorities and other key agencies and should be consistent with national and local emergency

Effective management implies definition of actions to be taken during normal operational conditions, of actions to be taken in specific “incident” situations where a loss of control of the system may occur and of procedures to be followed in emergency situations. Management procedures should be documented alongside system assessment, monitoring plans, supporting programmes and communication required to ensure safe operation of the system.

response arrangements. Key areas to be addressed in emergency response plans include:

- response actions, including increased monitoring;
- responsibilities of authorities internal and external to the organization;
- plans for emergency drinking-water supplies;
- communication protocols and strategies, including notification procedures (internal, regulatory body, media and public);
- mechanisms for increased public health surveillance.

Response plans for emergencies and unforeseen events involving microorganisms or chemicals should also include the basis for issuing boil water advisories (see [section 7.6.1](#)) and water avoidance advisories (see [section 8.7.10](#)). The objective of the advisory should be taken in the public interest. Therefore, the advisory should be issued after rapid, but careful, consideration of available information and conclusion that there is an ongoing risk to public health that outweighs any risk from the advice to boil or avoid water. The advisory will typically be managed by public health authorities. A decision to close a drinking-water supply carries an obligation to provide an alternative safe supply and is very rarely justifiable because of the adverse effects, especially to health, of restricting access to water. Specific actions in the event of a guideline exceedance or an emergency are discussed in [section 7.6](#) (microbial hazards) and [section 8.7](#) (chemical hazards); more general considerations are discussed in [section 6.8](#). “Practice” emergencies are an important part of the maintenance of readiness for emergencies. They help to determine the potential actions that can be taken in different circumstances for a specific water supply.

4.4.2 Supporting programmes

In addition to control measures, many actions are important in ensuring drinking-water safety. Activities that improve the management of drinking-water supplies that are consistent with the implementation of water safety planning are referred to as “supporting programmes”. These should also be documented in a WSP and implemented. Supporting programmes could involve:

- controlling access to treatment plants, catchments and reservoirs and implementing the appropriate security measures to prevent transfer of hazards from people when they do enter source water;
- developing verification protocols for the use of chemicals and materials in the drinking-water supply—for instance, to ensure the use of suppliers that participate in quality assurance programmes;
- using designated equipment for attending to incidents such as mains bursts (e.g. equipment should be designated for potable water work only and not for sewage work);
- training and educational programmes for personnel involved in activities that could influence drinking-water safety; training should be implemented as part of induction programmes and frequently updated;

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- studies to model the effects of changing temperature and precipitation scenarios on future water demand;
- user education programmes in informal settlements to enhance safe collection and transportation practices from tap stands.

Supporting programmes will consist almost entirely of items that drinking-water suppliers and handlers will ordinarily have in place as part of their normal operation. For most, the implementation of supporting programmes will involve:

- collation of existing operational and management practices;
- initial and, thereafter, periodic review and updating to continually improve practices;
- promotion of good practices to encourage their use;
- audit of practices to check that they are being used, including taking corrective actions in case of non-conformance.

Supporting programmes are activities that improve management of drinking-water supplies that are consistent with the implementation of water safety planning.

Codes of good operating and management practice and hygienic working practice are essential elements of supporting programmes. These are often captured within standard operating procedures. They include, but are not limited to:

- hygienic working practices in maintenance;
- training and competence of personnel involved in drinking-water supply;
- tools for managing the actions of staff, such as quality assurance systems;
- securing stakeholder commitment, at all levels, to the provision of safe drinking-water;
- education of communities whose activities may influence drinking-water quality;
- calibration of monitoring equipment;
- record keeping.

Comparison of one set of supporting programmes with the supporting programmes of other suppliers, through peer review, benchmarking and personnel or document exchange, can stimulate ideas for improved practice.

Supporting programmes can be extensive, be varied and involve multiple organizations and individuals. Many supporting programmes involve catchment protection measures and typically include aspects of land use control. Some catchment protection measures are engineered, such as effluent treatment processes and stormwater management practices that may be used as control measures.

4.4.3 Communication strategies

Effective communication is a key component of water safety planning and supports transparency, accountability, and public health protection. Communication strategies should include:

- procedures for promptly advising of any significant incidents within the drinking-water supply, including notification of the public health authority;

- summary information to be made available to consumers—for example, through annual reports and on the Internet;
- establishment of mechanisms to receive and actively address community complaints in a timely fashion.

The right of consumers to health-related information on the water supplied to them for domestic purposes is fundamental. However, in many communities, the simple right of access to information will not ensure that individuals are aware of the quality of the water supplied to them; furthermore, the probability of consuming unsafe water may be relatively high. The agencies responsible for monitoring should therefore develop strategies for disseminating and explaining the significance of health-related information. Further information on communication is provided in [section 5.5](#).

4.5 WSP review and improvement

4.5.1 Periodic review

Water safety planning should be regarded as a continuous improvement process. WSP development and implementation need to be regularly reviewed to ensure that the documentation is up to date and that the WSP is effective. Reviews should consider:

- data collected as part of monitoring processes;
- changes to water sources and catchments;
- changes to treatment, demand and distribution;
- implementation of improvement and upgrade programmes;
- revised procedures;
- emerging hazards and risks, including those relating to climate change;
- changes in the particular needs of vulnerable and disadvantaged populations.

By regularly reviewing records generated through operational monitoring and verification, an operator or manager can detect that a process is approaching an operational limit (e.g. gradual increases in filtered water turbidity may indicate that filter replacement may be required). Review of records can be instrumental in identifying trends and in making operational adjustments. Periodic review of WSP records is recommended so that trends can be noted and appropriate actions decided upon and implemented.

4.5.2 Post-incident review

WSPs should also be reviewed following significant incidents, “near misses” and emergencies, to ensure that, where possible, incidents do not recur and, where this is not possible, to reduce impacts. Post-incident reviews may identify areas for improvement and the need for revision of WSPs.

Following any significant incident, emergency or near miss, an investigation should be undertaken involving all concerned staff. Commensurate with the significance of the event, the investigation should consider factors such as:

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- the cause of the problem;
- how the problem was first identified or recognized;
- the most essential actions required;
- any communication problems that arose, and how they were addressed;
- the immediate and longer-term consequences;
- how well the emergency response plan functioned.

Appropriate documentation and reporting of the incident or emergency should also be established. The organization should learn as much as possible from the incident or emergency to improve preparedness and planning for future incidents. Review of the incident or emergency may indicate necessary amendments to the WSP and existing protocols.

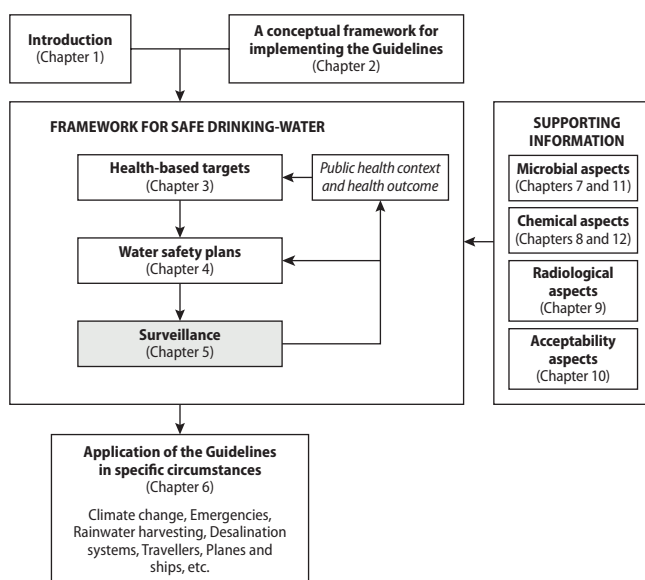
5

Surveillance

Drinking-water supply surveillance is “the continuous and vigilant public health assessment and review of the safety and acceptability of drinking-water supplies” (WHO, 1976). This surveillance contributes to the protection of public health by promoting improvement of the quality, quantity, accessibility, coverage, affordability and continuity of water supplies (known as service indicators) and

is complementary to the quality control function of the drinking-water supplier. Drinking-water supply surveillance does not remove or replace the responsibility of the drinking-water supplier to ensure that a drinking-water supply is of acceptable quality and meets predetermined health-based targets.

All members of the population receive drinking-water by some means—including the use of piped supplies with or without treatment and with or without pumping (supplied via domestic connection or public standpipe), delivery by tanker truck or carriage by beasts of burden or collection from groundwater sources (springs or wells) or surface sources (lakes, rivers and streams). It is important for the surveillance agency to build up a picture of the frequency of use of the different types of supply, especially as a preliminary step in the planning of a surveillance programme. There



is little to be gained from surveillance of piped water supplies alone if these are available to only a small proportion of the population or if they represent a minority of supplies.

Information alone does not lead to improvement. Instead, the effective management and use of the information generated by surveillance make possible the rational improvement of water supplies—where “rational” implies that available resources are used for maximum public health benefit.

Surveillance is an important element in the development of strategies for incremental improvement of the quality of drinking-water supply services. It is important that strategies be developed for implementing surveillance, collating, analysing and summarizing data and reporting and disseminating the findings and that the strategies are accompanied by recommendations for remedial action. Follow-up will be required to ensure that remedial action is taken.

Surveillance extends beyond drinking-water supplies operated by a discrete drinking-water supplier to include drinking-water supplies that are managed by communities and includes assurance of good hygiene in the collection and storage of household water.

The surveillance agency must have, or have access to, legal expertise in addition to expertise on drinking-water and water quality. Drinking-water supply surveillance is also used to ensure that any transgressions that may occur are appropriately investigated and resolved. In many cases, it will be more appropriate to use surveillance as a mechanism for collaboration between public health agencies and drinking-water suppliers to improve drinking-water supply than to resort to enforcement, particularly where the problem lies mainly with small drinking-water supplies.

The authorities responsible for drinking-water supply surveillance may be the public health ministry or other agency (see [section 1.2.1](#)), and their roles encompass four areas of activity:

- 1) public health oversight of formal drinking-water supplies;
- 2) public health oversight and information support to populations without access to formal drinking-water supplies, including supplies managed by communities and households;
- 3) consolidation of information from diverse sources to enable understanding of the overall drinking-water supply situation for a country or region as a whole as an input to the development of coherent public health-centred policies and practices;
- 4) participation in the investigation, reporting and compilation of outbreaks of waterborne disease.

A drinking-water supply surveillance programme should normally include processes for approval of water safety plans (WSPs). This approval will normally involve review of the system assessment, of the identification of appropriate control measures and supporting programmes and of operational monitoring and management plans. It should ensure that the WSP covers normal operating conditions and predictable incidents (deviations) and has contingency plans in case of an emergency or unplanned event.

The surveillance agency may also support the development of WSPs for small water supplies including community-managed supplies, as well as supporting their ongoing implementation. Such plans may be generic for particular technologies rather than specific for individual systems. They may also support proactive risk management by households, including household water treatment and storage.

5.1 Types of approaches

There are two types of approaches to surveillance of drinking-water quality: audit-based approaches and approaches relying on direct assessment. Implementation of surveillance will generally include a mixture of these approaches according to supply type and may involve using rolling programmes whereby systems are addressed progressively. Often it is not possible to undertake extensive surveillance of all community or household supplies. In these cases, well-designed surveys should be undertaken in order to understand the situation at the national or regional level.

5.1.1 Audit

In the audit approach to surveillance, assessment activities, including verification testing, are undertaken largely by the supplier, with third-party auditing to verify compliance. It is increasingly common that analytical services are procured from accredited external laboratories. Some authorities are also experimenting with the use of such arrangements for services such as sanitary inspection, sampling and audit reviews.

An audit approach requires the existence of a stable source of expertise and capacity within the surveillance agency in order to:

- review and approve new WSPs;
- undertake or oversee auditing of the implementation of individual WSPs as a programmed routine activity;
- respond to, investigate and provide advice on receipt of reports on significant incidents.

Periodic audit of the implementation of WSPs is required:

- at intervals (the frequency of routine audits will be dependent on factors such as the size of the population served and the nature and quality of source water and treatment facilities);
- following substantial changes to the source, the distribution or storage system or treatment processes;
- following significant incidents.

Periodic audit would normally include the following elements:

- examination of records to ensure that system management is being carried out as described in the WSP;
- ensuring that operational monitoring parameters are kept within operational limits and that compliance is being maintained;
- ensuring that verification programmes are operated by the water supplier (either through in-house expertise or through a third-party arrangement);

- assessment of supporting programmes and of strategies for improving and updating the WSP;
- in some circumstances, sanitary inspection, which may cover the whole of the drinking-water system, including sources, transmission infrastructure, treatment plants, storage reservoirs and distribution systems.

In response to reports of significant incidents, it is necessary to ensure that:

- the event is investigated promptly and appropriately;
- the cause of the event is determined and corrected;
- the incident and corrective action are documented and reported to appropriate authorities;
- the WSP is reassessed to avoid the occurrence of a similar situation.

The implementation of an audit-based approach places responsibility on the drinking-water supplier to provide the surveillance agency with information regarding system performance against agreed indicators. In addition, a programme of announced and unannounced visits by auditors to drinking-water suppliers should be implemented to review documentation and records of operational practice in order to ensure that data submitted are reliable. Such an approach does not necessarily imply that water suppliers are likely to falsify records, but it does provide an important means of reassuring consumers that there is true independent verification of the activities of the water supplier. The surveillance agency will normally retain the authority to undertake some analysis of drinking-water quality to verify performance or enter into a third-party arrangement for such analysis.

5.1.2 Direct assessment

It may be appropriate for the drinking-water supply surveillance agency to carry out independent testing of water supplies. Such an approach often implies that the agency has access to analytical facilities with staff trained to carry out sampling, analysis and sanitary inspection.

Direct assessment also implies that surveillance agencies have the capacity to assess findings and to report to and advise suppliers and communities. A surveillance programme based on direct assessment would normally include:

- specified approaches tailored to the type of drinking-water supply, for example, formal supplies serving large to small municipalities, community-managed supplies and household-managed systems;
- sanitary inspections to be carried out by qualified personnel;
- sampling to be carried out by qualified personnel;
- tests to be conducted using suitable methods by accredited laboratories or using approved field testing equipment and qualified personnel;
- procedures on reporting findings and follow-up to ensure that they have been acted on.

For small drinking-water supplies and where the development of in-house verification or third-party arrangements is limited, direct assessment may be used as the

principal system of surveillance. This may apply to drinking-water supplies in small towns by small-scale private sector operators or local government. Direct assessment may lead to the identification of requirements to amend or update the WSP, and the process to be followed when undertaking such amendments should be clearly identified.

Where direct assessment is carried out by the surveillance agency, it complements other verification testing of the water supplier. General guidance on verification testing, which is also applicable to surveillance through direct assessment, is provided in [section 4.3](#).

5.2 Adapting approaches to specific circumstances

5.2.1 Urban areas in developing countries

Drinking-water supply arrangements in urban areas of developing countries are typically complex. There can often be one or more large piped supplies with household and public connections, in combination with a range of alternative drinking-water supplies, including point sources and vended water. In these situations, the surveillance programme should take account of the different sources of drinking-water and the potential for deterioration in quality during collection, storage and use. Furthermore, the population will vary in terms of socioeconomic status and vulnerability to water-related disease.

In many situations, zoning the urban area on the basis of vulnerability and drinking-water supply arrangements is required. The zoning system should include all populations within the urban area, including informal and periurban settlements, regardless of their legal status, in order to direct resources to where greatest improvements (or benefits) to public health will be achieved. This provides a mechanism to ensure that non-piped drinking-water sources are also included within drinking-water supply surveillance activities.

Experience has shown that zoning can be developed using qualitative and quantitative methods and is useful in identifying vulnerable groups and priority communities where drinking-water supply improvements are required.

5.2.2 Small water supplies (including community-managed supplies)

Small drinking-water supplies, including those that are managed by communities, are found in most countries and may be the predominant form of drinking-water supply for large sections of the population. Small water supplies may include simple piped water systems or a range of point sources, such as boreholes with hand pumps, dug wells and protected springs. Although small water supplies are diverse, they tend to experience a common set of operational, managerial, technical and resourcing challenges.

The control of water quality and implementation of surveillance programmes for such supplies often face significant constraints. These typically include:

- limited capacity and skills within the community to undertake process control and verification; this may increase the need both for surveillance to assess the state of drinking-water supplies and for surveillance staff to provide training and support to community members;

- the very large number of widely dispersed supplies, which significantly increases overall costs in undertaking surveillance activities.

Furthermore, it is often small water supplies, including in rural areas, that present the greatest water quality problems.

Experience from both developing and developed countries has shown that surveillance of small water supplies including community-managed drinking-water supplies can be effective when well designed and when the objectives are geared more towards a supportive role to enhance management than towards enforcement of compliance.

Surveillance of small water supplies including community-managed drinking-water supplies requires a systematic programme that encompasses all aspects of the drinking-water supply to the population as a whole, including sanitary inspection (including catchment inspections) or WSP auditing. Surveillance should address variability in source water quality, treatment process efficacy and the quality of distributed or household-treated and household-stored water (see [section 5.2.3](#)).

Experience has also shown that the role of surveillance may include health education and health promotion activities to improve healthy behaviour towards management of drinking-water supply and sanitation. Participatory activities can include sanitary inspection by communities and, where appropriate, community-based testing of drinking-water quality using affordable field test kits and other accessible testing resources.

In the evaluation of overall strategies, the principal aim should be to derive overall lessons for improving water safety for all small water supplies including community-managed supplies, rather than relying on monitoring the performance of individual supplies.

Frequent visits to every individual supply may be impractical because of the very large numbers of such supplies and the limitations of resources for such visits. However, surveillance of large numbers of these supplies can be achieved through a rolling programme of visits. Commonly, the aim will be to visit each supply periodically (e.g. once every 3–5 years) using either stratified random sampling or cluster sampling to select specific supplies to be visited. During each visit, in addition to water quality analysis, a sanitary inspection or WSP audit will normally be done to provide insight to contamination and its causes.

During each visit, testing of water stored in the home may be undertaken in a sample of households. The objective for such testing is to determine whether contamination occurs primarily at the source or within the home. This will allow evaluation of the need for investment in supply improvement or education on good hygiene practices for household water collection, treatment and safe storage. Household testing may also be used to evaluate the impact of a specific hygiene education programme.

For further information on considerations for surveillance of small water supplies, see *Guidelines for drinking-water quality: small water supplies* (WHO, 2024).

5.2.3 Household treatment, storage and collection systems

Where water is handled during collection, transport, treatment or storage by households, it may be vulnerable to contamination, and sampling of household-stored water

is of interest in independent surveillance. It is often undertaken on a “survey” basis to develop insights into the extent and nature of prevailing problems. Surveillance systems managed by public health authorities for drinking-water supplies using household treatment and household storage containers are therefore recommended.

The principal focus of surveillance of household-based interventions will be assessment of their acceptance and impact through sample surveys so as to evaluate and inform overall strategy development and refinement. Systematic determination of continued, correct and effective use and management is recommended so that deficiencies in use and management can be identified and corrected by those responsible.

5.3 Adequacy of supply

As the drinking-water supply surveillance agency has an interest in the population at large, its interest extends beyond water quality in isolation to include all aspects of the adequacy of drinking-water supply for the protection of public health.

In undertaking an assessment of the adequacy of the drinking-water supply, the following basic service parameters of a drinking-water supply should normally be taken into consideration:

- *Accessibility*: the percentage of the population that has reasonable access to an improved drinking-water supply;
- *Quantity*: the proportion of the population with access to different levels of drinking-water supply (inadequate access, basic access, intermediate access, optimal access) as a proxy for the quantity of water used;
- *Quality*: whether the supply has regularly verified water quality and an approved WSP (see [chapter 4](#)) that has been validated and is subject to periodic audit to demonstrate compliance with relevant regulations (see [chapters 3 and 4](#));
- *Continuity*: the percentage of the time during which drinking-water is available (daily, weekly and seasonally);
- *Affordability*: the price of water paid by domestic consumers.

As the assessment of the quality of drinking-water is covered extensively throughout the Guidelines, including in [section 5.5.2](#), this section focuses on the other basic service parameters.

5.3.1 Accessibility

From the public health standpoint, the proportion of the population with sustained, reliable access to safe drinking-water is the most important single indicator of the overall success of a drinking-water supply programme.

There are a number of definitions of access (or coverage), many with qualifications regarding safety or adequacy. Sustainable Development Goal (SDG) target 6.1 calls for universal and equitable access to safe and affordable drinking-water for all. The target is monitored using the indicator “proportion of population using safely managed drinking water services”, which is defined as the use of an improved water source that is accessible on premises, available when needed, and free from faecal and priority chemical contamination (WHO & UNICEF, 2017). SDG target 1.4 also references drinking-water, by calling for “all men and women” to have equal access to

basic services, including “basic drinking water services”, which are defined as use of an improved water source, provided that collection time is not more than 30 minutes for a round trip, including queuing.

An improved drinking-water source is one that has the potential to deliver safe water by the nature of its construction and design. Improved sources are more likely than unimproved sources to supply drinking-water free from microbiological contamination, although microbiological contamination does occur in all types of water supply, particularly when they are inadequately managed. Improved and unimproved water supply technologies are summarized below:

- *Improved drinking-water sources:*
 - piped supplies (including household and yard connections, public taps and standpipes)
 - boreholes and tubewells
 - protected dug wells
 - protected springs
 - rainwater
 - water kiosks
 - packaged water
 - delivered water.
- *Unimproved drinking-water sources:*
 - unprotected dug wells
 - unprotected springs
 - surface water (rivers, reservoirs, lakes, ponds, streams, canals and irrigation channels).

Determining the proportion of a population with reliable access to drinking-water is an important function of a drinking-water surveillance agency. This task can be facilitated by establishing a common definition for reasonable access, appropriate to the local context, which may describe a minimum quantity of water supplied per person per day, together with a maximum tolerable distance or time to a source (see [section 5.3.2](#)).

The global SDG indicators identify different levels of accessibility: the basic service indicator calls for improved water supplies to be located within a 30-minute round trip travel time, including queuing, while the safely managed services indicator requires that the water collection point be located on-premises (i.e. within the household, yard or plot).

5.3.2 Quantity

The quantity of water used by households has an important influence on health. There is a basic human physiological requirement for water to maintain adequate hydration and an additional requirement for food preparation. There is a further requirement for water to support hygiene, both personal and household, which is necessary for health.

Estimates of the volume of water needed for health purposes vary widely. In deriving World Health Organization (WHO) guideline values, it is assumed that the

Table 5.1 Summary of water access, adequacy and level of health concern

Access level	Distance/time	Likely volumes of water collected	Public health risk from poor hygiene	Intervention priority and actions
Inadequate access	More than 1 km or more than 30 min total collection time	Very low: can be below minimum daily requirements for hydration	Very high Hygiene practice compromised Basic consumption may be compromised	Very high Provision of safely managed water at least to intermediate access Hygiene education
Basic access	100 m to 1 km or 5–30 min total collection time	Average quantity unlikely to exceed 20 litres per person per day	High Hygiene may be compromised Laundry and bathing may occur off-plot	High Provision of safely managed water at least to intermediate access Hygiene education
Intermediate access	Water provided on-plot through at least one tap, or within 100 m or 5 min total collection time	Average quantity approximately 50 litres per person per day	Medium Personal hygiene should not be compromised under usual conditions but may be compromised under outbreak conditions	Medium Hygiene promotion still yields health gains Encourage optimal access
Optimal access	Supply of water through multiple taps within the house	Average quantity will exceed 100 litres per person per day	Low Personal hygiene should not be compromised Household cleaning is also likely assured	Low Hygiene promotion still yields health gains

Source: *Domestic water quantity, service level and health, 2nd edition* (supporting document in [Annex 1](#))

daily per capita consumption of drinking-water is approximately 2 litres for adults, although actual consumption varies according to climate, activity level and diet. Based on currently available data, a minimum volume of 5.3 litres per person per day will provide sufficient water for hydration under most conditions. There is insufficient empirical evidence to define a minimum quantity of water necessary for food preparation or for hygiene. Experience and expert opinion suggest that 20 litres per person per day is often sufficient for drinking, cooking, food hygiene, handwashing and face washing. However other hygiene practices, including bathing and laundry, may not be assured. Further, where demands for water are increased—for example, due to increased hand hygiene in response to outbreaks of disease—20 litres per person per day may be insufficient; in many cases, running water from a tap will be necessary to support sufficient handwashing.

The quantity of water collected and used by households is primarily a function of the distance to the water supply or the total collection time required (including queuing). This broadly equates to access level. Four levels of access can be defined, as shown in [Table 5.1](#).

Access level is a useful and easily measured indicator that provides a valid proxy for the quantity of water collected by households and is the preferred indicator for surveillance. Available evidence indicates that health gains accrue from improving access level in three key stages: delivery of water within 1 km or 5–30 minutes of total collection time (“basic access”); water being supplied reliably on-plot, especially when running water is available (“intermediate access”); and water being available within the home through multiple taps (“optimal access”). The first and second categories of access broadly align with the access levels associated with basic and safely managed services, respectively. Although health gains do occur in these two categories, health concerns remain, particularly during outbreaks of disease when enhanced personal hygiene is needed. The volume of water collected may also depend on the continuity, reliability and cost of the water. Therefore, collection of data on these indicators is important.

5.3.3 *Continuity*

Interruptions to drinking-water supply, as a result of either intermittent sources or engineering inefficiencies, are a major determinant of the quantity and quality of drinking-water available to households. Analysis of data on continuity of supply requires the consideration of several components. Continuity can be classified as follows:

- year-round service from a reliable source with no interruption of flow at the tap or source;
- year-round service with frequent (daily or weekly) interruptions, of which the most common causes are:
 - water production that is insufficient to meet demand, requiring rationing of water;
 - restricted pumping regimes in pumped systems, whether planned or due to power failure or sporadic failure;
 - peak demand exceeding the flow capacity of the transmission mains or the capacity of the reservoir;
 - excessive leakage within the distribution system;
 - excessive demands on community-managed point sources;
 - breakdown of equipment (e.g. handpump parts);
- seasonal service variation resulting from source water fluctuation, which typically has three causes:
 - natural variation in source water volume during the year, which may be exacerbated by changes in climate (see [section 6.1](#));
 - volume limitation because of competition with other uses, such as irrigation;
 - periods of high turbidity when the source water may be untreatable;
- combined frequent and seasonal discontinuity.

These classifications reflect broad categories of continuity, which are likely to affect the quantity and quality of water available and thereby public health in different ways. Any interruption of service is likely to result in an increased risk of degradation of water quality, an increased risk of exposure to contaminated water and there-

fore an increased risk of waterborne disease. Managed discontinuity often results in low supply pressure and a consequent risk of in-pipe recontamination. Other consequences include reduced availability and lower volume use, which adversely affect hygiene. This may be a problem both when the discontinuity is predictable and when it is highly unpredictable. Household water storage may be necessary, and this may lead to an increase in the risk of contamination during such storage and associated handling. Discontinuity often forces users to obtain water from inferior and distant sources. As a consequence, in addition to the obvious reduction in water quality and quantity, time is lost in water collection, and other risks (e.g. to musculoskeletal health) may arise. This applies particularly to women, who disproportionately bear the burden of collecting and hauling water from off-premises sources.

The SDGs do not specify a minimum quantity of water to be available, but the safely managed drinking-water services indicator calls for water to be “available when needed”. For the purposes of global reporting, households are considered to have water available when needed if they report having “sufficient water” or that water is available “most of the time” (i.e. at least 12 hours per day or 4 days per week). However, although this metric is useful for global monitoring, it does not imply that this level of continuity is sufficient to realize public health gains, or that it represents a normative target. The indicators used for SDG monitoring are simplifications and do not capture all aspects of water services that are important from a normative perspective.

5.3.4 Affordability

Affordability, or economic accessibility, implies that individuals or households should be able to purchase water without compromising the purchase of any other basic needs (UNICEF & WHO, 2021). A critical component in assessing affordability is the cost of water paid by individuals and further information related to costs is described in these Guidelines.

The cost of water has an influence on the use of water and selection of water sources. Households with the lowest levels of access to safe water supply frequently pay more for their water than do households connected to a piped water system. The high cost of water may force households to use alternative sources of water of poorer quality that represent a greater risk to health, or to use multiple sources of water, which may have varying quality and therefore also pose a risk to health. Furthermore, high costs of water may reduce the volumes of water used by households, particularly where multiple sources are not available, which in turn may influence hygiene practices and increase risks of disease transmission.

When assessing cost, it is important to collect data on the price at the point of purchase. Where households are connected to the drinking-water supplier, this will be the tariff applied. Where water is purchased from, for example, public standpipes or trucked water, the price at the point of purchase may be very different from the drinking-water supplier tariff. Many alternative water sources (notably vendors) also involve costs, and these costs should also be considered. In addition to recurrent costs, the costs for initial acquisition of a connection for households connected to the drinking-water supplier or where households have their own private point source should be considered.

5.4 Planning and implementation

For drinking-water supply surveillance to lead to improvements in drinking-water supply, it is vital that the mechanisms for promoting improvement are recognized and used.

The focus of drinking-water supply-related improvement activities (whether these are establishment of regional or national priorities, hygiene education programmes or enforcement compliance) will depend on the nature of the drinking-water supplies and the types of problems identified. A list of mechanisms for drinking-water supply improvement based on the output of surveillance is given below:

- *Establishing national priorities:* When the most common problems and shortcomings in the drinking-water system have been identified, national strategies can be formulated for improvements and remedial measures; these might include changes in training (of managers, administrators, engineers or field staff), rolling programmes for rehabilitation or improvement or changes in funding strategies to target specific needs.
- *Establishing subnational/regional priorities:* Regional offices of drinking-water supply agencies can decide in which communities to work and which remedial activities are priorities; public health criteria should be considered when priorities are set.
- *Establishing hygiene education programmes:* Not all of the problems revealed by surveillance are technical in nature, and not all are solved by drinking-water suppliers; surveillance also looks at problems involving household practices, for example, including water collection, transport, treatment and storage. The solutions to many of these problems are likely to require educational and promotional activities.
- *Auditing of WSPs and upgrading:* Where WSPs are required in regulations, surveillance should include auditing of WSPs to assess risk management practices by the water supplier, inform progressive improvement and confirm that water suppliers are in compliance with regulations. Drinking-water systems and their associated WSPs should be upgraded where they are found to be deficient, although feasibility must be considered, and enforcement of upgrading should be linked to strategies for progressive improvement. Further information on WSP auditing is available in the supporting document *A practical guide to auditing water safety plans* ([Annex 1](#)).
- *Enabling community operation and maintenance:* Support should be provided by a designated authority to enable community members to be trained so that they are able to assume responsibility for the operation and maintenance of community-managed drinking-water supplies. Further information on supporting community-managed supplies and other types of small water supplies is available in the *Guidelines for drinking-water quality: small water supplies* (WHO, 2024).
- *Establishing public awareness and information channels:* Publication of information on public health aspects of drinking-water supplies, water quality and the performance of suppliers can encourage suppliers to follow good practices, mobilize public opinion and response and reduce the need for regulatory enforcement, which should be an option of last resort.

- *Implementing programmes for household water treatment and safe storage:* If information from surveillance reveals no or only basic access to water service, as defined in [Table 5.1](#), or unsafe supplied water, the implementation of programmes to promote household water treatment and safe storage may be advised to improve water quality and promote hygienic water management at the household level. These may be effective interim measures for provision of safer water supported by appropriate outreach, education and training activities and creating supply chains for appropriate household water treatment and safe storage technologies. Further information is available in [section 7.3.2](#).

In order to make best use of limited resources where surveillance is not yet practised, it is advisable to start with a basic programme that develops in a planned manner. Activities in the early stages should generate enough useful data to demonstrate the value of surveillance. Thereafter, the objective should be to progress to more advanced surveillance as resources and conditions permit.

The activities normally undertaken in the initial, intermediate and advanced stages of development of drinking-water supply surveillance are summarized as follows:

- *Initial phase:*
 - Establish requirements for institutional development.
 - Provide training for staff involved in the programme.
 - Define the role of participants (e.g. quality assurance/quality control by supplier, surveillance by public health authority).
 - Develop methodologies suitable for the area.
 - Commence routine surveillance in priority areas (including inventories).
 - Limit verification to essential parameters and known problem substances.
 - Establish reporting, filing and communication systems.
 - Advocate improvements according to identified priorities.
 - Establish reporting to local suppliers, communities, media and regional authorities.
 - Establish liaison with communities; identify community roles in surveillance and means of promoting community participation.
- *Intermediate phase:*
 - Train staff involved in the programme.
 - Establish and expand systematic routine surveillance.
 - Expand access to analytical capability (often by means of regional laboratories, national laboratories being largely responsible for analytical quality control and training of regional laboratory staff).
 - Undertake surveys for chemical contaminants using wider range of analytical methods.
 - Evaluate all methodologies (sampling, analysis, etc.).
 - Use appropriate standard methods (e.g. analytical methods, fieldwork procedures).
 - Develop capacity for statistical analysis of data.
 - Establish national database.

- Identify common problems and improve activities to address them at regional and national levels.
- Expand reporting to include interpretation at the national level.
- Draft or revise health-based targets as part of a framework for safe drinking-water.
- Use legal enforcement where necessary.
- Involve communities routinely in surveillance implementation.
- *Advanced phase:*
 - Provide further or advanced training for staff involved in the programme.
 - Establish routine surveillance for all health and acceptability parameters at defined frequencies.
 - Use a full network of national, regional and local laboratories (including analytical quality control).
 - Use national framework for drinking-water quality.
 - Improve water services on the basis of national and local priorities, hygiene education and enforcement of standards.
 - Establish regional database archives compatible with national database.
 - Disseminate data at all levels (local, regional and national).
 - Involve communities routinely in surveillance implementation.

5.5 Reporting and communicating

An essential element of a successful surveillance programme is the reporting of results to stakeholders. It is important to establish appropriate systems of reporting to all relevant bodies. Proper reporting and feedback will support the development of effective remedial strategies. The ability of the surveillance programme to identify and advocate interventions to improve water supply is highly dependent on the ability to analyse and present information in a meaningful way to different target audiences. The target audiences for surveillance information will typically include:

- public health officials at local, regional and national levels;
- water suppliers;
- local administrations;
- communities and water users;
- local, regional and national authorities responsible for development planning and investment.

5.5.1 Interaction with community and consumers

Community participation is a desirable component of surveillance, particularly for drinking-water supplies managed by communities and households. As primary beneficiaries of improved drinking-water supplies, community members have a right to take part in decision-making. The community represents a resource that can be drawn upon for local knowledge and experience. They are the people who are likely to first notice problems in the

The right of consumers to information on the safety of the water supplied to them for domestic purposes is fundamental.

drinking-water supply and therefore can provide an indication of when immediate remedial action is required. Communication strategies should include:

- provision of summary information to consumers (e.g. through annual reports or the Internet);
- establishment and involvement of consumer associations at local, regional and national levels.

In many communities, however, the simple right of access to information will not ensure that individuals are aware of the quality or safety of the water supplied to them. The agencies responsible for surveillance should develop strategies for disseminating and explaining the significance of results obtained.

It may not be feasible for the surveillance agency to provide feedback information directly to the entire community. Thus, it may be appropriate to use community organizations, where these exist, to provide an effective channel for providing feedback information to users. Some local organizations (e.g. local councils and community-based organizations, such as women's groups, religious groups and schools) have regular meetings in the communities that they serve and can therefore provide a mechanism of relaying important information to a large number of people within the community. Furthermore, by using local organizations, it is often easier to initiate a process of discussion and decision-making within the community concerning water quality. The most important element in working with local organizations is to ensure that the organization selected can access the whole community and can initiate discussion on the results of surveillance (see [sections 7.6.1](#) and [8.7](#)).

5.5.2 Regional use of data

Strategies for regional prioritization are typically of a medium-term nature and have specific data requirements. While the management of information at a national level is aimed at highlighting common or recurrent problems, the objective at a regional level is to assign a degree of priority to individual interventions. It is therefore important to derive a relative measure of health risk. Although this information cannot be used on its own to determine which systems should be given immediate attention (which would also require the analysis of economic, social, environmental and cultural factors), it provides an extremely important tool for determining regional priorities. It should be a declared objective to ensure that remedial action is carried out each year on a predetermined proportion of the systems classified as high risk.

At the regional level, it is also important to monitor the improvement in (or deterioration of) both individual drinking-water supplies and the supplies as a whole. In this context, simple measures, such as the mean sanitary inspection score of all systems, the proportion of systems with given degrees of faecal contamination, the population with different levels of service and the mean cost of domestic consumption, should be calculated yearly and changes monitored.

As shown in [Table 7.10](#) in [section 7.4](#), the aim should be to provide drinking-water that consistently contains no faecal indicator organisms, such as *Escherichia coli*. However, in many developing and developed countries, a high proportion of small drinking-water supplies, including those managed by communities and households,

fail to meet requirements for water safety, including the absence of *E. coli*. In such circumstances, it is important that realistic goals for progressive improvement are agreed upon and implemented. It is practical to classify water quality results in terms of an overall grading for water safety linked to priority for action, as illustrated in [Table 5.2](#).

Grading schemes may be of particular use in small water supplies where the frequency of testing is low and reliance on analytical results alone is especially inappropriate. Such schemes will typically take account of both analytical findings and results of the sanitary inspection or WSP audit through matrices such as those shown in [Tables 5.3](#) and [5.4](#). An example grading scheme based on a microbial water quality result and sanitary inspection score is shown in [Table 5.3](#). Another grading scheme, based on microbial water quality and WSP audit findings, is shown in [Table 5.4](#).

Table 5.2 Example of categorization of drinking-water systems on the basis of population size and quality rating in order to prioritize actions (see also [Table 7.10](#))

Quality of drinking-water system ^a	Proportion (%) of samples negative for <i>E. coli</i>		
	< 5000 population	5000–100 000 population	> 100 000 population
A	90	95	99
B	80	90	95
C	70	85	90
D	60	80	85

^a Quality decreases from A to D.

Table 5.3 Example matrix indicating priority of remedial action based on a sanitary inspection score and microbial water quality test result^{a, b}

		Sanitary inspection score ^c			
		0	1–4	5–8	≥9
<i>E. coli</i> classification per 100 mL sample ^d	<1				
	1–10				
	11–100				
	>100				

Low risk: continue routine management and surveillance	Intermediate risk: action needed	High risk: priority action needed	Very high risk: urgent action needed

^a Where there is a potential discrepancy between the results of the microbial water quality assessment and the sanitary inspection, further follow-up or investigation is required.

^b See *Guidelines for drinking-water quality: small water supplies* (WHO, 2024) section 5.3.5 for guidance on adapting this table.

^c The score is determined by the total number of yes responses to the sanitary inspection questions included in the sanitary inspection form checklist. This total provides an indication of the susceptibility of the water supply to contamination, primarily from human and animal faeces.

^d This could represent colony forming units or most probable numbers, depending on the method used.

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Table 5.4 Example matrix indicating priority for remedial action based on WSP audit scores and microbial water quality results^{a, b}

		WSP audit performance ^c					
		>95%	86–95%	76–85%	66–75%	51–65%	≤50%
Proportion of samples that have no detectable <i>E. coli</i> (per 100 mL) over a period of time ^d	≥95%						
	90–94%						
	85–89%						
	<85%						

Low risk: continue routine management and surveillance	Intermediate risk: action needed	High risk: priority action needed	Very high risk: urgent action needed

^a Where there is a potential discrepancy between the results of the microbial water quality assessment and the WSP audit, further follow-up or investigation is required.

^b See *Guidelines for drinking-water quality: small water supplies* (WHO, 2024) section 5.3.5 for guidance on adapting this table.

^c The score is calculated as the number of WSP audit points allocated divided by the total number of points available.

^d This could represent colony forming units or most probable numbers, depending on the method used.

Combined analysis of water quality data and sanitary inspection or WSP audit provides a more appropriate indication of risk level. Understanding relative risk allows surveillance agencies and other stakeholders to identify water supplies that require increased attention and guidance. It also provides a mechanism for setting priorities for action, allocating resources and assessing the impacts of improvement programmes.

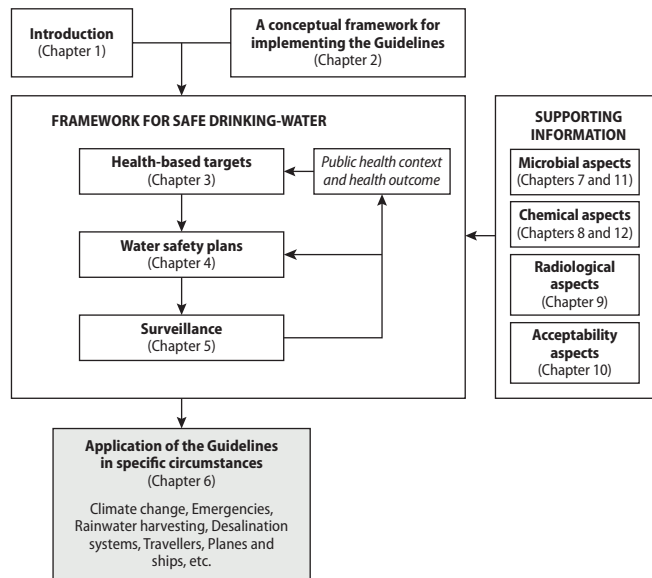
6

Application of the Guidelines in specific circumstances

These Guidelines provide a generally applicable approach to ensuring the safety of drinking-water supplied through large and small water supplies, including those managed by households, communities and professional entities. This chapter describes the application of the Guidelines in some commonly encountered circumstances and specific issues that should be taken into account in

each. The sections are not intended to stand alone, and reference is made to more comprehensive supporting documents that provide detailed guidance. In all the specific circumstances described below, the principles enshrined in water safety plans (WSPs) apply. However, the WSP should be tailored to the type of supply in each circumstance; for example, routine chemical and microbiological monitoring of rainwater may not be feasible at a household level, but preventive barriers are both applicable and achievable.

As indicated in [chapter 4](#), WSPs require careful consideration of possible hazards, and forward planning is one of the important requirements in ensuring that both the quantity and quality of water supplies are maintained. One of the significant concerns is climate change; all types of water supplies are being affected, including the specific circumstances discussed below.



6.1 Climate change: increasing threats from water scarcity, heavy rainfall and extreme events

Regional or localized droughts and heavy precipitation events and floods have always occurred, but they are increasing in frequency, and in some cases intensity, as a consequence of human-caused climate change. Anticipating and planning for these extreme weather events, such that sufficient quantities of safe water can be delivered to consumers without disruptions, are growing challenges for water suppliers and regulators. The effects of these climate extremes on water quality and quantity will be especially acute in areas with growing populations or where water demands are increasing. In such areas, existing water supplies typically are already stressed, and there is little, if any, water supply margin available in the event of a major or extended weather event. This may be a particular problem in regions with arid and semi-arid climates, such as parts of the Mediterranean, the Middle East, Australia and the south-western United States of America. Regions affected by flooding are also at risk. In some regions, drought may be rapidly followed by flooding, creating multiple, cascading threats.

Without action to reduce global heating, greater weather extremes will occur. Climate change is leading to more frequent and longer spells in extreme weather, with much higher peak temperatures, droughts, greater frequency of heavy precipitation, wildfires and violent storms, all of which may adversely affect drinking-water quality and, in some cases, water quantity. Changes in sea level from melting ice can affect coastal groundwater, causing salination, which may also occur as a result of over-abstraction. Windstorms will lead to storm surges that also degrade coastal water sources. With changes in water quantity come changes in water quality: greater or lesser runoff affects the sediment loading, chemical composition, total organic carbon content and microbial quality of water. These changes require modifications in water storage capacity and water treatment to ensure safe drinking-water. Changes in groundwater levels may also lead to altered mineral composition, and moves to deeper groundwater may tap into aquifers with high mineral content or high levels of specific constituents of concern for health.

To provide for adequate water quantity and quality in the event of these changes and extremes, natural supplies may need to be augmented in some areas, and more climate-resilient technologies and processes may need to be used. Water treatment systems may need to be upgraded and obtain greater storage capacity to be able to cope with greater microbial, turbidity and chemical loadings, and to improve the efficiency of water use during treatment. New sources of water may need to be developed, such as recycled wastewater or desalinated brackish water or seawater, and new strategies may need to be implemented, such as aquifer storage and recovery. In parallel, action will be required to improve water management, including detecting and reducing leaks, and reducing consumer demand.

WSPs should address climate change risks, particularly in the steps of assembling the team, assessing the system, preparing management procedures and developing supporting programmes (see [Figure 4.1](#)). For further information on the impacts of climate change on water supplies and practical information on how climate change impacts can be considered and addressed in the WSP process, see the supporting document *Climate-resilient water safety plans* ([Annex 1](#)).

6.2 Rainwater harvesting

Rainwater harvesting is widely practised at a household level but is increasingly being used on a larger community scale. Rainwater can provide an important source of drinking-water in some circumstances as well as a useful source of water for blending with other sources to reduce the levels of contaminants of health concern, such as arsenic and fluoride.

The development of formal WSPs at the household level may not always be practical, but promotion of sanitary inspection with simple good practice is important (see [chapter 4](#) and the sanitary inspection resources developed for rainwater harvesting systems in the supporting document *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#))). Well-designed rainwater harvesting systems with clean catchments, filter boxes, first-flush devices, covered cisterns and storage tanks, and treatment, as appropriate, supported by ongoing operations and maintenance, and good hygiene at point of use, can offer drinking-water with very low health risk.

Rainwater is initially relatively free from impurities, except those picked up by the rain from the atmosphere. However, the quality of rainwater may subsequently deteriorate during harvesting, storage and household use. Wind-blown dirt, leaves, faecal droppings from birds and other animals, insects and litter on the catchment areas, such as roofs and in cisterns, can contaminate rainwater, as can particles from the atmosphere, such as soot from burning materials like old tyres or ash from wildfires. Regular cleaning of catchment surfaces and gutters should be undertaken to minimize the accumulation of debris. Wire meshes or inlet filters should be placed over the top of downpipes to prevent leaves and other debris from entering storage containers and cleaned regularly to prevent clogging.

Materials used in the catchment and storage tank should be approved for use in contact with drinking-water and should not leach contaminants or cause taste, odour or discoloration. As rainwater is slightly acidic and very low in dissolved minerals, it can dissolve metals and other impurities from materials of the catchment and storage tank, resulting in unacceptably high concentrations of contaminants in the water. Most solid roof materials are suitable for collecting rainwater, but roofs with bitumen-based coatings are generally not recommended, as they may leach hazardous substances or cause taste problems. Where asbestos-containing roof materials are in place, the materials should be sealed with appropriate paint or resin to prevent fibres entering the water. Rainwater collected from asbestos roofing should be allowed to settle before use (i.e. allowing fibres to settle to the bottom of the container, before decanting off the water). Care should be taken to ensure that lead-based paints are not used on roof catchments. Mud or thatched roofs should be avoided where possible, as they can cause discoloration or deposition of particles in collected water, and they typically result in lower volumes of rainwater being collected.

Poor hygiene in water storage and abstraction from storage containers or at the point of use can also represent a health concern, but risks can be minimized by good design and practice. Faecal contamination is quite common, particularly in samples collected shortly after rainfall, but can be minimized by good practice. Higher microbial concentrations are generally found in the first flush of rainwater, decreasing as

the rain continues; therefore, microbial contamination is less in rainy seasons when catchments are frequently washed with fresh rainwater. A system to divert the contaminated first flow of rainwater from roof surfaces is necessary, and automatic devices that prevent the first flush of runoff from being collected in storage are recommended. If diverters are not available, a detachable downpipe can be used manually to provide the same result.

Storage tanks can present breeding sites for mosquitoes, including species that transmit dengue virus (see [section 8.6](#)). Covers discourage mosquito breeding and help to prevent faecal contaminants and sunlight from reaching the water (light can promote the growth of algae and cyanobacteria). Covers should be fitted, and openings need to be protected by mosquito-proof mesh. Cracks in the tank can result in contamination of stored water, and water loss, whereas water withdrawal using contaminated containers is a potential cause of both faecal and chemical contamination. Storage containers should preferably be fitted with a mechanism such as a tap or outlet pipe that enables hygienic abstraction of water.

Further treatment at the point of consumption may be applied to ensure better quality of drinking-water and reduce health risk. Solar water disinfection and point-of-use chlorination are examples of low-cost disinfection options for the treatment of stored rainwater. These and other household water treatment technologies are discussed in more detail in [sections 7.3.2](#) (microbial) and [8.4.4](#) (chemical).

6.3 Vended water

Vended water is common in many parts of the world where scarcity of supplies or lack of infrastructure limits access to suitable quantities of safe drinking-water. Although water vending is more common in developing countries, it also occurs in developed countries.

In the context of these Guidelines, water vending implies private vending of drinking-water, but does not include bottled or packaged water (which is considered in [section 6.15](#)) or water sold in bottles through vending machines.

Water vending may be undertaken by formal bodies, such as water utilities or registered associations, by contracted suppliers or by informal and independent suppliers. Where formal vending is practised, the water typically comes from treated utility supplies or registered sources and is supplied in tankers or from standpipes and water kiosks. Informal suppliers tend to use a range of sources, including surface water, dug wells and boreholes, and deliver small volumes for domestic use, often in containers loaded onto small carts or tanker trucks.

Both the quality and adequacy of vended supplies can vary significantly, and vended water has been associated with outbreaks of diarrhoeal disease (Hutin, Luby & Paquet, 2003). Water supplied to users should be suitable for drinking and comply with national or regional guidelines and regulatory requirements. The chemical and microbial quality of untreated or private sources of water should be tested to determine their suitability for use and to identify appropriate control measures, including treatment requirements. Surface water and some dug well and borehole waters are not suitable for drinking without treatment; disinfection is the minimum requirement, and filtration is often required when surface water is used.

In many developing countries, consumers purchase water from kiosks and then carry the water home in a variety of containers of varying size. Measures should be taken to protect vended water from contamination during transport as well as storage in the home, including transporting and storing water in containers that are clean, free from both faecal and chemical contamination and either enclosed or with narrow openings, ideally fitted with a dispensing device such as a spigot that prevents hand access and other sources of extraneous contamination. Good hygiene is required and should be supported by educational programmes.

In other cases, particularly in developed countries, vendors transport and deliver the water to users in tanker trucks. If large volumes are being transported, the addition of chlorine to provide a free residual concentration of at least 0.5 mg/l at the point of delivery to users is desirable. Tankers should also be used solely for water or, if this is not possible, should be thoroughly cleaned prior to use.

All components of systems associated with supplying and delivering vended water need to be designed and operated in a manner that protects water quality. Water storage containers, pipework and fittings should not include defects such as structural faults that allow leakage and permit the entry of contaminants. Cleanliness of storage containers, standpipes, taps and hoses needs to be maintained. Hoses used to transfer water at kiosks or used on carts and tanker trucks should be protected from contamination (e.g. by preventing contact of the ends with the ground) and drained when not in use. The area around standpipes should include drainage or be constructed in a manner to prevent pooling of water. Materials used in all components, including pipework, containers and hoses, need to be suitable for use in contact with drinking-water and should not result in contamination of the water with hazardous chemicals or with substances that could adversely affect its taste.

All components of water vending, including sources, methods of abstraction and transport, should be incorporated into a WSP. Where vendors are registered or have a contract with a water utility, implementation and operation of the WSP should be regularly checked by the utility. WSPs and the operation of water vendors should also be subject to independent surveillance.

6.4 Bulk water supply

Bulk water supplies can be either untreated or treated water, but usually there is limited or no choice in the provision of such supplies. They may be provided where one agency or company controls a large raw water source, usually surface water, and provides water to one or several other water suppliers. Bulk water supplies can be delivered by pipeline or tanker or using ships or fleets of road or rail tankers.

In all cases, it is important that the bulk supply is incorporated into the WSP of the receiving supply and treated as another source. Where bulk supplies of treated water have been used to provide support during a drought or emergency, it is vital that the receiving supplier takes steps to ensure that the water is safe before it is introduced into the receiving distribution system. At all stages, it is important that there is close communication between all parties involved and that the procedures and requirements are documented, understood and carried out with appropriate monitoring and verification.

The potential hazards from bulk water are similar to those from any water supply, but there are additional sources of contamination, such as inappropriate containers and materials and lack of sanitation and hygiene at bulk water filling connections or transfer points. Pipelines may be vulnerable to contamination along the transmission route, particularly if there is the potential for unapproved connections into the system.

Many of the requirements for bulk supply are the same as for any piped supply, such as using approved materials that will not adversely affect water quality. Where tankers are used, these should be of a suitable material and be clean and free from microbial and chemical contamination. To minimize contamination during filling of bulk water containers or water tankers and charging of water transmission pipelines, sanitary inspections and maintenance of sanitary conditions for water filling stations are necessary. These sites should have proper drainage to avoid standing water and flooding, should not be exposed to sources of contamination and should be secure, with access restricted to authorized personnel. At water filling and delivery points, nozzles and couplings should be protected from sources of contamination, including animals. Installation of protective coverings for filling and receiving connectors would help in this respect. Some plastic pipe materials are permeable to organic chemicals, and transfer of substances such as petroleum hydrocarbons could diminish the structural integrity of the pipe materials or render the water unpalatable to consumers. Such piping is most likely to be found in transfer hoses, so the cleanliness of the transfer points where tankers are used is vital, as is protection of the transfer area from spills of petroleum fuels.

Implementation of security measures to guard against intentional contamination and theft may also be warranted.

6.5 Desalination systems

Desalination is used to remove salts from brackish or saline surface water and groundwater to render it acceptable for human consumption or other uses. It is increasingly employed to provide drinking-water because of a growing scarcity of fresh water driven by population growth, overexploitation of water resources and climate change. Desalination facilities exist all over the world, particularly in the eastern Mediterranean region, with use increasing on all continents. Small-scale desalination is used to supply fresh water on ships and to provide additional fresh water in some hot and arid regions.

These Guidelines, including the use of WSPs, are fully applicable to desalinated water supply systems. Management of desalination includes consideration of specific aspects but these only represent variations in emphasis when developing and implementing WSPs, as described in these Guidelines.

Desalinated water has a very low total organic carbon content and low disinfectant demand, so disinfection by-products are generally of little concern, although brominated organics may occur owing to the presence of bromide in seawater. Membrane and distillation desalination processes are very efficient at removing higher molecular weight organic chemicals and virtually all inorganic chemicals, and volatile organic compounds are vented during thermal desalination processes. Where membranes are

used, boron and some smaller molecular weight organic substances may not be excluded, so it is important to establish the membrane capability. Because of the apparently high effectiveness of some of the processes used (especially distillation and reverse osmosis) in removing both microorganisms and chemical constituents, these processes may be employed as single-stage treatments or combined with only a low level of residual disinfectant. For further information, see the supporting document *Water treatment and pathogen control* ([Annex 1](#)). Pretreatment is largely in place to protect the desalination process, but it will also remove certain hazards present in brackish or saline waters.

Water produced by desalination is low in minerals and usually aggressive towards materials with which it comes into contact, such as materials used for distribution pipes, storage and plumbing. During post-treatment, the water must be stabilized or remineralized prior to distribution to reduce its corrosive nature. Stabilization is commonly achieved by adding chemical constituents such as calcium and magnesium carbonate along with pH adjustment or through blending with small volumes of mineral-rich waters. Seawater and spent seawater that has undergone electrolysis to form hypochlorite have been used for this purpose, but the latter practice has essentially ended because of the formation of bromate in the distributed water. Blending waters should be pretreated to ensure their microbial safety, because the post-desalination residual disinfectant level may be insufficient to control pathogens present in the blending water.

Desalinated water contains lower than usual concentrations of dissolved solids and essential elements such as calcium and magnesium, which are commonly found in water (see the supporting document *Calcium and magnesium in drinking-water*; [Annex 1](#)). Drinking-water typically contributes a small proportion to the recommended daily intake of essential elements, with most of the intake occurring through food. Fluoride would also be missing from desalinated water unless it were added prior to distribution, which may be considered by countries in which sugar consumption is high (WHO, 2005).

High temperatures of distributed water in warm climate areas and difficulty in maintaining disinfectant residuals during transport over long distances may lead to microbial aftergrowth, depending on nutrient availability. Although such growth is likely to be without health significance (see the supporting document *Heterotrophic plate counts and drinking-water safety*; [Annex 1](#)), it can contribute to problems of acceptability. The use of chloramines constitutes an advantageous alternative to free chlorine in distribution systems with long residence times and elevated temperatures, although nitrite formation by organisms in biofilms needs to be considered where chloramination is practised and excess ammonia is present.

Extensive information on desalination for safe drinking-water supply is available in the book *Desalination technology: health and environmental impacts* (Cotruvo et al., 2010) and the supporting document *Safe drinking-water from desalination* ([Annex 1](#)).

6.6 Potable reuse systems

Potable reuse of appropriately treated wastewater represents a source of drinking-water that is largely independent of the influence of climate change. Potable reuse,

like desalination, is increasing in response to the growing scarcity of fresh water driven by population growth, overexploitation of water resources and climate change. Potable reuse can involve direct supply of treated wastewater into drinking-water supplies or indirect supply through the planned addition of treated wastewater into water bodies (rivers, lakes, reservoirs or aquifers) for the purpose of augmenting existing sources of drinking-water.

These Guidelines, including the use of WSPs, are fully applicable to potable reuse schemes with no requirement for additional targets. Management of potable reuse includes consideration of specific aspects, but these only represent variations in emphasis when developing and implementing WSPs as described in these Guidelines. Specific aspects include the very poor quality of source waters; the associated complexity of potable reuse schemes, including the use of advanced water treatment processes; additional monitoring considerations; potential use of environmental buffers and engineered storages; and community acceptance. In some cases, wastewater management may be described in a separate sanitation safety plan, particularly where different entities are responsible for wastewater management and provision of drinking-water. Where both forms of safety plan are used, they will need to be coordinated to ensure that the overall scheme functions effectively.

Wastewater, by its nature, typically contains high concentrations of enteric pathogens, requiring the setting and meeting of appropriate performance targets to achieve production of safe drinking-water. Combinations of established treatment technologies can achieve the required performance targets.

Wastewater can also contain a broad range of chemicals, including domestic chemicals (e.g. cleaning compounds, garden products), chemicals excreted by people (e.g. pharmaceuticals, metabolites), personal care products, and chemicals used in wastewater and drinking-water treatment processes. Depending on the management and control of industrial discharges, wastewater can include significant concentrations of industrial chemicals. However, the impact of these chemicals can be minimized by applying appropriate industrial discharge controls. Metals and inorganic chemicals are generally present in moderate concentrations ($\mu\text{g/l}$ to mg/l) in untreated wastewater, and pharmaceuticals, personal care products and organic chemicals are generally present in lower concentrations (ng/l to $\mu\text{g/l}$). Treatment trains used for potable reuse often include specific processes such as reverse osmosis, nanofiltration or ozone/biologically activated carbon to reduce concentrations of potential chemical hazards and to provide protection against emerging chemicals of concern.

While potable reuse can be a practical source of drinking-water, potable reuse schemes are typically complex, and operators need to have sufficient resources and technical capabilities for successful operation.

For further guidance, see the supporting document *Potable reuse: Guidance for producing safe drinking-water* ([Annex 1](#)).

6.7 Dual piped water supply systems

In some locations, households and buildings served with a piped drinking-water supply may also receive piped water from an alternative source for non-potable purposes, creating a dual piped water supply system. The alternative water source is usually pro-

vided to reduce the use of high-quality water resources for non-potable uses (e.g. toilets, washing clothes, irrigation) or simply to conserve scarce water resources.

Non-potable piped supplies can potentially introduce health hazards, commonly through accidental cross-connections between potable and non-potable piped supplies. Measures to control health risks from dual piped supply systems include:

- use of good design practices that prevent cross-connections;
- unambiguous labelling of both systems to ensure that the non-potable supply is not mistaken for the potable supply;
- installation of the non-potable piped system only by qualified plumbers;
- regulation of non-potable piped systems by the authority responsible for drinking-water surveillance;
- public communication about the potential health risks from exposure to non-potable water through cross-connections and the dangers of modifying systems by inexperienced and non-certified individuals.

Increasingly in developed countries, dual systems are being installed at a household level or in public buildings. Guidance should be provided on installation, particularly where this is by non-certified individuals. Potable water supplied into the building should be fitted with a non-return valve in order to prevent backflow into the public water supply.

6.8 Emergencies and disasters

Safe water—for consumption and to enable adequate hygiene—is one of the most important public health requirements in most emergencies and disasters, along with adequate sanitation. The greatest waterborne risk to health comes from the transmission of faecal pathogens as a result of inadequate sanitation, hygiene and protection of drinking-water sources. Some important infectious diseases, such as those caused by the Ebola virus, mpox or SARS-CoV-2, do not have a faecal–oral transmission route but require sufficient and safe water for cleaning, caring for patients and good hand and environmental hygiene (WHO, 2020, 2021, 2024a, b). Some disasters, including those caused by or involving damage to chemical or nuclear industrial installations, spillage in transport or volcanic activity, may result in contamination by chemical or radiological hazards of concern. The circumstances of most large-scale emergencies will vary, and each will present its own peculiar problems and challenges.

Where a number of agencies are involved in disaster relief or overseeing an emergency, it is vital that there is good communication between the agencies and coordination of their activities. It is also important that the overall coordinators take advice from the experts in a particular field, such as water supply and sanitation.

This section considers primarily large-scale disasters and emergencies, although much of the information will apply to smaller-scale emergencies as well. For microbiological and chemical emergencies on a smaller scale in piped supplies, the relevant sections in [chapters 7 and 8](#) should be consulted. For emergencies involving radionuclides, see section 2 in the supporting document *Management of radioactivity in drinking-water* ([Annex 1](#)).

People displaced by conflict and natural disaster may move to an area where unprotected water sources are contaminated or safe storage capacities are not available. When population density is high and sanitation is inadequate, unprotected water sources in and around the temporary settlement are highly likely to become contaminated. A displaced population with low immunity due to malnutrition, as a consequence of food shortages or the burden of other diseases, is at an increased risk of an outbreak of waterborne disease.

Emergency planning initiatives should include three phases:

- 1) vulnerability assessments (which should be part of any WSP) to identify the critical elements of the existing systems that, if compromised, would result in major disruption of basic services;
- 2) mitigation/improvement plans to identify feasible actions to prevent or reduce the disruptive effects related to the loss of the vulnerable elements or facilities;
- 3) emergency preparedness, readiness and response plans to facilitate managing the crisis and restoring service should disruptions occur.

The key is to anticipate probable events, have plans in place, prepare to respond when needed, and have backup materials and facilities. As well, simulations should be conducted before the event so that the organization and its staff will be effective in the event of an emergency. Reference checklists to help strengthen water, sanitation and hygiene emergency preparedness and response in conflict settings have been developed by WHO (WHO EURO, 2025), and can be adapted for other contexts (also see [Annex 2](#) for additional resources).

Available sources of water are limited in most emergency situations, and providing a sufficient quantity of water of adequate quality for personal and domestic hygiene as well as for drinking and cooking is important. Detailed information may be found in Sphere Association (2018) and the supporting document *Domestic water quantity, service level and health* ([Annex 1](#)). Furthermore, it may be important to prioritize drinking-water to particular settings, such as health-care facilities (see [section 6.11](#)) and/or specific geographic regions (e.g. cholera hotspots). National drinking-water quality standards should therefore be flexible, taking into consideration the risks and benefits to health in the short and long terms. When standards are not being met, they should not excessively restrict water availability for hygiene, as this would often result in an increased overall risk of disease transmission.

There are a number of factors to take into consideration when providing drinking-water for a population affected by a disaster, including the following:

- *The quantity of water available and the reliability of supply:* These are likely to be the overriding concerns in most emergency situations, as it is usually easier to improve water quality than to increase its availability or to move the affected population closer to another water source. However, since ensuring sufficient quantities of water for consumption and hygiene is important, provision of increased quantities of water is common in emergency settings. This may involve rehabilitation or construction of new water supplies (e.g. network rehabilitation

or extensions, new boreholes) or, as a last resort, temporary transportation of water by trucks.

- *The equitability of access to water:* Even if sufficient water is available to meet minimum needs, additional measures may be needed to ensure that access is equitable. Unless water points are sufficiently close to their dwellings, people will not be able to collect enough water for their needs. Water may need to be rationed to ensure that everyone's basic needs are met.
- *Protecting the water source against contamination:* This should always be a priority in emergencies, whether or not disinfection of the water supply is considered necessary.
- *The need for disinfection:* Disinfection, maintaining an adequate disinfectant residual and, where necessary, pretreatment to reduce turbidity to as low as feasible in order to ensure the efficacy of disinfection are essential components in ensuring a safe drinking-water supply.
- *Acceptability:* It is important to ensure that drinking-water provided in emergencies is acceptable to the consumers in terms of taste, odour and appearance, or the consumers may resort to water from unprotected or untreated supplies.
- *The need for containers to collect and store water:* Hygienic containers are needed for the collection and storage of water to be used for drinking, cooking and washing. Further information on storage considerations for drinking-water can be found in [section 7.3.2](#).
- *The availability of bottled or packaged water:* Provision of bottled or packaged water from a reliable source is often an effective way to quickly provide safe, potable water in emergencies and disasters. Brewers and soft drink producers, if they are part of the emergency response plan, are often capable of converting their processes to produce bottled or packaged water in emergencies. This is particularly valuable if they have water treatment plants for ensuring the quality of water used as an ingredient in their processes. Management and monitoring processes should be in place to ensure safe drinking-water production.
- *Longer-term planning for continuing emergency situations:* When the acute phase of an emergency or disaster is over, consideration needs to be given to the longer-term provision of safe water and sanitation. Long-term planning at an early stage of a response can save costs (e.g. replacing water trucking with more sustainable water supply solutions, such as rehabilitation/construction of water networks or new boreholes). Long-term planning can also build local capacities (e.g. support for water safety planning to better identify and manage risks, including related to emergencies; training of local employees for more sustainable water quality monitoring systems). Finally, it can provide an opportunity to develop water, sanitation and hygiene services in under-resourced areas and for vulnerable populations; for example, ensuring sustainable interventions in cholera hotspots can prevent future outbreaks, in addition to providing an immediate response. This is particularly valuable for countries and regions (e.g. fragile and conflict-affected countries) that are experiencing recurrent or longer-term, chronic emergencies that can last several years or more.

In many emergency situations, water is collected from central water collection points, stored in containers and then transferred to cooking and drinking vessels by the affected people. It is important that people be aware of the risks to health from contamination of water from the point of collection to the moment of consumption and have the means to reduce or eliminate these risks. Detailed information may be found in Wisner & Adams (2002). Point-of-use treatment may be necessary; if so, ensuring that water treatment technologies meet WHO performance standards is important. A list of products that meet WHO standards can be found on the web pages of the WHO International Scheme to Evaluate Household Water Treatment Technologies.¹ Further information on point-of-use treatment and safe storage can be found in [section 7.3.2](#). The information in [section 6.12](#), including [Table 6.1](#) on drinking-water disinfection methods that can be used by travellers, may also be applied for temporary use in emergency situations.

Water safety should be assessed during emergencies including as part of any investigation of disease outbreaks. This should minimally include conducting sanitary inspections; monitoring microbial water quality; and monitoring water treatment processes, including disinfection. There may be a need to monitor other priority water quality parameters and evaluate hygiene promotion activities. The priority should be ensuring adequate disinfection (e.g. monitoring chlorine residual) in distribution systems, storage reservoirs, water collection points and households, including in high-density areas such as camps and periurban areas, and during outbreaks of diarrhoeal disease.

Monitoring and reporting systems should be designed and managed to ensure that action is swiftly taken to protect health. Health information should also be monitored to ensure that water quality can be rapidly investigated when it is suspected of contributing to a health problem so that treatment processes, particularly disinfection, can be modified as required.

Where large numbers of water samples need testing or analysis of a broad range of parameters is of interest, laboratory analysis is usually most appropriate. If the drinking-water supplier's laboratories or laboratories at environmental health offices and universities no longer function because of the disaster, a temporary laboratory may need to be set up. Where samples are transported to laboratories, appropriate handling is important to ensure meaningful results. Increased use of portable testing kits may also be considered, to monitor priority water quality parameters, such as *E. coli*/thermotolerant coliform count, free residual chlorine, pH and turbidity.

Workers should be trained in the correct procedures for collecting, labelling, packing and transporting samples and in supplying supporting information from sanitary inspections to help interpret water quality test results. For resources to support application of sanitary inspections, see the supporting publication *Sanitary inspection packages – a supporting tool for the Guidelines for drinking-water quality: small water supplies* ([Annex 1](#)). For guidance on methods of water sampling and

¹ <https://www.who.int/tools/international-scheme-to-evaluate-household-water-treatment-technologies/products-evaluated>

testing, see Bartram & Ballance (1996), APHA, AWWA & WEF (2005) and Snoad & Weber (2024).

For further information on considerations for providing safe drinking-water during emergencies and disasters, see Sphere Association (2018) and WHO/WEDC (2013).

6.9 Temporary water supplies

A number of waterborne disease outbreaks have occurred as a result of poor management and design of temporary water supplies, which are distributed water supplies for planned seasonal or time-limited events (e.g. festivals, markets and summer camps). Water supplies for holiday towns are not covered, because they are permanent supplies, although substantial seasonal variations in demand bring specific problems.

A systematic approach to drinking-water safety, including adequate quantity and quality, is needed for temporary water supplies. A WSP is an essential requirement in identifying the hazards and risks and developing good management procedures to deal with them. [Chapter 4](#) and other sections in [chapter 6](#) provide additional useful information. Where water is supplied through tankers, the requirements are the same as for vended water ([section 6.3](#)) and bulk water supplies ([section 6.4](#)).

A temporary water supply may be independent (i.e. not connected to any other water supply system and with its own facilities from source to tap) or dependent (i.e. receiving treated water from an existing water supply system but with independent distribution facilities). The risk of drinking-water contamination is usually lower in dependent systems, provided there is access to the technologies, expertise and management of the permanent system. A contract is often made between the organizer of an event (e.g. a festival) and a water supply entity, which should include the water quantity and quality supplied by the entity, the roles and responsibilities of each party in water quality management, the locations and frequency of water quality monitoring, sanitary inspection and surveillance by a health authority and the provision of adequate and properly sited sanitation. Coordination between an event organizer, a water supply entity and the relevant health authority is very important for ensuring drinking-water safety.

Temporary water supply systems can vary substantially in terms of their scale, period of operation, water use and fluctuations in demand, and these variations should be taken into consideration during the planning and design stages. In the case of an independent system, adequate consideration should also be given to the selection of a water source in terms of quantity, quality and treatment processes, and care should be taken not to adversely affect any other supply or water source. Where a temporary system is directly connected to a mains water supply, it is important to prevent the accidental contamination of the mains water supply through backflow during construction and operation of the temporary system. Water consumption for firefighting, hand washing and toilet flushing should be taken into account in estimating total and predictable variations in water demand where there are no other water sources available for such purposes.

Water quality targets for temporary supplies should be the same as those for permanent water supplies. Disinfection should be considered indispensable in a tem-

porary supply, and it is preferable to maintain a certain level of disinfectant (e.g. chlorine) residual at service taps. If the supply is not for potable uses, appropriate action should be taken to ensure that it is not used for drinking.

If a temporary water supply is used recurrently, it is essential to fully flush the entire system with water containing a higher than normal disinfectant residual before restarting. When planning installation on site, positioning of pipes, hoses and connections should take risks of contamination into account—for example, by avoiding the placement of hosing and fittings on the ground near sites of potential faecal contamination or storage tanks in direct sunlight where rising temperatures support microbial growth. It is also important to ensure that the facility has no defects, including leakage, that could cause the deterioration of water quality and that water quality at every service tap satisfies the required quality target. Important control measures during dismantling and transport of installations include emptying hoses, preferably drying them and storing them so that ingress of contamination is avoided. In all cases, the materials should be approved for use in contact with potable water.

Care should be taken in planning and designing wastewater management and disposal facilities, particularly to ensure that lavatories and disposal facilities are located so as to avoid any risk of adversely affecting source water quality or stored water. It is also important to prevent runoff from other areas, such as livestock pens, from entering the source. The source, treatment facilities and distribution reservoirs should be well protected from access by animals (e.g. bird faeces) and humans by covers or roofs.

A temporary system is usually more vulnerable to accidental and deliberate contamination than an existing permanent water supply system, and attention needs to be paid to security. All water treatment facilities should be thoroughly inspected at least every day. All of these procedures and requirements should be included in the operational management documents that are at the core of the WSP.

Signs are an important part of ensuring that water from taps is used appropriately and the protection of water sources and drinking-water infrastructure. The signs should be easily understood and used in conjunction with other barriers, such as fences.

Water quality and appearance should be routinely monitored at the service taps of a temporary water supply system. At the very least, water temperature and disinfectant residual should be monitored every day as simple rapid tests that act as indicators of possible problems. Other basic parameters that should be regularly monitored, if possible, include pH, conductivity, turbidity, colour and *Escherichia coli* (or thermo-tolerant coliforms). Routine sanitary inspection of a temporary water supply by the appropriate health authority is very important. If any problem related to water quality arises, remedial actions that are included in the management documents supporting the WSP should be taken promptly. If a temporary water supply system is to be used for a period of more than a few weeks, regular surveillance by the appropriate health authority should be implemented.

6.10 Buildings²

Drinking-water systems in buildings can be a significant source of contamination, and poor management of these systems has contributed to outbreaks of disease and illness. One of the challenges in ensuring water safety is that responsibility for many actions essential to the control of drinking-water quality in buildings is often outside the mandate of the drinking-water supplier. Roles and responsibilities of different stakeholders relating to the safe management of drinking-water systems within buildings can be influenced by a number of factors, including ownership of assets and rights of access. WSPs established for management of public water supplies are not typically extended to buildings, although the water supplier WSP may include a number of initiatives to ensure that backflow prevention is in place or to provide information to consumers on protecting their own water quality. In many cases, owners, managers or maintenance personnel are responsible for managing building water supplies, but awareness and application of drinking-water guidelines are often limited, and so educational supporting programmes may be required.

The design of water networks in buildings is variable, as influenced by the diversity of building types (e.g. schools, child-care facilities, residential buildings, hotels, sports facilities, factories, office blocks, museums, transport terminals), designs and water uses. Drinking-water systems in buildings are typically divided into hot and cold water networks and may be connected to water-based devices (e.g. cooling towers, boilers, swimming pools) or point-of-use equipment (e.g. washing machines).

General drinking-water safety is ensured by good management practices, including sound design, routine maintenance protocols, regular cleaning, temperature management and flow management (avoidance of stagnation). These practices should be incorporated in WSPs developed by building owners or managers. WSPs for buildings should address cold and hot drinking-water networks and consider water-based devices and point-of-use equipment. Regulatory or other appropriate authorities may provide guidance on the development and application of WSPs for drinking-water systems in buildings.

The regulator can specify compliance requirements for buildings in general or for specific types of buildings based on the level of risk. Schools, hotels and some other large buildings are high-risk environments because of both the complex nature of their drinking-water systems and the vulnerability of some users, occupants and visitors, and heightened vigilance in terms of operational monitoring, validation of control measures and verification is generally justified. Compliance may require that maintenance and monitoring programmes be carried out through a building-specific WSP. It may be appropriate to display maintenance and monitoring programmes and certification of compliance at a conspicuous location within the building. Compliance could be verified and certified by an independent auditor.

The principal hazard that may threaten drinking-water systems of buildings is ingress of contamination from external water supplies or through faults in the distribution system (including storage tanks). Unapproved and inappropriate fittings and materials can lead to the release of chemical substances from tanks, piping, jointing

² Hospitals, nursing care homes and other health-care facilities are discussed in [section 6.11](#).

and plumbing materials. The release may vary with the age of the material and the contact period; for example, first-draw water contains higher concentrations of lead or copper. Cross-connections with chemical storage containers, backflow from point-of-use equipment and cross-connections with non-potable supplies can lead to a range of contaminants entering drinking-water.

Where water is supplied directly to equipment in buildings, the potential for backflow into the mains network exists. This may be driven by high pressures generated in equipment connected to mains water supplies or by low pressures in the mains, but it can be prevented by fitting appropriate backflow prevention devices.

An additional problem not directly related to drinking-water is microbial growth (e.g. *Legionella* and mycobacteria), particularly in biofilms and within free-living amoebae on surfaces and in water-based devices that may lead to an inhalation hazard from aerosols produced by such devices. In large buildings, there is increased potential for growth of these bacteria in long water distribution systems, and maintenance of these systems needs particular attention. Growth of such bacteria can be controlled through measures including temperature control, maintaining a suitable disinfectant residual, minimizing periods of water stagnation and low flow rates, keeping systems clean and selecting plumbing materials that do not support microbial growth. Poor temperature control can occur in cold water systems through inadequate insulation and separation from hot water systems and in hot water systems in heating devices and storage containers, inappropriate location of tempering devices, long branch mains and dead ends (i.e. lengths of pipe, closed at one end, through which no water passes). Drinking-water temperatures should be maintained below 25 °C, where possible, and ideally below 20 °C. In building hot water systems, temperatures should be maintained above 60 °C in the water heaters and storages, and above 50–55 °C in all taps throughout the systems. However, maintaining hot water temperatures above 50 °C may represent a risk of scalding to young children, the elderly and other vulnerable groups. For further information on *Legionella* and mycobacteria in drinking-water, see [section 11.1](#) and the supporting documents *Legionella and the prevention of legionellosis* and *Pathogenic mycobacteria in water* ([Annex 1](#)).

Effective assessment of potential health hazards and risks requires documentation of the physical structure of water systems in buildings. This should be kept up to date and include hot and cold water networks, including materials used; point-of-entry treatment; point-of-use treatment, equipment and systems (e.g. for firefighting) connected to the drinking-water supply; and water-based devices supplied by the drinking-water system.

In undertaking an assessment of the building's distribution system, a range of specific issues must be taken into consideration that relate to ingress, introduction and proliferation of contaminants, including:

- the quality and management of external supplies;
- use of independent water supplies;
- intermittent supplies;
- pressure of water within the system;
- temperature of water (in both cold and hot water systems);

- integrity of storage tanks;
- areas subject to intermittent or seasonal use (e.g. hotels with seasonal occupancy, schools);
- cross-connections, especially in mixed systems;
- backflow prevention;
- system design to minimize dead/blind ends and other areas of potential stagnation;
- the use of materials and coatings approved for use with drinking-water.

The aim of a distribution system within a large building is to supply safe drinking-water at adequate pressure and flow. The quality of water entering building supplies will be ensured by a water utility or by the installation of point-of-entry devices typically managed by the building owner or operator. To maintain drinking-water quality, it is important to minimize transit times, low flows and low pressures.

Procedures should be established for repairs, renovations or extensions of systems to ensure that water safety is maintained, and all work, including changes to water systems, should be documented. Following work on the system, it would be appropriate to disinfect and flush.

Monitoring should focus on ensuring that control measures are working effectively. Where possible, this should include monitoring by maintenance personnel using field kits for parameters such as temperature, pH and disinfectant residuals. The frequency will vary depending on the size and use of the building, but it should be weekly in large buildings. Monitoring of drinking-water quality will be more frequent when the building is new or recently commissioned.

Independent surveillance is a desirable element in ensuring continued water safety within buildings and should be undertaken by the relevant health agency or other independent authority.

To ensure the safety of drinking-water within buildings, supportive activities of national regulatory agencies include:

- specific attention to application of codes of good practice (e.g. at commissioning and in contracting construction and rehabilitation);
- suitable education and training programmes for building owners and managers, engineers, plumbers and operators of water-based devices (e.g. cooling towers and evaporative condensers);
- regulation of the plumbing community and use of certified professionals;
- effective certification and use of materials and devices in the marketplace;
- codes of practice for design and operation of water-based devices.

For further guidance, see the supporting document *Water safety in buildings* ([Annex 1](#)).

6.11 Health-care facilities

Health-care facilities include hospitals, health centres and hospices, residential care, dental surgeries and dialysis units. Drinking-water in such facilities should be suitable for human consumption and for all usual domestic purposes, including personal

hygiene. However, it may not be suitable for all uses or for some patients, and further processing or treatment or other safeguards may be required.

Although microorganisms such as *Pseudomonas aeruginosa*, *Acinetobacter* and *Klebsiella* do not appear to represent a health concern through water consumption by the general population, including most patients in health-care facilities, they may be of concern for severely immunosuppressed persons, such as those with neutrophil counts below 500 per microlitre (see the supporting document *Heterotrophic plate counts and drinking-water safety* in [Annex 1](#), and [section 7.1.1](#) including [Tables 7.1](#) and [7.2](#) for a summary of potential pathogens of concern). These microorganisms also have the potential to cause infections of wounds, burns or eyes through exposure to contaminated water including from taps and sinks or bacteraemia through medical devices such as endoscopes and in dwelling catheters. Some can also cause pneumonia.

Health-care facilities may include environments that support the proliferation and dissemination of *Legionella* and mycobacteria (see [sections 6.10](#) and [11.1](#) and the supporting documents *Legionella and the prevention of legionellosis* and *Pathogenic mycobacteria in water*; [Annex 1](#)). Devices or installations that produce mists or sprays such as humidifiers, nebulizers and dental chair water lines can be sources of *Legionella*. Aspiration of contaminated water from birthing pools and chilled drinking-water from contaminated dispensers or from ice produced by contaminated ice machines can also be sources of *Legionella*. Ice produced by contaminated ice machines, processed water used for dialysis and aerosols from faulty heating-cooling units used during cardiothoracic surgery can be sources of mycobacteria.

Renal dialysis requires large volumes of water that is of higher quality than drinking-water. Water used for dialysis requires special processing to minimize the presence of microorganisms, endotoxins, cyanotoxins and chemical contaminants. There are special requirements regarding aluminium, which, in the past, has caused dialysis dementia, and dialysis patients are also sensitive to chloramines, which needs to be considered when chloramination is used to disinfect drinking-water supplies, particularly in areas where there are home dialysis patients. Of particular concern for bottle-fed infants is nitrate, which can cause methaemoglobinaemia after short-term exposure. Methaemoglobinaemia is complicated by the presence of microbial contamination and subsequent gastrointestinal infection, which can significantly increase the risk for bottle-fed infants (see [section 12.1](#) for more information).

All health-care facilities should have specific WSPs as part of their infection control programme. Such WSPs may be part of broader risk management efforts such as those supported through the *Water and Sanitation for Health Facility Improvement Tool* (see WHO & UNICEF, 2022). These plans should address issues such as water management quality and treatment requirements, regular monitoring, cleaning of specialized equipment and control of microbial growth in water systems and ancillary equipment. For further guidance, see the supporting document *Water safety in buildings* ([Annex 1](#)).

6.12 Safe drinking-water for travellers

The most common sources of exposure to disease-causing organisms for travellers are contaminated drinking-water and food that has been washed with contaminated

water. Diarrhoea is the most common symptom of waterborne infection, affecting a substantial proportion of travellers per year. Cases can occur even among people staying in high-quality resorts and hotels. In some parts of the world, tap or bottled water that has not been produced under proper conditions may not be safe, even if it is clear and colourless.

No single vaccine is capable of conferring general protection against infectious diarrhoea, which is caused by many different pathogens. It is important that travellers be aware of the possibility of illness and take appropriate steps to minimize the risks. Preventive measures while living or travelling in areas with questionable drinking-water quality include the following:

- Drink only bottled water or other beverages (carbonated beverages, pasteurized juices and milk) provided in sealed tamper-proof containers and bottled/canned by known manufacturers (preferably certified by responsible authorities). Hotel personnel or local hosts are often good sources of information about which local brands are safe.
- Drink water that has been treated effectively at point of use (e.g. through boiling, filtration or chemical disinfection as explained below) and stored in clean containers. Further information on storage considerations can be found in [section 7.3.2](#).
- Drink hot beverages such as coffee and tea that are made with boiled water and are kept hot and stored in clean containers.
- Avoid brushing teeth with water of questionable quality.
- Do not use ice unless it has been made from safe water.
- Avoid salads or other uncooked foods that may have been washed or prepared with unsafe water.

Water can be treated in small quantities by travellers to significantly improve its safety. Numerous simple treatment approaches and commercially available technologies are available to travellers to disinfect drinking-water for single-person or family use. The WHO International Scheme to Evaluate Household Water Treatment Technologies (<https://www.who.int/tools/international-scheme-to-evaluate-household-water-treatment-technologies/products-evaluated>) provides product-by-product assessments on treatment effectiveness. The Scheme highlights those ideal water treatment products that remove or inactivate all classes of pathogens and therefore are recommended for use by travellers. Water treatment products should be certified by a credible organization, and manufacturers' instructions should be followed carefully.

Bringing water to a rolling boil is the simplest and most effective way to kill all disease-causing pathogens, even in turbid water and at high altitudes. The hot water should be allowed to cool without the addition of ice. If the water is turbid and needs to be clarified for aesthetic reasons, this should be done before boiling.

Portable point-of-use membrane filtration products tested and rated effective against all pathogen classes (i.e. protozoa, bacteria and viruses) are available. Removal of viruses (20–100 nm) requires ultrafiltration, nanofiltration or reverse osmosis. Other point-of-use filtration devices have been tested and rated to remove protozoa and some bacteria, but not viruses. Filtration devices may require a pre-filter to remove suspended particles in order to avoid clogging the final filter.

Another treatment method that can be effective against all pathogen classes is point-of-use ultraviolet (UV) irradiation, in particular light-emitting diode devices. This approach to drinking-water disinfection is increasingly available for consumers.

Chemical disinfection of clear, non-turbid water is effective for killing bacteria and most viruses and some protozoa (but not, for example, *Cryptosporidium* oocysts). Certain chlorine-based or iodine-based compounds are most widely used for disinfection of drinking-water by travellers. Following chlorination or iodination, an activated carbon (charcoal) filter may be used to remove excess taste and odour from the water. The use of iodine is not recommended for use by infants, pregnant women, those with a history of thyroid disease, those with known hypersensitivity to iodine or those from areas of severe iodine deficiency. However, disinfection should not be compromised due to the public health significance of microbiologically unsafe water. Therefore if iodine is the only disinfectant available, use should be limited to as short a time as possible, and an alternative disinfectant sought. For members of the population with low risk, use of iodine disinfectants is recommended only for short-term use. Travellers intending to use iodine treatment daily for all water consumed for more than 3–4 weeks should use an effective post-disinfection iodine removal device (e.g. activated carbon) and consider consulting a physician beforehand. Silver is sometimes promoted as a disinfectant, but it is not recommended, as its efficacy is uncertain and it requires lengthy contact periods. (See the supporting document *Alternative drinking-water disinfectants*; [Annex 1](#), for more information.)

Suspended particles in water can reduce the effectiveness of chemical disinfectants and UV, and turbid water should be clarified by filtration or alternative methods before disinfection. Chemical products that combine clarification (coagulation and flocculation to remove particles) with chlorine disinfection are available.

Unless a treatment method is used that is effective against all pathogen classes, a combination of techniques is recommended. For example, filtration followed by chemical disinfection provides a multiple treatment barrier that can effectively remove protozoa in addition to killing bacteria and viruses. It is particularly important to protect against all pathogen classes for vulnerable subpopulations such as pregnant women, infants and immunocompromised people.

The treatment methods described here, with the exception of carbon filtration and reverse osmosis, will generally not reduce levels of most chemical contaminants in drinking-water. However, these are not usually of health concern in the short term.

Further information on household water treatment of microbial and chemical contaminants of water can be found in [sections 7.3.2](#) and [8.4.4](#), respectively. [Table 6.1](#) provides a summary of drinking-water disinfection methods that can be used by travellers.

Table 6.1 Drinking-water disinfection methods for use by travellers^a

Method	Recommendation	What it does	What it does <i>not</i> do
Boiling	Bring water to a rolling boil and allow to cool.	Kills all pathogens.	Does not remove turbidity/cloudiness. Does not provide residual chemical disinfectant, such as chlorine, to protect against re-contamination.
Portable filtering devices: 1. Ceramic filters 2. Carbon filters 3. Membrane filter (microfilter, ultrafilter, nanofilter and reverse osmosis) -type devices	Filter media pore size must be rated at 1 µm (absolute) or less to remove bacteria and protozoa; removal of viruses (20–100 nm) requires ultrafiltration, nanofiltration, or reverse osmosis. Note that water must be clear to prevent clogging of pores and seal integrity is critical to performance.	1 µm or less filter pore size will remove <i>Giardia</i> , <i>Cryptosporidium</i> and other protozoa. Approved reverse osmosis, ultra- and nanofiltration devices can remove almost all pathogens, including viruses. Some filters include a chemical disinfectant such as iodine or chlorine to kill microbes.	Most bacteria and viruses will not be removed by filters with a pore size larger than 1 µm. Microfilters may not remove viruses, especially from clear waters; additional treatment such as chemical disinfection or boiling/pasteurization may be needed to reduce viruses.
UV irradiation	Dose per manufacturer instructions.	Widely effective against bacteria, viruses, and protozoa; some viruses (e.g., adenovirus) are not inactivated as effectively.	Works only in non-turbid water. Does not provide residual chemical disinfectant, such as chlorine, to protect against re-contamination.
Chlorine compounds: 1. Unscented household bleach (sodium hypochlorite) 2. Sodium dichloroisocyanurate tablets 3. Calcium hypochlorite	For typical room temperature and water temperature of 25 °C, minimum contact time should be 30 min; increase contact time and/or dose for colder water—e.g. double time for each 10 °C less than 25 °C. Prepare according to instructions. Should be added to clear water or after settling or clarification to be most effective.	Effective for killing most bacteria and viruses. Longer contact time required to kill <i>Giardia</i> cysts, especially when water is cold.	Not effective against <i>Cryptosporidium</i> ; not as effective as iodine when using turbid water.

Table 6.1 (continued)

Method	Recommendation	What it does	What it does <i>not</i> do
Flocculant-chlorine tablets or sachets	Dose as per package directions.	Effective for killing or removing most waterborne pathogens (coagulant-flocculants partially remove <i>Cryptosporidium</i>).	Flocculated water must be decanted into a clean container, preferably through a clean fabric filter.
Iodine: 1. Tincture of iodine (2% solution) 2. Iodine (10% solution) 3. Iodine tablets 4. Iodinated (triiodide or pentaiodide) resins	25 °C—dose 5 drops per litre of water and use a minimum contact time of 30 min; increase contact time and/or dose for colder water. Prepare according to package instructions. Should be added to clear water or after settling or clarification to be most effective. <i>Caution:</i> Not recommended for pregnant women, for people with thyroid problems, infants and young children or for more than a few months' time unless a carbon filter or other effective process is included to remove excess iodine after iodine treatment.	Kills most pathogens. Longer contact time is required to kill <i>Giardia</i> cysts, especially when water is cold. Carbon filtration after an iodine resin will remove excess iodine from the water; replace the carbon filter regularly.	Not effective against <i>Cryptosporidium</i> .

^a Since performance also depends on the specifics of the method/device, it is preferable to select products whose performance has been independently tested and confirmed effective. For a list of products evaluated by the WHO International Scheme to Evaluate Household Water Treatment Technologies, including those that meet WHO performance criteria, see <https://www.who.int/tools/international-scheme-to-evaluate-household-water-treatment-technologies/products-evaluated>.

6.13 Aircraft and airports

The importance of water as a potential vehicle for infectious disease transmission on aircraft has been well documented. In general terms, the greatest microbial risks are those associated with ingestion of water that is contaminated with human and animal excreta. If the source of water used to replenish aircraft supplies is contaminated and adequate precautions are not taken, disease can be spread through the aircraft water if it is used for drinking or tooth cleaning. It is thus imperative that airports comply with the International Health Regulations (2005) and be provided with potable drinking-water from a source approved by the appropriate regulatory agency (WHO, 2016). Airports usually have special arrangements for managing water after it has entered the airport.

A potable water source is not a safeguard if the water is subsequently contaminated during transfer, storage or distribution in aircraft. A WSP covering water management within airports from receipt of the water through to its transfer to the aircraft (e.g. by water servicing vehicles or water bowsers), complemented by measures to ensure that water quality is maintained on the aircraft (e.g. safe materials and good practices in design, construction, operation and maintenance of aircraft systems), provides a framework for water safety in aviation.

In undertaking an assessment of the general airport/aircraft water distribution system, a range of specific issues must be taken into consideration, including:

- quality of source water and the need for additional treatment;
- design and construction of airport storage tanks and pipes;
- design and construction of water servicing vehicles;
- use of materials and fittings approved for contact with drinking-water at all stages;
- water loading techniques;
- any treatment systems on aircraft (e.g. ultraviolet disinfection);
- maintenance of on-board plumbing;
- prevention of cross-connections, including backflow prevention.

The airport authority has responsibility for safe drinking-water supply, including operational monitoring, until water is transferred to the aircraft operator. The primary emphasis of monitoring is to ensure that management processes are operating efficiently—for example, the source water quality is not compromised; all parts of the system, including hydrants, hoses and bowsers, are clean and in good repair; backflow prevention is in place; and any filters are clean. In addition, the system should be disinfected and flushed after maintenance or repairs, and the microbiological quality of the water should be checked, preferably before the system is returned to service.

Transfer of water into the aircraft and the aircraft drinking-water system also has the potential to introduce hazards, even if the water is of good quality up to this point. It is therefore important that staff involved be properly trained and understand the reasons for the precautions to be taken and the care required in preventing contamination. The precautions described in previous sections regarding transfer of drinking-water from a piped supply or from bowsers and tankers are essential, including maintaining the cleanliness of vehicles and transfer points. There is a significant

potential for aviation fuel to contaminate the system, and only small quantities of low molecular weight hydrocarbons can cause the water to be unacceptable. In addition, staff employed in drinking-water supply must not be engaged in activities related to aircraft toilet servicing without first taking all necessary precautions (e.g. thorough hand washing, change of outer garments). All of these requirements and procedures should be properly documented as part of the WSP for the airport water transfer system and should be made clear to airlines using the airport to ensure that they play their part as key stakeholders.

Independent surveillance is an important part of the WSP, because circumstances and equipment or staff may change, and the weakening of barriers or the introduction of new risks may not be noticed. This would include initial review and approval of the WSP, periodic review and direct assessment of the provisions and operation of the WSP, paying specific attention to the aircraft industry's codes of practice, the supporting document *Guide to hygiene and sanitation in aviation* ([Annex 1](#)) and airport health or airline regulations. It is also important that the response to any incident is recorded and reviewed and any lessons learnt incorporated into the WSP.

6.14 Ships

The importance of water as a vehicle for infectious disease transmission on ships has been clearly documented. In general terms, the greatest microbial risks are associated with ingestion of water that is contaminated with human and animal excreta. However, chemical contamination could also occur on ships as a result of contaminated bulk water being brought aboard in port, cross-connections on board or improper on-board treatment. The supporting document *Guide to ship sanitation* ([Annex 1](#)) describes the factors that can be encountered during water treatment, transfer, production, storage or distribution in ships and specific features of the organization of the supply and the regulatory framework. To this end, it is vital that all staff responsible for working with the potable water system are properly trained.

The organization of water supply systems covering shore facilities and ships differs considerably from conventional water transfer on land but is similar to that for airports. The port authority has responsibility for providing safe potable water for loading onto vessels. If water is suspected to have come from an unsafe source, the ship's master may have to decide if any additional treatment (e.g. hyperchlorination or filtration) is necessary. When treatment on board or prior to boarding is necessary, the treatment selected should be that which is best suited to the water and which is most easily operated and maintained by the ship's officers and crew.

Water is delivered to ships by hoses or transferred to the ship via water boats or barges. The transfer from shore to ship is a potential source of microbial or chemical contamination. In addition to shore-to-ship transfer of water and bulk storage on board ship, many ships use desalination (see [section 6.5](#)) to produce their own drinking-water.

In contrast to a shore facility, plumbing aboard ships consists of numerous piping systems carrying potable water, seawater, sewage and fuel and fitted into a relatively confined space. Piping systems are normally extensive and complex, making them difficult to inspect, repair and maintain. A number of waterborne outbreaks on ships

have been caused by contamination of potable water after it had been loaded onto the ship—for example, by sewage or bilge water when the water storage systems were not adequately designed and constructed. Potable water should be stored in one or more tanks that are constructed, located and protected so as to be safe against contamination. Potable water lines should be protected and located so that they will not be submerged in bilge water or pass through tanks storing non-potable liquids. It is important to design the system to prevent deterioration of water quality during distribution by minimizing stagnation and dead ends and to take into account ship movement, which increases the possibility of surge and backflow.

An overall assessment of the operation of the ship's water supply should be made, for which the final responsibility lies with the ship's master, who must ensure that all of the management processes in place are functioning efficiently. An important part of this process is ensuring that those crew who are responsible for the fresh drinking-water supply are properly trained and receive refresher training as appropriate. In developing a WSP and ensuring that the system is capable of supplying safe water, the following need to be considered:

- quality of source water if this is from a shore-based source along with the equipment and method of transfer from shore to ship;
- desalination equipment and processes where these are used, taking into consideration the points raised in [section 6.5](#);
- design and construction of storage tanks and pipework, including the use of approved materials and chemicals and clear colour coding of pipes for different purposes;
- minimization of dead ends and areas of stagnation, which may be managed by periodic flushing;
- filtration systems and other treatment systems on board the ship, including disinfection and delivery of residual disinfection;
- prevention of cross-connections and presence of working backflow prevention devices;
- maintenance of adequate water pressure within the system;
- presence of a disinfectant residual throughout the system.

The system needs to be checked regularly for cleanliness and repair, and parameters such as pH and disinfectant residual need to be checked daily. Where possible, checks on microbiological quality such as plate counts and faecal coliforms, even if only in port, help to ensure that the supply continues to deliver safe water. There also need to be suitable procedures in place to ensure safety after maintenance or repair, including specific disinfection of the system or the affected zone. Any indication of a problem, such as illness or taste or odour problems, should be immediately investigated and the system corrected if it is shown to be the source. In confined communities such as on ships, person-to-person spread of infectious disease is a major issue. Someone who has been working on the latrines and sanitation system on ships should not transfer to work on the drinking-water system without thorough hand washing and a change of outer clothing.

Independent surveillance is a desirable element in ensuring drinking-water safety on ships. This implies that there will be periodic audit and direct assessment and the review and approval of the WSP. Specific attention should be given to the shipping industry's codes of practice, the supporting document *Guide to ship sanitation* ([Annex 1](#)) and port health and shipping regulations. Independent surveillance should also include ensuring that any specific incidents that affect or might have affected water quality have been properly investigated and the lessons to be learnt are incorporated in the WSP.

6.15 Packaged drinking-water

Bottled water and water in containers are widely available in both industrialized and developing countries. Consumers purchase packaged drinking-water for reasons such as taste, convenience or fashion, but safety and potential health benefits are also important considerations.

Water is packaged for consumption in a range of vessels, including cans, laminated boxes and plastic bags, but it is most commonly supplied in glass or plastic bottles. Bottled water also comes in various sizes, from single servings to large carbuys holding up to 80 litres. Control of the quality of materials, containers and closures for bottled water is of special concern. Ozone is sometimes used for final disinfection prior to bottling because it does not impart a taste to the water. If the water contains naturally occurring bromide, this can lead to the formation of bromate unless care is taken to minimize its formation.

The Guidelines provide a basis for derivation of standards for all packaged waters. As with other sources of drinking-water, safety is pursued through a combination of safety management and end product quality standards and testing and is more readily achievable because batches can be held until results are available. The international framework for packaged water regulation is provided by the Codex Alimentarius Commission of the World Health Organization and the Food and Agriculture Organization of the United Nations.

The Codex Alimentarius Commission has developed a *Standard for natural mineral waters*—which describes the product and its compositional and quality factors, including prescribed treatments, limits for certain chemicals, hygiene, packaging and labelling—and an associated Code of Practice. It has also developed a *General standard for bottled/package drinking waters* and an associated Code of Practice, to cover packaged drinking-water other than natural mineral waters. Both relevant Codex standards refer directly to these Guidelines; the Codex standards for bottled/package water are directly equivalent to the guideline values established in these Guidelines. Under the Codex *Standard for natural mineral waters* and associated Code of Practice, natural mineral waters must conform to strict requirements, including collection and bottling without further treatment from a natural source, such as a spring or well. In comparison, the Codex *General standard for bottled/package drinking waters* and associated Code of Practice, include waters from other sources, in addition to springs and wells, and treatment to improve their safety and quality. The distinctions between these standards are especially relevant in regions where natural mineral waters have a long cultural history. For further information on

these standards and their associated Codes of Practice, see FAO/WHO (2001, 2011, 2019a, b).

The Codex Alimentarius Commission's *Code of hygienic practice for collecting, processing and marketing of natural mineral waters* provides guidance on a range of good manufacturing practices and provides a generic WSP applied to packaged drinking-water.

Some consumers believe that certain natural mineral waters have medicinal properties or offer other health benefits. Some such waters have higher mineral content, sometimes significantly higher than concentrations normally accepted in drinking-water. They often have a long tradition of use and are often accepted on the basis that they are considered foods rather than drinking-water per se. Although certain mineral waters may be useful in providing essential micronutrients, such as calcium and magnesium, these Guidelines do not make recommendations regarding minimum concentrations of essential elements because of the uncertainties surrounding mineral nutrition from drinking-water. Packaged waters with very low mineral content, such as distilled or demineralized waters, are also consumed. There is insufficient scientific information on the benefits or hazards of long-term consumption of very low mineral waters to allow any recommendations to be made (WHO, 2005; see also the supporting document *Calcium and magnesium in drinking-water*; [Annex 1](#)).

Another form of packaged water is ice that is intended for adding to drinks and which may come into contact with food to be eaten without cooking. Ice prepared and sold in this manner should be treated the same as any packaged water for potable use.

6.16 Food production and processing

The diverse uses of water in food production and processing have different water quality requirements. Uses include irrigation and livestock watering, as an ingredient or to wash or “refresh” foods (e.g. misting of salad vegetables in grocery stores, cleaning fish that are filleted), and in which contact between the water and foodstuff should be minimal (e.g. water for heating, cooling or cleaning).

The quality of water defined by the Guidelines is such that it is suitable for many processes used in the food industry. Some processes, however, have special water quality requirements to secure the desired characteristics of the product, and the Guidelines do not necessarily guarantee that such special requirements are met.

Poor quality drinking-water may have a severe impact in food processing and potentially on public health. The consequences of a failure to use water of suitable quality in food processing will depend on the use of the water and the subsequent processing of potentially contaminated materials. Variations in water quality that may be tolerated occasionally in drinking-water supply may be unacceptable for some uses in the food industry. These variations may result in a significant financial impact on food production—for example, through product recalls.

Water of a different quality from water used for drinking or as an ingredient may be fit for some purposes, provided that it does not compromise the safety of the final product for the consumer. A risk-based approach should be used, and the

fitness of the water for the intended purpose should be assessed. For guidance on determining whether water is “fit for purpose” for sourcing, use and reuse in different food production and processing scenarios, see FAO/WHO (2019c, 2021, 2023a, b, c, 2025) including the *Codex Guidelines for the safe use and reuse of water in food production and processing* (FAO/WHO, 2023a).

To reduce microbial contamination, specific treatments (e.g. heat, chemical disinfectants, preservatives) capable of minimizing or preventing proliferation of a range of pathogenic organisms of public health concern may be used in food processing. The effect of these treatments should be taken into account when assessing the impacts of deterioration in drinking-water quality on a food production or processing facility. For example, water that is used in canning will usually be heated to a temperature that is at least equivalent to pasteurization.

Information on deterioration of the microbial or chemical quality of a drinking-water supply should be promptly communicated to food and beverage production facilities.

For further information on disinfection of water for use in food production and processing with chlorine-containing disinfectants, see FAO/WHO (2009).

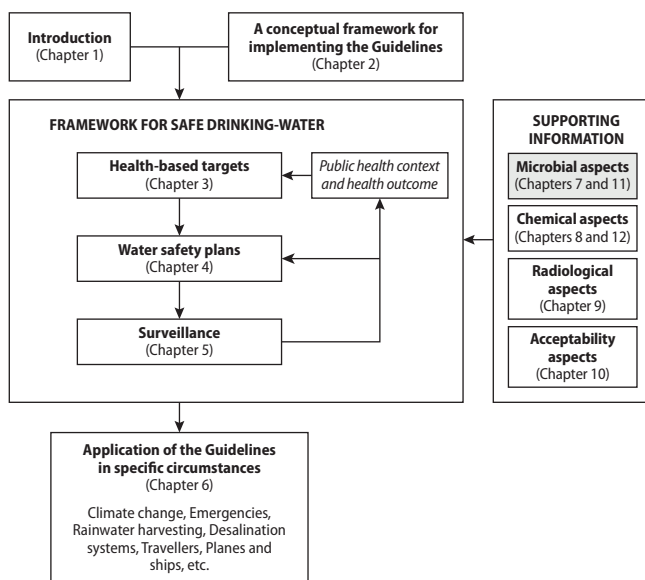
7

Microbial aspects

The greatest risk to public health from microbes in water is associated with consumption of drinking-water that is contaminated with human and animal excreta, although other sources and routes of exposure may also be significant.

Waterborne outbreaks have been associated with inadequate treatment of water supplies and unsatisfactory management of drinking-water distribution. For example, in distribution systems, such outbreaks have been linked to cross-connections, contamination during storage, low water pressure and intermittent supply. Waterborne outbreaks are preventable if an integrated risk management framework based on a multiple-barrier approach from catchment to consumer is applied. Implementing an integrated risk management framework to keep the water safe from contamination in distribution systems includes the protection of water sources, the proper selection and operation of drinking-water treatment processes, and the correct management of risks within the distribution systems (for further information, see the supporting document *Water safety in distribution systems*; [Annex 1](#)).

This chapter focuses on organisms for which there is evidence, from outbreak studies or from prospective studies in non-outbreak situations, of diseases being caused by ingestion of drinking-water, inhalation of water droplets or dermal contact



with drinking-water and their prevention and control. For the purpose of the Guidelines, these routes are considered waterborne.

[Chapter 11](#) (Microbial fact sheets) provides additional detailed information on individual waterborne pathogens, as well as on indicator microorganisms.

7.1 Microbial hazards associated with drinking-water

Infectious diseases caused by pathogenic bacteria, viruses and parasites (e.g. protozoa and helminths) are the most common and widespread health risk associated with drinking-water. The public health burden is determined by the severity and incidence of the illnesses associated with pathogens, their infectivity and the population exposed. In vulnerable subpopulations, disease outcome may be more severe.

Breakdown in water supply safety (source, treatment and distribution) may lead to large-scale contamination and potentially to detectable disease outbreaks. In some cases, low-level, potentially repeated contamination may lead to significant sporadic disease, but public health surveillance is unlikely to identify contaminated drinking-water as the source.

Infectious diseases caused by pathogenic bacteria, viruses, protozoa and helminths are the most common and widespread health risk associated with drinking-water.

Waterborne pathogens have several properties that distinguish them from other drinking-water contaminants:

- Pathogens can cause acute and also chronic health effects.
- Some pathogens can grow in the environment.
- Pathogens are discrete.
- Pathogens are often aggregated or adherent to suspended solids in water, and pathogen concentrations vary in time, so that the likelihood of acquiring an infective dose cannot be predicted from their average concentration in water.
- Exposure to a pathogen resulting in disease depends upon the dose, invasiveness and virulence of the pathogen, as well as the immune status of the individual.
- If infection is established, pathogens multiply in their host.
- Certain waterborne pathogens are also able to multiply in food, beverages or warm water systems, perpetuating or even increasing the likelihood of infection.
- Unlike many chemical agents, pathogens do not exhibit a cumulative effect.

Quantitative microbial risk assessment (QMRA), a mathematical framework for evaluating infectious risks from human pathogens, can assist in understanding and managing waterborne microbial hazards, especially those associated with sporadic disease.

7.1.1 Waterborne infections

The pathogens that may be transmitted through contaminated drinking-water are diverse in characteristics, behaviour and resistance. [Table 7.1](#) provides general information on pathogens that are of relevance for drinking-water supply management. Waterborne transmission of most of the pathogens listed has been confirmed by epi-

demiological studies. Part of the demonstration of pathogenicity involves reproducing the disease in suitable hosts. Experimental studies in which healthy adult volunteers are exposed to known numbers of pathogens provide information, but these data are applicable to only a part of the exposed population; extrapolation to more vulnerable subpopulations is an issue that remains to be studied in more detail. [Table 7.2](#) provides information on organisms that have been suggested to be transmitted through drinking-water but where evidence is weak or lacking. The spectrum of pathogens may change as a result of host, pathogen and environmental changes such as fluctuations in human and animal populations, reuse of wastewater, changes in lifestyles and medical interventions, population movement and travel, selective pressures for new pathogens and mutants or recombinations of existing pathogens. The immunity of individuals also varies considerably, whether acquired by contact with a pathogen or influenced by such factors as age, sex, state of health and living conditions.

For pathogens transmitted by the faecal–oral route, drinking-water is only one vehicle of transmission. Contamination of food, hands, utensils and clothing can also play a role, particularly when domestic sanitation and hygiene are poor. Improvements in the quality and availability of water, excreta management and general hygiene are all important in reducing faecal–oral disease transmission.

Microbial drinking-water safety is not related only to faecal contamination. Some organisms grow in piped water distribution systems (e.g. *Legionella*), whereas others occur in source waters (e.g. guinea worm [*Dracunculus medinensis*]) and may cause outbreaks and individual cases. Some other microbes (e.g. toxic cyanobacteria) require specific management approaches, which are covered elsewhere in these Guidelines (see [section 11.5](#)).

Although consumption of contaminated drinking-water represents the greatest risk, other routes of transmission can also lead to disease, with some pathogens transmitted by multiple routes (e.g. norovirus and non-tuberculosis mycobacteria) ([Figure 7.1](#)). Certain serious illnesses result from exposure to drinking-water through routes other than ingestion. These include legionellosis caused by inhalation of aerosols containing *Legionella* spp., which can multiply in warm waters including hot water systems in buildings, and illnesses caused by the amoebae *Naegleria fowleri* (primary amoebic meningoencephalitis), which are transmitted by the intranasal route and can also grow in warm water environments. Transmission of *Acanthamoeba* is primarily caused by cleaning or storing contact lenses with drinking-water, which can cause acanthamoebic keratitis.

Schistosomiasis (bilharzia) is a major parasitic disease of tropical and subtropical regions that is transmitted when the larval stage (cercariae), which is released by infected aquatic snails, penetrates the skin. It is primarily spread by contact with contaminated surface water. Ready availability of safe drinking-water contributes to disease prevention by reducing the need for contact with contaminated water resources—for example, when collecting water to carry to the home or when using water for bathing or laundry.

It is conceivable that unsafe drinking-water contaminated with soil or faeces could act as a carrier of other infectious parasites, such as *Balantidium coli* (balantidiasis) and certain helminths (species of *Fasciolopsis*, *Echinococcus*, *Spirometra*, *Ascaris*, *Trichuris*,

Table 7.1 Pathogens transmitted through drinking-water^a

Pathogen	Type species/ genus/group ^b	Health significance ^c	Strength of evidence ^d	Persistence in water supplies ^e	Resistance to chlorine ^f	Relative infectivity ^g	Important animal source
Bacteria							
<i>Burkholderia</i>	<i>B. pseudomallei</i>	Moderate	Moderate	Long, can multiply	Low	Depends on host susceptibility	No, although zoonotic transmission has been reported
<i>Campylobacter</i>	<i>C. jejuni</i>	High	Strong	Short	Low	Moderate to high	Yes
<i>Escherichia coli</i> – diarrhoeagenic ^h		High	Strong	Moderate	Low	Low	No, although zoonotic transmission has been reported
<i>E. coli</i> – enterohaemorrhagic	<i>E. coli</i> O157	High	Strong	Moderate	Low	Moderate	Yes
<i>Francisella</i>	<i>F. tularensis</i>	Moderate	Strong	Moderate, may be longer in biofilms and amoebae	Moderate	Low via ingestion, high via other routes	Yes
<i>Helicobacter</i>	<i>H. pylori</i>	Moderate	Moderate	Short, may be longer in biofilms and amoebae	Low	Unknown	No
<i>Legionella</i>	<i>L. pneumophila</i>	High	Strong, transmission primarily linked to inhalation of aerosols associated with drinking-water	Long, can multiply in biofilms and amoebae	Moderate	Depends on host susceptibility	No
Mycobacteria (non- tuberculous)	<i>Mycobacterium avium</i> complex	Low in community settings but higher in health care facilities	Moderate	Long, can multiply in biofilms and amoebae	High	Depends on host susceptibility	No

Table 7.1 (continued)

Pathogen	Type species/ genus/group ^b	Health significance ^c	Strength of evidence ^d	Persistence in water supplies ^e	Resistance to chlorine ^f	Relative infectivity ^g	Important animal source
<i>Salmonella</i> – non-typhoidal	<i>S. Enteritidis</i> <i>S. Typhimurium</i>	High	Strong	Moderate	Low	Moderate	Yes
<i>Salmonella</i> – typhoidal	<i>S. Paratyphi</i> <i>S. Typhi</i>	High	Strong	Short	Low	Low	No
<i>Shigella</i>	<i>S. flexneri</i> <i>S. dysenteriae</i>	High	Strong	Moderate	Low	Moderate to high	No
<i>Vibrio</i>	<i>V. cholerae</i> O1 and O139	High	Strong	Moderate to long ⁱ	Low to moderate	Low	No
<i>Yersinia</i>	<i>Y. enterocolitica</i>	Moderate	Moderate	Long, can multiply	Low	Likely low	Yes
Viruses							
Adenoviridae	Human adenoviruses (<i>Mastadenovirus</i>)	Moderate	Moderate	Long	Low	High	No, although zoonotic transmission may be possible
Caliciviridae	Noroviruses, (genogroup 2 type 4, GII.4)	High	Strong	Long	Low	High	No, although zoonotic transmission may be possible
	Sapoviruses	Moderate	Moderate	Likely long	Low	Likely high	No, although zoonotic transmission may be possible
Hepeviridae	Hepatitis E virus (<i>Paslahepevirus balayani</i>)	High	Strong	Likely long	Low to moderate	Likely high	Potentially
Picornaviridae	Enteroviruses	High	Moderate	Long	Low to moderate	Likely high Likely high	No No
	<i>Hepatovirus</i> (Hepatitis A virus)	High	Strong	Long	Moderate		
Reoviridae	<i>Rotavirus</i> type A	High	Strong	Long	Moderate	High	No, although zoonotic transmission may be possible

Table 7.1 (continued)

Pathogen	Type species/ genus/group^b	Health significance^c	Strength of evidence^d	Persistence in water supplies^e	Resistance to chlorine^f	Relative infectivity^g	Important animal source
Protozoa							
<i>Acanthamoeba</i>	<i>A. castellanii</i>	Moderate	Strong, transmission linked primarily to cleaning contact lenses with drinking- water	Long, can multiply	Moderate to high	Unknown	No
<i>Cryptosporidium</i>	<i>C. hominis</i> <i>C. parvum</i>	High	Strong	Long	High	High	Yes
<i>Cyclospora</i>	<i>C. cayetanensis</i>	High	Moderate	Long	Likely high	Likely high	No
<i>Entamoeba</i>	<i>E. histolytica</i>	High	Strong	Likely moderate to long	Moderate	Likely high	No
<i>Giardia</i>	<i>G. duodenalis</i> (<i>G. intestinalis</i> / <i>lamblia</i>)	High	Strong	Moderate to long	High	High	Yes
<i>Naegleria</i>	<i>N. fowleri</i>	High	Strong, transmission linked to recreational use or nasal ablution with tap water	Long, can multiply	Moderate	Unknown	No
<i>Toxoplasma</i>	<i>T. gondii</i>	High	Strong	Long	High	Likely high	Yes

Table 7.1 (continued)

Pathogen	Type species/ genus/group ^b	Health significance ^c	Strength of evidence ^d	Persistence in water supplies ^e	Resistance to chlorine ^f	Relative infectivity ^g	Important animal source
Helminths							
<i>Dracunculus</i>	<i>D. medinensis</i>	High	Strong, transmission linked to ingestion of surface water	Long ⁱ	Likely low to moderate	Unknown	No
<i>Fasciola</i>	<i>F. hepatica</i> <i>F. gigantica</i>	High	Moderate, transmission linked to ingestion of surface water	Long ⁱ	Likely high	Likely high	Yes

^a This table contains pathogens for which there is some evidence of community transmission through drinking-water. More information on these and other pathogens is presented in [chapter 11](#).

^b The type species listed (e.g. *L. pneumophila*) are those most commonly linked to waterborne transmission, but other species may also cause disease.

^c Health significance relates to the incidence and severity of disease, including association with drinking-water outbreaks.

^d Strength of evidence for transmission through drinking-water: strong, pathogen detected in water supply AND evidence from analytical epidemiological study (e.g. case-control study) supports hypothesis OR descriptive epidemiology supports hypothesis and excludes alternate explanations) OR known water quality failure AND evidence from analytical epidemiological study supports hypothesis; moderate, known water quality failure AND descriptive epidemiology supports hypothesis and excludes alternate explanations OR evidence from analytical epidemiological study (e.g. case-control study) supports hypothesis OR pathogen detected in water supply.

^e Detection period for viable stage in water at 20 °C: short, up to 1 week; moderate, 1 week to 1 month; long, over 1 month. In general, pathogens tend to persist longer at colder temperatures e.g. 4 °C; this longer survival can increase the risk of waterborne transmission and illness.

^f Within pathogen species and groups, there are likely to be variations in resistance, which could be further impacted by characteristics of the water supply and operating conditions. Resistance is based on 99% inactivation at 20 °C where, generally, low represents a Ct_{99} of < 1 min.mg/L, moderate 1–30 min.mg/L and high > 30 min.mg/L (where C = the concentration of free chlorine in mg/L and t = contact time in minutes) under the following conditions: the infective stage is freely suspended in water treated at conventional doses and contact times, and the pH is between 7 and 8. Although some microorganisms are moderately resistant to disinfection with chlorine, it can be effective provided adequate doses and contact times are applied. It should be noted that organisms that survive and grow in biofilms and within free-living protozoa in biofilms, such as *Legionella* and mycobacteria, will be protected from the disinfection process including chlorination.

^g From experiments with human volunteers, from epidemiological evidence and from experimental animal studies. High means infective doses can be 1–10² organisms or particles, moderate 10²–10⁴ and low > 10⁴.

^h Includes enteropathogenic, enterotoxigenic, enteroinvasive, diffusely adherent and enteroaggregative.

ⁱ *Vibrio cholerae*, *Dracunculus* and *Fasciola* may be endemic in certain regions and persist for long periods in association with copepods and other aquatic organisms.

Toxocara, *Necator*, *Ancylostoma*, *Strongyloides* and *Taenia*, but evidence is limited). However, for most of these, the normal mode of transmission is ingestion of faecally contaminated soil or eggs in food contaminated with faeces (in the case of *Taenia spp.*, ingestion of the larval cysticercus stage in uncooked pork) rather than ingestion of contaminated drinking-water.

There are other pathogens naturally present in the environment for which waterborne transmission evidence is limited or lacking, but which have the potential to cause disease in vulnerable subpopulations such as: the elderly or the very young, patients with burns or extensive wounds, those undergoing immunosuppressive therapy or those with acquired immunodeficiency syndrome (AIDS). Examples of such agents are *Pseudomonas aeruginosa* species of *Acinetobacter*, *Aeromonas*, *Klebsiella*, *Staphylococcus* and *Tsukamurella*. In addition, *Pseudomonas* and *Acinetobacter* have both been associated with a limited number of waterborne or water-related outbreaks in health-care facilities. More information on these organisms is listed in [Table 7.2](#) (and described in more detail in [chapter 11](#)).

Most of the human pathogens listed in [Table 7.1](#) (which are also described in more detail in [chapter 11](#)) are distributed worldwide; some, however, such as those causing outbreaks of cholera or guinea worm disease, are regional. *Dracunculus medinensis* has been nearly eradicated and cases have been restricted to a few African countries.

It is possible that there are pathogens not shown in [Table 7.1](#) that are also transmitted by drinking-water as pathogens continue to emerge or re-emerge.

7.1.2 Emerging issues

A number of developments are subsumed under the concept of “emerging issues” in drinking-water. Global changes, such as human development, population growth and movement and climate change (see [section 6.1](#)), exert pressures on the quality and quantity of water resources that may influence waterborne disease risks. Between 1972 and 1999, 35 new agents of disease were discovered, and many more have re-emerged after long periods of inactivity or are expanding into areas where they have not previously been reported (WHO, 2003). Zoonotic pathogens make up 75% of the emerging pathogens and are of increasing concern for human health, along with pathogens with strictly human-to-human transmission. Zoonotic pathogens pose the greatest challenges to ensuring the safety of drinking-water and ambient water, now and in the future (see the supporting document *Waterborne zoonoses*; [Annex 1](#)). For each emerging pathogen, whether zoonotic or not, it should be considered whether it can be transmitted through water and, if so, which prevention and control measures can be suggested to minimize this risk. Waterborne pathogens, emerging or not, should be addressed through water safety planning that includes protection of source waters from contamination by faecal wastes, application of adequate treatment processes and protection of drinking-water quality during distribution and storage.

Table 7.2 Microorganisms for which transmission through drinking-water has been suggested but for which evidence is weak or lacking^a

Microorganism	Type species/ genus/group ^b	Waterborne transmission evidence (or epidemiological features)	Presence and behaviour in water supplies ^c	Resistance to chlorine ^d
Bacteria				
<i>Acinetobacter</i>	<i>A. baumannii</i> particularly multi- drug resistant strains	Weak, possible issue in health-care facilities (non-gastrointestinal)	Common, persistence long and can multiply particularly in biofilms	Moderate
<i>Aeromonas</i>	<i>A. caviae</i> <i>A. dhakensis</i> <i>A. veronii</i>	Weak	Common, persistence long and can multiply, particularly in biofilms	Low
<i>Cronobacter</i>	<i>C. sakazakii</i>	None, infection associated primarily with infant formula	Detected, although rarely in piped supplies; persistence likely long	Likely low
<i>Klebsiella</i>	<i>K. pneumoniae</i>	None, possible issue in health-care facilities (non-gastrointestinal)	Detected, persistence long and can multiply, particularly in biofilms	Low
<i>Leptospira</i>	<i>L. interrogans</i>	Weak, infection caused primarily by contact with contaminated surface water associated with flooding	Persistence long	Low
<i>Pseudomonas</i>	<i>P. aeruginosa</i>	Can be an issue in health-care facilities (non-gastrointestinal) ^e	Common, persistence long and can multiply, particularly in biofilms	Low to moderate
<i>Staphylococcus</i>	<i>S. aureus</i> , particularly methicillin- resistant strains	None, possible issue in health-care facilities (non-gastrointestinal)	Common, persistence long and can multiply, particularly in biofilms	Low
<i>Tsukamurella</i>	<i>T. paurometabola</i>	None, possible issue in health-care facilities (non-gastrointestinal)	Detected, persistence likely long and can multiply, particularly in biofilms	Likely moderate

Table 7.2 (continued)

Microorganism	Type species/ genus/group^b	Waterborne transmission evidence (or epidemiological features)	Presence and behaviour in water supplies^c	Resistance to chlorine^d
Viruses				
Astroviridae	Astroviruses	Weak, although strong evidence linking transmission with inadequate sanitation	Detected, persistence long	Moderate
Coronaviridae	Severe acute respiratory syndrome (SARS) coronaviruses	None, drinking-water is an unlikely transmission route, although one outbreak of SARS and one of COVID-19 possibly linked to inhalation of wastewater droplets	Unlikely	Low
Filoviridae	Ebola virus Marburg virus	None, drinking-water is an unlikely transmission route	Unlikely	Low
Orthomyxoviridae	Influenza viruses	None	Unlikely but has been detected in wastewater and surface water	Low
Picornaviridae	Aichiviruses	Weak	Detected, persistence likely long	Likely moderate
	Parechoviruses	None	Not detected but has been detected in faecal waste, wastewater and surface water	Likely moderate
Protozoa				
<i>Balantiodos</i>	<i>B. coli</i>	Weak, one outbreak linked to contaminated surface water following a typhoon	Detected, persistence long	Likely high
<i>Blastocystis</i>	–	Weak	Detected, persistence long	High
<i>Cytoisopora</i>	<i>C. belli</i>	Weak	Detected, persistence likely long	Likely high
Microsporidia	–	Weak	Detected, persistence long	Moderate
Helminths				
Free-living nematodes (other than <i>Dracunculus medinensis</i>)	–	Not infectious but may harbour pathogenic microorganisms	Detected and can multiply	High

Table 7.2 (continued)

Microorganism	Type species/ genus/group ^b	Waterborne transmission evidence (or epidemiological features)	Presence and behaviour in water supplies ^c	Resistance to chlorine ^d
<i>Schistosoma</i>	<i>S. mansoni</i> <i>S. japonicum</i> <i>S. haematobium</i>	None, primarily spread by contact with contaminated surface water in communities with inadequate access to safe drinking-water	Life cycle involves humans or other animals and snails	Moderate

^a This table contains pathogens for which community transmission through drinking-water has been suggested, but for which evidence is weak or lacking. Except for *Pseudomonas*, transmission is also weak or lacking in water systems in health-care facilities. More information on these and other pathogens is presented in [chapter 11](#).

^b The type species listed (e.g. *K. pneumoniae*) are those most commonly linked to infections, but other species may also cause disease.

^c Detection period for viable stage in water at 20 °C: short, up to 1 week; moderate, 1 week to 1 month; long, over 1 month.

^d Within pathogen species and groups, there are likely to be variations in resistance, which could be further impacted by characteristics of the water supply and operating conditions. Resistance is based on 99% inactivation at 20 °C where, generally, low represents a Ct_{99} of < 1 min.mg/L, moderate 1–30 min.mg/L and high > 30 min.mg/L (where C = the concentration of free chlorine in mg/L and t = contact time in minutes) under the following conditions: the infective stage is freely suspended in water treated at conventional doses and contact times, and the pH is between 7 and 8. Although some microorganisms are moderately resistant to disinfection with chlorine, it can be effective provided adequate doses and contact times are applied. It should be noted that organisms that survive and grow in biofilms and within free-living protozoa in biofilms, such as *Pseudomonas aeruginosa*, will be protected from disinfection processes including chlorination.

^e A limited number of hospital outbreaks have been reported due to use of contaminated water in patient care. Non-touch taps and splashing from sink drains associated with water systems in health-care facilities have also been identified as probable risk factors for transmission.

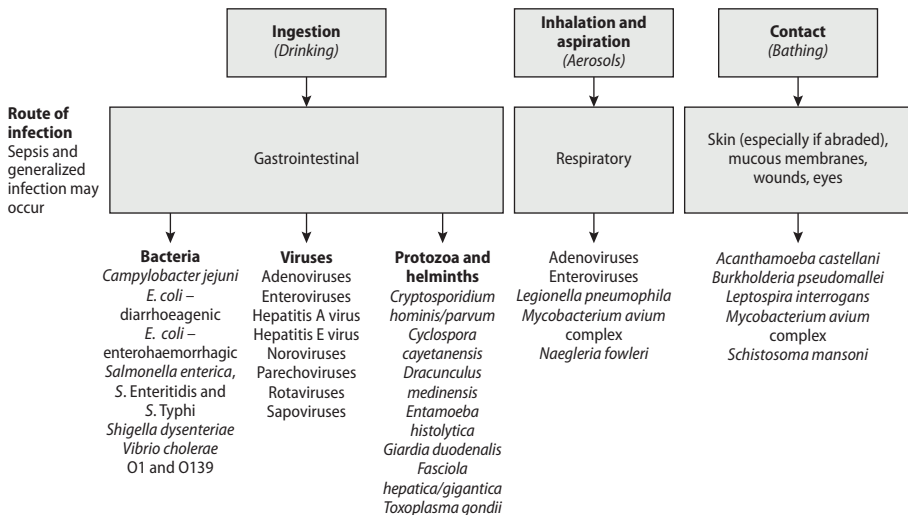


Figure 7.1 Transmission pathways for and examples of water-related pathogens

7.1.3 Persistence and growth in water

Waterborne pathogens, such as *Legionella*, may grow in water, whereas other host-dependent waterborne pathogens, such as noroviruses and *Cryptosporidium*, cannot grow in water, but are able to persist.

Host-dependent waterborne pathogens, after leaving the body of their host, gradually lose viability and the ability to infect. The rate of decay is usually exponential, and a pathogen will become undetectable after a certain period. Pathogens with low persistence must rapidly find new hosts and are more likely to be spread by person-to-person contact or poor personal hygiene than by drinking-water. Persistence is affected by several factors, of which temperature is the most important. Decay is usually faster at higher temperatures and may be mediated by the lethal effects of ultraviolet (UV) radiation in sunlight acting near the water surface.

Relatively high amounts of biodegradable organic carbon, together with warm waters and low residual concentrations of chlorine, can permit growth of *Legionella*, *Vibrio cholerae*, *Naegleria fowleri* and nuisance organisms in some surface waters and during water distribution (see also the supporting documents *Heterotrophic plate counts and drinking-water safety* and *Legionella and the prevention of legionellosis*; [Annex 1](#)).

Microbial water quality may vary rapidly and widely. Short-term peaks in pathogen concentration may increase disease risks considerably and may also trigger outbreaks of waterborne disease. Microorganisms can accumulate in sediments and are mobilized when water flow increases. Results of water quality testing for microbes are not normally available in time to inform management action and prevent the supply of unsafe water.

7.1.4 Public health aspects

Outbreaks of waterborne disease may affect large numbers of persons, and the first priority in developing and applying controls on drinking-water quality should be the control of such outbreaks. Available evidence also suggests that drinking-water can contribute to background rates of disease in non-outbreak situations, and control of drinking-water quality should therefore also address waterborne disease in the general community.

Experience has shown that systems for the detection of waterborne disease outbreaks are typically inefficient in countries at all levels of socioeconomic development, and failure to detect outbreaks is not a guarantee that they do not occur; nor does it suggest that drinking-water should necessarily be considered safe.

Some of the pathogens that are known to be transmitted through contaminated drinking-water lead to severe and sometimes life-threatening disease. Examples include typhoid, cholera, infectious hepatitis (caused by hepatitis A virus or hepatitis E virus) and disease caused by *Shigella* spp. and *E. coli* O157. Others such as *Campylobacter*, noroviruses and *Cryptosporidium* are typically associated with less severe outcomes, such as self-limiting diarrhoeal disease, although they can cause severe illness and, together with rotaviruses, cause a significant disease burden. Moreover, resistance to antibiotics is a concern for all of these bacteria.

The effects of exposure to pathogens are not the same for all individuals or, as a consequence, for all populations. Repeated exposure to a pathogen may be associated with a lower probability or severity of illness because of the effects of acquired

immunity. For some pathogens (e.g. hepatitis A virus), immunity is lifelong. In contrast, vulnerable subpopulations (e.g. the young, the elderly, pregnant women, the immunocompromised) may have a greater probability of illness or the illness may be more severe, including mortality. Not all pathogens have greater effects in all vulnerable subpopulations.

Not all infected individuals will develop symptomatic disease. The proportion of the infected population that is asymptomatic (including carriers) differs between pathogens and also depends on population characteristics, such as prevalence of immunity. Those with asymptomatic infections as well as patients during and after illness may all contribute to secondary spread of pathogens.

7.2 Health-based target setting

7.2.1 *Health-based targets applied to microbial hazards*

General approaches to health-based target setting are described in [section 2.1](#) and [chapter 3](#).

Sources of information on health risks may be from both epidemiology and QMRA, and typically both are employed as complementary sources. Development of health-based targets for many pathogens may be constrained by limitations in the data. Additional data, derived from both epidemiology and QMRA, are becoming progressively more available. Locally generated data will always be of great value in setting national targets.

Health-based targets may be set using a direct health outcome approach, where the waterborne disease burden is believed to be sufficiently high to allow measurement of the impact of interventions—that is, epidemiological measurement of reductions in disease that can be attributed to improvements in drinking-water quality.

Interpreting and applying information from analytical epidemiological studies to derive health-based targets for application at a national or local level require consideration of a number of factors, including the following questions:

- Are specific estimates of disease reduction or indicative ranges of expected reductions to be provided?
- How representative of the target population was the study sample in order to assure confidence in the reliability of the results across a wider group?
- To what extent will minor differences in demographic or socioeconomic conditions affect expected outcomes?

More commonly, QMRA is used as the basis for setting microbial health-based targets, particularly where the fraction of disease that can be attributed to drinking-water is low or difficult to measure directly through public health surveillance or analytical epidemiological studies.

For the control of microbial hazards, the most frequent form of health-based target applied is performance targets (see [section 3.3.3](#)), which are anchored to a predetermined tolerable burden of disease and established by applying QMRA taking into account raw water quality. Water quality targets (see [section 3.3.2](#)) are typically not developed for pathogens; monitoring finished water for pathogens is not considered

a feasible or cost-effective option because pathogen concentrations equivalent to tolerable levels of risk are typically less than 1 organism per 10^4 – 10^5 litres.

7.2.2 Reference pathogens

It is not practical, and there are insufficient data, to set performance targets for all potentially waterborne pathogens, including bacteria, viruses, protozoa and helminths. A more practical approach is to identify reference pathogens that represent groups of pathogens, taking into account variations in characteristics, behaviours and susceptibilities of each group to different treatment processes. Typically, different reference pathogens will be identified to represent bacteria, viruses, protozoa and helminths.

Selection criteria for reference pathogens include all of the following elements:

- waterborne transmission established as a route of infection;
- sufficient data available to enable a QMRA to be performed, including data on dose–response relationships in humans and disease burden;
- occurrence in source waters;
- persistence in the environment;
- sensitivity to removal or inactivation by treatment processes;
- infectivity, incidence and severity of disease.

Some of the criteria, such as environmental persistence and sensitivity to treatment processes, relate to the specific characteristics of the reference pathogens. Other criteria can be subject to local circumstances and conditions. These can include waterborne disease burden, which can be influenced by the prevalence of the organism from other sources, levels of immunity and nutrition (e.g. rotavirus infections have different outcomes in high- and low-income regions); and occurrence of the organism in source waters (e.g. presence of toxigenic *Vibrio cholerae* and *Entamoeba histolytica* is more common in defined geographical regions, whereas *Naegleria fowleri* is associated with warmer waters).

Selection of reference pathogens

The selection of reference pathogens may vary between different countries and regions and should take account of local conditions, including incidence and severity of waterborne disease and source water characteristics (see [section 7.3.1](#)). Evidence of disease prevalence and significance should be used in selecting reference pathogens. However, the range of potential reference pathogens is limited by data availability, particularly in regard to human dose–response models for QMRA.

Decision-making regarding selection of reference pathogens should be informed by all available data sources, including infectious disease surveillance and targeted studies, outbreak investigations and registries of laboratory-confirmed clinical cases. Such data can help identify the pathogens that are likely to be the biggest contributors to the burden of waterborne disease. It is these pathogens that may be suitable choices as reference pathogens and to consider when establishing health-based targets.

Viruses

Viruses are the smallest pathogens and hence are more difficult to remove by physical processes such as filtration. Specific viruses may be less sensitive to disinfection than bacteria and parasites (e.g. adenovirus is less sensitive to UV light). Viruses can persist for long periods in water. Infective doses are typically low. Viruses typically have a limited host range, and many are species specific. Most human enteric viruses are not carried by animals, although there are some exceptions, including specific strains of hepatitis E virus (Table 7.1).

Rotaviruses, enteroviruses and noroviruses have been identified as potential reference pathogens. Rotaviruses are the most important cause of gastrointestinal infection in children and can have severe consequences, including hospitalization and death, with the latter being far more frequent in low-income regions. There is a dose–response model for rotaviruses, but there is no routine culture-based method for quantifying infectious units. Typically, rotaviruses are excreted in very large numbers by infected patients, and waters contaminated by human waste could contain high concentrations. Several outbreaks of waterborne disease have been recorded.

Enteroviruses, including polioviruses, can cause mild febrile illness, but are also important causative agents of severe diseases, such as paralysis, meningitis and encephalitis, in children. There is a dose–response model for one type of enterovirus, and there is a routine culture-based analysis for measuring infective particles. Enteroviruses are excreted in very large numbers by infected patients, and waters contaminated by human waste could contain high concentrations. Noroviruses are a major cause of acute gastroenteritis in all age groups. Symptoms of illness are generally mild and rarely last longer than 3 days. Hence, the burden of disease per case is lower than for rotaviruses. Numerous outbreaks have been attributed to drinking-water. A dose–response model has been developed to estimate infectivity for several norovirus strains, but no culture-based method is available.

Bacteria

Bacteria are generally the group of pathogens that is most sensitive to inactivation by disinfection. Some free-living pathogens, such as *Legionella* and non-tuberculous mycobacteria, can grow in water environments, but enteric bacteria typically do not grow in water and survive for shorter periods than viruses or protozoa. Many bacterial species that are infective to humans are carried by animals.

There are a number of potentially waterborne bacterial pathogens with known dose–response models, including *Vibrio*, *Campylobacter*, *E. coli* O157, *Salmonella* and *Shigella*.

Toxigenic *Vibrio cholerae* can cause watery diarrhoea. When it is left untreated, as may be the case when people are displaced by conflict and natural disaster, case fatality rates are very high. The infective dose is relatively high. Large waterborne outbreaks have been described and keep occurring.

Campylobacter is an important cause of diarrhoea worldwide. Illness can produce a wide range of symptoms, including infrequent but serious long-term sequelae. Mortality is lower than some of the other bacterial pathogens (e.g. *Vibrio*, *Salmonella* Typhi). Compared with most other bacterial pathogens, the infective dose can be

relatively low, at less than 10 organisms but can be up to approximately 1000 organisms. It is relatively common in the environment, and waterborne outbreaks have been recorded.

Waterborne infection by *E. coli* O157 and other enterohaemorrhagic strains of *E. coli* is far less common than infection by *Campylobacter*, but the symptoms of infection are more severe, including haemolytic uraemic syndrome and death. The infective dose is moderate at approximately 200 organisms.

Shigella is also an important cause of diarrhoea worldwide. The infective dose is low to moderate and can be below 100 organisms. Waterborne outbreaks have been recorded.

Non-typhoidal *Salmonella* have caused a number of waterborne outbreaks, predominantly in groundwater and small rainwater systems, whereas *S. Typhi* have caused larger outbreaks of waterborne typhoid as well as endemic disease.

Protozoa

Protozoa are the group of pathogens that is least sensitive to inactivation by chemical disinfection. UV light irradiation is effective against *Cryptosporidium*, but *Cryptosporidium* is highly resistant to chlorine. Protozoa are of a moderate size ($> 2 \mu\text{m}$) and can be removed by physical processes. They can survive for long periods in water. They are moderately species specific. Livestock and humans can be sources of protozoa such as *Cryptosporidium* and *Giardia*, whereas humans are the sole reservoirs of pathogenic *Cyclospora* and *Entamoeba*. Infective doses are typically low.

There are dose–response models available for *Giardia* and *Cryptosporidium*. *Giardia* symptoms can be longer lasting than those of *Cryptosporidium*. However, *Cryptosporidium* is smaller than *Giardia* and hence more difficult to remove by physical processes; it is also more resistant to oxidizing disinfectants, and there is some evidence that it survives longer in water environments.

7.2.3 Quantitative microbial risk assessment

QMRA systematically combines available information on exposure (i.e. the number of pathogens ingested) and dose–response models to produce estimates of the probability of infection associated with exposure to pathogens in drinking-water. Epidemiological data on frequency of asymptomatic infections, duration and severity of illness can then be used to estimate disease burdens.

QMRA can be used to determine performance targets and as the basis for assessing the effects of improved water quality on health in the population and subpopulations. Mathematical modelling can be used to estimate the effects of low doses of pathogens in drinking-water on health.

Risk assessment, including QMRA, commences with problem formulation to identify all possible hazards and their pathways from sources to recipients. Human exposure to the pathogens (environmental concentrations and volumes ingested) and dose–response relationships for selected (or reference) organisms are then combined to characterize the risks. With the use of additional information (social, cultural, political, economic, environmental, etc.), management options can be prioritized. To encourage stakeholder support and participation, a transparent procedure and active

Table 7.3 Risk assessment paradigm for pathogen health risks

Step	Aim
1. Problem formulation and hazard identification	To identify all possible hazards associated with drinking-water that would have an adverse public health consequence, as well as their pathways from source(s) to consumer(s)
2. Exposure assessment	To determine the size and nature of the population exposed and the route, amount and duration of the exposure
3. Dose–response assessment	To characterize the relationship between exposure and the incidence of the health effect
4. Risk characterization	To integrate the information from exposure, dose–response and health interventions in order to estimate the magnitude of the public health problem and to evaluate variability and uncertainty

Source: Adapted from Haas, Rose & Gerba (1999)

risk communication at each stage of the process are important. An example of a risk assessment approach is outlined in [Table 7.3](#) and described below. For more detailed information on QMRA in the context of drinking-water safety, see the supporting document *Quantitative microbial risk assessment: application for water safety management*; [Annex 1](#).

Problem formulation and hazard identification

All potential hazards, sources and events that can lead to the presence of microbial pathogens (i.e. what can happen and how) should be identified and documented for each component of the drinking-water system, regardless of whether or not the component is under the direct control of the drinking-water supplier. This includes point sources of pollution (e.g. human and industrial waste discharges) as well as diffuse sources (e.g. those arising from agricultural and animal husbandry activities). Continuous, intermittent or seasonal pollution patterns should also be considered, as well as extreme and infrequent events, such as droughts and floods.

The broader sense of hazards includes hazardous scenarios, which are events that may lead to exposure of consumers to specific pathogenic microorganisms. In this, the hazardous event (e.g. peak contamination of source water with domestic wastewater) may be referred to as the hazard.

As a QMRA cannot be performed for each of the hazards identified, representative (or reference) organisms are selected that, if controlled, would ensure control of all pathogens of concern. Typically, this implies inclusion of at least one bacterium, virus, protozoan or helminth. In this section, *Campylobacter*, rotavirus and *Cryptosporidium* have been used as example reference pathogens to illustrate application of risk assessment and calculation of performance targets.

Exposure assessment

Exposure assessment in the context of drinking-water consumption involves estimation of the number of pathogens to which an individual is exposed, principally through ingestion. Exposure assessment inevitably contains uncertainty and must ac-

count for variability of such factors as concentrations of pathogens over time and volumes ingested.

Exposure can be considered as a single dose of pathogens that a consumer ingests at a certain point in time or the total amount over several exposures (e.g. over a year). Exposure is determined by the concentration of pathogens in drinking-water and the volume of water consumed.

It is rarely possible or appropriate to directly measure pathogens in drinking-water on a regular basis. More often, concentrations in raw waters are assumed or measured, and estimated reductions—for example, through treatment—are applied to estimate the concentration in the water consumed. Pathogen measurement, when performed, is generally best carried out at the location where the pathogens are at highest concentration (generally raw waters). Estimation of their removal by sequential control measures is generally achieved by the use of indicator organisms such as *E. coli* for enteric bacterial pathogens (see [Table 7.4](#); see also the supporting document *Water treatment and pathogen control* in [Annex 1](#)).

The other component of exposure assessment, which is common to all pathogens, is the volume of unboiled water consumed by the population, including person-to-person variation in consumption behaviour and especially consumption behaviour of vulnerable subpopulations. For microbial hazards, it is important that the unboiled volume of drinking-water, both consumed directly and used in food preparation, is used in the risk assessment, as heating will rapidly inactivate pathogens. This amount is lower than that used for deriving water quality targets, such as chemical guideline values.

The daily exposure of a consumer to pathogens in drinking-water can be assessed by multiplying the concentration of pathogens in drinking-water by the volume of drinking-water consumed (i.e. dose). For the purposes of the example model calculations, drinking-water consumption was assumed to be 1 litre of unboiled water per day, but location-specific data on drinking-water consumption are preferred.

Dose–response assessment

The probability of an adverse health effect following exposure to one or more pathogenic organisms is derived from a dose–response model. Available dose–response data have been obtained mainly from studies using healthy adult volunteers. However, adequate data are lacking for vulnerable subpopulations, such as children, the elderly and the immunocompromised, who may suffer more severe disease outcomes.

The conceptual basis for the dose–response model is the observation that exposure to the described dose leads to the probability of infection as a conditional event: for infection to occur, one or more viable pathogens must have been ingested. Furthermore, one or more of these ingested pathogens must have survived in the host's body. An important concept is the single-hit principle (i.e. that even a single pathogen may be able to cause infection and disease). This concept supersedes the concept of (minimum) infectious dose that is frequently used in older literature (see the supporting document *Hazard characterization for pathogens in food and water*; [Annex 1](#)).

In general, well-dispersed pathogens in water are considered to be Poisson distributed. When the individual probability of any organism surviving and starting infection is the same, the dose–response relationship simplifies to an exponential func-

tion. If, however, there is heterogeneity in this individual probability, this leads to the beta-Poisson dose–response relationship, where the “beta” stands for the distribution of the individual probabilities among pathogens (and hosts). At low exposures, such as would typically occur in drinking-water, the dose–response model is approximately linear and can be represented simply as the probability of infection resulting from exposure to a single organism (see the supporting document *Hazard characterization for pathogens in food and water*; [Annex 1](#)).

Risk characterization

Risk characterization brings together the data collected on exposure, dose–response and the incidence and severity of disease.

The probability of infection can be estimated as the product of the exposure to drinking-water and the probability that exposure to one organism would result in infection. The probability of infection per day is multiplied by 365 to calculate the probability of infection per year. In doing so, it is assumed that different exposure events are independent, in that no protective immunity is built up. This simplification is justified for low risks only, such as those discussed here.

Not all infected individuals will develop clinical illness; asymptomatic infection is common for most pathogens. The percentage of infected persons who will develop clinical illness depends on the pathogen, but also on other factors, such as the immune status of the host. Risk of illness per year is obtained by multiplying the probability of infection by the probability of illness given infection.

The low numbers in [Table 7.4](#) can be interpreted to represent the probability that a single individual will develop illness in a given year. For example, a risk of illness for *Campylobacter* of 2.2×10^{-4} per year indicates that, on average, 1 out of 4600 consumers would contract campylobacteriosis from consumption of drinking-water.

To translate the risk of developing a specific illness to disease burden per case, the metric disability-adjusted life year, or DALY, is used (see [Box 3.1](#) in [chapter 3](#)). This metric reflects not only the effects of acute end-points (e.g. diarrhoeal illness) but also mortality and the effects of more serious end-points (e.g. Guillain-Barré syndrome associated with *Campylobacter*). The disease burden per case varies widely. For example, the disease burden per 1000 cases of rotavirus diarrhoea is 480 DALYs in low-income regions, where child mortality frequently occurs. However, it is 14 DALYs per 1000 cases in high-income regions, where hospital facilities are accessible to the great majority of the population (see the supporting document *Quantifying public health risk in the WHO Guidelines for drinking-water quality*; [Annex 1](#)). This considerable difference in disease burden results in far stricter treatment requirements in low-income regions for the same raw water quality in order to obtain the same risk (expressed as DALYs per person per year). Ideally, the health outcome target of 10^{-6} DALY per person per year in [Table 7.4](#) should be adapted to specific national situations. In [Table 7.4](#), no accounting is made for effects on immunocompromised persons (e.g. cryptosporidiosis in patients with human immunodeficiency virus or AIDS), which is significant in some countries. [Section 3.2](#) gives more information on the DALY metric and how it is applied to derive a reference level of risk.

Table 7.4 Linking tolerable disease burden and raw water quality for reference pathogens: example calculation

River water (human and livestock pollution)	Units	<i>Cryptosporidium</i>	<i>Campylobacter</i>	Rotavirus ^a
Raw water quality (C_R)	Organisms per litre	10	100	10
Treatment effect needed to reach tolerable risk (PT)	Log_{10} reduction value	5.89	5.98	5.96
Drinking-water quality (C_D)	Organisms per litre	1.3×10^{-5}	1.05×10^{-4}	1.1×10^{-5}
Consumption of unheated drinking-water (V)	Litres per day	1	1	1
Exposure by drinking-water (E)	Organisms per day	1.3×10^{-5}	1.05×10^{-4}	1.1×10^{-5}
Dose–response (r) ^b	Probability of infection per organism	2.0×10^{-1}	1.9×10^{-2}	5.9×10^{-1}
Risk of infection ($P_{\text{inf},d}$)	Per day	2.6×10^{-6}	2.0×10^{-6}	6.5×10^{-6}
Risk of infection ($P_{\text{inf},y}$)	Per year	9.5×10^{-4}	7.3×10^{-4}	2.4×10^{-3}
Risk of (diarrhoeal) illness given infection ($P_{\text{ill inf}}$)	Probability of illness per infection	0.7	0.3	0.5
Risk of (diarrhoeal) illness (P_{ill})	Per year	6.7×10^{-4}	2.2×10^{-4}	1.2×10^{-3}
Disease burden (db)	DALY per case	1.5×10^{-3}	4.6×10^{-3}	1.4×10^{-2}
Susceptible fraction (f_s)	Percentage of population	100	100	6
Health outcome target (HT)	DALY per year ^c	1×10^{-6}	1×10^{-6}	1×10^{-6}
Formulas:		$C_D = C_R \div 10^{\text{PT}}$	$P_{\text{inf},d} = E \times r$	$\text{HT} = P_{\text{ill}} \times \text{db} \times f_s \div 100$
		$E = C_D \times V$	$P_{\text{ill}} = P_{\text{inf},y} \times P_{\text{ill inf}}$	

DALY, disability-adjusted life year

^a Data from high-income regions. In low-income regions, severity is typically higher (see the supporting document *Quantifying public health risk in the WHO Guidelines for drinking-water quality*; Annex 1).

^b Dose–response for *Campylobacter* and rotavirus from Haas, Rose & Gerba (1999) and for *Cryptosporidium* from the supporting document *Risk assessment of Cryptosporidium in drinking water* (Annex 1).

^c For a person drinking 1 litre per day (V).

Only a proportion of the population may be susceptible to some pathogens, because immunity developed after an initial episode of infection or illness may provide lifelong protection. Examples include hepatitis A virus and rotaviruses. It is estimated that in developing countries, all children above the age of 5 years are immune to rotaviruses because of repeated exposure in the first years of life. This translates to an average of 17% of the population being susceptible to rotavirus illness. In developed countries, rotavirus infection is also common in the first years of life, and the illness is diagnosed mainly in young children, but the percentage of young children as part of the total population is lower. This translates to an average of 6% of the population in developed countries being susceptible.

The uncertainty of the risk outcome is the result of the uncertainty and variability of the data collected in the various steps of the risk assessment. Risk assessment models should ideally account for this variability and uncertainty, although here we present only point estimates (see below).

It is important to choose the most appropriate point estimate for each of the variables. Theoretical considerations show that risks are directly proportional to the arithmetic mean of the ingested dose. Hence, arithmetic means of variables such as concentration in raw water, removal by treatment and consumption of drinking-water are recommended. This recommendation is different from the usual practice among microbiologists and engineers of converting concentrations and treatment effects to log values and making calculations or specifications on the log scale. Such calculations result in estimates of the geometric mean rather than the arithmetic mean, and these may significantly underestimate risk. Analysing site-specific data may therefore require going back to the raw data (i.e. counts and tested volumes) rather than relying on reported log-transformed values, as these introduce ambiguity.

Emergencies such as major storms and floods can lead to substantial deteriorations in source water quality, including large short-term increases in pathogen concentrations. These should not be included in calculations of arithmetic means. Inclusion will lead to higher levels of treatment being applied on a continuous basis, with substantial cost implications. It is more efficient to develop specific plans to deal with the events and emergencies (see [section 4.4](#)). Such plans can include enhanced treatment or (if possible) selection of alternative sources of water during an emergency.

7.2.4 Risk-based performance target setting

The process outlined above enables estimation of risk on a population level, taking account of raw water quality and impact of control. This can be compared with the reference level of risk (see [section 3.2](#)) or a locally developed tolerable risk. The calculations enable quantification of the degree of source protection or treatment that is needed to achieve a specified level of tolerable risk and analysis of the estimated impact of changes in control measures.

Performance targets are most frequently applied to treatment performance—that is, to determine the microbial reduction necessary to ensure water safety. A performance target may be applied to a specific system (i.e. formulated in response to local raw water characteristics) or generalized (e.g. formulated in response to raw water quality assumptions based on a certain type of source) (see also the supporting document *Water treatment and pathogen control*; [Annex 1](#)).

[Figure 7.2](#) illustrates the targets for treatment performance for a range of pathogens occurring in raw water. For example, 10 microorganisms per litre of raw water will lead to a performance target of 5.89 logs (or 99.999 87% reduction) for *Cryptosporidium* or of 5.96 logs (99.999 89% reduction) for rotaviruses in high-income regions to achieve 10^{-6} DALY per person per year (see also [Table 7.5](#) below). The difference in performance targets for rotaviruses in high- and low-income countries (5.96 and 7.96 logs; [Figure 7.2](#)) is related to the difference in disease severity caused by this organism. In low-income countries, the child case fatality rate is relatively high, and, as a con-

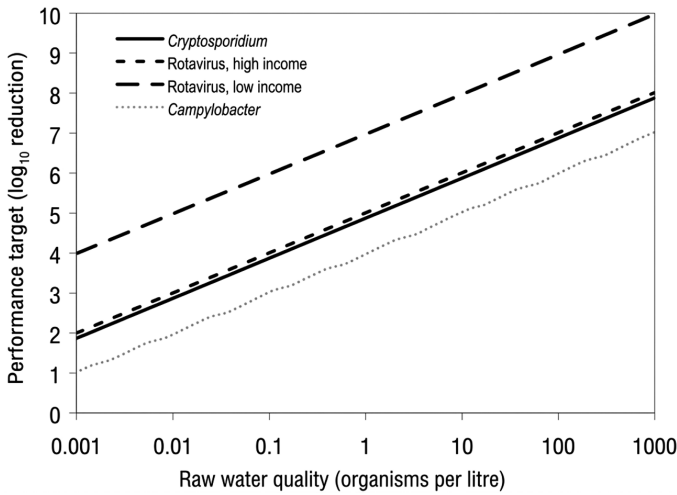


Figure 7.2 Performance targets for example bacterial, viral and protozoan pathogens in relation to raw water quality (to achieve 10^{-6} DALY per person per year)

sequence, the disease burden is higher. Also, a larger proportion of the population in low-income countries is under the age of 5 and at risk for rotavirus infection.

The derivation of these performance targets is described in [Table 7.5](#), which provides an example of the data and calculations that would normally be used to construct a risk assessment model for waterborne pathogens. The table presents data for representatives of the three major groups of pathogens (bacteria, viruses and protozoa) from a range of sources. These example calculations aim at achieving the reference level of risk of 10^{-6} DALY per person per year, as described in [section 3.2](#). The data in the table illustrate the calculations needed to arrive at a risk estimate and are not guideline values.

7.2.5 Presenting the outcome of performance target development

[Table 7.5](#) presents some data from [Table 7.4](#) in a format that is more meaningful to risk managers. The average concentration of pathogens in drinking-water is included for information. It is not a water quality target, nor is it intended to encourage pathogen monitoring in finished water. As an example, a concentration of 1.3×10^{-5} *Cryptosporidium* per litre (see [Table 7.4](#)) corresponds to 1 oocyst per 79 000 litres (see [Table 7.5](#)). The performance target (in the row “Treatment effect” in [Table 7.4](#)), expressed as a $\log_{10}$ reduction value, is the most important management information in the risk assessment table. It can also be expressed as a per cent reduction. For example, a 5.96 $\log_{10}$ unit reduction for rotaviruses corresponds to a 99.999 89% reduction.

7.2.6 Adapting risk-based performance target setting to local circumstances

The reference pathogens illustrated in the previous sections will not be priority pathogens in all regions of the world. Wherever possible, country- or site-specific informa-

Table 7.5 Health-based targets derived from example calculation in Table 7.4

	<i>Cryptosporidium</i>	<i>Campylobacter</i>	Rotavirus ^a
Organisms per litre in raw water	10	100	10
Health outcome target	10 ⁻⁶ DALY per person per year	10 ⁻⁶ DALY per person per year	10 ⁻⁶ DALY per person per year
Risk of diarrhoeal illness ^b	1 per 1500 per year	1 per 4600 per year	1 per 14 000 per year
Drinking-water quality	1 per 79 000 litres	1 per 9500 litres	1 per 90 000 litres
Performance target ^c	5.89 log ₁₀ units	5.98 log ₁₀ units	5.96 log ₁₀ units

^a Data from high-income regions. In low-income regions, severity is typically higher, but drinking-water transmission is unlikely to dominate.

^b For the susceptible population.

^c Performance target is a measure of log reduction of pathogens based on raw water quality.

tion should be used in assessments of this type. If no specific data are available, an approximate risk estimate can be based on default values (see Table 7.6 below).

Table 7.5 accounts only for changes in water quality derived from treatment and not from source protection measures, which are often important contributors to overall safety, affecting pathogen concentration and/or variability. The risk estimates presented in Table 7.4 also assume that there is no degradation of water quality in the distribution network. These may not be realistic assumptions under all circumstances, and it is advisable to take these factors into account wherever possible.

Table 7.5 presents point estimates only and does not account for variability and uncertainty. Full risk assessment models would incorporate such factors by representing the input variables by statistical distributions rather than by point estimates. However, such models are currently beyond the means of many countries, and data to define such distributions are scarce. Producing such data may involve considerable efforts in terms of time and resources, but will lead to much improved insight into the actual raw water quality and treatment performance.

The necessary degree of treatment also depends on the values assumed for variables that can be taken into account in the risk assessment model. One such variable is drinking-water consumption. Figure 7.3 shows the effect of variation in the consumption of unboiled drinking-water on the performance targets for *Cryptosporidium*. If the raw water concentration is 1 oocyst per litre, the performance target varies between 4.3 and 5.2 log₁₀ units if consumption values vary between 0.25 and 2 litres per day. Another variable is the fraction of the population that is susceptible. Some outbreak data suggest that in developed countries, a significant proportion of the population above 5 years of age may not be immune to rotavirus illness. Figure 7.4 shows the effect of variation in the susceptible fraction of the population. If the raw water concentration is 10 rotavirus particles per litre, the performance target increases from 5.96 to 7.18 as the susceptible fraction increases from 6% to 100%.

7.2.7 Health outcome targets

Health outcome targets that identify disease reductions in a community should be responded to by the control measures set out in water safety plans (WSPs) and associated

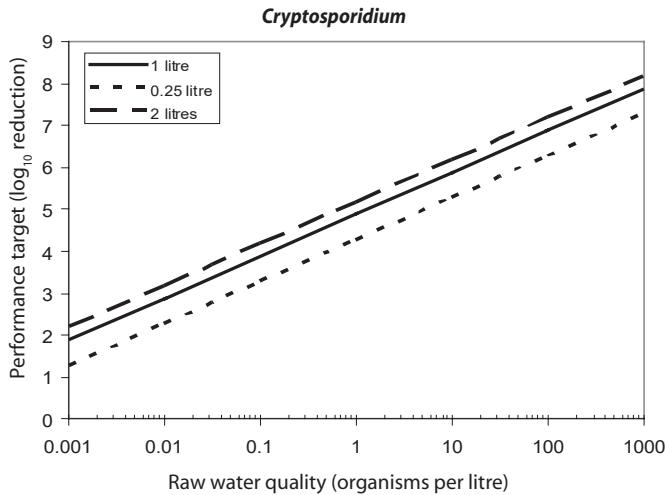


Figure 7.3 Performance targets for *Cryptosporidium* in relation to the daily consumption of unboiled drinking-water (to achieve 10^{-6} DALY per person per year)

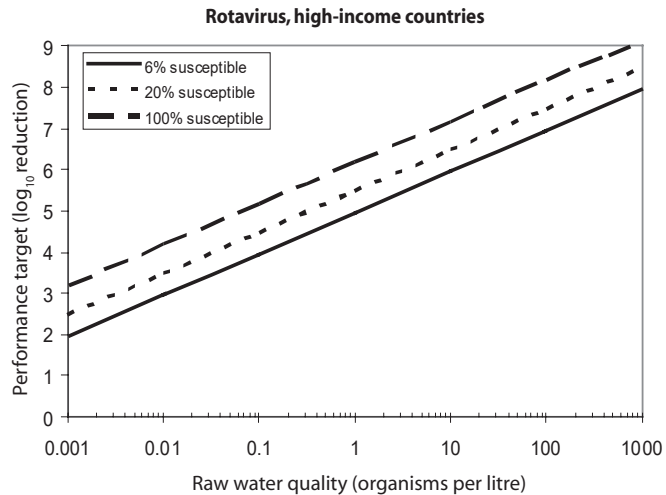


Figure 7.4 Performance targets for rotaviruses in relation to the fraction of the population that is susceptible to illness (to achieve 10^{-6} DALY per person per year)

water quality interventions at community and household levels. These targets would identify expected disease reductions in communities receiving the interventions.

The prioritization of water quality interventions should focus on those aspects that are estimated to contribute more than, for example, 5% of the burden of a given disease (e.g. 5% of total diarrhoea). In many parts of the world, the implementation of a water quality intervention that results in an estimated health gain of more than 5% would be considered extremely worthwhile. Directly demonstrating the health gains arising from improving water quality—as assessed, for example, by reduced *E. coli*

Table 7.6 Example occurrence of selected indicators and pathogens in faeces, wastewater and raw water (local data will vary)

Microbe	Number per gram of faeces	Number per litre in untreated wastewater	Number per litre in raw water
Faecal coliforms (<i>E. coli</i> and <i>Klebsiella</i>)	10 ⁷ (mostly non-pathogenic)	10 ⁶ –10 ¹⁰	100–100 000
<i>Campylobacter</i> spp.	10 ⁶	100–10 ⁶	100–10 000
<i>Vibrio cholerae</i> ^a	10 ⁶	100–10 ⁶	100–10 ⁸
Enteroviruses	10 ⁶	1–1000	0.01–10
Rotaviruses	10 ⁹	50–5000	0.01–100
<i>Cryptosporidium</i>	10 ⁷	1–10 000	0–1000
<i>Giardia intestinalis</i>	10 ⁷	1–10 000	0–1000

^a *Vibrio* can grow in the aquatic environment.

Sources: Feachem et al. (1983); Stelzer (1988); Jones, Betaieb & Telford (1990); Stampi et al. (1992); Koenraad et al. (1994); Gerba et al. (1996); AWWA (1999); Maier, Pepper & Gerba (2000); Metcalf & Eddy, Inc. (2003); Bitton (2005); Lodder & de Roda Husman (2005); Schijven & de Roda Husman (2006); Masini et al. (2007); Rutjes et al. (2009); Lodder et al. (2010)

counts at the point of consumption—may be possible where disease burden is high and effective interventions are applied and can be a powerful tool to demonstrate a first step in incremental drinking-water safety improvement.

Where a specified quantified disease reduction is identified as a health outcome target, it is advisable to undertake ongoing proactive public health surveillance among representative communities to measure the effectiveness of water quality interventions.

7.3 Occurrence and treatment of pathogens

As discussed in [section 4.2](#), system assessment involves determining whether the drinking-water supply chain as a whole can deliver drinking-water quality that meets identified targets. This requires an understanding of the quality of source water and the efficacy of control measures, such as treatment.

7.3.1 Occurrence

An understanding of pathogen occurrence in source waters is essential, because it facilitates selection of the highest-quality source for drinking-water supply, determines pathogen concentrations in source waters and provides a basis for establishing treatment requirements to meet health-based targets within a WSP.

By far the most accurate way of determining pathogen concentrations in specific catchments and other water sources is by analysing pathogen concentrations in water over a period of time, taking care to include consideration of seasonal variation and peak events such as storms. Direct measurement of pathogens and indicator organisms in the specific source waters for which a WSP and its target pathogens are being established is recommended wherever possible, because this provides the best estimates of microbial concentrations. However, resource limitations in many settings preclude this. In the absence of measured pathogen concentrations, an alternative interim approach is to make estimations based on available data, such as the results of sanitary

inspections or WSP audit findings combined with indicator testing. Examples of such approaches are shown in [Tables 5.3](#) and [5.4](#).

In the case of absence of data on the occurrence and distribution of human pathogens in water for the community or area of implementation, concentrations in raw waters can be inferred from observational data on numbers of pathogens per gram of faeces representing direct faecal contamination or from numbers of pathogens per litre of untreated wastewater ([Table 7.6](#)). Data from sanitary surveys can be used to estimate the impact of raw or treated wastewater discharged into source waters. In treated wastewater, the concentrations of pathogens may be reduced 10- to 100-fold or more, depending on the efficiency of the treatment process. The concentrations of pathogens in raw waters can be estimated from concentrations of pathogens in wastewater and the fraction of wastewater present in source waters. In addition, some indicative concentrations of pathogens in source waters are given that were measured at specific locations, but these concentrations may differ widely between locations.

From [Table 7.6](#), it may be clear that faecal indicator bacteria, such as *E. coli*, are always present at high concentrations in wastewater. Everybody sheds *E. coli*; nevertheless concentrations vary widely. Only infected persons shed pathogens; therefore, the concentrations of pathogens in wastewater vary even more. Such variations are due to shedding patterns, but they also depend on other factors, such as the size of the population discharging into wastewater and dilution with other types of wastewater, such as industrial wastewater. Conventional wastewater treatment commonly reduces microbial concentrations by one or two orders of magnitude before the wastewater is discharged into surface waters. At other locations, raw wastewater may be discharged directly, or discharges may occur occasionally during combined sewer overflows. Discharged wastewater is diluted in receiving surface waters, leading to reduced pathogen numbers, with the dilution factor being very location specific. Pathogen inactivation, die-off or partitioning to sediments may also play a role in pathogen reduction. These factors differ with the surface water body and climate. This variability suggests that concentrations of faecal indicators and pathogens vary even more in surface water than in wastewater.

Because of differences in survival, the ratio of pathogen to *E. coli* at the point of discharge will not be the same as farther downstream. A comparison of data on *E. coli* with pathogen concentrations in surface waters indicates that, overall, there is a positive relationship between the presence of pathogens in surface water and *E. coli* concentration, but that pathogen concentrations may vary widely from low to high at any *E. coli* concentration. Even the absence of *E. coli* is not a guarantee for the absence of pathogens or for pathogen concentrations to be below those of significance for public health.

The estimates based on field data in [Table 7.6](#) provide a useful guide to the concentrations of enteric pathogens in a variety of sources affected by faecal contamination. However, there are a number of limitations and sources of uncertainty in these data, including the following:

- Although data on pathogens and *E. coli* were derived from different regions in the world, they are by far mostly from high-income countries.
- There are concerns about the sensitivity and robustness of analytical techniques, particularly for viruses and protozoa, largely associated with the recoveries

achieved by techniques used to process and concentrate large sample volumes typically used in testing for these organisms.

- Numbers of pathogens were derived using a variety of methods, including culture-based methods using media or cells, molecular-based tests (such as polymerase chain reaction) and microscopy, and should be interpreted with care.
- The lack of knowledge about the infectivity of the pathogens for humans has implications in risk assessment and should be addressed.

7.3.2 Treatment

Understanding the efficacy of control measures includes validation (see [section 4.2.7](#)). Validation is important both in ensuring that treatment will achieve the desired goals (performance targets) and in assessing areas in which efficacy may be improved (e.g. by comparing performance achieved with that shown to be achievable through well-run processes). Water treatment could be applied in a drinking-water treatment plant (central treatment) to piped systems or in the home or at the point of use in settings other than piped supplies.

Central treatment

Waters of very high quality, such as groundwater from confined aquifers, may rely on protection of the source water and the distribution system as the principal control measures for provision of safe water. More typically, water treatment is required to remove or destroy pathogenic microorganisms. In many cases (e.g. poor quality surface water), multiple treatment stages are required, including, for example, coagulation, flocculation, sedimentation, filtration and disinfection. [Table 7.7](#) provides a summary of treatment processes that are commonly used individually or in combination to achieve microbial reductions (see also [Annex 5](#)). The minimum and maximum removals are indicated as $\log_{10}$ reduction values and may occur under failing and optimal treatment conditions, respectively.

The microbial reductions presented in [Table 7.7](#) are for broad groups or categories of microbes: bacteria, viruses and protozoa. This is because it is generally the case that treatment efficacy for microbial reduction differs among these microbial groups as a result of the inherently different properties of the microbes (e.g. size, nature of protective outer layers, physicochemical surface properties). Within these microbial groups, differences in treatment process efficiencies are smaller among the specific species, types or strains of microbes. Such differences do occur, however, and the table presents conservative estimates of microbial reductions based on the more resistant or persistent pathogenic members of that microbial group. Where differences in removal by treatment between specific members of a microbial group are great, the results for the individual microbes are presented separately in the table.

Treatment efficacy for microbial reduction can also differ when aggregating different treatment processes. Applying multiple barriers in treatment, for example in drinking-water treatment plants, may strengthen performance, as failure of one process does not result in failure of the entire treatment. However, both positive and negative interactions can occur between multiple treatment steps, and how these interactions affect the overall water quality and water treatment performance is not yet

completely understood. In positive interactions, the inactivation of a contaminant is higher when two steps are occurring together than when each of the steps occurs separately—as happens, for example, when coagulation and sedimentation are operating under optimal conditions, and there is an increase in performance of rapid sand filters. In contrast, negative interactions can occur when failure in the first step of the treatment process could lead to a failure of the next process—for example, if coagulation fails to remove organic material, this could lead to a reduced efficacy of subsequent disinfection and a potential increase in DBPs. An overall assessment of the drinking-water treatment performance, as part of the implementation of the WSP, will assist in understanding the efficacy of the multiple treatment processes to ensure the safety of the drinking-water supply.

Table 7.7 Reductions of bacteria, viruses and protozoa achieved by water treatment technologies at drinking-water treatment plants for large communities

Treatment process	Enteric pathogen group	Minimum removal (LRV)	Maximum removal (LRV)	Notes
Pretreatment				
Roughing filters	Bacteria	0.2	2.3	Depends on filter medium, coagulant
Storage reservoirs	Bacteria	0.7	2.2	Residence time > 40 days
	Protozoa	1.4	2.3	Residence time 160 days
Bank filtration	Viruses	> 2.1	8.3	Depends on travel distance, soil type, pumping rate, pH, ionic strength
	Bacteria	2	> 6	
	Protozoa	> 1	> 2	
Coagulation, flocculation and sedimentation				
Conventional clarification	Viruses	0.1	3.4	Depends on coagulation conditions
	Bacteria	0.2	2	
	Protozoa	1	2	
High-rate clarification	Protozoa	> 2	2.8	Depends on use of appropriate blanket polymer
Dissolved air flotation	Protozoa	0.6	2.6	Depends on coagulant dose
Lime softening	Viruses	2	4	Depends on pH and settling time
	Bacteria	1	4	
	Protozoa	0	2	
Filtration				
Granular high-rate filtration	Viruses	0	3.5	Depends on filter media and coagulation pretreatment; filtered water turbidity of ≤ 0.3 NTU in 95% of samples (and none to exceed 1 NTU) associated with 1–2 log reduction of viruses and 3 log reduction of <i>Cryptosporidium</i> ^a
	Bacteria	0.2	4.4	
	Protozoa	0.4	3.3	

Table 7.7 (continued)

Treatment process	Enteric pathogen group	Minimum removal (LRV)	Maximum removal (LRV)	Notes
Slow sand filtration	Viruses	0.25	4	Depends on presence of schmutzdecke, grain size, flow rate, operating conditions (mainly temperature, pH); filtered water turbidity of ≤ 1 NTU in 95% of samples (and none to exceed 5 NTU) associated with 1–2 log reduction of viruses and 2.5–3 log reduction of <i>Cryptosporidium</i> ^a
	Bacteria	2	6	
	Protozoa	0.3	> 5	
Precoat filtration	Viruses	1	1.7	If filter cake is present
	Bacteria	0.2	2.3	Depends on chemical pretreatment
	Protozoa	3	6.7	Depends on media grade and filtration rate
Membrane filtration: microfiltration, ultrafiltration, nanofiltration, reverse osmosis	Viruses	< 1	> 6.5	Varies with membrane pore size (microfilters, ultrafilters, nanofilters and reverse osmosis filters), integrity of filter medium and filter seals, and resistance to chemical and biological ("grow-through") degradation; maximum reductions associated with filtered water turbidity of < 0.1 NTU ^a
	Bacteria	1	> 7	
	Protozoa	2.3	> 7	
Primary disinfection^{b,c}				
Chlorine	Viruses	2 (Ct_{99} 2–30 min-mg/l; 0–10 °C; pH 7–9)		Free chlorine $\times$ contact time predicts efficacy; not effective against <i>Cryptosporidium</i> oocysts. Turbidity and chlorine-demanding solutes inhibit this process; hence, turbidity should be kept below 1 NTU to support effective disinfection. Where this is not practical, turbidities should be kept below 5 NTU with higher chlorine doses or contact times. ^a In addition to initial disinfection, the benefits of maintaining free chlorine residuals throughout distribution systems at or above 0.2 mg/l should be considered
	Bacteria	2 (Ct_{99} 0.04–0.08 min-mg/l; 5 °C; pH 6–7)		
	Protozoa	2 (Ct_{99} 25–245 min-mg/l; 0–25 °C; pH 7–8; mainly <i>Giardia</i>)		
Chlorine dioxide	Viruses	2 (Ct_{99} 2–30 min-mg/l; 0–10 °C; pH 7–9)		
	Bacteria	2 (Ct_{99} 0.02–0.3 min-mg/l; 15–25 °C; pH 6.5–7)		
	Protozoa	2 (Ct_{99} 100 min-mg/l)		

Table 7.7 (continued)

Treatment process	Enteric pathogen group	Minimum removal (LRV)	Maximum removal (LRV)	Notes
Ozone	Viruses	2 (Ct_{99} 0.006–0.2 min·mg/l)		Viruses generally more resistant than bacteria
	Bacteria	2 (Ct_{99} 0.02 min·mg/l)		
	Protozoa	2 (Ct_{99} 0.5–40 min·mg/l)		Depends on temperature; <i>Cryptosporidium</i> varies widely
UV irradiation	Viruses	4 (7–186 mJ/cm ²)		Effectiveness of disinfection depends on delivered fluence (dose), which varies with intensity, exposure time and UV wavelength. Excessive turbidity and certain dissolved species inhibit this process; hence, turbidity should be kept below 1 NTU to support effective disinfection. Where this is not practical, turbidities should be kept below 5 NTU with higher fluences ^a
	Bacteria	4 (0.65–230 mJ/cm ²)		
	Protozoa	4 (< 1–60 mJ/cm ²)		

Ct , product of disinfectant concentration and contact time; LRV, $\log_{10}$ reduction value.

^a See *Water quality and health – review of turbidity* (Annex 1).

^b Chemical disinfection: Ct values are given that achieve 2 LRV.

^c UV irradiation: UV dose range is given that achieves 4 LRV.

Sources: Chevrefils et al. (2006); Dullemeier et al. (2006); Hijnen, Beerendonk & Medema (2006); see also the supporting document *Water treatment and pathogen control* (Annex 1)

Further information about these water treatment processes, their operations and their performance for pathogen reduction in piped water supplies is provided in more detail in the supporting document *Water treatment and pathogen control* (Annex 1).

Household treatment

Household water treatment technologies are any of a range of devices or methods employed for the purposes of treating water in the home or at the point of use in other settings. These are also known as point-of-use or point-of-entry water treatment technologies (Cotruvo & Sobsey, 2006; Nath, Bloomfield & Jones, 2006; see also the supporting document *Managing water in the home*, Annex 1). Household water treatment technologies comprise a range of options that enable individuals and communities to treat collected water or contaminated piped water to remove or inactivate microbial pathogens. Many of these methods are coupled with safe storage of the treated water to preclude or minimize contamination after household treatment (Wright, Gundry & Conroy, 2003; Clasen et al., 2015).

Household water treatment and safe storage have been shown to significantly improve water quality and reduce waterborne infectious disease risks (Clasen et al., 2015). Household water treatment approaches may be important interim solutions to improve water quality where conventional water treatment is unavailable or inadequate, where people rely on untreated sources, or where stored water becomes contaminated because of unhygienic handling during transport or in the home. It may

be a compelling option for water treatment in outbreak situations or in humanitarian response (Lantagne & Clasen, 2012). Similar small technologies can also be used by travellers in areas where the drinking-water quality is uncertain (see also [section 6.12](#)). However, any household-based water treatment method must be used correctly and consistently over time to reduce waterborne exposures, suggesting user behaviour is a major potential limitation of household water treatment (Brown & Clasen, 2012). In practice, household water treatment's effectiveness as a public health intervention requires very high adherence that is difficult to achieve sustainably.

Not all household water treatment technologies are highly effective in reducing all classes of waterborne pathogens (bacteria, viruses, protozoa and helminths). Unless a treatment method is used that is effective against all pathogen classes, a combination of techniques (e.g. filtration followed by chemical disinfection) is recommended. Moreover, since performance also depends on the specifics of the methods/device, it is preferable to select a product that has been independently confirmed to meet performance or effectiveness criteria (see end of [section 7.3](#) for further information). Acceptability and adherence should also be considered carefully alongside microbiological performance in selecting household water treatment options (Bivins et al., 2019; Rao et al., 2025).

Definitions and descriptions of the various household water treatment technologies for microbial contamination follow:

- *Chemical disinfection:* Chemical disinfection of drinking-water includes any chlorine-based technology, such as chlorine dioxide, as well as ozone, some other oxidants and some strong acids and bases. Except for ozone, proper dosing of chemical disinfectants is intended to maintain a residual concentration in the water to provide some protection from post-treatment contamination during storage. Disinfection of household drinking-water in developing countries is done primarily with free chlorine, either in liquid form as hypochlorous acid (commercial household bleach or more dilute sodium hypochlorite solution between 0.5% and 1% hypochlorite marketed for household water treatment use) or in dry form as calcium hypochlorite or sodium dichloroisocyanurate. This is because these forms of free chlorine are convenient, relatively safe to handle, inexpensive and easy to dose. However, sodium trichloroisocyanurate and chlorine dioxide are also used in some household water treatment technologies. Proper dosing of chlorine for household water treatment is critical in order to provide sufficient free chlorine to maintain a residual during storage and use, important to prevent recontamination. Recommendations are to dose with free chlorine at about 2 mg/l to clear water (< 10 nephelometric turbidity units [NTU]) and twice that (4 mg/l) to turbid water (> 10 NTU). Although these free chlorine doses may lead to chlorine residuals that exceed the recommended chlorine residual for water that is centrally treated at the point of delivery, 0.2–0.5 mg/l, these doses are considered suitable for household water treatment to maintain a free chlorine residual of 0.2 mg/l in stored household water treated by chlorination. Further information on point-of-use chlorination can be found in the document *Preventing travellers' diarrhoea: how to make drinking water safe* (WHO, 2005).

As for central treatment, ozone for household water treatment must be generated on site, typically by corona discharge or electrolytically, both of which require electricity. As a result, ozone is not recommended for household water treatment because of the need for a reliable source of electricity to generate it, its complexity of generation and proper dosing in a small application, and its relatively high cost. Disinfection of drinking-water with iodine, which is also a strong oxidant, is generally not recommended for use by infants, pregnant women, those with a history of thyroid disease, nor for use by the general population for extended use unless the residual concentrations are controlled, because of concerns about adverse effects of excess intake on the thyroid gland (see also [section 6.12](#) and [Table 6.1](#)). Water treatment with bromine remains uncommon in small-scale applications, partly owing to concerns with the formation and the potential toxicity of brominated disinfection by-products (DBPs) that may exceed the risks of chlorinated DBPs. Various forms of silver (e.g. silver nanoparticles, silver nitrate) are sometimes promoted as a disinfectant, but it is not recommended as its efficacy is uncertain and it requires lengthy contact periods. More information about iodine, bromine and silver and their uses in drinking-water disinfection is available in the supporting document *Alternative drinking-water disinfectants: bromine, iodine and silver*; [Annex 1](#). Strong acids or bases are not recommended as chemical disinfectants for drinking-water, as they are hazardous chemicals that can alter the pH of the water to dangerously low or high levels. Strong acids or bases are not recommended as chemical disinfectants for drinking-water, as they are hazardous chemicals that can alter the pH of the water to dangerously low or high levels.

- *Membrane, porous ceramic or composite filters:* These are filters with defined pore sizes and include carbon block filters, porous ceramics, reactive membranes, polymeric membranes and fibre/cloth filters. They rely primarily on physical straining through a single porous surface or multiple surfaces having structured pores to physically remove and retain microbes by size exclusion. Some of these filters may also employ chemical antimicrobial or bacteriostatic surfaces (e.g. colloidal silver; see the previous paragraph) or chemical modifications to cause microbes to become adsorbed to filter media surfaces, to be inactivated or to inhibit growth. Cloth filters, such as those of sari cloth, have been recommended for reducing *Vibrio cholerae* in water. However, these filters reduce only vibrios associated with copepods, other large crustaceans or other large eukaryotes retained by the cloth. These cloths will not retain dispersed vibrios or other bacteria not associated with copepods, other crustaceans, suspended sediment or large eukaryotes, because the pores of the cloth fabric are much larger than the bacteria, allowing them to pass through. Most household filter technologies operate by gravity flow or by water pressure provided from a piped supply. However, some forms of ultrafiltration, nanofiltration and reverse osmosis filtration may require a reliable supply of electricity to operate.
- *Granular media filters:* Granular media filters include those containing sand or diatomaceous earth or others using discrete particles such as packed beds or layers of surfaces over or through which water is passed. These filters retain

microbes by a combination of physical and chemical processes, including physical straining, sedimentation and adsorption. Some may also employ chemically active antimicrobial or bacteriostatic surfaces or other chemical modifications. Other granular media filters are biologically active because they develop layers of microbes and their associated exopolymers on the surface of or within the granular medium matrix. This biologically active layer, called the *schmutzdecke* in conventional slow sand filters, retains microbes and often leads to their inactivation and biodegradation. A household-scale filter with a biologically active surface layer that can be dosed intermittently with water has been developed.

- *Solar disinfection:* There are a number of technologies using solar irradiation to disinfect water. Some use solar radiation to inactivate microbes in either dark or opaque containers by relying on heat from sunlight energy. Others, such as the solar water disinfection or SODIS system, use clear plastic containers penetrated by UV radiation from sunlight that rely on the combined action of the UV radiation, oxidative activity associated with dissolved oxygen and heat. Other physical forms of solar radiation exposure systems also employ combinations of these solar radiation effects in other types of containers, such as UV-penetrable plastic bags and panels.
- *UV irradiation:* A number of drinking-water treatment technologies use UV light radiation from UV lamps or UV light-emitting diodes (LEDs) to inactivate microbes. For household- or small-scale water treatment, most employ low-pressure mercury arc lamps producing monochromatic UV radiation at a germicidal wavelength of 254 nm. Typically, these technologies allow water in a vessel or in flow-through reactors to be exposed to the UV radiation from the UV lamps at sufficient dose (fluence) to inactivate waterborne pathogens. These systems need a reliable supply of electricity and have minimal maintenance requirements. Newer germicidal UV-LED devices (wavelengths of 255–285 nm) can be manufactured in small sizes, can be instantly cycled on/off, generate little heat, and can be run with a simple battery, solar cell or even a hand crank. Thus, UV-LED devices are appealing in communities where intermittent electricity is problematic.
- *Thermal (heat) technologies:* Thermal technologies are those whose primary mechanism for the destruction of microbes in water is heat produced by burning fuel. These include boiling and heating to pasteurization temperatures (typically > 63 °C for 30 minutes when applied to milk). The recommended procedure for water treatment is to raise the temperature so that a rolling boil is achieved, removing the water from the heat and allowing it to cool naturally, and then protecting it from post-treatment contamination during storage (see the supporting document *Boil water*; Annex 1). The previously-mentioned solar technologies using solar radiation for heat or for a combination of heat and UV radiation from sunlight are distinguished from this category.
- *Coagulation, precipitation and/or sedimentation:* Coagulation or precipitation is any device or method employing a natural or chemical coagulant or precipitant to coagulate or precipitate suspended particles, including microbes, to enhance their sedimentation. Sedimentation is any method for water treatment using the settling of suspended particles, including microbes, to remove them from the water.

This category includes simple sedimentation (i.e. that achieved without the use of a chemical coagulant). This method often employs a series of three pots or other water storage vessels in series, in which sedimented (settled) water is carefully transferred by decanting daily; by the third vessel, the water has been sequentially settled and stored a total of at least 2 days to reduce microbes. As microbial reductions via coagulation and precipitation and/or sedimentation are limited, these methods are not recommended for stand-alone drinking-water treatment. Rather, these methods may be used as a pretreatment step before filtration and/or chemical or UV disinfection.

- *Combination (multiple-barrier) treatment approaches:* These are any of the above technologies used together, either simultaneously or sequentially, for water treatment. These combination treatments include coagulation plus disinfection, media filtration plus disinfection or media filtration plus membrane filtration. Some are commercial single-use chemical products in the form of granules, powders or tablets containing a chemical coagulant, such as an iron or aluminium salt, and a disinfectant, such as chlorine. When added to water, these chemicals coagulate and flocculate impurities to promote rapid and efficient sedimentation and also deliver the chemical disinfectant (e.g. free chlorine) to inactivate microbes. Other combined treatment technologies are physical devices that include two or more stages of treatment, such as media or membrane filters or adsorbents to physically remove microbes and either chemical disinfectants or another physical treatment process (e.g. UV radiation) to kill any remaining microbes not physically removed by filtration or adsorption. Many of these combined household water treatment technologies are commercial products that can be purchased for household or other local use. It is important to choose commercial combination devices based on consideration of the treatment technologies that have been included in the device.

Estimated reductions of waterborne bacteria, viruses and protozoan parasites by several of the above-mentioned household water treatment technologies are summarized in [Table 7.8](#). Two categories of effectiveness are reported: baseline removals and maximum removals. Baseline removals are those typically expected in actual field practice when done by relatively unskilled persons who apply the treatment to raw waters of average and varying quality and where there are minimum facilities or supporting instruments to optimize treatment conditions and practices. Maximum removals are those possible when treatment is optimized by skilled operators who are supported with instrumentation and other tools to maintain the highest level of performance in waters of predictable and unchanging quality (e.g. a test water seeded with known concentrations of specific microbes). It should be noted that there are differences in the $\log_{10}$ reduction value performance of certain water treatment processes as specified for household water treatment in [Table 7.8](#) and for central treatment in [Table 7.7](#). These differences in performance by the same treatment technologies are to be expected, because central treatment is often applied to water that is of desirable quality for the treatment process, and treatment is applied by trained operators using properly engineered and operationally controlled processes. In contrast, household water treatment is often applied to waters having a range of water qualities, some of which are suboptimal for

best technology performance, and the treatment is often applied without the use of specialized operational controls by people who are relatively untrained and unskilled in treatment operations, compared with people managing central water treatment facilities. Further details on these treatment processes, including the factors that influence their performance and the basis for the $\log_{10}$ reduction value performance levels provided in Table 7.8, can be found in the supporting documents *Managing water in the home* and *Evaluating household water treatment options* (Annex 1).

Table 7.8 Reductions of bacteria, viruses and protozoa achieved by household water treatment technologies

Treatment process	Enteric pathogen group	Baseline removal (LRV)	Maximum removal (LRV)	Notes
Chemical disinfection				
Free chlorine disinfection	Bacteria	3	6	Free chlorine × contact time predicts efficacy; not effective against <i>Cryptosporidium</i> oocysts. Turbidity and chlorine-demanding solutes inhibit this process; hence, turbidity should be kept below 1 NTU to support effective disinfection. Where this is not practical, the aim should be to keep turbidities below 5 NTU, although disinfection should still be practised if 5 NTU cannot be achieved. At turbidities of more than 1 NTU, higher chlorine doses or contact times will be required ^a
	Viruses	3	6	
	Protozoa, non- <i>Cryptosporidium</i>	3	5	
	<i>Cryptosporidium</i>	0	1	
Membrane, porous ceramic or composite filtration				
Porous ceramic and carbon block filtration	Bacteria	2	6	Varies with pore size, flow rate, filter medium and inclusion of augmentation with silver or other chemical agents
	Viruses	1	4	
	Protozoa	4	6	
Membrane filtration (microfiltration, ultrafiltration, nanofiltration, reverse osmosis)	Bacteria	2 MF; 3 UF, NF or RO	4 MF; 6 UF, NF or RO	Varies with membrane pore size, integrity of filter medium and filter seals, and resistance to chemical and biological (“grow-through”) degradation; maximum reductions associated with filtered water turbidity of < 0.1 NTU ^a
	Viruses	0 MF; 3 UF, NF or RO	4 MF; 6 UF, NF or RO	
	Protozoa	2 MF; 3 UF, NF or RO	6 MF; 6 UF, NF or RO	
Fibre and fabric filtration (e.g. sari cloth filtration)	Bacteria	1	2	Particle or plankton association increases removal of microbes, notably copepod-associated guinea worm (<i>Dracunculus medinensis</i>) and plankton-associated <i>Vibrio cholerae</i> ; larger protozoa (> 20 µm) may be removed; ineffective for viruses, dispersed bacteria and small protozoa (e.g. <i>Giardia intestinalis</i> , 8–12 µm, and <i>Cryptosporidium</i> 4–6 µm)
	Viruses	0	0	
	Protozoa	0	1	

Table 7.8 (continued)

Treatment process	Enteric pathogen group	Baseline removal (LRV)	Maximum removal (LRV)	Notes
Granular media filtration				
Rapid granular, diatomaceous earth, biomass and fossil fuel-based (granular and powdered activated carbon, wood and charcoal ash, burnt rice hulls, etc.) filters	Bacteria	1	4+	Varies considerably with media size and properties, flow rate and operating conditions; some options are more practical than others for use in developing countries
	Viruses	1	4+	
	Protozoa	1	4+	
Household-level intermittently operated slow sand filtration	Bacteria	1	3	Varies with filter maturity, operating conditions, flow rate, grain size and filter bed contact time
	Viruses	0.5	2	
	Protozoa	2	4	
Solar disinfection				
Solar disinfection (solar UV radiation + thermal effects)	Bacteria	3	5+	Varies depending on oxygenation, sunlight intensity, exposure time, temperature, turbidity and size of water vessel (depth of water)
	Viruses	2	4+	
	Protozoa	2	4+	
UV light technologies using lamps				
UV irradiation	Bacteria	3	5+	Effectiveness of disinfection depends on delivered fluence (dose), which varies with intensity, exposure time and UV wavelength. Excessive turbidity and certain dissolved species inhibit this process; hence, turbidity should be kept below 1 NTU to support effective disinfection. Where this is not practical, turbidities should be kept below 5 NTU with higher fluences ^a
	Viruses	2	5+	
	Protozoa	3	5+	
Thermal (heat) technologies				
Thermal (e.g. boiling)	Bacteria	6	9+	Values are based on vegetative cells; spores are more resistant to thermal inactivation than are vegetative cells; treatment to reduce spores by boiling must ensure sufficient temperature and time
	Viruses	6	9+	
	Protozoa	6	9+	
Sedimentation				
Simple sedimentation	Bacteria	0	0.5	Effective due to settling of particle-associated and large (sedimentable) microbes; varies with storage time and particulates in the water
	Viruses	0	0.5	
	Protozoa	0	1	

Table 7.8 (continued)

Treatment process	Enteric pathogen group	Baseline removal (LRV)	Maximum removal (LRV)	Notes
Combination treatment approaches				
Flocculation plus disinfection systems (e.g. commercial powder sachets or tablets)	Bacteria	7	9	Some removal of <i>Cryptosporidium</i> possible by coagulation
	Viruses	4.5	6	
	Protozoa	3	5	

LRV, log₁₀ reduction value; MF, microfilter; NF, nanofilter; RO, reverse osmosis; UF, ultrafilter

^a See *Water quality and health – review of turbidity* (Annex 1).

The values in Table 7.8 do not account for post-treatment contamination of stored water, which may limit the effectiveness of some technologies where safe storage methods are not practised.

Validation, surveillance and certification of household water treatment and storage are recommended, just as they are for central water supplies and systems. The entities responsible for these activities for household water treatment systems may differ from those of central supplies. In addition, separate entities may be responsible for validation, independent surveillance and certification. Nevertheless, validation and surveillance as well as certification are critical for effective management of household and other point-of-use and point-of-entry drinking-water supplies and their treatment and storage technologies, just as they are for central systems (see sections 2.3 and 5.2.3).

Treatment technologies for non-piped water supplies that are manufactured by or obtained from commercial or other external sources should be certified to meet performance or effectiveness requirements or guidelines, preferably by an independent, accredited certification body. Products tested by the WHO International Scheme to Evaluate Household Water Treatment Technologies with data on their performance are available at the Scheme website (<https://www.who.int/tools/international-scheme-to-evaluate-household-water-treatment-technologies/products-evaluated>). If the treatment technologies are locally made and managed by the household itself, efforts to document effective construction and use and to monitor performance during use are recommended and encouraged. For further guidance to support monitoring and surveillance of household practices including water treatment, see the *Toolkit for monitoring and evaluating household water treatment and safe storage* (Annex 1).

Household storage and handling

Effective household water treatment should include safe storage, along with safe collection and transport. Households are responsible for maintaining water quality throughout these stages. Good hygiene practices are required, and should be supported through hygiene education.

Storage containers should be cleaned prior to water collection. Storage containers used should be safe for contact with drinking-water, for example, purpose-built safe storage containers or repurposed containers that were originally used for food or

beverages. Containers that previously held toxic substances such as bleach or pesticides should be avoided. Containers should be dedicated to drinking-water storage, and not used for other purposes, including storing water of lesser quality.

Ideally, containers should be designed to protect the water from contamination and also enable effective cleaning. For example, a bucket fitted with a tight-fitting lid and a spout for dispensing, or a narrow-mouthed, lidded container with a spout and an opening large enough to enable cleaning of the container, but still small enough to discourage scooping. Maintenance of a disinfectant residual in post-treatment water should also be considered.

Where such containers are impractical or unavailable, other water storage design options, such as large bottles, can also be considered. If water storage containers have large openings through which clean water is delivered but do not have a spout to dispense the water, utensils such as scoops or ladles could be used to dispense the water into containers for use, such as glasses, cups or smaller bottles. Such utensils should be cleaned and possibly sanitized on a regular basis. Direct hand contact with stored water should always be avoided to prevent contamination.

Storage containers should be cleaned and sanitized regularly. Spouts and related fittings should also be cleaned and maintained regularly. In addition, storage conditions should be considered: containers should be stored in a clean area, away from direct sunlight, ideally in a cool, dark, dry place, away from potential sources of contamination (e.g. raised off the ground, away from animals and young children).

7.4 Microbial monitoring

Microbial monitoring can be undertaken for a range of purposes, including:

- validation (see also [section 4.2.7](#));
- operational monitoring (see also [section 4.3.1](#));
- verification (see also [sections 2.4.1](#) and [4.3.2](#));
- surveillance (see [chapter 5](#));
- source water monitoring for identifying performance targets (see [sections 7.2](#) and [7.3.1](#));
- collecting data for QMRA (see also [section 7.2.3](#) and the supporting document *Quantitative microbial risk assessment: application to water safety management, Annex 1*).

Owing to issues relating to complexity, sensitivity of detection, cost and timeliness of obtaining results, testing for specific pathogens is generally limited to assessing raw water quality as a basis for identifying performance targets and validation, where monitoring is used to determine whether a treatment or other process is effective in removing target organisms. Very occasionally, pathogen testing may be performed to verify that a specific treatment or process has been effective. However, microbial testing included in verification, operational and surveillance monitoring is usually limited to testing for indicator organisms.

Different methods can be employed for the detection of bacteria, viruses, protozoan parasites and helminths in water. The use of some methods, such as microscopy, relies on detection of the whole particle or organism. Other methods, such as

molecular amplification using polymerase chain reaction (PCR), target the genomic material, deoxyribonucleic acid (DNA) or ribonucleic acid (RNA). Still other methods, such as immunological detection methods (e.g. enzyme-linked immunosorbent assay [ELISA]), target proteins. Culture-based methods, such as broth cultures or agar-based bacterial media and cell cultures for viruses and phages, detect organisms by infection or growth.

Culture in broth or on solid media is largely applied to determine the number of viable bacteria in water. The best known examples are culture-based methods for indicators such as *E. coli*. Viruses can be detected by several methods. Using cell culture, the number of infectious viruses in water can be determined. Alternatively, viral genomes can be detected by use of PCR. Protozoan parasites are often detected by immunomagnetic separation in combination with immunofluorescence microscopy. PCR can also be applied. Helminths are generally detected using microscopy.

In source investigation associated with waterborne infectious disease outbreaks, microbial hazards are generally typed by use of PCR, which can be followed by sequencing analysis to improve the precision of identification. One innovative approach is metagenome analysis (i.e. sequencing nucleic acid obtained directly from environmental samples). This can detect a multitude of microbial hazards in a water sample.

It is important to recognize that the different methods measure different properties of microorganisms. Culture-based methods detect living organisms, whereas microscopy, detection of nucleic acid and immunological assays measure the physical presence of microorganisms or components of them, and do not necessarily determine if what is detected is alive or infectious. This creates greater uncertainty regarding the significance of the human health risk compared with detection by culture-based methods. When using non-culture methods that do not measure in units indicative of culturability or infectivity, assumptions are often made about the fraction of pathogens or components detected that represent viable and infectious organisms.

The concept of using organisms such as *E. coli* as indicators of faecal pollution is a well-established practice in the assessment of drinking-water quality. The criteria determined for such faecal indicators are that they should not be pathogens themselves and they should:

- be universally present in faeces of humans and animals in large numbers;
- not multiply in natural waters;
- persist in water in a similar manner to faecal pathogens;
- be present in higher numbers than faecal pathogens;
- respond to treatment processes in a similar fashion to faecal pathogens;
- be readily detected by simple, inexpensive culture methods.

These criteria reflect an assumption that the same organism could be used as an indicator of both faecal pollution and treatment/process efficacy. However, it has become clear that one indicator cannot fulfil these two roles and that a range of organisms should be considered for different purposes (Table 7.9). For example, heterotrophic bacteria can be used as operational indicators of disinfection effectiveness and distribution system cleanliness; *Clostridium perfringens* and coliphage can be used to validate the effectiveness of treatment systems.

Table 7.9 Use of indicator organisms in monitoring

Microorganism(s)	Type of monitoring		Verification and surveillance
	Validation of process	Operational	
<i>E. coli</i> (or thermotolerant coliforms)	Not applicable	Not applicable	Faecal indicator
Total coliforms	Not applicable	Not applicable	Not applicable
Heterotrophic plate counts	Indicator for effectiveness of disinfection of bacteria	Indicator for effectiveness of disinfection processes and cleanliness and integrity of distribution systems	Not applicable
<i>Clostridium perfringens</i> ^a	Indicator for effectiveness of disinfection and physical removal processes for viruses and protozoa	Not applicable	Not applicable ^b
Coliphages <i>Bacteroides fragilis</i> phages Enteric viruses	Indicator for effectiveness of disinfection and physical removal processes for viruses	Not applicable	Not applicable ^b

^a Use of *Clostridium perfringens* for validation will depend on the treatment process being assessed.

^b Could be used for verification where source waters are known to be contaminated with enteric viruses and protozoa or where such contamination is suspected as a result of impacts of human faecal waste.

Escherichia coli has traditionally been used to monitor drinking-water quality, and it remains an important parameter in monitoring undertaken as part of verification or surveillance. Thermotolerant coliforms can be used as an alternative to the test for *E. coli* in many circumstances. Water intended for human consumption should contain no faecal indicator organisms. In the majority of cases, monitoring for *E. coli* or thermotolerant coliforms provides a high degree of assurance because of their large numbers in polluted waters.

However, increased attention has focused on the shortcomings of traditional indicators, such as *E. coli*, as indicator organisms for enteric viruses and protozoa. Viruses and protozoa more resistant to conventional environmental conditions or treatment technologies, including filtration and disinfection, may be present in treated drinking-water in the absence of *E. coli*. Retrospective studies of waterborne disease outbreaks have shown that complete reliance on assumptions surrounding the absence or presence of *E. coli* may not ensure safety. Under certain circumstances, it may be desirable to include more resistant microorganisms, such as bacteriophages and/or bacterial spores, as indicators of persistent microbial hazards. Their inclusion in monitoring programmes, including control and surveillance programmes, should be evaluated in relation to local circumstances and scientific understanding. Such circumstances could include the use of source water known to be contaminated with enteric viruses and parasites or where such contamination is suspected as a result of the impacts of human and livestock waste.

Further discussion on indicator organisms is contained in the supporting document *Assessing microbial safety of drinking water* (Annex 1).

Table 7.10 Guideline values for verification of microbial quality^a (see also [Table 5.2](#))

Organisms	Guideline value
All water directly intended for drinking	
<i>E. coli</i> or thermotolerant coliform bacteria ^{b,c}	Must not be detectable in any 100 ml sample
Treated water entering the distribution system	
<i>E. coli</i> or thermotolerant coliform bacteria ^b	Must not be detectable in any 100 ml sample
Treated water in the distribution system	
<i>E. coli</i> or thermotolerant coliform bacteria ^b	Must not be detectable in any 100 ml sample

^a Immediate investigative action must be taken if *E. coli* are detected.

^b Although *E. coli* is the more precise indicator of faecal pollution, the count of thermotolerant coliform bacteria is an acceptable alternative. If necessary, proper confirmatory tests must be carried out. Total coliform bacteria are not acceptable as an indicator of the sanitary quality of water supplies, particularly in tropical areas, where many bacteria of no sanitary significance occur in almost all untreated supplies.

^c It is recognized that in the great majority of rural water supplies, especially in developing countries, faecal contamination is widespread. Especially under these conditions, medium-term targets for the progressive improvement of water supplies should be set.

[Table 7.10](#) presents guideline values for verification of the microbial quality of drinking-water. Individual values should not be used directly from the table. The guideline values should be used and interpreted in conjunction with the information contained in these Guidelines and other supporting documentation.

A consequence of variable susceptibility to pathogens is that exposure to drinking-water of a particular quality may lead to different health effects in different populations. For derivation of national standards, it is necessary to define reference populations or, in some cases, to focus on specific vulnerable subpopulations. National or local authorities may wish to apply specific characteristics of their populations in deriving national standards.

7.5 Methods of detection of faecal indicator organisms

Analysis for faecal indicator organisms provides a sensitive, although not the most rapid, indication of pollution of drinking-water supplies. Because the growth medium and the conditions of incubation, as well as the nature and age of the water sample, can influence the species isolated and the count, microbiological examinations may have variable accuracy. This means that the standardization of methods and of laboratory procedures is of great importance if criteria for the microbial quality of water are to be uniform in different laboratories and internationally.

International standard methods should be evaluated under local circumstances before being adopted. Established standard methods are available, such as those of the International Organization for Standardization (ISO) ([Table 7.11](#)) or methods of equivalent efficacy and reliability. It is desirable that established standard methods be used for routine examinations. Whatever method is chosen for detection of *E. coli* or thermotolerant coliforms, the importance of “resuscitating” or recovering environmentally damaged or disinfectant-damaged strains must be considered.

Table 7.11 International Organization for Standardization (ISO) standards for detection and enumeration of faecal indicator organisms in water

ISO standard	Title (water quality)
6461-1:1986	Detection and enumeration of the spores of sulfite-reducing anaerobes (clostridia)—Part 1: Method by enrichment in a liquid medium
6461-2:1986	Detection and enumeration of the spores of sulfite-reducing anaerobes (clostridia)—Part 2: Method by membrane filtration
7704:2023	Requirements for the performance testing of membrane filters use for direct enumeration of microorganisms by culture methods
9308-1:2014	Enumeration of <i>Escherichia coli</i> and coliform bacteria—Part 1: Membrane filtration method for waters with low bacterial background flora
9308-2:2012	Enumeration of <i>Escherichia coli</i> and coliform bacteria—Part 2: Most probable number method
9308-3:1998	Detection and enumeration of <i>Escherichia coli</i> and coliform bacteria—Part 3: Miniaturized method (most probable number) for the detection and enumeration of <i>E. coli</i> in surface and waste water
10705-1:1995	Detection and enumeration of bacteriophages—Part 1: Enumeration of F-specific RNA bacteriophages
10705-2:2000	Detection and enumeration of bacteriophages—Part 2: Enumeration of somatic coliphages
10705-3:2003	Detection and enumeration of bacteriophages—Part 3: Validation of methods for concentration of bacteriophages from water
10705-4:2001	Detection and enumeration of bacteriophages—Part 4: Enumeration of bacteriophages infecting <i>Bacteroides fragilis</i>

7.6 Identifying local actions in response to microbial water quality problems and emergencies

During an emergency in which there is evidence of faecal contamination of the drinking-water supply, it may be necessary either to modify the treatment of existing sources or to temporarily use alternative sources of drinking-water. It may be necessary to increase disinfection at source, following treatment or during distribution.

If microbial quality cannot be maintained, it may be necessary to advise consumers to boil the water during the emergency (see [section 7.6.1](#)). Initiating superchlorination and undertaking immediate corrective measures may be preferable where the speed of response is sufficient to prevent significant quantities of contaminated water from reaching consumers.

During outbreaks of potentially waterborne disease or when faecal contamination of a drinking-water supply is detected, the concentration of free chlorine should be increased to greater than 0.5 mg/l throughout the system as a minimum immediate response. It is most important that decisions are taken in consultation with public health authorities and, where appropriate, civil authorities (see also [sections 4.4.3, 6.8 and 8.7](#)).

7.6.1 *Boil water advisories*

Boil water advisories share many features with water avoidance advisories used in the event of serious chemical contamination (see [section 8.7](#)). Water suppliers in conjunction with public health authorities should develop protocols for boil water orders. Protocols should be prepared prior to the occurrence of incidents and incorporated within management plans. Decisions to issue advisories are often made within a short period of time, and developing responses during an event can complicate decision-making, compromise communication and undermine public confidence. In addition to the information discussed in [section 4.4.3](#), the protocols should deal with:

- criteria for issuing and rescinding advisories;
- information to be provided to the general public and specific groups;
- activities affected by the advisory.

Protocols should identify mechanisms for the communication of boil water advisories. The mechanisms may vary, depending on the nature of the supply and the size of the community affected, and could include:

- media releases through television, radio and newspapers;
- telephone, e-mail and fax contact of specific facilities, community groups and local authorities;
- posting of notices in conspicuous locations;
- personal delivery;
- mail delivery.

The methods chosen should provide a reasonable surety that all of those affected by the advisory, including residents, workers and travellers, are notified as soon as possible.

Boil water advisories should indicate that the water can be made safe by bringing it to a rolling boil. After boiling, the water should be allowed to cool down on its own without the addition of ice. This procedure is effective at all altitudes and with turbid water. The types of event that should lead to consideration of boil water advisories include:

- substantial deterioration in source water quality;
- major failures associated with treatment processes or the integrity of distribution systems;
- inadequate disinfection;
- detection of pathogens or faecal indicator organisms in drinking-water;
- epidemiological evidence suggesting that drinking-water is responsible for an outbreak of illness.

Boil water advisories are a serious measure that can have substantial adverse consequences. Advice to boil water can have negative public health consequences through scalding and increased anxiety, even after the advice is rescinded. In addition, not all consumers will follow the advice issued, even at the outset; if boil water advisories are issued frequently or are left in place for long periods, compliance will decrease. Hence, advisories should be issued only after careful consideration of all available information by the public health authority and the incident response team and conclusion that

there is an ongoing risk to public health that outweighs any risk from the advice to boil water. For example, where microbial contamination is detected in samples of drinking-water, factors that should be considered in evaluating the need for an advisory include:

- reliability and accuracy of results;
- vulnerability of source water to contamination;
- evidence of deterioration in source water quality;
- source water monitoring results;
- results from operational monitoring of treatment and disinfection processes;
- disinfectant residuals;
- physical integrity of the distribution system.

The available information should be reviewed to determine the likely source of the contamination and the likelihood of recurrence or persistence.

When issued, a boil water advisory should be clear and easily understood by recipients, or it may be ignored. Advisories should normally include a description of the problem, potential health risks and symptoms, activities that are affected, investigative actions and corrective measures that have been initiated, as well as the expected time to resolve the problem. If the advisory is related to an outbreak of illness, specific information should be provided on the nature of the outbreak, the illness and the public health response.

Boil water advisories should identify both affected and unaffected uses of drinking-water supplies. Generally, the advisory will indicate that unboiled water should not be used for drinking, preparing cold drinks, making ice, preparing or washing food or brushing teeth. Unless heavily contaminated, unboiled water will generally be safe for bathing (providing swallowing of water is avoided) and washing clothes. A boil water advisory could include specific advice for vulnerable subpopulations, such as pregnant women and others who might be immunocompromised. Specific advice should also be provided to facilities such as dental clinics, dialysis centres, doctors' offices, hospitals and other health-care facilities, child-care facilities, schools, food suppliers and manufacturers, hotels, restaurants and operators of public swimming pools and spas.

Provision of alternative supplies of drinking-water, such as bottled water or bulk water, should be considered when temporary boil water advisories are in place. The protocols should identify sources of alternative supplies and mechanisms for delivery.

Protocols should include criteria for rescinding boil water advisories. Depending on the reason for issuing the advisory, the criteria could include one or more of the following:

- evidence that source water quality has returned to normal;
- correction of failures associated with treatment processes or distribution systems;
- correction of faults in disinfection processes and restoration of normal disinfectant residuals;

- where detection of microbial contamination in drinking-water initiated the advisory, evidence that this contamination has been removed or inactivated;
- evidence that sufficient mains flushing or water displacement has removed potentially contaminated water and biofilms;
- epidemiological evidence indicating that an outbreak has concluded.

When boil water advisories are rescinded, information should be provided through similar channels and to the same groups that received the original advice. In addition, operators/managers or occupants of large buildings and buildings with storage tanks should be advised of the need to ensure that storages and extensive internal distribution systems are thoroughly flushed before normal uses are restored.

7.6.2 Actions following an incident

It is important that any incident be properly investigated and remedial action instigated to prevent its recurrence. The WSP will require revision to take into account the experience gained, and the findings may also be of importance in informing actions regarding other water supplies to prevent a similar event from occurring elsewhere. Where appropriate, epidemiological investigations by the health authority will also help to inform actions for the future.

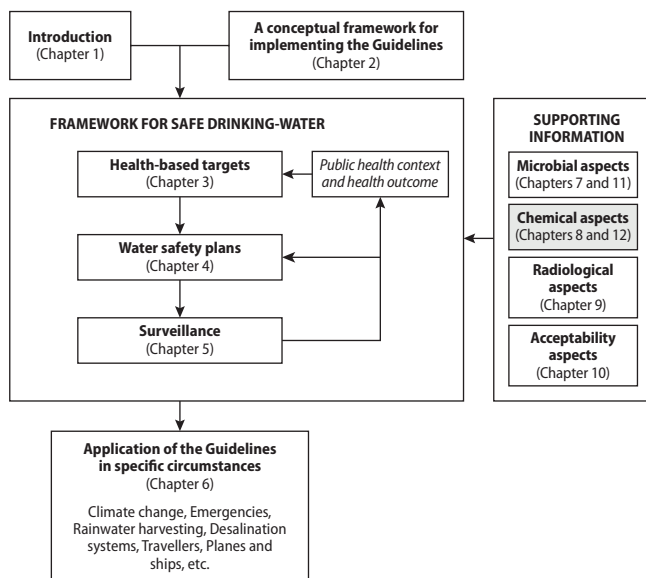
8

Chemical aspects

Most chemicals arising in drinking-water are of health concern only after extended exposure of years, rather than months. The principal exception is nitrate. Typically, changes in water quality occur progressively, except for those substances that are discharged or leach intermittently to flowing surface waters or ground-water supplies from, for example, contaminated landfill sites.

In some cases, there are groups of chemicals that arise from related sources—for example, disinfection by-products (DBPs)—and it may not be necessary to set standards for all of the DBPs for which there are guideline values. If chlorination is practised, the trihalomethanes (THMs) and haloacetic acids (HAAs) will be the main DBPs. If bromide is present, brominated as well as chlorinated DBPs will be produced. Maintaining THM and HAA concentrations below the guideline values by controlling precursor compounds will provide adequate control over other chlorination by-products.

Several of the inorganic elements for which guideline values have been established are recognized to be essential elements in human nutrition. No attempt has been made here at this time to define a minimum desirable concentration of such substances in drinking-water, although the issue of nutritional essentiality is considered during the guideline development process.



Fact sheets for individual chemical contaminants are provided in [chapter 12](#). For those contaminants for which a guideline value or health-based value has been established (see [section 8.2](#)), the fact sheets include a brief toxicological overview of the chemical, the basis for guideline derivation, treatment performance and analytical limit of detection. More detailed chemical reviews are available (<https://www.who.int/teams/environment-climate-change-and-health/water-sanitation-and-health/chemical-hazards-in-drinking-water>).

8.1 Chemical hazards in drinking-water

A few chemical contaminants have been shown to cause adverse health effects in humans as a consequence of prolonged exposure through drinking-water. However, this is only a very small proportion of the chemicals that may reach drinking-water from various sources.

The lists of chemicals addressed in these Guidelines do not imply that all of these chemicals will always be present or that other chemicals not addressed will be absent.

The substances considered here have been assessed for possible health effects, and guideline values have been established only on the basis of health concerns. Additional consideration of the potential effects of chemical contaminants on the acceptability (i.e. taste, odour and appearance) of drinking-water to consumers is included in [chapter 10](#). Some substances of health concern have effects on the acceptability of drinking-water that would normally lead to rejection of the water at concentrations significantly lower than those of health concern. For such substances, no formal guideline value is usually proposed, but a health-based value (see [section 8.2](#)) may be needed, for instance, in order to assist in judging the response required when problems are encountered and in some cases to provide reassurance to health authorities and consumers with regard to possible health risks.

Regulators are required to establish health-based targets that must be met through water safety plans (WSPs). In the case of chemical contaminants, these are normally based on the guideline value, which is, in turn, based on health-related endpoints. In this case, the guideline value and the local water quality target are similar, but not necessarily identical, because the latter value may need to be adjusted to take into account local sociocultural, economic and environmental/geological circumstances, as indicated in [section 2.6](#). Guideline values provide a benchmark for the development of local water quality targets for chemicals (usually a national standard expressing a maximum allowable concentration). Guideline values may not directly reflect the target of 10^{-6} disability-adjusted life year (DALY), as these are frequently derived based on evidence indicating a no-adverse effect or negligible risk level. Some guideline values are based on extrapolation of the risk of cancer from exposures at which this can be measured to low exposures where measurement is currently not possible.

In [section 2.6](#), it is stated that “In developing national drinking-water standards based on these Guidelines, it will be necessary to take account of a variety of environmental, social, cultural, economic, dietary and other conditions affecting potential exposure. This may lead to national standards that differ appreciably from these

Guidelines.” This is particularly applicable to chemical contaminants, for which there is a long list, and setting standards for, or including, all of them in monitoring programmes is neither feasible nor desirable.

The probability that any particular chemical may occur in significant concentrations in any particular setting must be assessed on a case-by-case basis. The presence of certain chemicals may already be known within a particular country, but others may be more difficult to assess.

It is important that chemical contaminants be prioritized so that the most important in the country or local region are considered for inclusion in national standards and monitoring programmes.

In most countries, whether developing or industrialized, water sector professionals are likely to be aware of a number of chemicals that are present in significant concentrations in some drinking-water supplies. A body of local knowledge that has been built up by practical experience over a period of time is invaluable. Hence, the presence of a limited number of chemical contaminants in drinking-water is usually already known in many countries and in many local systems. Significant problems, even crises, can occur, however, when chemicals posing high health risk are widespread but their presence is unknown, because their long-term health effect is caused by chronic exposure as opposed to acute exposure. Such has been the case of arsenic in groundwater in Bangladesh and West Bengal, India, for example.

For many contaminants, there will be exposure from sources other than drinking-water, and this may need to be taken into account when setting, and considering the need for, standards. It may also be important when considering the need for monitoring. In some cases, drinking-water will be a minor source of exposure, and controlling levels in water will have little impact on overall exposure. In other cases, controlling a contaminant in water may be the most cost-effective way of reducing exposure. Drinking-water monitoring strategies should therefore not be considered in isolation from other potential routes of exposure to chemicals in the environment.

The scientific basis for each of the guideline values is summarized in [chapter 12](#). This information is important in helping to adapt guideline values to suit national requirements or for assessing the health significance of a contaminant that is of a higher concentration than the guideline value.

Chemical contaminants in drinking-water may be categorized in various ways; however, the most appropriate is to consider the primary source of the contaminant—that is, to group chemicals according to where control may be effectively exercised. This aids in the development of approaches that are designed to prevent or minimize contamination, rather than those that rely primarily on the measurement of contaminant levels in final waters.

In general, approaches to the management of chemical hazards in drinking-water vary between those where the source water is a significant contributor (with control effected, for example, through source water selection, pollution control, treatment or blending) and those from materials and chemicals used in the production and distribution of drinking-water (controlled by process optimization or product specification). In these Guidelines, chemicals are therefore divided into five major source groups, as shown in [Table 8.1](#).

Table 8.1 Categorization of source of chemical constituents

Source of chemical constituents	Examples of sources
Naturally occurring	Rocks, soils and the effects of the geological setting and climate; eutrophic water bodies (also influenced by sewage inputs and agricultural runoff)
Industrial sources and human dwellings	Mining (extractive industries) and manufacturing and processing industries, sewage (including a number of contaminants of emerging concern), solid wastes, urban runoff, fuel leakages
Agricultural activities	Manures, fertilizers, intensive animal practices and pesticides
Water treatment or materials in contact with drinking-water	Coagulants, DBPs, piping materials
Pesticides used in water for public health	Larvicides used in the control of insect vectors of disease

Categories may not always be clear-cut. The group of naturally occurring contaminants, for example, includes many inorganic chemicals that are found in drinking-water as a consequence of release from rocks and soils by rainfall, some of which may become problematical where there is environmental disturbance, such as in mining areas.

8.2 Derivation of chemical guideline values and health-based values

In order for a particular chemical constituent to be evaluated to determine whether a guideline value or health-based value should be derived, one of the following criteria must be satisfied:

- There is credible evidence of occurrence of the chemical in drinking-water, combined with evidence of actual or potential toxicity.
- The chemical is of significant international concern.

Guideline values are derived for many chemical constituents of drinking-water. A guideline value normally represents the concentration of a constituent that does not result in any significant risk to health over a lifetime of consumption. A number of provisional guideline values have been established at concentrations that are reasonably achievable to meet practical treatment approaches or at the limit of detection of widely available analytical approaches; in these cases, the guideline value is above the concentration that would normally represent the calculated health-based value. Guideline values are also designated as provisional when there is a high degree of uncertainty in the toxicological and health data (see also [section 8.2.5](#)).

For some chemicals, no formal guideline value is proposed, on the grounds that occurrence is only at concentrations well below those that would be of concern for health. Establishing a formal guideline value for such substances could encourage some Member States to incorporate the value into their national standards when this is neither necessary nor appropriate. However, to provide guidance for Member States

should the chemical be found in drinking-water or in source water in the hazard identification phase of developing a WSP, a health-based value has been determined.

In addition, health-based values for acute exposures are now being developed for a small number of substances that may be implicated in emergency situations as a result of a spill, usually to surface water sources. The derivation of these acute health-based values is explained in [section 8.7.5](#).

Where data are inadequate to establish a formal guideline value, a provisional reference value may be proposed. These values are derived from the limited data and are too uncertain to be used for developing regulations or standards. Nevertheless these “bounding values” may be useful to guide actions by Member States in the event of need.

An explanation of these different values and considerations for their application are outlined in [Table 8.2](#).

It is not appropriate to set guideline values for pesticides used for vector control because the public health benefit from preventing these diseases (e.g. malaria, dengue) outweighs the minimal risk posed by ingesting these chemicals when applied at the rates registered for use in drinking-water by the relevant regulatory authority and/or the WHO prequalification programme (for more information on WHO's prequalification programme, see [section 8.6](#)). Nevertheless, it is valuable to provide information regarding their safety in use. Accordingly, information to support a safety assessment has been included for some pesticides that may potentially be used in drinking-water contexts, which may include application in drinking-water sources and containers (see [section 8.6](#)).

8.2.1 Approaches taken

Two approaches to the derivation of guideline values are used: one for “threshold chemicals” and the other for “non-threshold chemicals” (mostly genotoxic carcinogens).

It is generally considered that the initiating event in the process of genotoxic chemical carcinogenesis is the induction of a mutation in the genetic material (deoxyribonucleic acid [DNA]) of somatic cells (i.e. cells other than ova or sperm) and that there is a theoretical risk at any level of exposure (i.e. no threshold). In contrast, there are carcinogens that are capable of producing tumours in experimental animals or humans without exerting a genotoxic activity, but acting through an indirect mechanism. It is generally believed that a demonstrable threshold dose exists for non-genotoxic carcinogens.

In deriving guideline values for carcinogens, consideration is given to the potential mechanisms by which the substance may cause cancer, in order to decide whether a threshold or non-threshold approach should be used (see [sections 8.2.2](#) and [8.2.3](#)).

The evaluation of the potential carcinogenicity of chemical substances is usually based on long-term laboratory animal studies. Sometimes data are available on carcinogenicity in humans, mostly from occupational exposure.

Table 8.2 Use and types of chemical values

Type of value	Comments	Application
Guideline value	Represents the concentration of a constituent that does not result in any significant risk to health. This is for a lifetime of consumption unless indicated.	For adaptation into national drinking-water standards if warranted by local circumstances including data on occurrence.
Health-based value	Represents the concentration of a constituent that does not result in any significant risk to health over a lifetime of consumption unless indicated.	Where a provisional guideline value is presented along with the health-based value, the provisional guideline value is recommended for adaptation into national drinking-water standards because the provisional guideline value takes into account practical issues associated with monitoring, management or competing risks. See Table 8.4 for more information on provisional guideline values. Health-based values may also be presented without a provisional guideline value. These values can be useful when there is a local concern, such as to inform temporary advisories or limits during a spill situation or to communicate health risks in the case of contaminants that can cause aesthetic issues at lower levels than the health-based value.
Provisional reference value	As above, but derived with limited data for a very limited number of substances.	Allows for an interim risk assessment, as above, but with greater uncertainty and therefore not to be used for developing regulations or standards.

On the basis of the available evidence, the International Agency for Research on Cancer (IARC) categorizes chemical substances with respect to their potential carcinogenic risk into the following groups:

- Group 1: the agent is carcinogenic to humans
- Group 2A: the agent is probably carcinogenic to humans
- Group 2B: the agent is possibly carcinogenic to humans
- Group 3: the agent is not classifiable as to its carcinogenicity to humans
- Group 4: the agent is probably not carcinogenic to humans

According to IARC, these classifications represent a first step in carcinogenic risk assessment, which leads to a second step of quantitative risk assessment where possible. In establishing guideline values for drinking-water, the IARC evaluation of carcinogenic compounds, where available, is taken into consideration.

8.2.2 Threshold chemicals

For most kinds of toxicity, it is believed that there is a dose below which no adverse effect will occur. For chemicals that give rise to such toxic effects, a tolerable daily intake

(TDI) should be derived as follows, using the most sensitive end-point in the most relevant study, preferably involving administration in drinking-water:

$$\text{TDI} = \frac{\text{NOAEL or LOAEL or BMDL}}{\text{UF and/or CSAF}}$$

where:

- NOAEL = no-observed-adverse-effect level
- LOAEL = lowest-observed-adverse-effect level
- BMDL = lower confidence limit on the benchmark dose
- UF = uncertainty factor
- CSAF = chemical-specific adjustment factor

The guideline value (GV) is then derived from the TDI as follows:

$$\text{GV} = \frac{\text{TDI} \times \text{bw} \times \text{P}}{\text{C}}$$

where:

- bw = body weight (see below)
- P = fraction of the TDI allocated to drinking-water
- C = daily drinking-water consumption (see below)

Tolerable daily intake

The TDI is an estimate of the amount of a substance in food and drinking-water, expressed on a body weight basis (milligram or microgram per kilogram of body weight), that can be ingested over a lifetime without appreciable health risk, and with a margin of safety.

Acceptable daily intakes (ADIs) are established for food additives and pesticide residues that occur in food for necessary technological purposes or plant protection reasons. For chemical contaminants, which usually have no intended function in drinking-water, the term “tolerable daily intake” is more appropriate than “acceptable daily intake”, as it signifies permissibility rather than acceptability.

Over many years, the Joint Food and Agriculture Organization of the United Nations (FAO)/World Health Organization (WHO) Expert Committee on Food Additives (JECFA) and the Joint FAO/WHO Meeting on Pesticide Residues (JMPR) have developed certain principles in the derivation of ADIs (FAO/WHO, 2009). These principles have been adopted, where appropriate, in the derivation of TDIs used in developing guideline values for drinking-water quality.

As TDIs are regarded as representing a tolerable intake for a lifetime, they are not so precise that they cannot be exceeded for short periods of time. Short-term exposure to levels exceeding the TDI is not a cause for concern, provided the individual's intake averaged over longer periods of time does not appreciably exceed the level set. The large uncertainty factors generally involved in establishing a TDI (see below) serve to provide assurance that exposure exceeding the TDI for short periods is unlikely to

have any deleterious effects upon health. However, consideration should be given to any potential acute effects that may occur if the TDI is substantially exceeded for short periods of time.

No-observed-adverse-effect level and lowest-observed-adverse-effect level

The NOAEL is defined as the highest dose or concentration of a chemical in a single study, found by experiment or observation, that causes no detectable adverse health effect. Wherever possible, the NOAEL is based on long-term studies, preferably of ingestion in drinking-water. However, NOAELs obtained from short-term studies and studies using other sources of exposure (e.g. food, air) may also be used.

If a NOAEL is not available, a LOAEL may be used, which is the lowest observed dose or concentration of a substance at which there is a detectable adverse health effect. When a LOAEL is used instead of a NOAEL, an additional uncertainty factor is normally applied (see below).

Benchmark dose

Increasingly, the preferred approaches for the derivation of TDIs/ADIs for threshold effects include the benchmark dose (BMD) or the lower confidence limit on the benchmark dose (BMDL) (IPCS, 1994). When appropriate data for mathematical modelling of dose–response relationships are available, BMDLs are used as alternatives to NOAELs in the calculation of health-based guideline values. In such a case, use of the BMDL could eliminate the need for application of an additional uncertainty factor to the LOAEL. The BMDL is the lower confidence limit of the dose that produces a small increase (e.g. 5% or 10%) in the level of adverse effects. The BMDL is derived on a quantitative basis using data from the entire dose–response curve for the critical effect rather than from a single dose at the NOAEL or LOAEL and accounts for the statistical power and quality of the data (IPCS, 2009).

Uncertainty factors

The application of uncertainty or safety factors has been traditionally and successfully used in the derivation of ADIs and TDIs for food additives, pesticides and environmental contaminants. The derivation of these factors requires expert judgement and careful consideration of the available scientific evidence.

In the derivation of guideline values, uncertainty factors are applied to the NOAEL, LOAEL or BMD/BMDL for the response considered to be the most biologically significant.

In relation to exposure of the general population, the NOAEL or BMD/BMDL for the critical effect in experimental animals is normally divided by an uncertainty factor of 100. This comprises two 10-fold factors, one for interspecies differences and one for interindividual variability in humans (Table 8.3). Extra uncertainty factors may be incorporated to allow for database deficiencies and for the severity or irreversibility of effects.

Factors lower than 10 are used, for example, for interspecies variation when humans are known to be less sensitive than the experimental animal species studied. Inadequate studies or databases include those where a LOAEL is used instead of a NOAEL and studies considered to be shorter in duration than desirable. Situations

Table 8.3 Source of uncertainty in derivation of guideline values

Source of uncertainty	Uncertainty factor
Interspecies variation (extrapolating from experimental animals to humans)	1–10
Intraspecies variation (accounting for individual variations within humans)	1–10
Adequacy of studies or database	1–10
Nature and severity of effect	1–10

in which the nature or severity of effect might warrant an additional uncertainty factor include studies in which the end-point is malformation of a fetus or in which the end-point determining the NOAEL is directly related to possible carcinogenicity. In the latter case, an additional uncertainty factor is usually applied for carcinogenic compounds for which the guideline value is derived using a TDI approach rather than a theoretical risk extrapolation approach.

For substances for which the uncertainty factors are equal to or greater than 1000, guideline values are designated as provisional in order to emphasize the higher level of uncertainty inherent in these values. A high uncertainty factor indicates that the guideline value may be considerably lower than the concentration at which health effects would actually occur in a real human population. Guideline values with high uncertainty are more likely to be modified as new information becomes available.

The selection and application of uncertainty factors are important in the derivation of guideline values for chemicals, as they can make a considerable difference in the values set. For contaminants for which there is sufficient confidence in the database, the guideline value is derived using a small uncertainty factor. For most contaminants, however, there is greater scientific uncertainty, and a relatively large uncertainty factor is used. The use of uncertainty factors enables the particular attributes of the chemical and the data available to be considered in the derivation of guideline values.

Use of chemical-specific adjustment factors instead of uncertainty factors

Approaches to the derivation of TDIs are increasingly being based on understanding of a chemical's mode of action in order to reduce reliance on default assumptions. This approach provides a departure from the use of default uncertainty factors (such as a simple 10 for interspecies variation and 10 for intraspecies variation) and relies on the use of quantitative toxicokinetic and toxicodynamic data to derive CSAFs for use in interspecies and intraspecies extrapolations (IPCS, 2005; Bhat et al., 2017). Previously, CSAFs were called “data-derived uncertainty factors”. The part of the CSAF approach that is at present best developed is the use of physiologically based pharmacokinetic models to replace the default values for extrapolation between species and between differing routes of exposure (e.g. inhalation to oral).

Relative source allocation

Drinking-water is usually not the only source of human exposure to the chemicals for which guideline values have been derived. In many cases, the exposure to or intake of chemical contaminants from drinking-water is much lower than that from other sources, such as food, air and consumer products. Some consideration of the

proportion of the ADI or TDI that may be attributed to different sources is therefore needed in developing guideline values and risk management strategies. This approach ensures that total daily intake from all sources (including drinking-water containing concentrations of the chemical at or near the guideline value) does not exceed the ADI or TDI.

Wherever possible or in an ideal situation, derivation of guideline values uses data on the proportion of total daily intake normally ingested in drinking-water (based on mean levels in food, drinking-water, consumer products, soil and air), or data on intakes estimated on the basis of physical and chemical properties of the substances of concern. As the primary sources of exposure to chemicals are generally food (e.g. pesticide residues) and water, it is important to quantify, whenever possible, the exposures from both sources. To inform this process, it is desirable to collect as much high-quality data as possible on food intake in different parts of the world as possible. The data collected can then be used to estimate the proportion of the intake that comes from food and the proportion that comes from drinking-water. However, for most contaminants, data from the various exposure sources, most notably food and drinking-water, are available only from developed countries.

In the absence of adequate exposure data or where documented evidence is available regarding widespread presence in one or more of the other media (i.e. air, food, soil or consumer products), the normal allocation of the total daily intake to drinking-water is 20% (floor value), which reflects a reasonable level of exposure based on broad experience, while still being protective (Krishnan & Carrier, 2013). This value reflects a change from the previous allocation of 10%, which was found to be excessively conservative. As chemicals are progressively reassessed, overall exposure will be reconsidered, and a change in the default allocation factor from 10% to 20% will be made, if appropriate. Therefore, not all older guideline values reflect this change. In some circumstances, there is clear evidence that water is the main (and possibly only) source of exposure, such as for some of the DBPs; the allocation in such cases may be as high as 80% (ceiling value), which still allows for some exposure from other sources (Krishnan & Carrier, 2013). Where chemical and context-specific allocation factors can be developed using exposure data or models, the allocation factor applied should still be bounded by the floor and ceiling values (i.e. 20–80%).

For pesticides, even when available food exposure data suggest that exposure via this route is minimal, the default allocation factor of 20% is used to account for the fact that available food exposure data do not generally include information from developing countries, where exposure via this route may be higher.

A detailed explanation of the reasoning behind the choice of allocation factor is an essential component of the evaluation. This assists Member States in making appropriate decisions about incorporating or adapting guideline values into national standards where local circumstances need to be taken into account. It also provides assistance in making decisions regarding potential risks when a guideline value is exceeded. As a general principle, efforts should be made to keep contaminant concentrations as low as possible and not allow increases up to the guideline value.

Although the values chosen are, in most cases, sufficient to account for additional routes of intake (i.e. inhalation and dermal absorption) of contaminants in water,

under certain circumstances (e.g. limited ventilation), authorities may wish to take inhalation and dermal exposure into account in adapting the guideline values to local conditions (see [section 8.2.9](#)).

Some elements are essential for human nutrition. In developing guideline values and in considering allocation factors, it is necessary to take into account the recommended minimum daily intake and exposures from food and to ensure that the allocation does not result in an apparent conflict with essentiality.

Default assumptions

There is variation in both the volume of water consumed daily and the body weight of consumers. It is therefore necessary to apply some assumptions in order to determine a guideline value. The default assumption for consumption by an adult is 2 litres of water per day, whereas the default assumption for body weight is 60 kg.

In some cases, the guideline value is based on children, where they are considered to be particularly vulnerable to a particular substance. In this event, a default intake of 1 litre is assumed for a body weight of 10 kg; where the most vulnerable group is considered to be bottle-fed infants, an intake of 0.75 litre is assumed for a body weight of 5 kg.

Significant figures

The calculated ADI or TDI is used to derive the guideline value, which is usually rounded to one significant figure. In calculating the guideline value, the unrounded ADI or TDI value should be used.

The guideline value is generally rounded to one significant figure to reflect the uncertainty in, for example, experimental animal toxicity data, exposure assumptions made and the uncertainty factors selected. In a few cases, rounding to two significant figures is appropriate because the practical impact of rounding depends on the units; for example, rounding from 1.5 to 2.0 µg/l has less influence on treatment requirements than rounding from 1.5 to 2.0 mg/l. These are considered on a case-by-case basis.

The general rounding rule for mid-way values (x.5) is to round up, in line with common convention. Examples for rounding to one significant figure are as follows: 1.25 becomes 1, 0.73 becomes 0.7 and 1.5 becomes 2.

8.2.3 *Non-threshold chemicals*

In the case of compounds considered to be genotoxic carcinogens, guideline values are normally determined using a mathematical model. Although several models exist, the linearized multistage model is generally adopted. Other models are considered more appropriate in certain cases. These models compute an estimate of risk at a particular level of exposure, along with upper and lower bounds of confidence on the calculation, which may include zero at the lower bound. Guideline values are conservatively presented as the concentrations in drinking-water associated with an estimated upper-bound excess lifetime cancer risk of 10^{-5} (or one additional case of cancer per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). This value does not equate to the number of cases of

cancer that will be caused by exposure to the substance at this level. It is the maximum potential risk, taking into account large uncertainties. It is highly probable that the actual level of risk is less than this, even approaching zero, but risks at low levels of exposure cannot be experimentally verified. The recognition that the cancer risk may approach zero or be indistinguishable from zero stems from the uncertainties associated with mechanisms of carcinogenesis, including the role of the chemical in the cancer process and the possibility of detoxification and repair mechanisms. Member States may consider that a different level of hypothetical risk is more appropriate to their circumstances, and values relating to risks of 10^{-4} or 10^{-6} additional cancer cases over a lifetime of exposure may be determined by respectively multiplying or dividing the guideline value by 10.

The mathematical models used for deriving guideline values for non-threshold chemicals cannot be verified experimentally, and they do not usually take into account a number of biologically important considerations, such as pharmacokinetics, pre-systemic and metabolic detoxification, DNA repair or protection by the immune system. They also assume the validity of a linear extrapolation of very high dose exposures in test animals to very low dose exposures in humans. As a consequence, the models used are conservative (i.e. err on the side of caution). The guideline values derived using these models should be interpreted differently from TDI-derived values because of the lack of precision of the models. Moderate short-term exposure to levels exceeding the guideline value for non-threshold chemicals does not significantly affect the risk.

8.2.4 Data quality

The following factors were taken into account in assessing the quality and reliability of available information:

- Oral studies are preferred (in particular, drinking-water studies), using the pure substance with appropriate dosing regime and a good quality clinical biochemistry and histopathology.
- The database should be sufficiently broad that all potential toxicological end-points of concern have been identified.
- The quality of the studies is such that they are considered reliable; for example, there has been adequate consideration of confounding factors in epidemiological studies.
- There is reasonable consistency between studies; the end-point and study used to derive a guideline value do not contradict the overall weight of evidence.
- For inorganic substances, there is some consideration of speciation in drinking-water.
- There is appropriate consideration of multimedia exposure in the case of epidemiological studies.

In the development of guideline values, existing international approaches are carefully considered. In particular, previous risk assessments developed by the International Programme on Chemical Safety (IPCS) in Environmental Health Criteria monographs and Concise International Chemical Assessment Documents, IARC, JMPR and JECFA are reviewed. These assessments are relied upon except where new

information justifies a reassessment, but the quality of new data is critically evaluated before it is used in any risk assessment. Where international reviews are not available, other sources of data are used in the derivation of guideline values, including published reports from peer-reviewed open literature, national reviews recognized to be of high quality, information submitted by governments and other interested parties and, to a limited extent, unpublished proprietary data (primarily for the evaluation of pesticides).

8.2.5 Provisional guideline values

The use and designation of provisional guideline values are outlined in [Table 8.4](#).

For non-threshold substances, in cases in which the concentration associated with an upper-bound excess lifetime cancer risk of 10^{-5} is not feasible as a result of inadequate analytical or treatment technology, a provisional guideline value (designated A or T, respectively) is recommended at a practicable level.

8.2.6 Chemicals with effects on acceptability

Some substances of health concern have effects on the taste, odour or appearance of drinking-water that would normally lead to rejection of water at concentrations significantly lower than those of concern for health. Such substances are not normally appropriate for routine monitoring. However, guideline values have been established for some substances that may cause taste or odour in drinking-water at concentrations much lower than the guideline values because there is such a wide range in the ability of consumers to detect them by taste or odour. For such substances, a fact sheet and health-based guideline value (see [chapter 12](#)) are presented in the usual way. In the fact sheet, the relationship between concentrations relevant to health and those relevant to the acceptability of the drinking-water is explained. In tables of guideline values, the health-based guideline values are designated with a “C”. For other substances, health-based guideline values may be needed, for instance, in order to assist in judging the response that is required when problems are encountered and in some cases to provide reassurance to health authorities and consumers with regard to possible health risks.

Table 8.4 Use and designation of provisional guideline values

Situations where a provisional guideline applies	Designation	
Significant scientific uncertainties regarding derivation of health-based guideline value	P	
Calculated health-based value is below the achievable analytical quantification level	A	<i>(Guideline value is set at the achievable quantification level)</i>
Calculated health-based value is below the level that can be achieved through practical treatment methods	T	<i>(Guideline value is set at the practical treatment level)</i>
Calculated health-based value may be exceeded as a result of disinfection procedures	D	<i>(Guideline value is set considering possible health effects and the need to maintain adequate disinfection. Adequate disinfection of drinking-water remains paramount)</i>

8.2.7 Chemicals not included in the Guidelines

Additional information on many chemicals not included in these Guidelines is available from several credible sources, including WHO Environmental Health Criteria monographs and Concise International Chemical Assessment Documents (<https://www.who.int/teams/environment-climate-change-and-health/chemical-safety-and-health/health-impacts>), chemical risk assessment reports from JMPR, JECFA and IARC and published documents from a number of national sources, such as the United States Environmental Protection Agency. Although these information sources may not have been reviewed for these Guidelines, they have been peer reviewed and provide readily accessible information on the toxicology of many additional chemicals. They can help drinking-water suppliers and health officials decide upon the significance (if any) of a detected chemical and on the response that might be appropriate.

8.2.8 Mixtures

Chemical contaminants of drinking-water supplies are present with numerous other inorganic and organic constituents. The guideline values are calculated separately for individual substances, without specific consideration of the potential for interaction of each substance with other compounds present. Synergistic interactions between substances are usually selective and very limited, especially at the very low levels usually encountered in drinking-water. The large margin of uncertainty incorporated in the majority of the guideline values is considered to be sufficient to account for potential interactions. In addition, the majority of contaminants will not be continuously present at concentrations at or near their guideline value.

For many chemical contaminants, mechanisms of toxicity are different; consequently, there is no reason to assume that interactions occur. There may, however, be occasions when a number of contaminants with similar toxicological mechanisms are present at levels near their respective guideline values. In such cases, decisions concerning appropriate action should be made, taking into consideration local circumstances. Unless there is evidence to the contrary, it is appropriate to assume that the toxic effects of these compounds are additive. A pragmatic approach in considering the additive approach is to base any regulation or guidance on the basis that the sum (hazard index) of the concentration of each component divided by its standard or guideline value (hazard quotient) should not exceed 1. This approach is recommended in the Guidelines to be applied, for example, to nitrate/nitrite and the trihalomethanes. Where chemicals are present that act on the same target organ with a dissimilar toxicity mechanism, an additive approach is also usually assumed and a modified hazard index approach is considered appropriate. No examples for application of this approach are currently included in the Guidelines. Further information on chemical mixtures is provided in the supporting document *Chemical mixtures in source water and drinking-water* (Annex 1).

8.2.9 Adapting guideline values to local circumstances

In order to account for the variations in exposure from different sources in different parts of the world, default values, generally between 20% and 80%, are used to make an allocation of the TDI to drinking-water in setting guideline values for many

chemicals. Where relevant exposure data are available, authorities are encouraged to develop context-specific guideline values that are tailored to local circumstances and conditions. For example, in areas where the intake of a particular contaminant in drinking-water is known to be much greater than that from other sources (e.g. air and food), it may be appropriate to allocate a greater proportion of the TDI to drinking-water to derive a guideline value more suited to the local conditions.

Daily water intake can vary significantly in different parts of the world, seasonally and particularly where consumers are involved in manual labour in hot climates. Local adjustments to the daily water consumption value may be needed in setting local standards, as in the case of fluoride, for example. For most other substances, the drinking-water intake range is very small (perhaps a factor of 2–4) compared with the much larger range in the toxicological uncertainty factors; hence, no such adjustment is necessary.

Volatile substances in water may be released to the atmosphere in showering and through a range of other household activities. Under such circumstances, inhalation may become a significant route of exposure. Some substances may also be absorbed through the skin during bathing, but this is not usually a major source of uptake. For those substances that are particularly volatile, such as chloroform, the correction factor would be approximately equivalent to a doubling of exposure, which is small in relation to the uncertainties inherent in the derivation of guideline values. However, in some parts of the world, houses have a very low rate of ventilation, and authorities may wish to take inhalation exposure into account in adapting the guideline values to local conditions, although other uncertainty factors used in the quantitative assessments may render this unnecessary. Where such exposure is shown to be important for a particular substance (i.e. high volatility, low ventilation rates and high rates of showering/bathing), it may be appropriate to adjust the guideline value accordingly.

Further considerations on adapting chemical guideline values is provided in the supporting document *Developing drinking-water quality regulations and standards* ([Annex 1](#)).

8.3 Analytical achievability

As noted above, guideline values are not set at concentrations of substances that cannot reasonably be measured. In such circumstances, provisional guideline values are set at the reasonable analytical limits.

Guidance provided in this section and in [Annex 4](#) is intended to assist readers to select appropriate analytical methods for specific circumstances. In carrying out hazard identification and risk assessment and for verification and auditing of the WSP for chemical contaminants, it is usually necessary to carry out some analysis. It is important that appropriate facilities are available to ensure that suitable methods are used in carrying out chemical analysis.

Various collections of “standard” or “recommended” methods for water analysis are published by a number of national and international agencies. It is often thought that adequate analytical accuracy can be achieved provided that all laboratories use the same standard method. Experience shows that this is not always the case, as a variety of factors may affect the accuracy of the results. Examples include reagent purity,

apparatus type and performance, degree of modification of the method in a particular laboratory and the skill and care of the analyst. These factors are likely to vary both between laboratories and over time in an individual laboratory. Moreover, the precision and accuracy that can be achieved with a particular method frequently depend upon the adequacy of sampling and nature of the sample (“matrix”). While it is not essential to use standard methods, it is important that the methods used are properly validated and their precision and accuracy determined before significant decisions are made based on the results. In the case of “nonspecific” variables such as taste, odour, colour and turbidity, the result is method specific, and this needs to be considered when using the data to make comparisons.

A number of considerations are important in selecting methods:

- The overriding consideration is that the method chosen is demonstrated to have the required accuracy. Other factors, such as speed and convenience, should be considered only in selecting among methods that meet this primary criterion.
- Of primary importance is the expertise and diligence of the laboratories performing the analyses. They must utilize auditable quality control and quality assurance procedures for their results to be credible. External certification is highly desirable.
- There are a number of markedly different procedures for measuring and reporting the errors to which all methods are subject. This complicates and prejudices the effectiveness of method selection, and suggestions for standardizing such procedures have been made. It is therefore desirable that details of all analytical methods are published together with performance characteristics that can be interpreted unambiguously.
- If the analytical results from one laboratory are to be compared with those from others or with a numerical standard, it is obviously preferable for them not to have any associated systematic error. In practice, this is not possible, but each laboratory should select methods whose systematic errors have been thoroughly evaluated and shown to be acceptably small.

A qualitative ranking of analytical methods based on their degree of technical complexity is given in [Table 8.5](#) for inorganic chemicals and in [Table 8.6](#) for organic chemicals. These groups of chemicals are separated, as the analytical methods used differ greatly. The higher the ranking, the more complex the process in terms of equipment or operation. In general, higher rankings are also associated with higher total costs.

Analytical achievabilities, based on detection limits, of the inorganic and organic chemicals for which guideline values have been established are given in [Annex 4](#), by source category.

Many kinds of field test kits are available to measure the concentrations of various chemicals in water. These are generally used for compliance examinations as well as for operational monitoring of drinking-water quality. Although the field test kits have the advantage of being simple to use in non-laboratory environments and are often available at relatively low prices, their analytical accuracy is generally less than that of the methods shown in [Tables 8.5](#) and [8.6](#). However, when properly used, they provide

Table 8.5 Ranking of complexity of analytical methods for inorganic chemicals

Ranking	Example of analytical methods
1	Volumetric method, colorimetric method
2	Electrode method
3	Ion chromatography
4	High-performance liquid chromatography
5	Flame atomic absorption spectrometry
6	Electrothermal atomic absorption spectrometry
7	Inductively coupled plasma atomic emission spectrometry
8	Inductively coupled plasma mass spectrometry

Table 8.6 Ranking of complexity of analytical methods for organic chemicals

Ranking	Example of analytical methods
1	High-performance liquid chromatography
2	Gas chromatography
3	Gas chromatography–mass spectrometry
4	Headspace gas chromatography–mass spectrometry
5	Purge-and-trap gas chromatography Purge-and-trap gas chromatography–mass spectrometry

valuable tools for rapidly assessing numerous contaminants in a non-formal laboratory setting at low cost compared with commercial laboratory tests. It is therefore necessary to check the validity of the field test kit before applying it.

A brief description of the analytical methods listed in [Tables 8.5](#) and [8.6](#) is provided in [Annex 4](#).

8.4 Treatment

As noted above, where a health-based guideline value cannot be achieved by reasonably practicable treatment, then the guideline value is designated as provisional and set at the concentration that can be reasonably achieved through treatment.

Collection, treatment, storage and distribution of drinking-water involve deliberate additions of numerous chemicals to improve the safety and quality of the finished drinking-water for consumers (direct additives). In addition, water is in constant contact with pipes, valves, taps and tank surfaces, all of which have the potential to impart additional chemicals to the water (indirect additives). The chemicals used in water treatment or from materials in contact with drinking-water are discussed in more detail in [section 8.5.4](#).

8.4.1 Treatment performance

Treatment performance varies according to local conditions and circumstances. The ability to achieve a guideline value within a drinking-water supply depends on a number of factors, including:

- the concentration of the chemical in the raw water;
- control measures employed throughout the drinking-water system;
- nature of the raw water (groundwater or surface water, presence of natural organic matter and inorganic solutes and other components, such as turbidity);
- treatment processes already installed.

If a guideline value cannot be met with the existing system, then additional treatment may need to be considered, or water might need to be obtained from alternative sources.

Table 8.7 Ranking of technical complexity and cost of water treatment processes

Ranking	Examples of treatment processes
1	Simple chlorination Plain filtration (rapid sand, slow sand)
2	Prechlorination plus filtration Aeration
3	Chemical coagulation Process optimization for control of DBPs
4	Granular activated carbon treatment Ion exchange
5	Ozonation
6	Advanced oxidation processes Membrane treatment

The cost of achieving a guideline value will depend on the complexity of any additional treatment or other control measures required. It is not possible to provide general quantitative information on the cost of achieving individual guideline values. Treatment costs (capital and operating) will depend not only on the factors identified above, but also on issues such as plant throughput; local costs for labour, civil and mechanical works, chemicals and electricity; life expectancy of the plant; and so on. Guideline values may be progressively achieved in the long term through less capital-intensive non-treatment options, such as through agreements with land users to reduce application of chemicals (fertilizers, pesticides, etc.)

A qualitative ranking of treatment processes based on their degree of technical complexity is given in [Table 8.7](#). The higher the ranking, the more complex the process in terms of plant or operation. In general, higher rankings are also associated with higher costs.

[Annex 5](#) summarizes the treatment processes that are capable of removing chemical contaminants of health significance. The tables in [Annex 5](#) include only those chemicals, by source category, for which some treatment data are available and for which guideline values have been established.

The tables in [Annex 5](#) are provided to help inform decisions regarding the ability of existing treatment to meet guidelines and what additional treatment might need to be installed. They have been compiled on the basis of published literature, which includes mainly laboratory experiments, some pilot plant investigations and relatively few full-scale studies of water treatment processes. Consequently:

- Many of the treatments outlined are designed for larger treatment plants and may not necessarily be appropriate for smaller treatment plants or individual-type treatment. In these cases, the choice of technology must be made on a case-by-case basis.
- The information is probably “best case”, as the data would have been obtained under laboratory conditions or with a carefully controlled plant for the purposes of experimentation.
- Actual process performance will depend on the concentration of the chemical in the raw water and on general raw water quality. For example, chlorination and removal of organic chemicals and pesticides using activated carbon or ozonation will be impaired if there is a high concentration of natural organic matter.
- For many contaminants, potentially several different processes could be appropriate, and the choice between processes should be made on the basis of technical complexity and cost, taking into account local circumstances. For example, membrane processes can remove a broad spectrum of chemicals, but simpler and cheaper alternatives are effective for the removal of most chemicals.
- It is normal practice to use a series of unit processes (e.g. coagulation, sedimentation, filtration, chlorination) to achieve desired water quality objectives. Each of these may contribute to the removal of chemicals. It may be technically and economically advantageous to use a combination of processes (e.g. ozonation plus granular activated carbon or membranes) to remove particular chemicals.
- The effectiveness of potential processes should be assessed using laboratory or pilot plant tests on the actual raw water concerned. These tests should be of sufficient duration to identify potential seasonal or other temporal variations in contaminant concentrations and process performance.
- These treatment technology characterizations are estimates and are not comprehensive, but are intended to provide some indications of the types of technologies that have shown greater or lesser capabilities for removing the indicated chemicals from drinking-water.

A brief description of the various treatment processes referred to in [Table 8.7](#) is included in [Annex 5](#).

8.4.2 Process control measures for disinfection by-products

All chemical disinfectants produce inorganic or organic DBPs that may be of concern.

The principal DBPs formed during chlorination are THMs, HAAs, haloketones and haloacetonitriles, as a result of chlorination of naturally occurring organic precursors such as humic substances. Monochloramine produces lower THM concentrations than chlorine but produces other DBPs, including cyanogen chloride.

Chlorine and ozone oxidize bromide to produce hypohalous acids, which react with precursors to form brominated THMs. A range of other DBPs, including aldehydes and carboxylic acids, may also be formed. Of particular

In attempting to control DBP concentrations, it is of paramount importance that the efficiency of disinfection is not compromised and that a suitable residual level of disinfectant is maintained throughout the distribution system.

concern is bromate, formed by the oxidation of bromide. Bromate may also be present in some sources of hypochlorite, but usually at concentrations that will give rise to levels in final water that are below the guideline value.

The main by-products from the use of chlorine dioxide are chlorite ion, which is an inevitable decomposition product, and chlorate ion. Chlorate is also produced in hypochlorite as it ages.

The basic strategies that can be adopted for reducing the concentrations of DBPs are:

- changing the process conditions (including removal of precursor compounds prior to application);
- using a different chemical disinfectant with a lower propensity to produce by-products with the source water;
- using non-chemical disinfection;
- removing DBPs prior to distribution.

Changes to process conditions

The formation of THMs during chlorination can be reduced by removing precursors prior to contact with chlorine—for example, by installing or enhancing coagulation (this may involve using higher coagulant doses or lower coagulation pH values than are applied conventionally). DBP formation can also be reduced by lowering the applied chlorine dose; if this is done, it must be ensured that disinfection is still effective.

The pH value during chlorination affects the distribution of chlorinated by-products. Reducing the pH lowers the THM concentration, but at the expense of increased formation of HAAs. Conversely, increasing the pH reduces HAA production but leads to increased THM formation.

The formation of bromate during ozonation depends on several factors, including concentrations of bromide and ozone and the pH. It is not practicable to remove bromide from raw water, and it is difficult to remove bromate once formed, although granular activated carbon filtration has been reported to be effective under certain circumstances. Bromate formation can be minimized by using lower ozone dose, shorter contact time and a lower residual ozone concentration. Operating at lower pH (e.g. pH 6.5) followed by raising the pH after ozonation also reduces bromate formation, and addition of ammonia can also be effective. Addition of hydrogen peroxide can either increase or decrease bromate formation, depending on the point at which it is applied and local treatment conditions.

Changing disinfectants

It may be feasible to change disinfectant in order to achieve guideline values for DBPs. The extent to which this is possible will be dependent on the raw water quality and installed treatment (e.g. for precursor removal).

It may be effective to change from chlorine to monochloramine to provide a secondary disinfectant residual within distribution, in order to reduce THM formation and subsequent development within the distribution system. Although monochloramine provides a more stable residual within distribution, it is a less powerful disinfectant and should not be used as a primary disinfectant.

Chlorine dioxide can be considered as a potential alternative to both chlorine and ozone disinfection, although it does not provide a residual effect, as chlorine would. The main concerns with chlorine dioxide are with the residual concentrations of chlorine dioxide and the by-products chlorite and chlorate. These can be addressed by controlling the dose of chlorine dioxide at the treatment plant.

Non-chemical disinfection

Ultraviolet (UV) irradiation or membrane processes can be considered as alternatives to chemical disinfection. UV is particularly effective at inactivating *Cryptosporidium*, which is extremely resistant to chlorination. Neither of these provides any residual disinfection, and it may be considered appropriate to add a small dose of a persistent disinfectant such as chlorine or monochloramine to act as a preservative during distribution.

Removing DBPs prior to distribution

It is technically feasible to remove DBPs prior to distribution; however, this is the least attractive option for controlling DBP concentrations. Strategies for DBP control include source control, precursor removal, use of alternative disinfectants and removal of DBPs by technologies such as air stripping, activated carbon, UV light and advanced oxidation. These processes would need to be followed by a further disinfection step to guard against microbial contamination and to ensure a residual concentration of disinfectant within distribution.

8.4.3 Treatment for corrosion control

Corrosion is the partial dissolution of the materials constituting the treatment and supply systems, tanks, pipes, valves and pumps. In certain circumstances, all water can be corrosive. Corrosion may lead to structural failure, leaks, loss of capacity and deterioration of chemical and microbial water quality. The internal corrosion of pipes and fittings can have a direct impact on the concentration of water constituents, including lead and copper. Corrosion control is therefore an important aspect of the management of a drinking-water system for safety.

Corrosion control involves many parameters, including the concentrations of calcium, bicarbonate, carbonate and dissolved oxygen, as well as pH. The detailed requirements differ depending on water quality and the materials used in the distribution system. The pH controls the solubility and rate of reaction of most of the metal species involved in corrosion reactions. It is particularly important in relation to the formation of a protective film at the metal surface. For some metals, alkalinity (carbonate and bicarbonate) and calcium (hardness) also affect corrosion rates.

Characterizing corrosivity

Most of the indices that have been developed to characterize the corrosion potential of waters are based on the assumption that water with a tendency to deposit a calcium carbonate scale on metal surfaces will be less corrosive. The Langelier index is the difference between the actual pH of a water and its "saturation pH", this being the pH at which a water of the same alkalinity and calcium hardness would be at equilibrium

with solid calcium carbonate. Waters with a positive Langelier index are capable of depositing calcium carbonate scale from solution.

There is no corrosion index that applies to all materials, and corrosion indices, particularly those related to calcium carbonate saturation, have given mixed results. The parameters related to calcium carbonate saturation status are, strictly speaking, indicators of the tendency to deposit or dissolve calcium carbonate (calcite) scale, not indicators of the “corrosivity” of a water. For example, there are many waters with a negative Langelier index that are non-corrosive and many with a positive Langelier index that are corrosive. Nevertheless, there are many documented instances of the use of saturation indices for corrosion control based on the concept of laying down a protective “eggshell” scale of calcite in iron pipes. In general, waters with high pH, calcium and alkalinity are less corrosive, and this tends to be correlated with a positive Langelier index. However, these calcium carbonate precipitation indices are not necessarily considered to be good corrosion predictors for copper systems.

The ratio of the chloride and sulfate concentrations to the bicarbonate concentration (Larson ratio) has been shown to be helpful in assessing the corrosiveness of water to cast iron and steel. A similar approach has been used in studying zinc dissolution from brass fittings—the Turner diagram.

Water treatment for corrosion control

To control corrosion in water distribution networks, the methods most commonly applied are adjusting pH, increasing the alkalinity or hardness or adding corrosion inhibitors, such as polyphosphates, silicates and orthophosphates. The quality and maximum dose to be used should be in line with specifications for such water treatment chemicals. Although pH adjustment is an important approach, its possible impact on other aspects of water supply technology, including disinfection, must always be taken into account.

It is not always possible to achieve the desired values for all parameters. For example, the pH of hard waters cannot be increased too much, or softening will occur. The application of lime and carbon dioxide to soft waters can be used to increase both the calcium concentration and the alkalinity to at least 40 mg/l as calcium carbonate.

More detailed information on the corrosion of various metals commonly used in water treatment and distribution systems can be found in [Annex 5](#).

8.4.4 Household treatment

The chemicals of greatest health concern in some natural waters are usually excess natural fluoride, nitrate/nitrite and arsenic.

Some commercial water treatment technologies are available for small applications for the removal of chemical contaminants. For example, anion exchange using activated alumina or iron-containing products will effectively reduce excess fluoride concentrations. Bone char has also been used to reduce fluoride concentrations. Arsenic is also removed by anion exchange processes similar to those employed for fluoride. Nitrates and nitrites, which are frequently present due to sewage contamination or agricultural runoff, are best managed by protecting the source water from contamination. They are difficult to remove, although disinfection will oxidize nitrite, the more toxic form, to nitrate. In addition, disinfection will sanitize the water and reduce

the risk of gastrointestinal infection, which is a risk factor for methaemoglobinaemia caused by excess nitrate/nitrite exposure of infants up to approximately 3–6 months of age.

Cation exchange water softening is widely used in homes to remove excess hardness due to high calcium or magnesium, and it can also remove metals including iron and radium.

Synthetic and natural organic chemicals can be removed by granular activated carbon or carbon block technologies. The treatment systems must be well managed and replaced regularly, because their effectiveness is eventually lost, depending upon the types of contaminating chemicals and their concentrations in the water. Reverse osmosis technologies have general applicability for removal of most organic and inorganic chemicals; however, there is some selectivity, and also there is a significant amount of water wastage when low-pressure units are used in small-volume applications.

8.5 Guideline values for individual chemicals, by source category

8.5.1 *Naturally occurring chemicals*

There are a number of sources of naturally occurring chemicals in drinking-water. All natural water contains a range of inorganic and organic chemicals. Inorganic chemicals derive from the rocks and soil through which water percolates or over which it flows. Organic chemicals derive from the breakdown of plant material or from algae and other microorganisms that grow in the water or on sediments. Most of the naturally occurring chemicals for which guideline values have been derived or that have been considered for guideline value derivation are inorganic. Toxins produced by cyanobacteria (blue-green algae; see [section 11.5](#))—that is, “cyanotoxins”—are the only organic substances for which guideline values have been derived. Cyanobacteria occur widely in lakes, reservoirs, ponds and slow-flowing rivers. They produce a wide range of compounds, some of which show activity in *in vitro* bioassays. Four key groups—anatoxin-a variants, cylindrospermopsins, microcystins and saxitoxins (see also the corresponding [chapter 12](#) fact sheets)—account for much, but not all, of the toxicity to humans (and other vertebrates) and are frequently reported. Whereas anatoxins, microcystins and saxitoxins are primarily found within cells, major fractions of cylindrospermopsins are released into water. The toxicity of different structural variants within a cyanotoxin group varies widely, and further cyanotoxins may yet be recognized. Mass developments of cyanobacteria (“blooms”) also cause a high organic load that challenges treatment. Therefore, preventing blooms is the preferred control option.

The approach to dealing with naturally occurring chemicals will vary according to the nature of the chemical and the source. For inorganic contaminants that arise from rocks and sediments, it is important to screen possible water sources to determine whether the source is suitable for use or whether it will be necessary to treat the water to remove the contaminants of concern along with microbial contaminants. In some cases, where a number of sources may be available, dilution or blending of the water containing high levels of a contaminant with a water containing much lower levels may achieve the desired result.

Table 8.8 Naturally occurring chemicals for which guideline values have not been established

Chemical	Reason for not establishing a guideline value	Remarks
Inorganic		
Bromide	Occurs in drinking-water at concentrations well below those of health concern	
Chloride	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Hardness	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Hydrogen sulfide	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Iron	Not of health concern at levels causing acceptability problems in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Molybdenum	Occurs in drinking-water at concentrations well below those of health concern	
pH	Not of health concern at levels found in drinking-water	An important operational water quality parameter
Potassium	Occurs in drinking-water at concentrations well below those of health concern	
Sodium	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Sulfate	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Total dissolved solids	Not of health concern at levels found in drinking-water	May affect acceptability of drinking-water (see chapter 10)
Organic		
Anatoxins	Available data inadequate to permit derivation of health-based guideline value. However, a reference value ^a of 0.03 mg/l has been proposed.	The reference value is for total anatoxins (sum of all congeners, free plus cell-bound), for short-term exposure ^b

^a A provisional reference value may be useful to guide actions by Member States when there is reason for local concern. However, the value is too uncertain to be used for developing regulations or standards. See [section 8.2](#) for more information.

^b Also applicable for acute exposure. See the chemical fact sheet in [chapter 12](#) for considerations for bottle-fed infants.

A number of the most important chemical contaminants (i.e. those that have been shown to cause adverse health effects as a consequence of exposure through drinking-water) fall into the category of naturally occurring chemicals. Some naturally occurring chemicals have other primary sources and are therefore discussed in other sections of this chapter.

Guideline values have not been established for the naturally occurring chemicals listed in [Table 8.8](#) for the reasons indicated in the table. However health-based values have been developed for a number of these chemicals in order to provide guidance to Member States when there is a reason for local concern, although, as noted in the table, in many circumstances the acceptability of drinking-water is the overriding

Table 8.9 Guideline values for naturally occurring chemicals that are of health significance in drinking-water

Chemical	Guideline value		Remarks
	µg/l	mg/l	
Inorganic			
Arsenic	10 (A, T)	0.01 (A, T)	
Barium	1300	1.3	
Boron	2400	2.4	
Chromium	50	0.05	For total chromium
Fluoride	1500	1.5	Volume of water consumed and intake from other sources should be considered when setting national standards
Manganese	80 (P)	0.08 (P)	For total manganese. Aesthetic as well as health aspects should be considered when setting national standards
Selenium	40 (P)	0.04 (P)	
Uranium	30 (P)	0.03 (P)	Only chemical, not radiological, aspects of uranium addressed
Organic			
Cylindrospermopsins	0.7 (P)	0.0007 (P)	
	3 (P)	0.003 (P)	For short-term exposure ^a Both values are for total cylindrospermopsins (sum of all congeners, free plus cell-bound)
Microcystins	1 (P)	0.001 (P)	
	12 (P)	0.012 (P)	For short-term exposure ^a Both values are for total microcystins (sum of all congeners, free plus cell-bound)
Saxitoxins	3	0.003	For acute exposure
			For total saxitoxins (sum of all congeners, free plus cell-bound)

A, provisional guideline value because calculated guideline value is below the achievable quantification level; P, provisional guideline value because of uncertainties in the health database; T, provisional guideline value because calculated guideline value is below the level that can be achieved through practical treatment methods, source protection, etc.

^a See the respective chemical fact sheet in [chapter 12](#) for considerations for bottle-fed infants.

concern (for further information on guideline values and health-based values, see [section 8.2](#)). Fact sheets are included in [chapter 12](#).

Guideline values have been established for the naturally occurring chemicals listed in [Table 8.9](#), which meet the criteria for inclusion. Fact sheets are included for each in [chapter 12](#).

8.5.2 Chemicals from industrial sources and human dwellings

Chemicals from industrial sources can reach drinking-water directly from discharges or indirectly from diffuse sources arising from the use and disposal of materials and products containing the chemicals. In some cases, inappropriate handling and disposal may lead to contamination (e.g. degreasing agents that are allowed to reach groundwater). Some of these chemicals, particularly inorganic substances, may also

be encountered as a consequence of natural contamination, but this may also be a by-product of industrial activity, such as mining, that changes drainage patterns. Many of these chemicals are used in small industrial units within human settlements, and, particularly where such units are found in groups of similar enterprises, they may be a significant source of pollution. Petroleum oils are widely used in human settlements, and improper handling or disposal can lead to significant pollution of surface water and groundwater. Where plastic pipes are used, the smaller aromatic molecules in petroleum oils can sometimes penetrate the pipes where they are surrounded by earth soaked in the oil, with subsequent pollution of the local water supply.

A number of chemicals can reach water as a consequence of disposal of general household chemicals; in particular, a number of heavy metals may be found in domestic wastewater. Where wastewater is treated, these will usually partition out into the sludge. Some chemicals that are widely used both in industry and in materials used in a domestic setting are found widely in the environment (e.g. di(2-ethylhexyl)-phthalate), and these may be found in water sources, although usually at low concentrations.

Some chemicals that reach drinking-water from industrial sources or human settlements have other primary sources and are therefore discussed in other sections of this chapter. Where latrines and septic tanks are poorly sited, these can lead to contamination of drinking-water sources with nitrate (see [section 8.5.3](#)).

Identification of the potential for contamination by chemicals from industrial activities and human dwellings requires assessment of activities in the catchment and of the risk that particular contaminants may reach water sources. The primary approach to addressing these contaminants is prevention of contamination by encouraging good practices. However, if contamination has occurred, then it may be necessary to consider the introduction of treatment.

Guideline values have not been established for the chemicals listed in [Table 8.10](#) for the reasons indicated in the table. However, health-based values have been developed for a number of these chemicals in order to provide guidance to Member States when there is a reason for local concern (for further information on guideline values and health-based values, see [section 8.2](#)). Fact sheets for each are included in [chapter 12](#).

Guideline values have been established for the chemicals listed in [Table 8.11](#), which meet all of the criteria for inclusion. Fact sheets for each are included in [chapter 12](#).

8.5.3 Chemicals from agricultural activities

Chemicals are used in agriculture on crops and in animal husbandry. Nitrate may be present as a consequence of tillage when there is no growth to take up nitrate released from decomposing plants, from the application of excess inorganic or organic fertilizer and in slurry from animal production. Most chemicals that may arise from agriculture are pesticides, although their presence will depend on many factors, and not all pesticides are used in all circumstances or climates. Contamination can result from application and subsequent movement following rainfall or from inappropriate disposal methods.

Table 8.10 Chemicals from industrial sources and human dwellings for which guideline values have not been established

Chemical	Reason for not establishing a guideline value
Beryllium	Rarely found in drinking-water at concentrations of health concern
Cyanide	Occurs in drinking-water at concentrations well below those of health concern, except in emergency situations following a spill to a water source
1,3-Dichlorobenzene	Available data inadequate to permit derivation of health-based guideline value
1,1-Dichloroethane	Available data inadequate to permit derivation of health-based guideline value
1,1-Dichloroethene	Occurs in drinking-water at concentrations well below those of health concern
Di(2-ethylhexyl)adipate	Occurs in drinking-water at concentrations well below those of health concern
Hexachlorobenzene	Occurs in drinking-water at concentrations well below those of health concern
Methyl tertiary-butyl ether	Any guideline that would be derived would be significantly higher than concentrations at which methyl tertiary-butyl ether would be detected by odour
Monochlorobenzene	Occurs in drinking-water at concentrations well below those of health concern, and health-based value would far exceed lowest reported taste and odour threshold
Nitrobenzene	Rarely found in drinking-water at concentrations of health concern
Organotins	
TBT, TPT, DBT, DOT ^a	Occurs in drinking-water at concentrations well below those of health concern
Other organotins ^b	Available data inadequate to permit derivation of health-based guideline value
Petroleum products	Taste and odour will in most cases be detectable at concentrations below those of health concern, particularly with short-term exposure
Trichlorobenzenes (total)	Occur in drinking-water at concentrations well below those of health concern, and health-based value would exceed lowest reported odour threshold
1,1,1-Trichloroethane	Occurs in drinking-water at concentrations well below those of health concern

^a Tributyltin, triphenyltin, dibutyltin and di-*n*-octyltin, respectively.^b Also excluding monomethyltin, dimethyltin and dimethyltin dichloride, which are covered in [Table 8.16](#).

Table 8.11 Guideline values for chemicals from industrial sources and human dwellings that are of health significance in drinking-water

Chemicals	Guideline value		Remarks	
	µg/l	mg/l		
Inorganic				
Cadmium	3	0.003	For inorganic mercury	
Mercury	6	0.006		
Organic				
Benzene	10 ^a	0.01 ^a	Derived using TDI approach as well as linear multistage modelling	
Carbon tetrachloride	4	0.004		
1,2-Dichlorobenzene	1000 (C)	1 (C)		
1,4-Dichlorobenzene	300 (C)	0.3 (C)		
1,2-Dichloroethane	30 ^a	0.03 ^a		
1,2-Dichloroethene	50	0.05		
Dichloromethane	20	0.02		
Di(2-ethylhexyl)phthalate	8	0.008		
1,4-Dioxane	50 ^a	0.05 ^a		
Edetic acid	600	0.6		Applies to the free acid
Ethylbenzene	300 (C)	0.3 (C)		
Hexachlorobutadiene	0.6	0.0006		
Nitrilotriacetic acid	200	0.2		
Pentachlorophenol	9 ^a (P)	0.009 ^a (P)		
Styrene	20 (C)	0.02 (C)		
Tetrachloroethene	100	0.1		
Toluene	700 (C)	0.7 (C)		
Trichloroethene	8	0.008		
Xylenes	500 (C)	0.5 (C)		

C, concentrations of the substance at or below the health-based guideline value may affect the appearance, taste or odour of the water, leading to consumer complaints; P, provisional guideline value because of uncertainties in the health database

^a For non-threshold substances, the guideline value is the concentration in drinking-water associated with an upper-bound excess lifetime cancer risk of 10⁻⁵ (one additional case of cancer per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). Concentrations associated with estimated upper-bound excess lifetime cancer risks of 10⁻⁴ and 10⁻⁶ can be calculated by multiplying and dividing, respectively, the guideline value by 10.

Some pesticides are also used in non-agricultural circumstances, such as the control of weeds on roads and railway lines. These pesticides are also included in this section.

Guideline values have not been established for the chemicals listed in [Table 8.12](#), as a review of the literature on occurrence or credibility of occurrence in drinking-water has shown evidence that the chemicals do not occur in drinking-water.

Table 8.12 Chemicals from agricultural activities excluded from guideline value derivation

Chemical	Reason for exclusion
Amitraz	Degrades rapidly in the environment and is not expected to occur at measurable concentrations in drinking-water supplies
Chlorobenzilate	Unlikely to occur in drinking-water
Chlorothalonil	Unlikely to occur in drinking-water
Cypermethrin	Unlikely to occur in drinking-water
Deltamethrin	Unlikely to occur in drinking-water
Diazinon	Unlikely to occur in drinking-water
Dinoseb	Unlikely to occur in drinking-water
Ethylene thiourea	Unlikely to occur in drinking-water
Fenamiphos	Unlikely to occur in drinking-water
Formothion	Unlikely to occur in drinking-water
Hexachlorocyclohexanes (mixed isomers)	Unlikely to occur in drinking-water
MCPB ^a	Unlikely to occur in drinking-water
Methamidophos	Unlikely to occur in drinking-water
Methomyl	Unlikely to occur in drinking-water
Mirex	Unlikely to occur in drinking-water
Monocrotophos	Has been withdrawn from use in many countries and is unlikely to occur in drinking-water
Oxamyl	Unlikely to occur in drinking-water
Phorate	Unlikely to occur in drinking-water
Propoxur	Unlikely to occur in drinking-water
Pyridate	Not persistent and only rarely found in drinking-water
Pyriproxyfen	Unlikely to occur in drinking-water ^b
Quintozene	Unlikely to occur in drinking-water
Toxaphene	Unlikely to occur in drinking-water
Triazophos	Unlikely to occur in drinking-water
Trichlorfon	Unlikely to occur in drinking-water

^a 4-(4-chloro-o-tolyloxy)butyric acid.^b The use of pyriproxyfen as a larvicide for public health purposes is discussed further in [section 8.6](#).

Guideline values have not been established for the chemicals listed in [Table 8.13](#) for the reasons indicated in the table. However, health-based values and, in some cases, acute health-based values have been developed for a number of these pesticides in order to provide guidance to Member States when there is a reason for local concern such as an emergency or spill situation (for further information on guideline values and health-based values, see [section 8.2](#)). Fact sheets for each are included in [chapter 12](#).

Table 8.13 Chemicals from agricultural activities for which guideline values have not been established

Chemical	Reason for not establishing a guideline value
Ammonia	Occurs in drinking-water at concentrations well below those of health concern
Bentazone	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Carbaryl	Occurs in drinking-water at concentrations well below those of health concern
1,3-Dichloropropane	Available data inadequate to permit derivation of health-based guideline value
Dichlorvos	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Dicofol	Unlikely to occur in drinking-water or drinking-water sources ^a
Diquat	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Endosulfan	Occurs in drinking-water at concentrations well below those of health concern
Fenitrothion	Occurs in drinking-water at concentrations well below those of health concern
Glyphosate and AMPA ^b	Occur in drinking-water at concentrations well below those of health concern
Heptachlor and heptachlor epoxide	Occur in drinking-water at concentrations well below those of health concern
Malathion	Occurs in drinking-water at concentrations well below those of health concern
MCPA ^c	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Methyl parathion	Occurs in drinking-water at concentrations well below those of health concern
Parathion	Occurs in drinking-water at concentrations well below those of health concern
2-Phenylphenol and its sodium salt	Occurs in drinking-water at concentrations well below those of health concern
Propanil	Readily transformed into metabolites that are more toxic; a guideline value for the parent compound is considered inappropriate, and there are inadequate data to enable the derivation of guideline values for the metabolites

^a Although dicofol does not fulfil one of the three criteria for evaluation in the Guidelines, a background document has been prepared, and a health-based value has been established, in response to a request from Member States for guidance.

^b Aminomethylphosphonic acid.

^c (2-Methyl-4-chlorophenoxy)acetic acid.

Guideline values have been established for the chemicals listed in [Table 8.14](#), which meet the criteria for inclusion (see [section 8.2](#)). Fact sheets for each are included in [chapter 12](#).

Table 8.14 Guideline values for chemicals from agricultural activities that are of health significance in drinking-water

Chemical	Guideline value		Remarks
	µg/l	mg/l	
Non-pesticides			
Nitrate (as NO ₃ ⁻)	50 000	50	Based on short-term effects, but protective for long-term effects
Nitrite (as NO ₂ ⁻)	3 000	3	Based on short-term effects, but protective for long-term effects
Pesticides used in agriculture			
Alachlor	20 ^a	0.02 ^a	
Aldicarb and its sulfoxide and sulfone metabolites	10	0.01	
Aldrin and dieldrin	0.03	0.000 03	For combined aldrin plus dieldrin
Atrazine and its chloro- <i>s</i> -triazine metabolites	100	0.1	
Carbofuran	7	0.007	
Chlordane	0.2	0.000 2	
Chlorotoluron	30	0.03	
Chlorpyrifos	30	0.03	
Cyanazine	0.6	0.000 6	
2,4-D ^b	30	0.03	Applies to free acid
2,4-DB ^c	90	0.09	
1,2-Dibromo-3-chloropropane	1 ^a	0.001 ^a	
1,2-Dibromoethane	0.4 ^a (P)	0.000 4 ^a (P)	
1,2-Dichloropropane	40 (P)	0.04 (P)	
1,3-Dichloropropene	20 ^a	0.02 ^a	
Dichlorprop	100	0.1	
Dimethoate	6	0.006	
Endrin	0.6	0.000 6	
Fenoprop	9	0.009	
Hydroxyatrazine	200	0.2	Atrazine metabolite
Isoproturon	9	0.009	
Lindane	2	0.002	
Mecoprop	10	0.01	
Methoxychlor	20	0.02	
Metolachlor	10	0.01	

Table 8.14 (continued)

Chemical	Guideline value		Remarks
	µg/l	mg/l	
Molinate	6	0.006	
Pendimethalin	20	0.02	
Simazine	2	0.002	
2,4,5-T ^d	9	0.009	
Terbutylazine	7	0.007	
Trifluralin	20	0.02	

P, provisional guideline value because of uncertainties in the health database

^a For substances that are considered to be carcinogenic, the guideline value is the concentration in drinking-water associated with an upper-bound excess lifetime cancer risk of 10^{-5} (one additional cancer per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). Concentrations associated with estimated upper-bound excess lifetime cancer risks of 10^{-4} and 10^{-6} can be calculated by multiplying and dividing, respectively, the guideline value by 10.

^b 2,4-Dichlorophenoxyacetic acid.

^c 2,4-Dichlorophenoxybutyric acid.

^d 2,4,5-Trichlorophenoxyacetic acid.

Guideline values and health-based values are protective against health effects resulting from lifetime exposure. Small exceedances for short periods would not normally constitute a health emergency. In the event of a spill, a higher allocation of the ADI to drinking-water could be justified. Alternatively, in cases where acute health-based values have been derived, normally based on JMPR evaluations, these may provide useful guidance (for further information, see [section 8.7.5](#)).

Routine monitoring of pesticides is generally not considered necessary. Member States should consider local usage and potential situations such as spills in deciding whether and where to monitor. In the event that monitoring results show levels above the guideline value or health-based value on a regular basis, it is advisable that a plan be developed and implemented to address the situation.

As a general principle, efforts should be made to keep the concentration of pesticides in water as low as possible, and to not allow concentrations to increase up to the guideline value or health-based value.

8.5.4 Chemicals used in water treatment or from materials in contact with drinking-water

Chemicals used in water treatment and chemicals arising from materials in contact with water may give rise to contaminants in the final water.

Some substances are deliberately added to water in the course of treatment (direct additives), some of which may be inadvertently retained in the finished water (e.g. salts, coagulant polymer residues or monomers). Chloramine and chlorine disinfectant residuals, for example, are deliberate additives, and their presence confers a benefit. Others, such as DBPs, are generated during chemical interactions between disinfectant chemicals and substances normally in water ([Table 8.15](#)). Chlorination by-products and other DBPs may also occur in swimming pools, from which exposure by inhalation and skin absorption will be of greater importance (WHO, 2006).

Table 8.15 Disinfection by-products present in disinfected waters (based on IPCS, 2000)

Disinfectant	Significant organohalogen products	Significant inorganic products	Significant non-halogenated products
Chlorine/ hypochlorous acid (hypochlorite)	THMs, HAAs, haloaceto-nitriles, chloral hydrate, chloropicrin, chlorophenols, <i>N</i> -chloramines, halo-furanones, bromohydrins	Chlorate (mostly from hypochlorite use)	Aldehydes, cyanoalkanoic acids, alkanolic acids, benzene, carboxylic acids, <i>N</i> -nitrosodimethylamine
Chlorine dioxide		Reduced primarily to chlorite, chlorate and chloride in drinking-water, and to chlorite and chloride upon ingestion; the provisional guideline values for chlorite and chlorate are protective for potential toxicity from chlorine dioxide	Unknown
Chloramine	Haloacetonitriles, cyanogen chloride, organic chloramines, chloramino acids, chloral hydrate, halo ketones	Nitrate, nitrite, chlorate, hydrazine	Aldehydes, ketones, <i>N</i> -nitrosodimethylamine
Ozone	Bromoform, monobromoacetic acid, dibromoacetic acid, dibromoacetone, cyanogen bromide	Chlorate, iodate, bromate, hydrogen peroxide, hypobromous acid, epoxides, ozonates	Aldehydes, ketoacids, ketones, carboxylic acids
Sodium dichloroisocyanurate	As for chlorine/hypochlorous acid (hypochlorite)		Cyanuric acid

Other chemicals, such as lead or copper from pipes or brass taps and chemicals leaching from coatings, may be taken up from contact with surfaces during treatment or distribution (indirect or unintentional additives).

Some chemicals used in water treatment (e.g. aluminium) or in materials in contact with drinking-water (e.g. styrene) have other principal sources and are therefore discussed in detail in other sections of this chapter.

Many of these additives, both direct and indirect or unintentional, are components of processes for producing safe drinking-water. The approach to monitoring and management is preferably through control of the material or chemical. It is important to optimize treatment processes and to ensure that such processes remain optimized in order to control residuals of chemicals used in treatment and to control the formation of DBPs. Inadvertent contamination caused by poor quality materials is best controlled by applying specifications governing the composition of the products themselves rather than by setting limits on the quality of finished water, whereas contamination due to the inappropriate use of additives can be addressed by guidance

on use. Similarly, regulations on the quality of pipe can avoid possible contamination of water by leachable materials. Control of contamination from in situ applied coatings requires suitable codes of practice on their application in addition to controls on the composition of materials.

Numerous national and third-party evaluation and approval systems for additives and materials for contact with drinking-water exist throughout the world; however, many countries do not have or operate such systems. Governments and other organizations should consider establishing or adapting additive management systems and setting product quality standards and guidance on use that would apply to determining acceptable water contact products. Ideally, harmonized standards between countries or reciprocal recognition would reduce costs and increase access to such standards (see also [section 1.2.9](#)).

Guideline values have not been established for the chemicals listed in [Table 8.16](#) for the reasons indicated in the table. However, health-based values have been developed for a number of these chemicals in order to provide guidance to Member States when there is a reason for local concern (for further information on guideline values and health-based values, see [section 8.2](#)). Fact sheets for each are included in [chapter 12](#).

Guideline values have been established for the chemicals listed in [Table 8.17](#), which meet the criteria for inclusion. Fact sheets for each are included in [chapter 12](#).

Indicator substances for monitoring chlorination by-products

Although guideline values have been established for a number of chlorination by-products, data from drinking-water supplies indicate that THMs and HAAs are adequate as indicators of the majority of chlorination by-products. The most appropriate means of controlling chlorination by-products is to remove the organic precursors, which are largely of natural origin. Measurement of THMs and, if appropriate, HAAs (e.g. where water is chlorinated at a low pH) can be used to optimize treatment efficiency and to establish the boundaries of other operational parameters that can be used to monitor treatment performance. In these circumstances, monitoring frequencies of other chlorination by-products can be reduced. Although total organohalogen does not correlate well with either THMs or HAAs, it is a measure of total chlorination by-products and may be another potential indicator for operational purposes.

In all circumstances, disinfection efficiency should not be compromised in trying to meet guidelines for DBPs, including chlorination by-products, or in trying to reduce concentrations of these substances.

Contaminants from storage and generation of hypochlorite solutions

Sodium hypochlorite solutions slowly decompose—more rapidly at warmer temperatures—to produce chlorate and chlorite ions. As the solution ages and the available chlorine concentration decreases, it is necessary to dose more product to achieve the desired residual chlorine concentration, with a consequent increase in the amounts of chlorate and chlorite added to the treated water. The decomposition of solid calcium hypochlorite is much slower, and consequently contamination is less likely to be significant. However, if calcium hypochlorite solutions are prepared and stored before use, then decomposition to form chlorate and chlorite would also occur.

Table 8.16 Chemicals used in water treatment or materials in contact with drinking-water for which guideline values have not been established

Chemical	Reason for not establishing a guideline value
Disinfectants	
Chlorine dioxide	Reduced primarily to chlorite, chlorate and chloride in drinking-water, and to chlorite and chloride upon ingestion; the provisional guideline values for chlorite and chlorate are protective for potential toxicity from chlorine dioxide
Dichloramine	Available data inadequate to permit derivation of health-based guideline value
Iodine	Available data inadequate to permit derivation of health-based guideline value. Additionally, occurrence in drinking-water is usually low. Although higher levels of exposure may occur when iodine is used as a drinking-water disinfectant at the point of use, extended periods of exposure to iodine through water disinfection are unlikely.
Silver	Available data inadequate to permit derivation of health-based guideline value and usually occurs in drinking-water at concentrations well below those of health concern. However, a reference value ^a of 0.1 mg/l has been proposed in cases of local concern.
Trichloramine	Available data inadequate to permit derivation of health-based guideline value
Disinfection by-products	
Bromochloroacetate	Available data inadequate to permit derivation of health-based guideline value
Bromochloroacetonitrile	Available data inadequate to permit derivation of health-based guideline value
Chloral hydrate	Occurs in drinking-water at concentrations well below those of health concern
Chloroacetones	Available data inadequate to permit derivation of health-based guideline values for any of the chloroacetones
2-Chlorophenol	Available data inadequate to permit derivation of health-based guideline value
Chloropicrin	Available data inadequate to permit derivation of health-based guideline value
Cyanogen chloride	Occurs in drinking-water at concentrations well below those of health concern
Dibromoacetate	Available data inadequate to permit derivation of health-based guideline value
2,4-Dichlorophenol	Available data inadequate to permit derivation of health-based guideline value
Formaldehyde	Occurs in drinking-water at concentrations well below those of health concern
Monobromoacetate	Available data inadequate to permit derivation of health-based guideline value
MX ^b	Occurs in drinking-water at concentrations well below those of health concern
Trichloroacetonitrile	Available data inadequate to permit derivation of health-based guideline value

Table 8.16 (continued)

Chemical	Reason for not establishing a guideline value
Contaminants from treatment chemicals	
Aluminium	The health-based value exceeds practicable levels based on optimization of the coagulation process in drinking-water plants using aluminium-based coagulants: 0.1 mg/l or less in large drinking-water supplies and 0.2 mg/l or less in small supplies
Contaminants from pipes and fittings	
Asbestos	No consistent evidence that ingested asbestos is hazardous to health and available data inadequate to permit derivation of health-based guideline value
Fluoranthene ^c	Occurs in drinking-water at concentrations well below those of health concern
Inorganic tin	Occurs in drinking-water at concentrations well below those of health concern
Organotins MMT, DMT, DMTc ^d	Unnecessary since their use as stabilizers in polyvinyl chloride and chlorinated polyvinyl chloride are normally controlled by product specification
Zinc	Not of health concern at levels found in drinking-water ^e

^a A provisional reference value may be useful to guide actions by Member States in the event of need. However, the value is too uncertain to be used for developing regulations or standards. See [section 8.2](#) for more information.

^b 3-Chloro-4-dichloromethyl-5-hydroxy-2(5H)-furanone.

^c See fact sheet on polynuclear aromatic hydrocarbons.

^d Monomethyltin, dimethyltin and dimethyltin dichloride, respectively. Other organotins are covered in [Table 8.10](#).

^e May affect acceptability of drinking-water (see [chapter 10](#)).

Table 8.17 Guideline values for chemicals used in water treatment or materials in contact with drinking-water that are of health significance in drinking-water

	Guideline value ^a		
Chemical	µg/l	mg/l	Remarks
Disinfectants			
Chlorine	5 000 (C)	5 (C)	For free chlorine. For effective disinfection, there should be a residual concentration of free chlorine of ≥ 0.5 mg/l after at least 30 min contact time at pH < 8.0. A chlorine residual should be maintained throughout the distribution system. At the point of delivery, the minimum residual concentration of free chlorine should be 0.2 mg/l.
Monochloramine	3 000	3	
Sodium dichloroisocyanurate	50 000	50	As sodium dichloroisocyanurate
	40 000	40	As cyanuric acid

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Table 8.17 (continued)

Chemical	Guideline value ^a		Remarks
	µg/l	mg/l	
Disinfection by-products			
Bromate	10 ^a (A, T)	0.01 ^a (A, T)	The sum of the ratio of the concentration of each to its respective guideline value should not exceed 1
Bromodichloromethane	60 ^a	0.06 ^a	
Bromoform	100	0.1	
Chlorate	700 (D)	0.7 (D)	
Chlorite	700 (D)	0.7 (D)	
Chloroform	300	0.3	
Dibromoacetonitrile	70	0.07	
Dibromochloromethane	100	0.1	
Dichloroacetate	50 ^a (D)	0.05 ^a (D)	
Dichloroacetonitrile	20 (P)	0.02 (P)	
Monochloroacetate	20	0.02	
N-Nitrosodimethylamine	0.1	0.0001	
Trichloroacetate	200	0.2	
2,4,6-Trichlorophenol	200 ^a (C)	0.2 ^a (C)	
Trihalomethanes			
Contaminants from treatment chemicals			
Acrylamide	0.5 ^a	0.0005 ^a	
Epichlorohydrin	0.4 (P)	0.0004 (P)	
Contaminants from pipes and fittings			
Antimony	20	0.02	Staining of laundry and sanitary ware may occur below guideline value
Benzo[a]pyrene	0.7 ^a	0.0007 ^a	
Copper	2000	2	
Lead	10 (A, T)	0.01 (A, T)	Based on long-term effects, but protective for short-term effects
Nickel	70	0.07	
Vinyl chloride	0.3 ^a	0.0003 ^a	

A, provisional guideline value because calculated guideline value is below the achievable quantification level; C, concentrations of the substance at or below the health-based guideline value may affect the appearance, taste or odour of the water, leading to consumer complaints; D, provisional guideline value because disinfection is likely to result in the guideline value being exceeded; P, provisional guideline value because of uncertainties in the health database; T, provisional guideline value because calculated guideline value is below the level that can be achieved through practical treatment methods, source control, etc.

^a For substances that are considered to be carcinogenic, the guideline value is the concentration in drinking-water associated with an upper-bound excess lifetime cancer risk of 10^{-5} (one additional case of cancer per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). Concentrations associated with estimated upper-bound excess lifetime cancer risks of 10^{-4} and 10^{-6} can be calculated by multiplying and dividing, respectively, the guideline value by 10.

Sodium hypochlorite is manufactured by electrolysis of sodium chloride dissolved in water, which would naturally also contain small concentrations of sodium bromide. This results in the presence of bromate in the sodium hypochlorite solution and will contribute bromate to the treated water. The quality and acceptability of sodium hypochlorite will partly be a function of the concentration of the bromate residue. Industrial-grade product may not be acceptable for drinking-water applications. The sodium bromide naturally present in sodium chloride will also be oxidized to form bromate in systems using on-site electrochemical generation of hypochlorite.

Contaminants from use of ozone and chlorine dioxide

The use of ozone can lead to elevated bromate concentrations through oxidation of bromide present in the water. As a general rule, the higher the bromide concentration in the water, the more bromate that is produced.

Chlorine dioxide solutions can contain chlorate as a result of reactions that compete with the desired reaction for generation of chlorine dioxide. Chlorite ion is an inevitable decomposition product from the use of chlorine dioxide; typically, 60–70% of the applied dose is converted to chlorite in the treated water.

8.5.5 Chemicals of emerging concern

Pharmaceuticals

Pharmaceuticals can be introduced into water sources in sewage by excretion from individuals using these chemicals, from uncontrolled drug disposal (e.g. discarding drugs into toilets) and from agricultural runoff from livestock manure. They have become chemicals of emerging concern to the public because of their potential to reach drinking-water.

The specific types of pharmaceuticals and their metabolites in water sources can differ between countries or regions depending on social, cultural, technological and agricultural factors. Urban and rural areas may exhibit important differences in the occurrence and concentrations of these chemicals as a result of different usage patterns. The local physical and chemical characteristics of source waters can also affect the occurrence levels of pharmaceuticals by influencing their natural degradation.

Most occurrence data in drinking-water and source water have resulted from targeted investigations, rather than from systematic monitoring. Advancements in the sensitivity and accuracy of detection technologies and methodologies have led to increasing detection of trace amounts of pharmaceuticals, ranging from concentrations in the nanogram per litre to low microgram per litre range (although largely less than 0.1 µg/l) in drinking-water, surface water and groundwater. Higher concentrations of these contaminants are found in wastewater treatment effluents or wastewater discharges from poorly controlled manufacturing facilities.

The concentrations of pharmaceuticals found in drinking-water are typically orders of magnitude less than the lowest therapeutic doses. Therefore, exposure to individual compounds in drinking-water is unlikely to have appreciable adverse impacts on human health. Formal guideline values are therefore not proposed in these Guidelines.

Routine monitoring for pharmaceuticals in drinking-water and additional or specialized drinking-water treatment to reduce the concentrations of pharmaceuticals

in drinking-water are not considered necessary. However, where local circumstances indicate a potential for elevated concentrations of pharmaceuticals in drinking-water, investigative monitoring and surveys of impacted water sources can be undertaken to assess possible exposure. If undertaken, these surveys should be quality assured and should target pharmaceuticals that are of local significance—i.e. those that are commonly prescribed and used or manufactured locally. Based on the risk assessment, screening values can be developed to assess the potential risks from exposure through drinking-water, and possible control measures could be considered within the context of WSPs. Practical difficulties with implementing monitoring programmes include lack of standardized sampling and analysis protocols, high costs and limited availability of technologies needed to detect the diverse range of pharmaceuticals that may be present.

Effective treatment of pharmaceuticals depends on the physicochemical properties of the specific compounds. Typically, conventional treatment processes are less effective than advanced treatment processes for the removal of many organic compounds, particularly those that are more water soluble.

Preventive measures, such as rational drug use and education of prescribers and the public to reduce disposal and discharges to the environment, will likely reduce human exposure.

Further information is available in *Pharmaceuticals in drinking-water* (see [Annex 1](#)).

8.6 Pesticides used in water for public health purposes

For those diseases transmitted to humans by invertebrate organisms, referred to as vectors, vector control intervention programmes to control/mitigate populations or interaction with humans are part of critical first line health systems aimed to prevent disease. The use of vector control interventions is vital in many countries, especially where other preventive and treatment measures may not be possible. The use of vector control products has been instrumental in preventing the transmission of malaria and other major vector-borne diseases such as dengue fever, and schistosomiasis. There are occasions when vectors, particularly mosquitoes, breed in containers used for the storage and collection of drinking-water. Although actions should be taken to prevent access of vectors to or breeding of vectors in these containers, this is not always possible or may not always be fully effective, and use of mosquito larvicides may be indicated in certain settings.

It is important to note that not all larvicides are appropriate for use in drinking-water or containers that may contain drinking-water. Only those larvicide products that have been assessed and registered for use in drinking-water in accordance with local, national or regional regulatory requirements should be used. The list of prequalified products established by WHO may be an important point of reference in identifying quality assured products (see <https://extranet.who.int/prequal/vector-control-products/prequalified-product-list>). Products on the list have been found to meet WHO standards for quality, safety and efficacy. The list will continually be updated if data are submitted by manufacturers for WHO assessment and the supporting information demonstrates that the product and its uses meet the standards.

While it is not appropriate to set guideline values for pesticides used for vector control, it is valuable to provide information to support water and health authorities to assess the safety of drinking-water containing larvicides used for this purpose. Accordingly, these Guidelines provide information on acceptable daily intakes (ADIs) of larvicide active ingredients that have been known to be, or considered to be used, as larvicides in drinking-water (see [Table 8.18](#)). In assessing the safety of vector control pesticides in drinking-water, an assessment could be made of the potential exposure compared with the ADI as shown in [Box 8.1](#).

Where exposure is expected to exceed the ADI for small children and/or bottle-fed infants, considerations should be given to using an alternative safe source of drinking-water (e.g. bottled water that is certified by the responsible authorities) for these population groups, for a period after application where this is practical. However, exceeding the ADI (for all population groups) does not necessarily mean this will result in adverse health effects. The diseases spread by vectors are significant causes of morbidity and mortality. It is therefore important to achieve an appropriate balance between the intake of the pesticide from drinking-water and the control of disease carrying insects. It is stressed that every effort should be made to keep overall exposure and the concentration of any larvicide no greater than that authorized/recommended by the relevant authority and as low as possible commensurate with efficacy. Only products registered by the relevant authority should be used and application should be used in accordance with label recommendations.

Member States should consider the use of larvicides within the context of their broad vector control strategy. The use of larvicides should only be part of a comprehensive management plan for household water storage and domestic waste management that does not rely exclusively on larviciding by insecticides, but also includes other environmental management measures and social behaviour change. Nevertheless, it would be valuable to obtain actual data on exposure to these substances under field conditions in order to carry out a more refined assessment of margins of exposure.

In addition to the use of larvicides approved for drinking-water application to control disease vector insects, other control measures should also be considered. For example, the stocking of fish of appropriate varieties (e.g. larvae-eating mosquito-fish and predatory copepods) in water bodies may adequately control infestations and breeding of mosquitoes in those bodies. Other mosquito breeding areas where water collects should be managed by draining, especially after rainfall. For guidance on designing, implementing and evaluating larval source management programmes to control *Anopheles* and *Aedes* mosquito vectors, see the *Operational manual on larval source management* (WHO, 2026). This manual includes an overview of the main types of larvicides including information on types of formulations and advantages and disadvantages of the different types.

The larvicides referenced in the operational manual and previously assessed for these Guidelines are listed in [Table 8.18](#). Dichlorodiphenyltrichlorethane (DDT) has been used for public health purposes in the past. It is being reintroduced (but not for water applications) in some areas to control malaria-carrying mosquitoes. Its guideline value is shown in [Table 8.19](#).

Table 8.18 Pesticides potentially used for public health purposes for which guideline values have not been derived^a

Pesticide	Reason for not establishing a guideline value	ADI for potential use in assessing health risks (mg/kg bw) ^b
<i>Bacillus</i> ^c <i>thuringiensis israelensis</i> (Bti)	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	Not applicable (biological agent) ^d
Diflubenzuron	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.02
Methoprene	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.09
Novaluron	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.01
Pirimiphos-methyl	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.03
Pyriproxyfen	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.1
Spinosad	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0–0.02
Temephos	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water	0.023 ^e

^a Inclusion in this table does not indicate WHO endorsement. Only those larvicide products that have been assessed and registered for use in drinking-water in accordance with local, national or regional regulatory requirements should be used. The list of prequalified products established by WHO (<https://extranet.who.int/prequal/vector-control-products/prequalified-product-list>) may be an important point of reference in identifying quality assured products. Products on the list, including one containing polydimethylsiloxane, have been found to meet WHO standards for quality, safety and efficacy.

^b Information on ADIs for these and other pesticides including permethrin, can be found within JMPR reports, which can be accessed at <https://apps.who.int/pesticide-residues-jmpr-database/>. The ADIs are also summarized in WHO publications on these larvicides in drinking-water (Annex 2). The ADIs are a reference point for water and health authorities to assess the safety of drinking-water containing these pesticides used for vector control. An assessment could be made of the potential exposure compared with the ADI (see Box 8.1). However, exceeding the ADI does not necessarily mean that this will result in adverse health effects. It is recommended that water authorities consult with health authorities and pesticide authorities to assess significance of detection. An assessment on exposure considering previously known recommended dosages can be found in these background documents. While these publications have been withdrawn as background documents to these Guidelines for which chapter 12 fact sheets were previously developed, they may be a useful source of information for water and health authorities in assessing human health risks of vector control pesticides in drinking-water, particularly when exposure is estimated to be close to the ADI.

^c The *Operational manual on larval source management* (WHO, 2026) also references *Bacillus sphaericus*.

^d Bti itself is not considered to pose a hazard to humans through drinking-water (WHO, 2009).

^e This is a TDI rather than an ADI, as JMPR considered that the database was insufficiently robust to serve as the basis for establishing an ADI for temephos. For the purposes of these Guidelines, a TDI has been calculated from the lowest oral NOAEL in the critical study identified by JMPR (WHO, 2009).

Box 8.1 Assessing human health risks of vector control pesticides in drinking-water

In assessing the safety of vector control pesticides in drinking-water, an assessment could be made of the estimated exposure compared with the dose in drinking-water at the ADI as follows.

Estimated exposure = dosage $\times$ C

where

dosage: maximum dosage recommended by manufacturer (mg/L) (as a conservative assumption) or concentration detected in drinking-water

C = daily drinking-water consumption (2 L for adults, 1 L for children, 0.75 L for bottle-fed infants)

Dose in drinking-water at the ADI = ADI $\times$ bw

where

ADI (mg/kg bw)

bw = body weight (60 kg for adults, 10 kg for children, 5 kg for infants)

C = daily drinking-water consumption (2 L for adults, 1 L for children, 0.75 L for bottle-fed infants)

An example of an assessment is shown for diflubenzuron below^a

JMPR established an ADI for diflubenzuron at 0–0.02 mg/kg bw.

The upper-limit dose in drinking-water at the ADI is the following:

Adult: 0.02 mg/kg bw $\times$ 60 kg = 1.2 mg

Child: 0.02 mg/kg bw $\times$ 10 kg = 0.2 mg

Bottle-fed infant: 0.02 mg/kg bw $\times$ 5 kg = 0.1 mg

Exposure to diflubenzuron in drinking-water is assumed to be 0.25 mg/L based on the maximum recommended dosage in drinking-water.

This is equivalent to the following maximum doses in drinking-water:

Adult: 0.25 mg/L $\times$ 2L = 0.5 mg

Child: 0.25 mg/L $\times$ 1L = 0.25 mg

Bottle-fed infant: 0.25 mg/L $\times$ 0.75L = 0.19 mg

Exposure to diflubenzuron in drinking-water is not considered a risk for adults considering that the estimated dose is approximately 40% of the upper limit of the ADI.

Consideration should also be given to the fact that diflubenzuron is unlikely to remain in solution at the maximum recommended applied dose, and therefore, the actual levels of exposure are likely to be much lower than those calculated. Nevertheless as a precautionary measure, consideration should be given to using an alternative safe source of drinking-water for bottle-fed infants for a period after an application of diflubenzuron, where this is practical.

^a This example is based on information in *Diflubenzuron in drinking-water: use for vector control in drinking-water sources and containers* (WHO, 2008). While this and the other larvicide drinking-water documents have been withdrawn as background documents for the preparation of these Guidelines for which [chapter 12](#) fact sheets were developed, they may be a useful source of information for water and health authorities in assessing human health risks of vector control pesticides in drinking-water, particularly when exposure is estimated to be close to the ADI.

Table 8.19 Guideline values for pesticides that were previously used for public health purposes and are of health significance in drinking-water

Pesticides previously used for public health purposes	Guideline value	
	µg/l	mg/l
DDT and metabolites	1	0.001

8.7 Identifying local actions in response to chemical water quality problems and emergencies

It is difficult to give comprehensive guidance concerning emergencies in which chemicals cause massive contamination of the drinking-water supply, caused either by accident or by deliberate action. Most of the guideline values recommended in these Guidelines (see [section 8.5](#) and [Annex 3](#)) relate to a level of exposure that is regarded as tolerable throughout life. Acute toxic effects are considered for a limited number of chemicals. The length of time for which exposure to a chemical far in excess of the guideline value would have adverse effects on health will depend upon factors that vary from contaminant to contaminant. In an emergency situation, the public health authorities should be consulted about appropriate action.

The exceedance of a guideline value may not result in a significant or increased risk to health. Therefore, deviations above the guideline values in either the short or long term may not necessarily mean that the water is unsuitable for consumption. The amount by which, and the period for which, any guideline value can be exceeded without affecting public health depends upon the specific substance involved, and acceptability judgements need to be made by qualified health officials. However, exceedance should be a signal:

- as a minimum, to investigate the cause with a view to taking remedial action as necessary;
- to consult the authority responsible for public health for advice on suitable action, taking into account the intake of the substance from sources other than drinking-water, the toxicity of the substance, the likelihood and nature of any adverse effects and the practicality of remedial measures.

If a guideline value is to be exceeded by a significant amount or for more than a few days, it may be necessary to act rapidly so as to ensure that health protective action is taken and to inform consumers of the situation so that they can act appropriately.

The primary aim with regard to chemical contaminants when a guideline value is exceeded or in an emergency is to prevent exposure of the population to toxic concentrations of pollutants. However, in applying the Guidelines under such circumstances, an important consideration is that, unless there are appropriate alternative supplies of drinking-water available, maintenance of adequate quantities of water is a high priority. In the case of an incident in which chemical contaminants are spilt into a source water and enter a drinking-water supply or enter a supply through treatment or during distribution, the primary aim is to minimize the risk of adverse effects without unnecessarily disrupting the use of the water supply.

This section of the Guidelines can be used to assist evaluation of the risks associated with a particular situation and—especially if a guideline value exists or an authoritative risk assessment is available from an alternative source—support appropriate decision-making on short- and medium-term actions. The approaches proposed provide a basis for discussion between various authorities and for judging the urgency of taking further action.

Normally, a specific review of the situation will be required and should call on suitable expertise. It is important to take local circumstances into account, including the availability of alternative water supplies and exposure to the contaminant from other sources, such as food. It is also important to consider what water treatment is applied or available and whether this will reduce the concentration of the substance.

Where the nature of contamination is unknown, expert opinion should be sought as quickly as possible to identify the contaminants, to determine what actions can be taken to prevent the contaminants from entering the supply and to minimize the exposure of the population and so minimize any potential for adverse effects.

A WSP should include planning for response to both predictable events and undefined “emergencies”. Such planning facilitates rapid and appropriate response to events when they occur (see [section 4.4](#)).

Consideration of emergency planning and planning for response to incidents in which a guideline value is exceeded, covering both microbial and chemical contaminants, is discussed in [section 4.4](#). Broader discussion of actions in emergency situations can be found in [section 6.8](#) and, for microbial contamination, [section 7.6](#).

8.7.1 Trigger for action

Triggers for action may include:

- detection of a spill by, or reporting of a spill to, the drinking-water supplier;
- an alarm raised by the observation of items, such as chemical drums, adjacent to a vulnerable part of the drinking-water supply;
- the detection of a substance in the water;
- a sudden change to water treatment;
- consumer complaints (e.g. an unusual odour, taste or discoloration).

8.7.2 Investigating the situation

Each incident is unique, and it is therefore important to determine associated facts, including what the contaminant is; what the likely concentration is, and by how much the guideline value has been exceeded, if at all; and the potential duration of the incident. These are important in determining the actions to be taken.

8.7.3 Talking to the right people

In any emergency, it is important that there be good communication between the various authorities, particularly the water supplier and health authorities. It will usually be the health authorities that make the final decisions, but knowledge of the water supply and the nature of the supply is vital in making the most appropriate decisions. In addition, timely and clear communication with consumers is a vital part of successfully handling drinking-water problems and emergencies.

Liaison with key authorities is discussed in [section 4.4](#). It is particularly important to inform the public health authority of any exceedance or likely exceedance of a guideline value or other conditions likely to affect human health and to ensure that the public health authority is involved in decision-making. In the event of actions that require all consumers to be informed or where the provision of temporary supplies of drinking-water is appropriate, civil authorities should also be involved. Planning for these actions is an important part of the development of WSPs. Involving the public health authorities at an early stage enables them to obtain specialist information and to make the appropriate staff available.

8.7.4 Informing the public

Consumers may be aware of a potential problem with the safety of their drinking-water because of media coverage, their own senses or informal networks. Lack of confidence in the drinking-water or the authorities may drive consumers to alternative, potentially less safe sources. Not only do consumers have a right to information on the safety of their drinking-water, but they have an important role to play in assisting the authorities in an incident by their own actions and by carrying out the necessary measures at the household level. Trust and goodwill from consumers are extremely important in both the short and long term.

The health authorities should be involved whenever a decision to inform the public of health-based concerns or advice to adopt health protection measures such as boiling of water may be required. Such guidance needs to be both timely and clear.

8.7.5 Evaluating the significance to public health and individuals

In assessing the significance of an exceedance of a guideline value, account should be taken of:

- information underpinning the guideline value derivation;
- local exposure to the substance of concern through other routes (e.g. food);
- any sensitive subpopulations;
- locally relevant protective measures to prevent the chemical from entering the source water or supply in the case of a spill.

Information underpinning guideline value derivation

The derivation of guideline values for chemical contaminants is described in [section 8.2](#).

Most guideline values are derived by calculating a TDI or using an existing TDI or ADI. A proportion of the TDI or ADI is then allocated to drinking-water to make allowance for exposure from other sources, particularly food. This allocation is often 20%, but it may be as low as 1% or as high as 80%. In many circumstances, a review of likely local sources of exposure may identify that sources other than drinking-water are less significant than assumed and that a larger proportion of total exposure can be safely allocated to drinking-water. The fact sheets in [chapter 12](#) and background documents on all chemicals addressed in these Guidelines (<https://www.who.int/teams/environment-climate-change-and-health/water-sanitation-and-health/chemical-hazards-in-drinking-water>) provide further information on likely sources of

the chemicals concerned, including their allocation factors. When rapid decision-making is required for such chemicals, it is possible to allow 100% of the TDI to come from drinking-water for a short period (e.g. a few days) while undertaking a more substantive review. In the event that there is significant exposure from other sources or exposure is likely to be for more than a few days, then it is possible to allocate more than the allocation used in the guideline value derivation, but no more than 100%.

In some cases, the guideline value is derived from epidemiological or clinical studies in humans. In most cases (e.g. benzene, barium), these relate to long-term exposure, and short-term exposure to concentrations higher than the guideline value are unlikely to be of significant concern; however, it is important to seek expert advice. In other cases of guideline values derived from epidemiological studies, the associated health effects are acute in nature. For example:

- The guideline value for nitrate is 50 mg/L, (as nitrate ion), to be protective of the health of the most sensitive subpopulation, bottle-fed infants. This guideline value is based on the absence of adverse health effects (methaemoglobinaemia and thyroid effects) at concentrations below 50 mg/L in epidemiological studies. Although the guideline value is based on short-term effects, it is protective for long-term effects and in other population groups, such as older children and adults. Methaemoglobinaemia is complicated by the presence of microbial contamination and subsequent gastrointestinal infection, which can increase the risk for this group significantly. Authorities should therefore be all the more vigilant that water to be used for bottle-fed infants is microbiologically safe when nitrate is present at concentrations near or above the guideline value. It is also particularly important to ensure that these infants are not currently exhibiting symptoms of gastrointestinal infection (diarrhoea). In addition, because excessive boiling of water to ensure microbiological safety can concentrate levels of nitrate in the water, care should be taken to ensure that water is heated only until the water reaches a rolling boil. In extreme situations, alternative sources of water (e.g. bottled water) can be used.
- The guideline value for copper is also based on short-term exposure but is intended to protect against direct gastric irritation, which is a concentration-dependent phenomenon. The guideline value may be exceeded, but there will be an increasing risk of consumers suffering from gastrointestinal irritation as the concentration increases above the guideline value. The occurrence of such irritation can be assessed in exposed populations.

In some cases, the guideline value is derived from a cancer risk estimate derived from studies in laboratory animals. In these cases, short-term (a few months to a year) exposure to concentrations up to 10 times the guideline value would result in only a small increase in estimated risk of cancer. Because the estimate of risk varies over a wide range, there may be no, or a very small, increase in risk. In such a circumstance, accepting a 10-fold increase in the guideline value for a short period would have no discernible impact on the risk over a lifetime. However, care would be needed to determine whether other toxicological end-points more relevant for short-term exposure, such as neurotoxicity, would become significant.

Health-based values for short-term exposures are now being developed for a small number of substances that are used in significant quantities and are frequently implicated in an emergency as a consequences of spills, usually to surface water sources. The methodology used in the derivation of these health-based values is described below.

Health-based values for use in emergencies

Health-based values for acute and short-term exposures (called acute and short-term health-based values) can be derived for any chemicals that are used in significant quantities and are involved in an emergency, such as a spill into surface water sources.

JMPR has provided guidance on the setting of acute reference doses (ARfDs) for pesticides (Solecki et al., 2005). These ARfDs can be used as a basis for deriving acute health-based values for pesticides in drinking-water, and the general guidance can also be applied to derive ARfDs for other chemicals. The JMPR ARfD is usually established to cover the whole population, and must be adequate to protect the embryo or fetus from possible in utero effects. An ARfD based on developmental (embryo/fetal) effects, which applies to women of childbearing age only, may be conservative and not relevant to other population subgroups.¹

The ARfD can be defined as the amount of a chemical, normally expressed on a body weight basis, that can be ingested in a period of 24 hours or less without appreciable health risk to the consumer. Most of the scientific concepts applicable to the setting of ADIs or TDIs for chronic exposure apply equally to the setting of ARfDs. The toxicological end-points most relevant for a single or 1-day exposure should be selected. For ARfDs for pesticides, possible relevant end-points include haematotoxicity (including methaemoglobin formation), immunotoxicity, acute neurotoxicity, liver and kidney toxicity (observed in single-dose studies or early in repeated-dose studies), endocrine effects and developmental effects. The most relevant or adequate study in which these end-points have been determined (in the most sensitive species or most vulnerable subgroup) is selected, and NOAELs are established. The most relevant end-point providing the lowest NOAEL is then used in the derivation of the ARfD. Uncertainty factors are used to extrapolate from experimental animal data to the average human and to allow for variation in sensitivity within the human population. An ARfD derived in such a manner can then be used to establish an acute health-based value by allocating 100% of the ARfD to drinking-water, as follows:

$$\text{Acute health-based value} = \frac{\text{ARfD} \times \text{bw} \times \text{P}}{\text{C}}$$

where:

- bw = body weight (60 kg for adults, 10 kg for children, 5 kg for infants)
- P = fraction of the ARfD allocated to drinking-water (100%)
- C = daily drinking-water consumption (2 L for adults, 1 L for children, 0.75 L for bottle-fed infants)

¹ ARfDs established for pesticides by JMPR may be found at <http://apps.who.int/pesticide-residues-jmpr-database>.

However, available data sets do not allow the accurate evaluation of the acute toxicity for a number of compounds of interest. If appropriate single-dose or short-term data are lacking, an end-point from a repeated-dose toxicity study can be used. This is likely to be a more conservative approach, and this should be clearly stated in the health-based value derivation.

When a substance has been spilt into a drinking-water source, contamination may be present for a period longer than 24 hours, but is not usually present for longer than a few days. Under these circumstances, the use of data from repeated-dose toxicity studies is appropriate to derive a short-term health-based value (using the approach outlined in [section 8.2.2](#)). As the period of exposure used in these studies will often be much longer than a few days, this, too, is likely to be a conservative approach.

Where there is a need for a rapid response, and suitable data are not available to establish an ARfD but a guideline value or health-based value is available for the chemical of concern, a pragmatic approach would be to allocate a higher proportion of the ADI or TDI to drinking-water. As the ADI or TDI is intended to be protective of lifetime exposure, small exceedances of the ADI or TDI for short periods will not be of significant concern for health. In these circumstances, it would be reasonable to allow 100% of the ADI or TDI to come from drinking-water for a short period.

Assessing locally relevant sources of the substance of concern through other routes of exposure

The most useful sources of information regarding local exposure to substances through food and, to a lesser extent, air and other environmental routes are usually government departments dealing with food and environmental pollution. Other sources of information may include universities. In the absence of specific data, the Guidelines background documents consider the sources of exposure and give a generic assessment that can be used to make a local evaluation as to the potential use of a chemical and whether this would be likely to enter the food-chain. Further information is available in the supporting document *Chemical safety of drinking-water* ([Annex 1](#)).

Sensitive subpopulations

In some cases, there may be a specific subpopulation that is at greater risk from a substance than the rest of the population. These usually relate to high exposure relative to body weight (e.g. bottle-fed infants) or a particular sensitivity (e.g. fetal haemoglobin and nitrate/nitrite). However, some genetic subpopulations may show greater sensitivity to particular toxicity (e.g. glucose-6-phosphate dehydrogenase-deficient groups and oxidative stress on red blood cells). If the potential exposure from drinking-water in an incident is greater than the ADI or TDI or exposure is likely to be extended beyond a few days, then this would require consideration in conjunction with health authorities. In such circumstances, it may be possible to target action to avoid exposure of the specific group concerned, such as supplying bottled water for bottle-fed infants.

Specific mitigation measures affecting risk assessment

Such measures relate to actions taken locally or on a household basis that can have an impact on the presence of a particular contaminant. For example, the presence of a substance that is volatile or heat labile will be affected by heating the water for cooking or the preparation of beverages. Where such measures are routinely undertaken by the exposed population, the risk assessment may be modified accordingly. Alternatively, such steps can be used on a household basis to reduce exposure and allow the continued use of the supply without interruption.

8.7.6 Determining appropriate action

Determining appropriate action means that various risks will need to be balanced. The interruption of water supply to consumers is a serious step and can lead to risks associated with contamination of drinking-water stored in the household with pathogens and limiting use for purposes of hygiene and health protection. Issuing a “do not drink” notice may allow the use of the supply for hygiene purposes such as showering or bathing, but creates pressure on consumers and authorities to provide a safe alternative for drinking and cooking. In some cases, this option will be expensive and could divert resources from other, more important issues. Appropriate action will always be decided on a case-by-case basis in conjunction with other authorities, including the health protection and civil authorities, who may be required to participate in informing consumers, delivering alternative supplies or supervising the collection of water from bowzers and tankers. Responding to a potential risk to health from a chemical contaminant should not lead to an increase in overall health risk from disruption of supply, microbial contaminants or other chemical contaminants.

8.7.7 Consumer acceptability

Even though, in an emergency, supplying water that contains a substance present at higher concentrations than would normally be desirable may not result in an undue risk to health, the water may not be acceptable to consumers. A number of substances that can contaminate drinking-water supplies as a consequence of spills can give rise to severe problems with taste or odour. Under these circumstances, drinking-water may become so unpalatable as to render the water undrinkable or to cause consumers to turn to alternative drinking-water sources that may present a greater risk to health. In addition, water that is clearly contaminated may cause some consumers to feel unwell due to a perception of poor water quality. Consumer acceptability may be the most important factor in determining the advice given to consumers about whether or not the water should be used for drinking or cooking.

8.7.8 Ensuring remedial action, preventing recurrence and updating the WSP

The recording of an incident, the decisions taken and the reasons for them are essential parts of handling an incident. The WSP, as discussed in [chapter 4](#), should be updated in the light of experience. This would include making sure that problem areas identified during an incident are corrected. Where possible, it would also mean that

the cause of the incident is dealt with to prevent its recurrence. For example, if the incident has arisen as a consequence of a spill from industry, the source of the spill can be advised as to how to prevent another spill and the information passed on to other similar industrial establishments.

8.7.9 Mixtures

A spill may contain more than one contaminant of potential health concern (see [section 8.2.8](#)). Under these circumstances, it will be important to determine whether the substances present interact. Where the substances have a similar mechanism or mode of action, it is appropriate to consider them as additive. This may be particularly true of some pesticides, such as atrazine and simazine. In these circumstances, appropriate action must take local circumstances into consideration. Specialist advice should generally be sought.

8.7.10 Water avoidance advisories

Water avoidance advisories share many features with boil water advisories (see [section 7.6.1](#)), but are less common. Like boil water advisories, they are a serious measure that should be instituted only when there is evidence that an advisory is necessary to reduce a substantial public health risk. In cases where alternative sources of water are recommended, particular consideration should be given to the potential for microbial hazards in those alternative sources. Water avoidance advisories are applied when the parameter of concern is not susceptible to boiling or when risks from dermal contact or inhalation of the contaminant are also significant. Water avoidance advisories may also be issued when an unknown agent or chemical substance is detected in the distribution system. It is important that the water avoidance advisories include the information that boiling is ineffective or insufficient to reduce the risk.

As with the case of boil water advisories, water suppliers in conjunction with public health authorities should develop protocols for water avoidance advisories. Protocols should be prepared before any incident occurs and incorporated within WSPs. Decisions to issue advisories are often made within a short period of time, and developing responses during an event can complicate decision-making, compromise communication and undermine public confidence.

In addition to the information discussed in [section 4.4.3](#), the protocols should provide information to the general public and specific groups on the following:

- criteria for issuing and rescinding an advisory;
- activities impacted by the advisory;
- alternative sources of safe water for drinking and other domestic uses.

Protocols should identify mechanisms for the communication of water avoidance advisories. The mechanisms may vary, depending on the nature of the supply and the size of the community affected, and could include:

- media releases through television, radio and newspapers;
- telephone, e-mail and fax contact of specific facilities, community groups and local authorities;

- posting of notices in conspicuous locations;
- personal delivery;
- mail delivery.

The methods chosen should provide a reasonable assurance that all of those affected by the advisory, including residents, workers and travellers, are notified as soon as possible.

The issuing of a water avoidance advisory may be necessary, for example, following contamination—for example, chemical or radiological—as a result of accidental, natural or malicious origin that leads to:

- a significant exceedance of a guideline value, which may pose a threat to health from short-term exposure;
- concentrations of a chemical with no guideline value that may pose a threat to health from short-term exposure;
- significant odour or taste that has no identified source or that will give rise to significant public anxiety.

When issued, water avoidance advisories should provide information on the same issues included in boil water advisories (see [section 7.6.1](#)), although recommendations relating to affected uses and users will vary, depending on the nature of the problem. For example, for elevated concentrations of contaminants that are of concern only from a drinking or cooking perspective, the public could be advised to avoid using the water for drinking, food preparation, preparing cold drinks, making ice and hygienic uses, such as tooth brushing. Where the advisory applies to elevated levels of chemicals that can cause skin or eye irritation or gastrointestinal upsets, the public could be advised not to use the water for drinking, cooking, tooth brushing or bathing/showering. Alternatively, specific water avoidance advice might be issued where the contamination might affect subgroups of the population—for example, pregnant women or bottle-fed infants.

As for boil water advisories, specific advice may need to be issued for dentists, doctors, hospitals and other health-care facilities, child-care facilities, schools, food suppliers and manufacturers, hotels, restaurants and operators of public swimming pools.

Water avoidance advisories do not equate to cessation of supply; water will generally be suitable for flushing toilets and other uses, such as clothes washing. However, suitable alternative supplies of drinking-water, such as bottled water and carted or tankered water, will be required for drinking and other domestic uses.

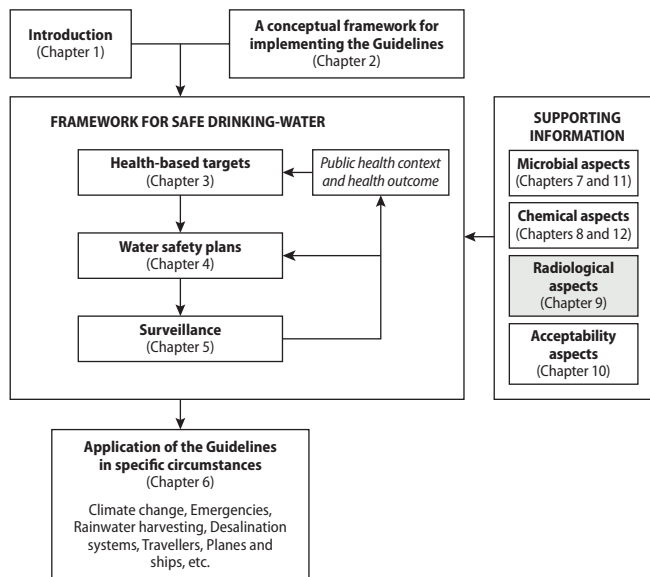
Criteria for rescinding water avoidance advisories will generally be based on evidence that the source of elevated concentrations of hazardous contaminants has been removed, that distribution systems have been appropriately flushed and that the water is safe for drinking and other uses. In buildings, the flushing would extend to storages and internal plumbing systems.

9

Radiological aspects

Drinking-water may contain radioactive substances (“radionuclides”) that could present a risk to human health. These risks are normally small compared with the risks from micro-organisms and chemicals that may be present in drinking-water. Except in extreme circumstances, the radiation dose resulting from the ingestion of radionuclides in drinking-water is much lower than that received from other sources of radiation. The objective of this chapter is to provide criteria with which to assess the safety of drinking-water with respect to its radionuclide content and to provide guidance on reducing health risks by taking measures to decrease radionuclide concentrations, and therefore radiation doses, in situations where this is considered necessary.

In terms of health risk assessment, the Guidelines do not differentiate between radionuclides that occur naturally and those that arise from human activities. However, in terms of risk management, a differentiation is made because, in principle, human-made radionuclides are often controllable at the point at which they enter the water supply. Naturally occurring radionuclides, in contrast, can potentially enter the water supply at any point, or at several points, prior to consumption. For this reason, naturally occurring radionuclides in drinking-water are often less amenable to control.



Naturally occurring radionuclides in drinking-water usually give radiation doses higher than those provided by artificially produced radionuclides and are therefore of greater concern. Radiological risks are best controlled through a preventive risk management approach following the framework for safe drinking-water (see [chapter 2](#)) and the water safety plan approach (see [chapter 4](#)). When considering what action to take in assessing and managing radiological risks, care should be taken to ensure that scarce resources are not diverted away from other, more important public health concerns.

The screening levels and guidance levels for radioactivity presented in these Guidelines are based on the latest recommendations of the International Commission on Radiological Protection (ICRP, 2007).

Some drinking-water supplies, in particular those sourced from groundwater, may contain radon, a radioactive gas. Although radon can enter indoor air in buildings through its release from water from taps or during showering, the most significant source of radon in indoor air is the underlying soil and building materials. An evaluation of international research data (UNSCEAR, 2000) has concluded that, on average, 90% of the dose attributable to radon in drinking-water comes from inhalation rather than ingestion. Consequently, the setting of screening levels and guidance levels to limit the dose from ingestion of radon contained in drinking-water is not usually necessary. The screening measurements for gross alpha and gross beta activities will include the contribution from radon progeny, which is the principal source of dose from ingestion of radon present in drinking-water supplies. This is further discussed in [section 9.7](#).

For further information on radionuclides in drinking-water, including more detailed guidance on considerations for developing drinking-water standards and actions to take when WHO criteria have been exceeded, see the supporting document *Management of radioactivity in drinking-water* ([Annex 1](#)).

9.1 Sources¹ and health effects of radiation exposure

Radioactivity from several naturally occurring and human-made sources is present throughout the environment. Some chemical elements present in the environment are naturally radioactive. These are found in varying amounts in soils, water, indoor and outdoor air and even within our bodies, and so exposure to them is inevitable. In addition, Earth is constantly bombarded by high-energy particles originating both from the sun and from outside the solar system. Collectively, these particles are referred to as cosmic radiation. Everybody receives a dose from cosmic radiation, which is influenced by latitude, longitude and height above sea level.

The use of radiation in medicine for diagnosis and treatment is the largest human-made source of radiation exposure today. The testing of nuclear weapons, routine discharges from industrial and medical facilities and accidents such as Chernobyl have added human-made radionuclides to our environment.

¹ When the term “source” appears in this chapter without any other reference, it is used in the context of “radiation source”. For any other purpose, additional information is provided (e.g. “water source”).

Box 9.1 Key terms, quantities and units

Becquerel (Bq)—The becquerel is the unit of radioactivity in the International System of Units (abbreviated SI from the French *Système international d'unités*), corresponding to one radioactive disintegration per second. In the case of drinking-water, it is usual to talk about the activity concentration, expressed in units of Bq/l.

Effective dose—When radiation interacts with body tissues and organs, the radiation dose received is a function of factors such as the type of radiation, the part of the body affected and the exposure pathway. This means that 1 Bq of radioactivity will not always deliver the same radiation dose. A unit called “effective dose” has been developed to take account of the differences between different types of radiation so that their biological impacts can be compared directly. The effective dose is expressed in SI units called sieverts (Sv). The sievert is a very large unit, and it is often more practical to talk in terms of millisieverts (mSv). There are 1000 mSv in 1 Sv.

Effective half-life—Radioisotopes have a “physical” half-life, which is the period of time it takes for one half of the atoms to disintegrate. Physical half-lives for various radioisotopes can range from a few microseconds to billions of years. When a radioisotope is present in a living organism, it may be excreted. The rate of this elimination is influenced by biological factors and is referred to as the “biological” half-life. The effective half-life is the actual rate of halving the radioactivity in a living organism as determined by both the physical and biological half-lives. Whereas for certain radionuclides, the biological processes are dominant, for others, physical decay is the dominant influence.

Radioactivity is described using different quantities and units (see [Box 9.1](#)). The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR, 2008) has estimated that the global average annual dose per person from all sources of radiation in the environment is approximately 3.0 mSv/year. Of this, 80% (2.4 mSv) is due to naturally occurring sources of radiation, 19.6% (almost 0.6 mSv) is due to the use of radiation for medical diagnosis and the remaining 0.4% (around 0.01 mSv) is due to other sources of human-made radiation (see [Figure 9.1](#)). There can be large variability in the dose received by individual members of the population, depending on where they live, their dietary preferences and other lifestyle choices. Individual radiation doses can also differ depending on medical treatments and occupational exposures. Annual average doses and typical ranges of individual doses from naturally occurring sources are presented in [Table 9.1](#) (UNSCEAR, 2008).

9.1.1 Radiation exposure through ingestion of drinking-water

Water sources can contain radionuclides of natural and artificial origin (i.e. human-made):

- Natural radionuclides, including potassium-40, and those of the thorium and uranium decay series, in particular radium-226, radium-228, uranium-234, uranium-238, lead-210 and polonium-210, can be found in water in varying amounts, as a result of either natural processes (e.g. absorption from the soil) or technological processes involving naturally occurring radioactive materials (e.g. the mining and processing of mineral sands or phosphate fertilizer production).

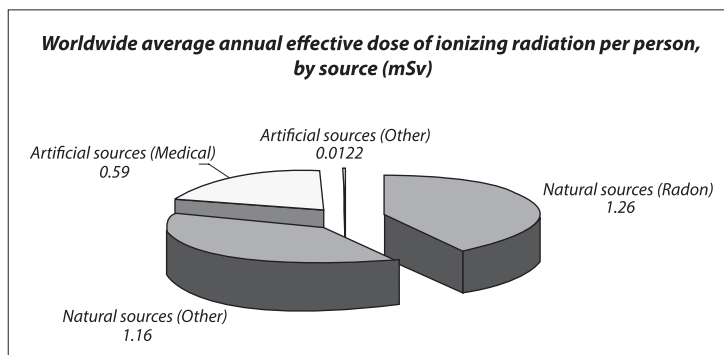


Figure 9.1 Distribution of average radiation exposure for the world population

Table 9.1 Average radiation dose from naturally occurring sources

Source	Worldwide average annual effective dose (mSv)	Typical annual effective dose range (mSv)
External exposure		
Cosmic rays	0.39	0.3–1 ^a
Terrestrial radiation (outdoors and indoors)	0.48	0.3–1 ^b
Internal exposure		
Inhalation (mainly radon)	1.26	0.2–10 ^c
Ingestion (food and drinking-water)	0.29	0.2–1 ^d
Total	2.4	1–13

^a Range from sea level to high ground elevation.

^b Depending on radionuclide composition of soil and building material.

^c Depending on indoor accumulation of radon gas.

^d Depending on radionuclide composition of foods and drinking-water; the drinking-water contribution to the ingested dose is typically 0.05 mSv.

Source: Adapted from UNSCEAR (2008)

- Human-made radionuclides may be present in water from several sources, such as
 - radionuclides discharged from nuclear fuel cycle facilities (e.g. helium-3, chromium-51, cobalt-60);
 - manufactured radionuclides (produced and used in unsealed form in medicine or industry) as a result of regular or incidental discharges (e.g. iodine-131, technetium-99m);
 - radionuclides released in the past into the environment, including drinking-water sources (e.g. caesium-137, strontium-90, plutonium-239, plutonium-240).

9.1.2 Radiation-induced health effects through drinking-water

Radiation protection is based on the assumption that any exposure to radiation involves some level of risk. For prolonged exposures, as is the case for ingestion of drinking-

water containing radionuclides over extended periods of time, evidence of an increased cancer risk in humans is available at doses above 100 mSv (Brenner et al., 2003; UNSCEAR, 2012). Below this dose, an increased risk is difficult to identify through epidemiological studies, because of the required large sample sizes, although some large studies have shown increased cancer risk at doses lower than 100 mSv (Brenner et al., 2003; Rühm, Laurier & Wakeford, 2022). It is assumed that there is a linear relationship between exposure and risk, with no threshold value below which there is no risk (UNSCEAR, 2010). The WHO strategy to control the presence of radionuclides in drinking-water is based on an “individual dose criterion” (IDC) in terms of an annual effective dose of 0.1 mSv/year. This level of effective dose represents a very low level of risk that is not expected to give rise to any detected adverse health effect (see [Box 9.3](#)).

9.2 Rationale for screening levels and guidance levels

The current Guidelines are based on the approach proposed by the ICRP in situations of prolonged radiation exposure of the public. According to the ICRP, in planned exposure situations (see [Box 9.2](#)), it is prudent to restrict the prolonged component of the individual dose to 0.1 mSv in any given year (ICRP, 2000). It is recognized that exposure to radionuclides in drinking-water may be a consequence of a planned exposure situation, but is more likely to be from an existing exposure situation. Rather than adopt a different approach depending on whether or not the radionuclides are naturally occurring or human-made, a pragmatic and conservative approach was adopted, with an IDC of 0.1 mSv from 1 year’s consumption of drinking-water, regardless of the origin of the radionuclides (see [Box 9.3](#)).

Screening levels and guidance levels are conservative and should not be interpreted as mandatory limits. Exceeding a guidance level should be taken as a trigger for further investigation, but not necessarily as an indication that the drinking-water is unsafe.

Box 9.2 Radiation exposure situations

The ICRP (2007) distinguishes between three types of radiation exposure situations—planned, existing and emergency exposure situations:

- A **planned exposure** situation is a situation that arises from the planned operation of a radiation source or from a planned activity that results in an exposure to a radiation source (e.g. exposure to a radiation source during a medical procedure for diagnosis or treatment).
- An **existing exposure** situation is a situation that already exists when a decision on the need for control has to be taken (e.g. exposure to indoor radon in dwellings).
- An **emergency exposure** situation is a situation that arises as a result of an accident, a malicious act or any other unexpected event. The criteria included in the present Guidelines do not apply during emergency exposure situations (see [chapter 6](#)).

Exposure to radionuclides due to ingestion of drinking-water in non-emergency situations is regarded as an existing exposure situation (IAEA, 2014).

Box 9.3 Individual dose criterion (IDC) and health risks

The additional risk to health from exposure to an annual dose of 0.1 mSv (i.e. the IDC) associated with the intake of radionuclides from drinking-water is considered to be low for the following reasons:

- Individual doses from natural radioactivity in the environment vary widely. The worldwide average is about 2.4 mSv/year, but, in some parts of the world, average doses can be up to 10 times higher. An IDC of 0.1 mSv/year therefore represents a small addition compared with these values.
- The nominal risk coefficient for radiation-induced cancer incidence is approximately $5.5 \times 10^{-2}/\text{Sv}$ (ICRP, 2007). Multiplying this by an IDC of 0.1 mSv/year from drinking-water gives an estimated annual cancer risk of approximately 5.5×10^{-6} .

The IDC provides the basis for the development of the operational criteria—that is, the screening levels and guidance levels included in these Guidelines—that can be applied by water suppliers and authorities. The WHO screening levels and guidance levels can be adapted in national standards for radionuclides in drinking-water considering the national context. Adaptation may particularly be considered where there are elevated levels of naturally occurring radionuclides in groundwater and minimal options for alternative water sources or water treatment. In such situations, the Basic Safety Standards (BSS) reference level should be considered. See [section 9.3.3](#) for more information on the BSS reference level, as well as *Management of radioactivity in drinking-water* ([Annex 1](#)) for further information on considerations for developing national standards.

In the second edition of the Guidelines, the IDC of 0.1 mSv/year was based on screening levels for gross alpha activity and gross beta activity of 0.1 Bq/l and 1 Bq/l, respectively. This IDC represents less than 5% of the average annual dose attributable to radiation of natural origin (see [section 9.1](#)). Subsequent experience indicated that, in practice, the 0.1 mSv annual dose would usually not be exceeded if the gross alpha activity was equal to or below 0.5 Bq/l. For this reason, in the third edition of the Guidelines, the IDC was based on screening levels of 0.5 Bq/l for gross alpha activity and 1 Bq/l for gross beta activity. This change was carried forward to the current edition of the Guidelines. These values have been derived to cover the most common radionuclides in drinking-water (see [section 9.1.1](#)) and their contributions to the radiation dose from the consumption of drinking-water (see [section 9.3.1](#)).

9.3 Monitoring and assessment for dissolved radionuclides

The recommended assessment methodology for controlling radionuclide health risks from drinking-water is illustrated in [Figure 9.2](#) and summarized in [Box 9.4](#).

9.3.1 Screening of drinking-water supplies

The process of identifying individual radionuclides in drinking-water and determining their concentration may be time-consuming and expensive. Because, in most circumstances, the concentrations are low, such detailed analysis is normally not justified for routine monitoring. A more practical approach, particularly if resources are limited, is to use a screening procedure, where the total radioactivity present in the

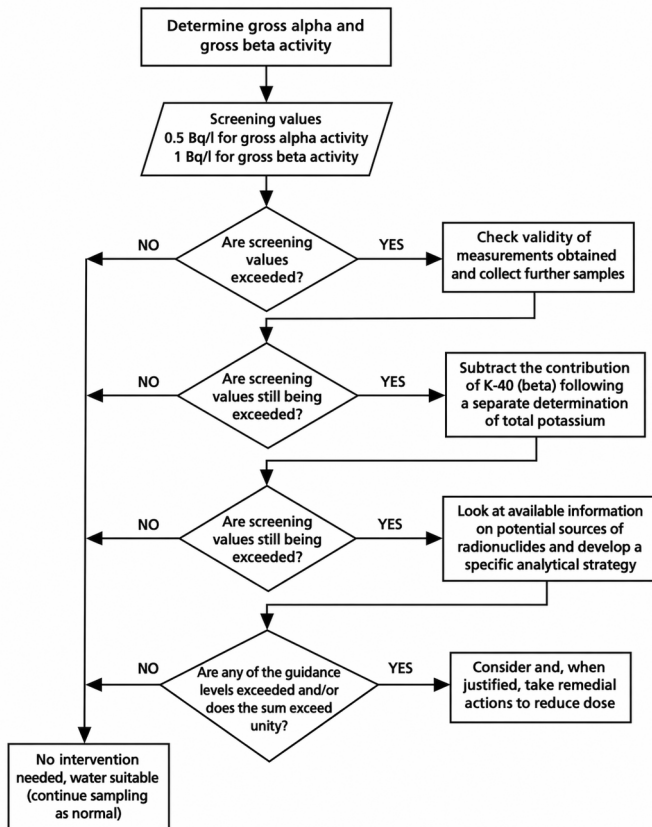


Figure 9.2 Application of screening and guidance levels for radionuclides in drinking-water

form of alpha and beta radiation is first determined, without regard to the identity of specific radionuclides.

These measurements are suitable as a preliminary screening approach to determine whether further radioisotope-specific analysis is necessary. They can also be used for detecting changes in the radiological characteristics of the drinking-water source as well as for identifying spatial and/or temporal trends in the radionuclide content of drinking-water.

Screening levels for drinking-water, below which no further action is required, are 0.5 Bq/l for gross alpha activity and 1 Bq/l for gross beta activity. If neither of these values is exceeded, the IDC of 0.1 mSv/year will usually not be exceeded. The use of these screening levels is recommended, as this maximizes both the reliability and the cost-effectiveness of assessing the radionuclide content of drinking-water.

Box 9.4 Recommended assessment methodology

The recommended assessment methodology for controlling radionuclide health risks from drinking-water involves four steps:

1. An IDC¹ of 0.1 mSv from 1 year's consumption of drinking-water is adopted.
2. Initial screening is undertaken for both gross alpha activity and gross beta activity. If the measured activity concentrations are below the screening levels of 0.5 Bq/l for gross alpha activity and 1 Bq/l for gross beta activity, no further action is required.
3. If either of the screening levels is consistently exceeded, the concentrations of individual radionuclides should be determined and compared with the guidance levels (see [Table 9.2](#)).
4. The outcome of this further evaluation may indicate that no action is required or that further evaluation is necessary before a decision can be made on the need for measures to reduce the dose.

For further guidance on actions to take when screening levels or guidance levels are exceeded, see *Management of radioactivity in drinking-water* ([Annex 1](#)).

However, the IDC of 0.1 mSv/year could be slightly or even significantly exceeded, even if the screening levels are not exceeded, in the uncommon situation where some specific radionuclides are the only significant contributors to the total gross activity concentration. This can happen for certain radionuclides, such as radium-226 and polonium-210 among the alpha emitters, and radium-228 and lead-210 among the beta emitters. If the local geology and hydrology indicate that these radionuclides may be present, the individual radionuclides should be measured and compared with the guidance levels.

Furthermore, radionuclides emitting low-energy beta radiation, such as tritium, and some gaseous or volatile radionuclides, such as iodine, will not be detected by standard gross beta activity measurements. Routine analysis for these radionuclides is not necessary, but, if there are any reasons for believing that they may be present, radionuclide-specific sampling and measurement techniques should be used.²

Gross beta measurements include a contribution from potassium-40, a beta emitter that occurs naturally in a fixed ratio to stable potassium. Potassium is an essential element for humans and is absorbed mainly from ingested food. If the screening level of 1 Bq/l for gross beta is exceeded, the contribution of potassium-40 to beta activity should be subtracted following a separate determination of total potassium. The beta activity of potassium-40 per unit of weight of stable potassium is 27.9 Bq/g, which is the factor that should be used to calculate the beta activity due to potassium-40.

¹ In the European Drinking Water Directive (Euratom, 2013), this parameter is called the indicative dose (ID), and the same value of 0.1 mSv/year is adopted.

² References for analytical methods and treatment technologies specific to radionuclides are provided in [Annex 6](#).

9.3.2 Strategy for assessing drinking-water if screening levels are exceeded

If either of the screening levels are consistently exceeded, then the specific radionuclides should be identified and their individual activity concentrations measured. This will allow the contribution from each radionuclide to the IDC to be calculated. If the following additive formula is satisfied, then no further action is required:

$$\sum_i \frac{C_i}{GL_i} \leq 1$$

where:

- C_i = the measured activity concentration of radionuclide i
- GL = the guidance level (see [Tables 9.2](#) and [A6.1 in Annex 6](#)) of radionuclide i that, at an intake of 2 litres/day¹ for 1 year, will result in an effective dose of 0.1 mSv/year

If any of the guidance levels is exceeded, then the sum will exceed unity. The sum may also exceed unity even if none of the individual guidance levels is exceeded. Where the sum exceeds unity for a single sample, the IDC of 0.1 mSv/year would be exceeded only if the exposure to the same measured concentrations were to continue for a full year. In practice, activity concentrations often vary throughout the year. *Hence, such a result does not in itself imply that the water is unsuitable for consumption.*

9.3.3 Strategy for assessing drinking-water if guidance levels are exceeded

An annual dose at the IDC of 0.1 mSv is a small percentage of the average radiation dose received by any individual. Both the screening levels and guidance levels, which are based on the IDC, are highly conservative values that allow national authorities to determine, without further consideration, that the drinking-water is fit for consumption from a radiological viewpoint. National experiences have shown that the vast majority of water supplies comply with these criteria. The screening levels and guidance levels should not be interpreted as a limit above which drinking-water is unsafe for consumption, but they represent useful operational tools to put into practice the optimization principle.

Occasionally, the situation may arise where the guidance levels are consistently exceeded for one or a combination of specific radionuclides. National authorities will then need to conduct further investigations and make a decision regarding the need to implement remedial measures or to place some restriction on the continued use of the water supply for drinking purposes. Ideally, the decision should be made in consultation with all the relevant authorities—that is, those involved in drinking-water quality and in radiation protection, and civil authorities that may be required to participate in informing consumers, delivering alternative supplies or supervising the collection of water from bowsers and tankers.

¹ In certain situations, where national or regional consumption rates are known, the guidance level could be adjusted to take this into account.

Table 9.2 Guidance levels for common^a natural and artificial radionuclides

Category	Radionuclide	Dose coefficient (Sv/Bq)	Guidance level ^b (Bq/l)
Natural occurring radioactive isotope that starts the uranium decay series ^c	Uranium-238	4.5×10^{-8}	10
Natural occurring radioactive isotopes belonging to the uranium decay series	Uranium-234	4.9×10^{-8}	1
	Thorium-230	2.1×10^{-7}	1
	Radium-226	2.8×10^{-7}	1
	Lead-210	6.9×10^{-7}	0.1
	Polonium-210	1.2×10^{-6}	0.1
Natural occurring radioactive isotope that starts the thorium decay series	Thorium-232	2.3×10^{-7}	1
Natural occurring radioactive isotopes belonging to the thorium decay series	Radium-228	6.9×10^{-7}	0.1
	Thorium-228	7.2×10^{-8}	1
Artificial radionuclides that can be released to the environment as part of the fission products found in reactor emissions or nuclear weapons tests	Caesium-134 ^d	1.9×10^{-8}	10
	Caesium-137 ^d	1.3×10^{-8}	10
	Strontium-90 ^d	2.8×10^{-8}	10
Artificial radionuclide that can be released to the environment as a fission product (see above). It is also used in nuclear medicine procedures and thus can be released into water bodies through sewage effluent.	Iodine-131 ^{d,e}	2.2×10^{-8}	10
Radioactive isotope of the hydrogen produced artificially as a fission product from nuclear power reactors and nuclear weapons tests. It may be naturally present in the environment in a very small amount. Its presence in a water source suggests potential industrial contamination.	Tritium ^e	1.8×10^{-11}	10 000
Naturally occurring radioactive isotope widely distributed in nature and present in organic compounds and in the human body.	Carbon-14	5.8×10^{-10}	100
Artificial isotope formed in nuclear reactors that also exists in trace quantities in <i>natural</i> uranium ores.	Plutonium-239 ^d	2.5×10^{-7}	1
Artificial isotope by-product formed in nuclear reactors.	Americium-241 ^d	2.0×10^{-7}	1

^a This list is not exhaustive. In certain circumstances, other radionuclides should be investigated (see [Annex 6](#)).

^b Guidance levels were rounded to the nearest order of magnitude by averaging the log scale values (to 10^n if the calculated value was below 3×10^n and to 10^{n+1} if the value was 3×10^n or above). For example, if the calculated value was 2.8 Bq/l (i.e. 2.8×10^0 , as is the case for uranium-234), the guidance level was rounded to 10^0 (i.e. = 1), whereas, if the calculated value was 3 Bq/l, (i.e. 3×10^0 , as is the case for uranium-238), or above, the guidance level was rounded to 10^1 (i.e. = 10). There may be situations where it is useful to use the unrounded guidance levels to calculate the IDC, particularly considering that the rounded guidance levels are conservative in most cases. See section 1.5.2 in *Management of radioactivity in drinking-water* ([Annex 1](#)) for guidance on when to consider using unrounded guidance levels, as well as a table for the unrounded guidance levels for common radionuclides.

^c Separate guidance levels are provided for individual uranium radioisotopes in terms of radioactivity (i.e. expressed as Bq/l). The provisional guideline value for total content of uranium in drinking-water is 30 µg/l based on its chemical toxicity, which is predominant compared with its radiological toxicity (see [chapter 12](#)).

Table 9.2 (continued)

^d These radionuclides either may not occur in drinking-water in normal situations or may be found at doses that are too low to be of significance to public health. Therefore, they are of lower priority for investigation following an exceedance of a screening level.

^e Although iodine and tritium will not be detected by standard gross activity measurements and routine analysis for these radionuclides is not necessary, if there are any reasons for believing that they may be present, radionuclide-specific sampling and measurement techniques should be used. This is the reason for including them in this table.

From a radiological point of view, one of the key considerations is the extent to which the guidance levels are exceeded. The International Basic Safety Standards (BSS) for Radiation Protection and Safety of Radiation Sources (IAEA, 2014) address drinking-water in the chapter on existing exposure situations and contain a requirement that, generally, the annual effective dose received from the consumption of drinking-water should not exceed a value of about 1 mSv.¹ This reference level should not be regarded either as an “acceptable” dose or as a dose limit, and all reasonable efforts should be made to minimize the doses received as part of the optimization process. Each situation will be different, and non-radiological factors, such as the costs of remediation and the availability of other drinking-water supplies, will need to be taken into account in reaching a final decision. In general, however, restrictions on the use of a drinking-water supply are not justified from a public health perspective when the dose is at or below the BSS reference level unless other supplies of safe drinking-water in the same area are available. Restrictions on the water supply may be considered at higher levels, but it is important that radiological risks are balanced with other risks, including the risk of not having a water supply. National authorities also need to be aware that radionuclides such as uranium are chemically toxic, and the allowable concentrations in drinking-water may be determined by a radioisotope’s toxicological rather than its radioactive properties (see [chapter 12](#)).

9.3.4 Sampling frequency

Criteria for monitoring radiological contamination of drinking-water should be developed, taking into account available resources and the potential for radiological risks. It should not detract from the adequate assessment and management of microbial and chemical risks. New water supplies should be sampled to determine their suitability for drinking-water, whereas existing supplies would need monitoring occasionally. If the water supply is adequately characterized and measured concentrations are consistently below screening levels, then sampling frequency should be reduced. However, if sources of potential radionuclide contamination exist nearby or are expected to be changing rapidly with time, then the sampling should be more frequent. Sampling frequency should be maintained, or even increased, if concentrations are approaching the screening levels or if the sum of ratios of the observed concentrations of individual radionuclides to their guidance levels approaches unity (see below). A graded approach to sampling frequency should be developed commensurate with the degree

¹ IAEA International Safety Standards Series No. GSR Part 3, IAEA, Vienna (IAEA, 2014). These standards are co-sponsored by eight international organizations, including WHO.

of contamination, the source of supply (i.e. surface water or groundwater), the size of the population served, the expected variability of radionuclide concentrations and the availability and results of historical monitoring records. International standards are available relating to the assessment of radiological water quality, including sampling procedures (e.g. preservation and handling of samples) and programmes (Standards Australia & Standards New Zealand, 1998; ISO, 2006, 2009, 2023, 2024).

9.4 Guidance levels for radionuclides commonly found in drinking-water

Guidance levels established for naturally occurring and human-made radionuclides most commonly detected in drinking-water supplies as well as for human-made radionuclides potentially relevant for prolonged exposure situations resulting from past nuclear emergency situations are presented in [Table 9.2](#). The respective dose coefficients for adults are also presented (IAEA, 1996; ICRP, 1996).

The guidance level for each radionuclide in [Table 9.2](#) represents the concentration that, if present in the drinking-water consumed throughout the year, would result in an individual dose of 0.1 mSv.

The guidance levels were calculated using dose coefficients for adults. Insufficient evidence was found to introduce separate guidance levels for different age groups. Although infants and children consume a lower mean volume of drinking-water, the age-dependent dose coefficients for children are higher than those for adults, accounting for higher uptake or metabolic rates. In the case of prolonged contamination of the water source, an assessment of doses to infants and children may be considered.

The guidance levels apply to routine (“normal”) operational conditions of existing or new drinking-water supplies. They do not apply during an emergency exposure situation involving the release of radionuclides into the environment. However, the guidance levels apply again once the relevant authorities have declared an end to the emergency exposure situation. Additional guidance is provided in [section 6.8](#), the supporting document *Management of radioactivity in drinking-water* ([Annex 1](#)), and in several other publications (IAEA, 2011, 2015, 2018; ICRP, 2020).

The guidance levels for radionuclides in drinking-water were calculated using the following equation:

$$GL = \frac{IDC}{h_{ing} \times q}$$

where:

- GL = guidance level of radionuclide in drinking-water (Bq/l)
- IDC = individual dose criterion, equal to 0.1 mSv/year for this calculation
- h_{ing} = dose coefficient for ingestion by adults (mSv/Bq)
- q = annual ingested volume of drinking-water, assumed to be 730 litres/year (equivalent to the standard World Health Organization drinking-water consumption rate of 2 litres/day)

9.5 Analytical methods

9.5.1 *Measuring gross alpha and gross beta activity concentrations*

To analyse drinking-water for gross alpha and gross beta activities (excluding radon and other volatile radionuclides), the most common approach is to evaporate a known volume of the sample to dryness and measure the activity of the residue. As alpha radiation is easily absorbed within a thin layer of solid material, the reliability and sensitivity of the method for alpha determination may be reduced in samples with high total dissolved solids (TDS) content (ISO, 2017, 2018a).

The determination of gross beta activity using the evaporation method includes the contribution from potassium-40. An additional analysis of total potassium is therefore required if the gross beta screening value is exceeded; in this case, the contribution of potassium-40 needs to be subtracted from the gross beta value (see [section 9.3.1](#) and [Figure 9.2](#)).

The co-precipitation technique (APHA et al., 2017) excludes the contribution due to potassium-40; therefore, determination of total potassium is not necessary. This method is not applicable to assessment of water samples containing certain fission products, such as caesium-137. However, under normal circumstances, concentrations of fission products in drinking-water supplies are extremely low.

The use of liquid scintillation counters capable of performing alpha-beta discrimination is increasing and provides an alternative for gross alpha and gross beta determination (ISO, 2018b; WHO, 2018).

Where possible, standardized methods should be used to determine concentrations of gross alpha and gross beta activities. Standardized methods for this analysis are listed in [Table 9.3](#).

9.5.2 *Measuring specific radionuclides*

If either of the gross alpha and gross beta screening levels is exceeded, then the specific radionuclides should be identified and their individual activity concentrations measured.

References for analytical methods for specific radionuclides are provided in [Annex 6](#). Information on measuring radon concentrations in water is provided in [section 9.7.4](#).

9.6 Remedial measures

If the IDC of 0.1 mSv/year is exceeded consistently, then the options available to the regulatory authority to reduce the dose should be examined. Where remedial measures are contemplated, any strategy considered should first be justified (in the sense that it achieves a net benefit). Any decision that alters the radiation exposure situation should do more good than harm. This means that by reducing existing exposure, it will achieve sufficient individual or societal benefit to offset the detriment it causes (ICRP, 2007).

Once the remedial action is justified, then protection should be optimized in accordance with the recommendations of ICRP (2007). The principle of optimization of protection implies that the likelihood of incurring exposures, the number of

Table 9.3 Methods for the analysis of gross alpha and gross beta activities in drinking-water

Method (reference)	Preparation technique	Application
International Organization for Standardization:		
ISO 9696 for gross alpha counting on thick sources (ISO, 2017)	Evaporation	Water with TDS less than 0.1 g/l, non-volatile radionuclides
ISO 9697 for gross beta counting on thick sources (ISO, 2018a)	Evaporation	
ISO 10704 for gross alpha and gross beta counting on thin sources (ISO, 2019)	Evaporation or co-precipitation	
ISO 11704 for gross alpha and gross beta using liquid scintillation counting (ISO, 2018b)	None	Water with TDS less than 5 g/l and when no correction for colour quench is necessary, non-volatile radionuclides below 80 °C
American Public Health Association 7110C for gross alpha and gross beta (APHA et al., 2017)	Co-precipitation	Surface water and groundwater (TDS is not a factor)

Note: Detection limits for these different standardized methods depend on a number of factors, mainly the detection efficiency of the equipment used and the counting time; they will also vary between laboratories depending on the exact protocols used. The detection limits for the analytical methods presented in the table can be expected to be in the range 1.4–340 mBq/l for gross alpha and 0–424 mBq/l for gross beta (Jobbágy et al., 2012).

people exposed and the magnitude of their individual doses should all be kept as low as reasonably achievable, taking economic and societal factors into account.

When source water contains unacceptably high concentrations of radionuclides, control options include use of an alternative supply, controlled blending with another source or additional water treatment. Treatment plants with a combination of coagulation, sedimentation and sand filtration processes may remove up to 100% of the suspended radioactivity present in raw waters. Lime–soda ash softening plants can also remove practically all of the suspended radioactivity, depending on the radionuclide and on the proportion of radioactivity that might be associated with particulates.

A comprehensive review of the removal of dissolved radionuclides by water treatment processes has been undertaken (Brown, Hammond & Wilkins, 2008). The results summarized in that report are reproduced in [Table 9.4](#). References for treatment technologies specific to radionuclides are provided in [Annex 6](#).

9.7 Radon

9.7.1 Radon in air and water

Uranium, radium and radon are all soluble in water. Radon present in surface waters, such as lakes and rivers, is readily released into outdoor air by agitation as it passes over rocks and soils. Groundwater from wells and boreholes usually contains higher radon concentrations than surface waters. In some extreme circumstances, very high radon concentrations can be found in drinking-water supplies from these sources (see [Box 9.5](#)).

Table 9.4 Treatment performance for some common radionuclides^a

Element	Coagulation	Sand filtration	Activated carbon	Precipitation softening	Ion exchange	Reverse osmosis
Strontium	xx	xx	x	xxxx	xxx	xxxx
Iodine	xx	xx	xxx	x	xxx	xxxx
Caesium	xx	xx	x	xx	xxx	xxxx
Radium	xx	xxx	xx	xxxx	xxxx	xxxx
Uranium	xxxx	x	xx	xxxx	xxxx	xxxx
Plutonium	xxxx	xx	xxx	x	xxxx	xxxx
Americium	xxxx	xx	xxx	x	xxxx	xxxx
Tritium	Not possible to remove					

^a x = 0–10% removal; xx = 10–40% removal; xxx = 40–70% removal; xxxx = > 70% removal.

Box 9.5 Radon in drinking-water

- Some groundwater supplies may contain elevated concentrations of radon. High radon concentrations are seldom found in surface drinking-water supplies.
- Radon dissolved in drinking-water can be released into indoor air. Normally, a higher radon dose is received from inhaling the radon and radon progeny compared with their ingestion.
- Radon released from drinking-water is not the only source of radon in indoor air. Where high indoor radon concentrations exist, the underlying soil and building materials, rather than the drinking-water, are normally the predominant sources.
- Straightforward and effective techniques exist to reduce the concentration of radon in drinking-water supplies.
- In deciding whether or not to take steps to reduce the concentration of radon in drinking-water supplies, it is important to take account of the contribution of other sources of radon to the total radiation dose. Any action should be both justified and optimized and take account of local conditions.

Radon is soluble in water, its solubility decreasing rapidly with an increase in temperature. When a tap or shower is turned on, some of the dissolved radon is released into indoor air. This adds to the radon present from other sources and will give rise to a radiation dose when inhaled.

An evaluation of international research data (UNSCEAR, 2000) has concluded that, on average, 90% of the dose attributable to radon in drinking-water comes from inhalation rather than ingestion. Therefore, controlling the inhalation pathway rather than the ingestion pathway is the most effective way to control doses from radon in drinking-water.

The percentage of radon present in drinking-water that is released into indoor air will depend on local conditions, such as the total consumption of water in the house, the volume of the house and its ventilation rate, and is likely to be highly variable. It has been estimated that a radon concentration of 1000 Bq/l in drinking-water

discharged from a tap or shower will, on average, increase the radon concentration by 100 Bq/m³ in indoor air (NAS, 1999; European Commission, 2001; Health Canada, 2009). This contribution is not constant, as it occurs only while the water is being discharged through the tap or shower. Radon in air also comes from other sources, and normally the underlying soil and building materials, rather than the drinking-water, are the predominant sources.

9.7.2 Health risks from radon

Epidemiological studies have clearly shown that long-term exposure to high radon concentrations in indoor air increases the risk of lung cancer (WHO, 2009). Radon ingested in drinking-water will give a radiation dose to the lining of the stomach. Scientific studies however, have not shown a definitive link between consumption of drinking-water containing radon and an increased risk of stomach cancer (Ye et al., 1998; Auvinen et al., 2005; WHO, 2009).

9.7.3 Guidance on radon in drinking-water supplies

As the dose from radon present in drinking-water is normally received from inhalation rather than ingestion, it is more appropriate to measure the radon concentration in air than in drinking-water.

The WHO reference level for radon concentration in indoor air is 100 Bq/m³ in dwellings. If this level cannot be reached under prevailing country-specific conditions, the level should not exceed 300 Bq/m³, corresponding to an annual dose of approximately 10 mSv (WHO, 2009). This recommendation is consistent with the International BSS (IAEA, 2014) and with the most recent recommendations of the ICRP (2009).

If a country establishes a national standard for radon in drinking-water, screening levels for radon in water should be set on the basis of the national reference level for radon in air and the distribution of radon in the national housing stock. Where high radon concentrations are identified in indoor air, this is nearly always due to ingress of radon from the soil rather than degassing from the drinking-water supply. Nevertheless, in circumstances where high radon concentrations might be expected in drinking-water, it is prudent to measure for radon and, if high concentrations are identified, consider whether measures to reduce the concentrations are justified.

The concentration of radon in groundwater supplies can vary considerably. Consequently, in situations where high radon concentrations have been identified or are suspected, the frequency of gross alpha and gross beta measurements may need to be increased so that the presence of long-lived radon progeny (notably lead-210 and polonium-210), which can be major contributors to dose, can be assessed and monitored on an ongoing basis.

9.7.4 Measuring radon in drinking-water

There are difficulties in deriving activity concentrations of radon in drinking-water because of the ease with which radon is released from water during handling. Stirring and transferring water from one container to another will release dissolved radon.

Water that has been left to stand will have reduced radon activity, and boiling will also completely release radon from the water into the air. A variety of methods can be used to measure radon in water, including liquid scintillation counting, which is a sensitive and widely used method (WHO, 2009). Samples should ideally be taken at the point of use. This can be complemented by taking measurements at the source to understand the potential for radon to be in drinking-water and to inform decisions on remedial actions.

9.7.5 Decreasing radon concentrations in drinking-water

Reasonably simple measures are available to decrease radon concentrations in drinking-water by aeration. High-performance aeration is an effective means for the removal of radon in groundwater supplies and can achieve up to 99.9% removal. However, these methods may create a large source of airborne radon. Adsorption via granular activated carbon, with or without ion exchange, can also achieve high radon removal efficiencies, but is less efficient and requires large amounts of granular activated carbon. Other possibilities are blending the water with water that has very low radon concentrations or storing¹ the water in tanks or basins to allow the radioactive decay of radon gas before distribution to consumers. A combination of these methods is frequently adopted.

Decreasing radon in drinking-water should be considered in the overall context of radon exposure. Radon exposure from ingested water is small compared with exposure from inhalation. Controlling overall exposure should therefore most likely focus on reducing radon concentrations in the indoor air from radon entering buildings from the ground and/or building materials.

9.8 Risk communication

9.8.1 Reporting results

The analytical results for each sample should contain the following information:

- sample identification code;
- sample collection date and time;
- standard analytical methods used or brief description of any non-standard analytical methods used;
- identification of the radionuclides or type of radioactivity and total radioactivity determined;
- measurement-based concentration or activity value calculated using the appropriate blank for each radionuclide;
- estimates of the counting uncertainty;

¹ Storing is not effective for radium-226-rich waters because, in such cases, radon is continuously produced by the decay of its parent radium-226; radon is usually in equilibrium with radium-226 throughout the storage period, regardless of its duration. Furthermore, if storing water is used to reduce its radon content, attention should be paid to the accumulation of long-lived radon progeny (i.e. lead-210 and polonium-210).

- a minimum detectable concentration for each radionuclide or parameter analysed;
- estimate of total projected uncertainty of the reported result, including the contributions from all the parameters within the analytical method (i.e. counting and other random and systematic uncertainties or errors).

9.8.2 *Communicating risks*

Communicating radiation risks clearly and effectively includes identifying target audiences (e.g. public, policy-makers and decision-makers) and tailoring the messages to them (WHO, 2002). Risk has different meaning for different people, but, in general, risk communication requires a description of the likelihood of harm and its severity.

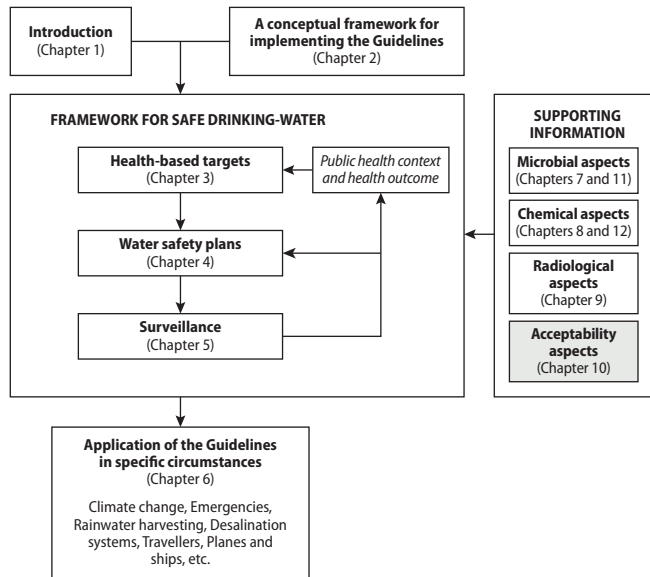
Risk communication with the public should utilize plain language. The technical lexicon of radiation protection is not readily understood by non-specialists (Picano, 2008). In some situations, comparisons are helpful to explain radiation risks (e.g. placing possible health risks from ingestion of drinking-water in the context of risk associated with exposure to natural radiation in different parts of the world). It should be clearly explained that guidance levels should not be interpreted as mandatory limits and that exceeding a guidance level may be taken as a trigger for further investigation, but it is not necessarily an indication that the drinking-water is unsafe.

The persons in charge of communicating risk should be skilled in interpersonal communication, able to convey empathy, effective listeners and respectful of people's concerns. They should be knowledgeable about the topic area with which they are dealing and be able to answer basic questions about the current as well as possible future risks. Guidance on radiation risk communication is provided elsewhere (USEPA, 2007; WHO, 2009).

Acceptability aspects: Taste, odour and appearance

The provision of drinking-water that is not only safe but also acceptable in appearance, taste and odour is of high priority. Water that is aesthetically unacceptable will undermine the confidence of consumers, will lead to complaints and, more importantly, could lead to the use of water from sources that are less safe.

To a large extent, consumers have no means of judging the safety of their drinking-water themselves, but their attitude towards their drinking-water supply and their drinking-water suppliers will be affected to a considerable extent by the aspects of water quality that they are able to perceive with their own senses. It is natural for consumers to regard with suspicion water that appears dirty or discoloured or that has an unpleasant taste or smell, even though these characteristics may not in themselves be of direct consequence to health.



The appearance, taste and odour of drinking-water should be acceptable to the consumer. Water that is aesthetically unacceptable can lead to the use of water from sources that are aesthetically more acceptable, but potentially less safe.

Some substances of health concern have effects on the taste, odour or appearance of drinking-water that would normally lead to rejection of the water at concentrations significantly lower than those of concern for health.

The concentration at which constituents are objectionable to consumers is variable and dependent on individual and local factors, including the quality of the water to which the community is accustomed and a variety of social, environmental and cultural considerations. Guideline values have not been established for constituents influencing water quality

Guideline values have not been established for constituents influencing water quality that have no direct link to adverse health impacts.

that have no direct link to adverse health impacts. However, guideline values have been established for some substances that may cause taste or odour in drinking-water at much lower concentrations than the guideline value because there is such a wide range in the ability of consumers to detect them by taste or odour. In the summaries in this chapter and the fact sheets in [chapter 12](#), reference is made to levels likely to give rise to complaints from consumers. These are not precise numbers, and tastes or odours may be detectable by consumers at lower or higher levels, depending on individual and local circumstances.

It is important to consider whether existing or proposed water treatment and distribution practices can affect the acceptability of drinking-water and to manage change and operations to minimize the risk of problems for acceptability as well as health. For example, chloramination that is not properly managed can lead to the formation of trichloramines, which can cause unacceptable taste and odour. Other problems may be indirect, such as the disturbance of internal pipe deposits and biofilms when the flow is disturbed or changed in distribution systems.

It is not normally appropriate to directly regulate or monitor substances of health concern whose effects on the acceptability of water would normally lead to rejection of the water at concentrations significantly lower than those of concern for health; rather, these substances may be addressed through a general requirement that water be acceptable to the majority of consumers. For such substances, a formal guideline value is not usually derived, but a health-based value is derived in order to assist in judging the response that is needed when problems are encountered and in some cases to provide reassurance to health authorities and consumers with regard to possible health risks. In the fact sheets in [chapter 12](#), this is explained, and information on acceptability is described. In the tables of guideline values (see [chapter 8](#) and [Annex 3](#)), for those chemicals for which health-based guideline values were derived, the guideline value is designated with a “C”, with a footnote explaining that while the substance is of health significance, water would normally be rejected by consumers at concentrations well below the health-based guideline value. Monitoring of such substances should be undertaken in response to consumer complaints.

Taste and odour can originate from natural inorganic and organic chemical contaminants and biological sources or processes (e.g. aquatic microorganisms), from contamination by synthetic chemicals, from corrosion or as a result of problems with water treatment (e.g. chlorination). Taste and odour may also develop during storage and distribution as a result of microbial activity.

Taste and odour in drinking-water may be indicative of some form of pollution or of a malfunction during water treatment or distribution. It may therefore be an indication of the presence of potentially harmful substances. The cause should be investigated and the appropriate health authorities should be consulted, particularly if there is a sudden or substantial change.

Colour, cloudiness, particulate matter and visible organisms may also be noticed by consumers and may create concerns about the quality and acceptability of a drinking-water supply.

10.1 Biologically derived contaminants

There are a number of diverse organisms that often have no public health significance but which are undesirable because they produce taste and odour. As well as affecting the acceptability of the water, they indicate that water treatment and/or the state of maintenance and repair of the distribution system are insufficient.

Actinomycetes and fungi

Actinomycetes and fungi can be abundant in surface water sources, including reservoirs, and they can also grow on unsuitable materials in the water supply distribution systems, such as rubber. They can produce geosmin, 2-methyl isoborneol and other substances, resulting in objectionable tastes and odours in the drinking-water.

Cyanobacteria and algae

High amounts of phytoplankton (i.e. cyanobacteria and eukaryotic algae) in reservoirs and in surface waters used as drinking-water sources may impede coagulation and filtration, causing discoloration and turbidity of water after filtration. Some of these organisms can also produce geosmin, 2-methyl isoborneol and other chemicals, which have taste thresholds in drinking-water of a few nanograms per litre. Some other cyanobacterial products—cyanotoxins—are of direct health significance (see [section 8.5.1](#)), but their production does not seem to be linked to the production of taste and odour chemicals.

Invertebrate animal life¹

Invertebrate animals are naturally present in many water resources used as sources for the supply of drinking-water and often infest shallow, open wells. Small numbers of invertebrates may also pass through water treatment works where the barriers to particulate matter are not completely effective and colonize filters or the distribution system. Their motility may enable them and their larvae to penetrate filters at the treatment works and vents on storage reservoirs.

The types of invertebrates concerned can be considered, for control purposes, as belonging to two groups. First, there are free-swimming organisms in the water itself or on water surfaces, such as the crustaceans *Gammarus pulex* (freshwater shrimp), *Crangonyx pseudogracilis*, *Cyclops* spp. and *Chydorus sphaericus*. Second, there are

¹ The section was drawn largely from chapter 6 of the supporting document *Safe piped water* ([Annex 1](#)).

other invertebrates that either move along surfaces or are anchored to them (e.g. water louse [*Asellus aquaticus*], snails, zebra mussel [*Dreissena polymorpha*], other bivalve molluscs and the bryozoan *Plumatella* sp.) or inhabit slimes (e.g. *Nais* spp., nematodes and the larvae of chironomids). In warm weather, slow sand filters can sometimes discharge the larvae of gnats (*Chironomus* and *Culex* spp.) into the water. In certain circumstances, these can reproduce parthenogenetically (i.e. asexual reproduction), which can exacerbate the problem in service reservoirs and distribution.

Many of these invertebrates can survive, deriving food from bacteria, algae and protozoa in the water or present on slimes on pipe and tank surfaces. Few water distribution systems are completely free of animals at all times. However, the density and composition of invertebrate populations vary widely, from heavy infestations, including readily visible species that are objectionable to consumers, to sparse occurrences of microscopic species.

The presence of invertebrates has largely been regarded by piped drinking-water suppliers in temperate regions as an acceptability problem, either directly or through their association with discoloured water. Large invertebrate populations also indicate high levels of organic material that may give rise to other water quality issues, such as microbial growth. In tropical and subtropical countries, in contrast, there are species of aquatic invertebrates that act as secondary hosts for parasites. For example, the small crustacean *Cyclops* is the intermediate host of the guinea worm (*Dracunculus medinensis*) (see [sections 7.1.1](#) and [11.4](#)). However, there is no evidence that guinea worm transmission occurs from piped drinking-water supplies. The presence of invertebrates in drinking-water, especially if visible, raises consumer concern about the quality of the drinking-water supply and should be controlled.

Penetration of waterworks and mains is more likely to be a problem when high-rate filtration processes are used, but problems can arise even at well-run treatment works. Regular cleaning of water mains by flushing and/or swabbing will usually control infestation.

Treatment of invertebrate infestations in piped distribution systems is discussed in detail in chapter 6 of the supporting document *Safe piped water* ([Annex 1](#)).

Iron bacteria

In waters containing ferrous and manganous salts, oxidation by iron bacteria (or by exposure to air) may cause rust-coloured deposits on the walls of tanks, pipes and channels and carry-over of deposits into the water.

10.2 Chemically derived contaminants

Aluminium

Naturally occurring aluminium as well as aluminium salts used as coagulants in drinking-water treatment are the primary sources of aluminium in drinking-water. The presence of aluminium at concentrations in excess of 0.1–0.2 mg/l often leads to consumer complaints as a result of deposition of aluminium hydroxide floc and the exacerbation of discoloration of water by iron. It is therefore important to optimize

treatment processes in order to minimize any residual aluminium entering the distribution system. Under good operating conditions, aluminium concentrations of less than 0.1 mg/l are achievable in many circumstances. Available evidence does not support the derivation of a health-based guideline value for aluminium in drinking-water (see [sections 8.5.4](#) and [12.1](#)).

Ammonia

The threshold odour concentration of ammonia at alkaline pH is approximately 1.5 mg/l, and a taste threshold of 35 mg/l has been proposed for the ammonium cation. Ammonia is not of direct relevance to health at these levels, and no health-based guideline value has been proposed (see [sections 8.5.3](#) and [12.1](#)). However, ammonia does react with chlorine to reduce free chlorine and to form chloramines.

Chloramines

Chloramines, such as monochloramine, dichloramine and trichloramine (nitrogen trichloride), are generated from the reaction of chlorine with ammonia. Among chloramines, monochloramine is the only useful chlorine disinfectant, and chloramination systems are operated to minimize the formation of dichloramine and trichloramine. Higher chloramines, particularly trichloramine, are likely to give rise to taste and odour complaints, except at very low concentrations.

For monochloramine, no odour or taste was detected at concentrations between 0.5 and 1.5 mg/l. However, slight organoleptic effects within this range and odour and taste thresholds of 0.65 and 0.48 mg/l have been reported. For dichloramine, the organoleptic effects between 0.1 and 0.5 mg/l were found to be “slight” and “acceptable”. Odour and taste thresholds of 0.15 and 0.13 mg/l were reported, respectively. An odour threshold of 0.02 mg/l has been reported for trichloramine, and it has been described as “geranium”.

A guideline value for monochloramine has been established (see [sections 8.5.4](#) and [12.1](#)).

Chloride

High concentrations of chloride give a salty taste to water and beverages. Taste thresholds for the chloride anion depend on the associated cation and are in the range of 200–300 mg/l for sodium, potassium and calcium chloride. Concentrations in excess of 250 mg/l are increasingly likely to be detected by taste, but some consumers may become accustomed to low levels of chloride-induced taste. No health-based guideline value is proposed for chloride in drinking-water (see [sections 8.5.1](#) and [12.1](#)).

Chlorine

Most individuals are able to taste or smell chlorine in drinking-water at concentrations well below 5 mg/l, and some at levels as low as 0.3 mg/l. The taste threshold for chlorine is below the health-based guideline value of 5 mg/l (see [sections 8.5.4](#) and [12.1](#)).

Chlorobenzenes

Taste and odour thresholds of 10–20 µg/l and odour thresholds ranging from 40 to 120 µg/l have been reported for monochlorobenzene. A health-based guideline value has not been derived for monochlorobenzene (see [sections 8.5.2](#) and [12.1](#)), although the health-based value that could be derived far exceeds the lowest reported taste and odour threshold in water.

Odour thresholds of 2–10 and 0.3–30 µg/l have been reported for 1,2- and 1,4-dichlorobenzene, respectively. Taste thresholds of 1 and 6 µg/l have been reported for 1,2- and 1,4-dichlorobenzene, respectively. The health-based guideline values of 1 mg/l derived for 1,2-dichlorobenzene and of 0.3 mg/l for 1,4-dichlorobenzene (see [sections 8.5.2](#) and [12.1](#)) far exceed the lowest reported taste and odour thresholds for these compounds.

Odour thresholds of 10, 5–30 and 50 µg/l have been reported for 1,2,3-, 1,2,4- and 1,3,5-trichlorobenzene, respectively. A taste and odour threshold concentration of 30 µg/l has been reported for 1,2,4-trichlorobenzene. A health-based guideline value was not derived for trichlorobenzenes, although the health-based value that could be derived (see [sections 8.5.2](#) and [12.1](#)) exceeds the lowest reported odour threshold in water of 5 µg/l.

Chlorophenols

Chlorophenols generally have very low taste and odour thresholds. The taste thresholds in water for 2-chlorophenol, 2,4-dichlorophenol and 2,4,6-trichlorophenol are 0.1, 0.3 and 2 µg/l, respectively. Odour thresholds are 10, 40 and 300 µg/l, respectively. If water containing 2,4,6-trichlorophenol is free from taste, it is unlikely to present a significant risk to health (see [sections 8.5.4](#) and [12.1](#)). Microorganisms in distribution systems may sometimes methylate chlorophenols to produce chlorinated anisoles, for which the odour threshold is considerably lower.

Colour

Drinking-water should ideally have no visible colour. Colour in drinking-water is usually due to the presence of coloured organic matter (primarily humic and fulvic acids) associated with the humus fraction of soil. Colour is also strongly influenced by the presence of iron and other metals, either as natural impurities or as corrosion products. It may also result from the contamination of the water source with industrial effluents and may be the first indication of a hazardous situation. The source of colour in a drinking-water supply should be investigated, particularly if a substantial change has taken place.

Most people can detect colour above 15 true colour units (TCU) in a glass of water. Levels of colour below 15 TCU are often acceptable to consumers. High colour from natural organic carbon (e.g. humics) could also indicate a high propensity to produce by-products from disinfection processes. No health-based guideline value is proposed for colour in drinking-water.

Copper

Copper in a drinking-water supply usually arises from the corrosive action of water leaching copper from copper pipes in buildings. High levels of dissolved oxygen have

been shown to accelerate copper corrosion in some cases. Concentrations can vary significantly with the period of time the water has been standing in contact with the pipes; for example, first-draw water would be expected to have a higher copper concentration than a fully flushed sample. High concentrations can interfere with the intended domestic uses of the water. Staining of sanitary ware and laundry may occur at copper concentrations above 1 mg/l. At levels above 5 mg/l, copper also imparts a colour and an undesirable bitter taste to water. Although copper can give rise to taste, it should be acceptable at the health-based guideline value of 2 mg/l (see [sections 8.5.4, 12.1](#) and [A5.3](#) in [Annex 5](#)).

Dissolved oxygen

The dissolved oxygen content of water is influenced by the source, raw water temperature, treatment and chemical or biological processes taking place in the distribution system. Depletion of dissolved oxygen in water supplies can encourage the microbial reduction of nitrate to nitrite and sulfate to sulfide. It can also cause an increase in the concentration of ferrous iron in solution, with subsequent discoloration at the tap when the water is aerated. No health-based guideline value is recommended. However, very high levels of dissolved oxygen may exacerbate corrosion of metal pipes.

Ethylbenzene

Ethylbenzene has an aromatic odour; the reported odour threshold in water ranges from 2 to 130 µg/l. The lowest reported odour threshold is 100-fold lower than the health-based guideline value of 0.3 mg/l (see [sections 8.5.2](#) and [12.1](#)). The taste threshold ranges from 72 to 200 µg/l.

Hardness

Hardness caused by calcium and magnesium is usually indicated by precipitation of soap scum and the need for excess use of soap to achieve cleaning. Consumers are likely to notice changes in hardness. Public acceptability of the degree of hardness of water may vary considerably from one community to another. The taste threshold for the calcium ion is in the range of 100–300 mg/l, depending on the associated anion, and the taste threshold for magnesium is probably lower than that for calcium. In some instances, consumers tolerate water hardness in excess of 500 mg/l.

Depending on the interaction of other factors, such as pH and alkalinity, water with a hardness above approximately 200 mg/l may cause scale deposition in the treatment works, distribution system and pipework and tanks within buildings. It will also result in high soap consumption and subsequent “scum” formation. On heating, hard waters form deposits of calcium carbonate scale. Soft water, but not necessarily cation exchange softened water, with a hardness of less than 100 mg/l, may, in contrast, have a low buffering capacity and so be more corrosive for water pipes.

No health-based guideline value is proposed for hardness in drinking-water (see the supporting document *Calcium and magnesium in drinking-water*; [Annex 1](#)).

Hydrogen sulfide

The taste and odour thresholds of hydrogen sulfide in water are estimated to be between 0.05 and 0.1 mg/l. The “rotten eggs” odour of hydrogen sulfide is particularly noticeable in some groundwaters and in stagnant drinking-water in the distribution system, as a result of oxygen depletion and the subsequent reduction of sulfate by bacterial activity.

Sulfide is oxidized rapidly to sulfate in well-aerated or chlorinated water, and hydrogen sulfide levels in oxygenated water supplies are normally very low. The presence of hydrogen sulfide in drinking-water can be easily detected by the consumer and requires immediate corrective action. It is unlikely that a person could consume a harmful dose of hydrogen sulfide from drinking-water; hence, a health-based guideline value has not been derived for this compound (see [sections 8.5.1](#) and [12.1](#)).

Iron

Anaerobic groundwater may contain ferrous iron at concentrations up to several milligrams per litre without discoloration or turbidity in the water when directly pumped from a well. On exposure to the atmosphere, however, the ferrous iron oxidizes to ferric iron, giving an objectionable reddish-brown colour to the water.

Iron also promotes the growth of “iron bacteria”, which derive their energy from the oxidation of ferrous iron to ferric iron and in the process deposit a slimy coating on the piping. At levels above 0.3 mg/l, iron stains laundry and plumbing fixtures. There is usually no noticeable taste at iron concentrations below 0.3 mg/l, although turbidity and colour may develop. No health-based guideline value is proposed for iron (see [sections 8.5.4](#) and [12.1](#)).

Manganese

At levels as low as 0.02 mg/l, manganese as insoluble manganese oxides in water supplies may cause discoloured water and staining of laundry and plumbing fixtures. The presence of manganese in drinking-water, like iron, may lead to accumulation of deposits in the distribution system. Even at a concentration of 0.02 mg/l, manganese may deposit in pipes as manganese oxides, which may slough off as a black precipitate. In contrast, soluble manganese(II) is colourless and is visually undetectable at concentrations as high as 506 mg/l, the maximum tested concentration.

The health-based guideline value of 0.08 mg/l for manganese (see [section 12.1](#)) is higher than the acceptability threshold of insoluble manganese. However, since dissolved manganese can also be released at the point of use or collection, the presence of discoloured water cannot be used reliably to assess if manganese is present. Therefore, aesthetic as well as health aspects should be considered when setting regulations and standards for drinking-water quality.

Petroleum oils

Petroleum oils can give rise to the presence of a number of low molecular weight hydrocarbons that have low odour thresholds in drinking-water. Benzene, toluene, ethylbenzene and xylenes (BTEX) are considered individually in this section, as health-

based guideline values have been derived for these chemicals. However, a number of other hydrocarbons, particularly alkylbenzenes such as trimethylbenzene, may give rise to a very unpleasant “diesel-like” odour at concentrations of a few micrograms per litre. There is experience indicating that the taste threshold of a mixture of low molecular weight aromatic hydrocarbons is lower than the threshold of individual substances. Diesel is a particularly rich source of such substances.

pH and corrosion

Although pH usually has no direct impact on consumers, it is one of the most important operational water quality parameters. Careful attention to pH control is necessary at all stages of water treatment to ensure satisfactory water clarification and disinfection (see the supporting document *Safe piped water*; [Annex 1](#)). For effective disinfection with chlorine, the pH should preferably be less than 8; however, lower-pH water (approximately pH 7 or less) is more likely to be corrosive. The pH of the water entering the distribution system must be controlled to minimize the corrosion of water mains and pipes in household water systems. Alkalinity and calcium management also contribute to the stability of water and control its aggressiveness to pipes and appliances. Failure to minimize corrosion can result in the contamination of drinking-water and in adverse effects on its taste and appearance. The optimum pH required will vary in different supplies according to the composition of the water and the nature of the construction materials used in the distribution system, but it is usually in the range 6.5–8.5 (see [section 8.4.3](#)). Extreme values of pH can result from accidental spills, treatment breakdowns and insufficiently cured cement mortar pipe linings or cement mortar linings applied when the alkalinity of the water is low. No health-based guideline value has been proposed for pH (see [section 12.1](#)).

Sodium

The taste threshold concentration of sodium in water depends on the associated anion and the temperature of the solution. At room temperature, the average taste threshold for sodium is about 200 mg/l. No health-based guideline value has been derived (see [sections 8.5.1](#) and [12.1](#)), as the contribution from drinking-water to daily intake is small.

Styrene

Styrene has a sweet/sickly odour, and reported odour thresholds for styrene in water range from 0.004 to 2.6 mg/l, depending on temperature. Styrene may therefore be detected in water at concentrations below its health-based guideline value of 0.02 mg/l (see [sections 8.5.2](#) and [12.1](#)).

Sulfate

The presence of sulfate in drinking-water can cause noticeable taste, and very high levels might cause a laxative effect in unaccustomed consumers. Taste impairment varies with the nature of the associated cation; taste thresholds have been found to range from 250 mg/l for sodium sulfate to 1000 mg/l for calcium sulfate. It is generally

considered that taste impairment is minimal at levels below 250 mg/l. No health-based guideline value has been derived for sulfate (see [sections 8.5.1](#) and [12.1](#)).

Synthetic detergents

In many countries, persistent types of anionic detergent have been replaced by others that are more easily biodegraded, and hence the levels found in water sources have decreased substantially. The concentration of detergents in drinking-water should not be allowed to reach levels giving rise to either foaming or taste problems. The presence of any detergent may indicate contamination of source water with sewage or ingress of detergent solution into the distribution system, as a result of back-flow, for example.

Toluene

Toluene has a sweet, pungent, benzene-like odour. The reported taste threshold ranges from 0.04 to 0.12 mg/l. The reported odour threshold for toluene in water ranges from 0.024 to 0.17 mg/l. Toluene may therefore affect the acceptability of water at concentrations below its health-based guideline value of 0.7 mg/l (see [sections 8.5.2](#) and [12.1](#)).

Total dissolved solids

The palatability of water with a total dissolved solids (TDS) level of less than about 600 mg/l is generally considered to be good; drinking-water becomes significantly and increasingly unpalatable at TDS levels greater than about 1000 mg/l. The presence of high levels of TDS may also be objectionable to consumers, owing to excessive scaling in water pipes, heaters, boilers and household appliances. No health-based guideline value for TDS has been proposed (see [sections 8.5.1](#) and [12.1](#)).

Turbidity

Turbidity, typically expressed as nephelometric turbidity units (NTU), describes the cloudiness of water caused by suspended particles (e.g. clay and silts), chemical precipitates (e.g. manganese and iron), organic particles (e.g. plant debris) and organisms. Turbidity can be caused by poor source water quality, poor treatment and, within distribution systems, disturbance of sediments and biofilms or the ingress of dirty water through main breaks and other faults. At high levels, turbidity can lead to staining of materials, fittings and clothes exposed during washing, in addition to interfering with the effectiveness of treatment processes (see [Tables 7.7](#) and [7.8](#) in [chapter 7](#)).

Increasing turbidity reduces the clarity of water to transmitted light. Below 4 NTU, turbidity can be detected only using instruments, but at 4 NTU and above, a milky-white, muddy, red-brown or black suspension can be visible. Large drinking-water supplies should consistently produce water with no visible turbidity (and should be able to achieve 0.5 NTU before disinfection at all times and average 0.2 NTU or less). However, small supplies, particularly those where resources are limited, may not be able to achieve such levels.

Visible turbidity reduces the acceptability of drinking-water. Although most particles that contribute to turbidity have no health significance (even though they

may indicate the presence of hazardous chemical and microbial contaminants), many consumers associate turbidity with safety and consider turbid water as being unsafe to drink. This response is exacerbated when consumers have been used to receiving high-quality filtered water. If consumers lose confidence in a drinking-water supply, they may drink less water or use lower turbidity alternatives that may not be safe. Any complaints about unexpected turbidity should always be investigated because they could reflect significant faults or breaches in distribution systems.

Further information is available in *Water quality and health – review of turbidity* (see [Annex 1](#)).

Xylenes

Xylene concentrations in the range of 0.3 mg/l produce a detectable taste and odour. The odour threshold for xylene isomers in water has been reported to range from 0.02 to 1.8 mg/l. The lowest odour threshold is well below the health-based guideline value of 0.5 mg/l for xylene (see [sections 8.5.2](#) and [12.1](#)).

Zinc

Zinc imparts an undesirable astringent taste to water at a taste threshold concentration of about 4 mg/l (as zinc sulfate). Water containing zinc at concentrations in excess of 3–5 mg/l may appear opalescent and develop a greasy film on boiling. Although drinking-water seldom contains zinc at concentrations above 0.1 mg/l, levels in tap water can be considerably higher because of the zinc used in older galvanized plumbing materials; this may also be an indicator of elevated cadmium from such older material. No health-based guideline value has been proposed for zinc in drinking-water (see [sections 8.5.4](#) and [12.1](#)).

10.3 Treatment of taste, odour and appearance problems

In many cases, aesthetic problems will be prevented by optimizing conventional treatment processes such as coagulation, sedimentation and chlorination. If specific treatment is deemed necessary, aeration, granular or powdered activated carbon and ozonation are generally effective in removing organic chemicals and some inorganic chemicals, such as hydrogen sulfide, that cause tastes and odours (see [Annex 5](#)).

Tastes and odours caused by disinfectants are best controlled through careful operation of the disinfection process and pretreatment to remove precursors.

Manganese can be removed by several treatment methods, including oxidation followed by filtration (see [section 12.1](#), and [Table A5.1](#) in [Annex 5](#) for more information). Techniques for removing hydrogen sulfide include aeration, granular activated carbon, filtration and oxidation. Ammonia can be removed by biological nitrification. Precipitation softening or cation exchange can reduce hardness. Other taste- and odour-causing inorganic chemicals (e.g. chloride and sulfate) are generally not amenable to treatment (see the supporting document *Chemical safety of drinking-water*; [Annex 1](#)).

10.4 Temperature

Cool water is generally more palatable than warm water, and temperature will have an impact on the acceptability of a number of other inorganic constituents and chemical contaminants that may affect taste. High water temperature enhances the growth of microorganisms and may increase problems related to taste, odour, colour and corrosion.

Microbial fact sheets

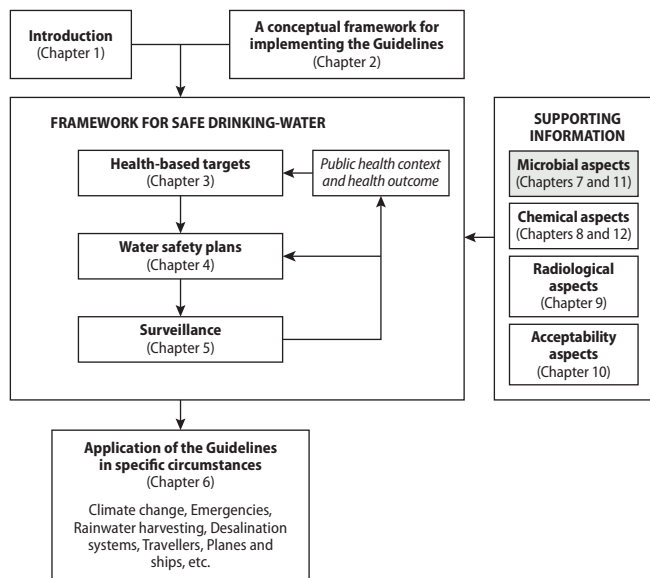
Fact sheets are provided on potential waterborne pathogens as well as on indicator microorganisms.

The waterborne microorganisms potentially causing illness include:

- bacteria, viruses, protozoa and helminths identified in [Table 7.1](#) and [Figure 7.1](#);
- other pathogens, for which transmission through drinking-water has been suggested but for which evidence is weak or lacking, identified in [Table 7.2](#);
- hazardous cyanobacteria.

The human health effects caused by waterborne transmission vary in severity from mild gastroenteritis to severe and sometimes fatal diarrhoea, dysentery, hepatitis and typhoid fever. Contaminated water can be the source of large outbreaks of disease, including cholera, dysentery and cryptosporidiosis; for the majority of waterborne pathogens, however, there are other important sources of infection, such as person-to-person contact and food.

Most waterborne pathogens are introduced into drinking-water supplies in human or animal faeces, do not grow in water and initiate infection in the gastrointestinal tract following ingestion (enteric pathogens). However, other pathogens including *Legionella*, non-tuberculosis mycobacteria, *Burkholderia pseudomallei*, *Acanthamoeba*



spp. and *Naegleria fowleri* are environmental organisms that can grow in water and soil. Besides ingestion, other routes of transmission can include inhalation and aspiration, leading to infections of the respiratory tract (e.g. *Legionella*, non-tuberculosis mycobacteria), and contact, leading to infections at sites as diverse as the skin and brain (e.g. *Naegleria fowleri*, *Burkholderia pseudomallei*, non-tuberculosis mycobacteria).

Of all the waterborne pathogens, the helminth *Dracunculus medinensis* is unique in that it is the only pathogen that is solely transmitted through ingestion of contaminated water. The provision of a safe drinking-water supply is key to eradicating dracunculiasis.

Bacteria, viruses and protozoa for which transmission through drinking-water has been suggested but for which evidence is weak or lacking (summarized in [Table 7.2](#)) can be grouped based on common characteristics. Common features of bacteria in this category are an ability to grow in water (e.g. *Acinetobacter*, *Klebsiella*, *Pseudomonas*) or survive for long periods (*Leptospira*), raising the possibility of drinking-water transmission. Except for *Aeromonas*, these bacteria are not transmitted by the faecal–oral route and do not cause gastrointestinal disease. Most are associated with potential issues in health-care facilities (*Acinetobacter*, *Klebsiella*, *Pseudomonas*, *Staphylococcus* and *Tsukamurella*). Common features of a number of viruses and all protozoa in this group are that they have been detected in drinking-water or surface water and are transmitted by the faecal–oral route (e.g. astroviruses, parechoviruses, *Balantoides* microsporidia). The last group are highly pathogenic zoonotic viruses (e.g. coronaviruses, filoviruses) that are not transmitted by the faecal–oral route and for which there is no evidence that drinking-water is a source.

Of the helminths included in [Table 7.2](#), while *Schistosoma* is not a waterborne pathogen, a safe drinking-water supply can contribute to disease prevention by reducing the use of infested water for domestic purposes such as washing clothes.

The fact sheets on potential pathogens include information on human health effects, epidemiological features, sources and occurrence, routes of transmission and the significance of drinking-water as a source of infection. The fact sheets on microorganisms that can be used as indicators of the effectiveness of control measures or of the potential presence of pathogenic microorganisms provide information on indicator value, source and occurrence, application and significance of detection.

11.1 Bacterial pathogens

Most bacteria with moderate to strong evidence of waterborne transmission are enteric pathogens. A number of enteric bacteria are zoonotic with important animal sources including *Campylobacter*, enterohaemorrhagic *Escherichia coli* (EHEC) and non-typhoidal *Salmonella*. However, there are also some free-living environmental pathogens, such as *Legionella*, *Burkholderia pseudomallei* and non-tuberculosis mycobacteria. These pathogens are generally present in low concentrations in drinking-water sources, but can grow in drinking-water distribution systems particularly in biofilms. Enteric bacteria are transmitted by ingestion of contaminated drinking-water while free-living bacteria are typically transmitted by inhalation, aspiration and contact (e.g. bathing), with infections occurring in the respiratory tract, in skin lesions or in the brain. Increasing incidence of antimicrobial resistant enteric bacteria

including strains with multiple resistances represents a significant public health concern. For further information including an overview of treatment considerations for bacteria, see [section 7.2.2](#).

Acinetobacter

Significance in drinking-water

Acinetobacter are of low health significance in drinking-water. Although *Acinetobacter* are often detected in treated drinking-water supplies, there is no evidence linking transmission with drinking-water for the general population. However, transmission of non-gastrointestinal infections by *A. baumannii* to susceptible patients in health-care facilities through drinking-water may be possible. Water-related outbreaks in these settings have been associated with water baths and room humidifiers, rather than drinking-water ingestion.

General description

Acinetobacter are Gram-negative, short, rod-shaped, non-spore-forming bacteria with some motility. Although there are several species, only some are documented human pathogens. The most common examples are *A. baumannii*, *A. calcoaceticus* and *A. lwoffii*.

Human health effects

Some species of *Acinetobacter*, most notably multidrug-resistant (MDR) *A. baumannii*, are opportunistic pathogens that regularly cause infections, illness and death, predominantly in susceptible patients in health-care facilities. Health effects of *Acinetobacter* include infections of the urinary tract, eyes, skin and wounds in addition to pneumonia, bacteraemia, endocarditis and secondary meningitis. The emergence and rapid spread of MDR *A. baumannii* has increased the health impact and significance of this organism.

Epidemiology

Acinetobacter are an important cause of illness and death in susceptible health-care patients. Infections are predisposed by factors such as co-morbidities, burns, major surgery and weakened immune systems. Community-acquired illness, especially pneumonia in people who have a history of alcohol abuse or have chronic obstructive pulmonary disease, occurs globally, particularly in tropical regions during warm and humid months.

Transmission

In health-care facilities, person-to-person transmission through direct contact with wounds or burns, inhalation or contact with contaminated fomites, including medical devices and equipment, are the likely exposure pathways for *Acinetobacter* in susceptible individuals. Although *Acinetobacter* have been found in drinking-water, ingestion of water is not expected to be a source of infection. However, water-related outbreaks have been associated with water baths and room humidifiers in health-care facilities. Outbreaks in health-care facilities can establish an endemic presence and

potentially be spread to community populations through hospital wastewater, sewage and ambient waters, although transmission factors in community settings are poorly understood. The human infectivity of MDR *A. baumannii* and other species is uncertain; there are no definitive quantitative dose–response data available in humans or animal models.

Detection methods

Several culture-based methods are available to isolate *Acinetobacter* species and strains, but further confirmation by antimicrobial susceptibility testing is needed to identify human pathogens. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Acinetobacter are widespread in soil and water environments, with frequent detection, including MDR *A. baumannii* and other species implicated in human illness, in surface waters. *Acinetobacter* have also been found to be widespread in drinking-water. *Acinetobacter* species, particularly *A. baumannii*, are important bacteria in biofilms in premise plumbing, especially in health-care facilities. Most species and strains detected outside health-care facilities are not human pathogens. *Acinetobacter* include species that can grow and persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a water safety plan (WSP) (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies and building water systems will control *Acinetobacter*. These include application of adequate treatment processes and effective biofilm management in distribution systems and building water systems, including health-care facilities. WSPs for health-care facilities need to also consider the susceptibilities of patients. *Acinetobacter* are moderately resistant to oxidizing disinfectants such as chlorine, but doses and contact times typically used in drinking-water treatment plants should provide adequate inactivation of planktonic *Acinetobacter*. As *Acinetobacter* are non-faecal, environmental organisms that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

Bibliography

WHO (2025) *Acinetobacter. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Aeromonas

Significance in drinking-water

Aeromonas are of low health significance in drinking-water. The evidence linking transmission of gastroenteritis with drinking-water is weak. Occasionally, the same or similar genotypes of *Aeromonas* have been reported in drinking-water supplies and cases of gastrointestinal illness. The presence of high concentrations of *Aeromonas* in stored and distributed drinking-water can be associated with complaints about taste and odour.

General description

Aeromonas are Gram-negative, rod-shaped, non-spore-forming bacteria. They are divided into two main groups: the psychrophilic, non-motile aeromonads, which are rarely pathogenic in humans, and the mesophilic, motile aeromonads. The latter includes many species considered to be of potential human health significance, especially *A. caviae*, *A. dhakensis* and *A. veronii*.

Human health effects

Aeromonas are opportunistic pathogens. They can cause asymptomatic infections and clinical illnesses. Symptoms include septicaemia, infections of wounds, burns and soft tissue, gastroenteritis, respiratory tract infections and pneumonia, and peritonitis. People with cirrhosis are prone to peritonitis, and immunocompromised patients are particularly susceptible to septicaemia.

Epidemiology

The disease burden from *Aeromonas* in humans is uncertain due to the range of illnesses they cause, the variety of exposure routes and differences in human host susceptibility. The most reported health effects are wound and soft tissue infections, gastrointestinal illnesses (in children) and septicaemia (particularly in immunocompromised patients). Multidrug-resistant *Aeromonas* are an emerging concern; aquatic environments may play a role in their dissemination.

Transmission

Human exposure can occur from food, fresh and estuarine water, wastewater, sediments, soil and animals such as fish, shellfish and poultry. Transmission can occur by the faecal–oral route, but evidence that ingestion is a route of exposure is weak. Transmission can also occur extra-intestinally through trauma (e.g. wounds, burns) following exposure to water or soil containing *Aeromonas*.

Although there are some data available, the infectivity of *Aeromonas* remains uncertain. A human volunteer study reported that approximately 25% of exposed subjects shed the bacteria and approximately 4% became ill with diarrhoea after ingesting doses of 10^{10} colony forming units. However, based on reported water- and foodborne disease outbreaks, only a low dose (0.9 of a colony forming unit) of some strains of *Aeromonas* may cause infection.

Detection methods

Culture-based and molecular methods are available for *Aeromonas* (see [section 7.4](#)).

Source and occurrence

Aeromonas are widespread in soil and water environments and are frequently found in source water. *Aeromonas*, including human pathogenic species, have been detected in surface water, groundwater and drinking-water. The species present are influenced by environmental type and characteristics. *Aeromonas* can grow and persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies will control *Aeromonas*. These include application of adequate treatment processes and effective biofilm management in distribution systems. Disinfection processes including chlorination are effective for inactivating *Aeromonas*. The presence of *Aeromonas* in stored and distributed drinking-water is generally considered a nuisance condition that could be indicative of inadequate management practices. As *Aeromonas* are environmental organisms and can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

Bibliography

WHO (2025) *Aeromonas. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Burkholderia pseudomallei

Significance in drinking-water

There is moderate evidence for waterborne transmission of *Burkholderia pseudomallei*, with a number of melioidosis outbreaks possibly associated with untreated or poorly treated drinking-water supplies in two countries.

General description

Burkholderia are Gram-negative, non-spore-forming, rod-shaped bacteria. There are over 80 species, with the principal human pathogens being *B. pseudomallei*, the species of concern for waterborne transmission, and the *B. cepacia* complex, which can cause pulmonary infections in cystic fibrosis patients.

Human health effects

B. pseudomallei can cause melioidosis. Disease manifestations include pneumonia, cutaneous or visceral abscesses and septicaemia. Reported mortality rates vary from 14% in Australia (based on a review published in 2021) to approximately 40% in Thailand (based on reviews published in 2010 and 2019). *B. pseudomallei* infections are commonly asymptomatic.

Epidemiology

The worldwide incidence of melioidosis is unknown due to difficulties in diagnosis and limited systematic reporting and surveillance. The disease is generally restricted to the tropics and subtropics. Host factors play a significant role in susceptibility to disease, with the majority of cases occurring in people with impaired immunocompetence. Risk factors include diabetes, excessive alcohol consumption, chronic pulmonary and renal disease, thalassemia, glucocorticoid therapy and cancer. Incidence of the disease is highest in adults, but it can occur in children. Melioidosis is seasonal; a review published in 2021 reported that 80% or more cases in Australia occurred in the wet season.

Transmission

The most common route of infection is contact with contaminated soil or environmental water sources through pre-existing wounds. Inhalation and ingestion of contaminated water can also lead to melioidosis. Drinking-water is not a common source of melioidosis, but outbreaks have been reported. Person-to-person or zoonotic transmission of *B. pseudomallei* is uncommon, but isolated cases have been reported. The infectious dose is unknown. The incubation period is typically 1–21 days, but there are reports of extended periods over many years.

Detection methods

A culture-based method is available for *Burkholderia*, but it lacks selectivity; molecular methods are also available (see [section 7.4](#)).

Source and occurrence

B. pseudomallei is a common soil and water organism, particularly in the tropics and subtropics. Surveys in northern Australia and south-east Asia detected *B. pseudomallei* in up to 32% of soil samples, while a survey of groundwater detected the organism in 33% of samples. *B. pseudomallei* may be present in drinking-water distribution systems due to ingress of environmental organisms together with survival and/or growth in biofilms. Being an environmental organism, it can grow and persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *B. pseudomallei* should include application and maintenance of drinking-water disinfection, and protection of distribution systems to prevent ingress of contaminated soil and water. Control strategies to reduce biofilm formation can also reduce the potential for survival of *B. pseudomallei* in distribution systems. Chlorine and ultraviolet (UV) light are effective drinking-water disinfectants against *B. pseudomallei*. As *B. pseudomallei* is a non-faecal, environmental organism that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Burkholderia pseudomallei*. *Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Campylobacter

Significance in drinking-water

Campylobacter are major zoonotic waterborne pathogens, with strong evidence from multiple outbreaks linking transmission to drinking-water. Most outbreaks are linked to small groundwater systems contaminated with human sewage or animal faeces after heavy rain or disinfection failure. In some cases, the same strains were isolated from infected patients and the drinking-water source.

General description

Campylobacter are Gram-negative, curved spiral rod-shaped bacteria that can be motile or non-motile. Of the more than 30 species, *C. jejuni* accounts for most of the infections, with *C. coli* being a less-frequent source. Emerging species, including *C. lari*, *C. fetus*, *C. concisus*, *C. upsaliensis* and *C. ureolyticus*, have been associated with gastrointestinal disease.

Human health effects

Campylobacter gastroenteritis (campylobacteriosis) may cause mild to severe illness or be asymptomatic. Symptoms include abdominal pain, acute watery diarrhoea, vomiting, chills and fever. Illness is usually self-limiting with symptoms persisting from a few days to around 2 weeks (average of 6 days). Estimates of case fatality ratios from studies in 2012 and 2013 range from <0.01% to 8.8%, depending on the population studied.

In low-income countries, campylobacteriosis in children has been associated with malnutrition, enteric dysfunction and impaired growth. Infrequent post-infection syndromes include reactive arthritis, meningitis and Guillain–Barré syndrome (a substantial proportion of cases of the latter have been attributed to preceding *Campylobacter* infections). There is increasing evidence associating campylobacteriosis with inflammatory bowel disease and irritable bowel syndrome. In severe cases, treatment of campylobacteriosis with antibiotics may be beneficial. However, multiple resistant strains have been reported.

Epidemiology

Campylobacter are one of the most important causes of diarrhoeal diseases worldwide. The burden of campylobacteriosis is greatest in young children, with incidence and mortality rates higher in low-income settings. In high-income settings, reported cases of campylobacteriosis have increased. *Campylobacter* are a major cause of diarrhoea among travellers from high-income settings to low- to middle-income settings. Immunocompromised individuals are more vulnerable to infection, relapse and complications. In temperate climates, infections are more common in summer than in winter.

Transmission

Campylobacter are transmitted primarily by the faecal–oral route, with poultry, wild-fowl, cattle and other domestic animals, pets and humans all being reservoirs. Food is the most common source of exposure, but other important sources include drinking-water and contact with infected animals. There have been a limited number of reports on transmission through recreational contact, although person-to-person transmission is uncommon.

C. jejuni is moderately to highly infective, with estimated doses associated with 50% infection rate (ID_{50}) values ranging from below 10 to 890 organisms. There is some evidence, particularly in adults, that infection could occur at low doses, while high doses are required for illness. Acquired immunity could be a contributing factor. The incubation period is usually 2–5 days. Pathogen shedding can persist for a few hours to 3 weeks after infection.

Detection methods

Standard culture-based methods for *Campylobacter* are available, but they are complex compared to detection methods for *E. coli*. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Campylobacter are found in a broad range of water and wastewater environments. *Campylobacter* have been frequently detected in surface waters and have been detected in vulnerable groundwater and undisinfected drinking-water. Prevalence is influenced by rainfall, water temperature, and the presence of faecal material including from poultry, waterfowl and human sewage (untreated or partially treated). The persistence of *Campylobacter* in water is considered to be short (up to 1 week at 20 °C), with survival influenced by temperature and other environmental conditions.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Campylobacter* should include protection of source waters from contamination by human and animal faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes and protection of drinking-water quality during distribution. Drinking-water should be protected from bird faeces during storage. Chlorine and UV light are effective drinking-water disinfectants against *Campylobacter*. *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of *Campylobacter* in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2024) *Campylobacter. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Cronobacter sakazakii

Significance in drinking-water

Cronobacter sakazakii is of low health significance in drinking-water. There is no evidence linking transmission with drinking-water.

General description

C. sakazakii (formerly classified as *Enterobacter sakazakii*) is a motile, Gram-negative, non-spore-forming, rod-shaped coliform bacterium.

Human health effects

C. sakazakii is an opportunistic pathogen. It has been associated with sporadic cases (or focal outbreaks) of sepsis, meningitis and necrotizing enterocolitis, primarily among neonates. In other age groups, illness occurs most often as wound or urinary tract infections, but sometimes bloodstream infections, pneumonia, osteomyelitis and splenic abscesses are reported. Asymptomatic infections can occur, especially in healthy people. *C. sakazakii* found in drinking-water and other environmental media are not resistant to most antibiotics relevant to human health.

Epidemiology

Estimates of disease burden have been made in a limited number of countries, but the magnitude of the disease burden globally is not known. Most infections are seen in low-birth-weight infants (i.e. less than 2 kg), infants born prematurely (i.e. less than 37 weeks of gestation) or within the first 2 months of life. *C. sakazakii* infections have also been reported in other age groups; elderly people and immunocompromised individuals are at greatest risk and experience more severe symptoms.

Transmission

There is no evidence of water- or excreta-borne transmission of *C. sakazakii*. There have also been no confirmed reports of infections spreading vertically from mother to child or through person-to-person contact or more generally via environmental transmission routes other than ingestion of contaminated powdered infant formula. The infectivity of *C. sakazakii* in humans is unknown, but dose–response studies in infant mice bioassays have reported infectivity to be as low as 100 organisms.

Detection methods

Culture-based and molecular methods are available for *C. sakazakii* (see [section 7.4](#)).

Source and occurrence

C. sakazakii is widespread in soil and water environments. It has been detected in surface water and groundwater, but only rarely found in treated drinking-water. The persistence of *C. sakazakii* is expected to be long in water (over 1 month at 20 °C) since it occurs naturally in the environment and persists in infant formula and other foods.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies and building water systems will control *C. sakazakii*. These include application of adequate treatment processes and effective biofilm management in distribution systems and building water systems. Although information on the efficacy of disinfection has not been reported, based on similarities to other members of the coliform group, disinfection processes including chlorination are expected to be effective for inactivating *C. sakazakii*. As *C. sakazakii* is an environmental organism that can persist and grow in water, *E. coli* (or thermo-tolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water.

Bibliography

WHO (2025) *Cronobacter sakazakii*. *Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

***Escherichia coli* – diarrhoeagenic**

Significance in drinking-water

Diarrhoeagenic *Escherichia coli* are important waterborne pathogens, with strong evidence linking transmission to contaminated drinking-water. Most of the evidence is

based on investigations of outbreaks associated with enterotoxigenic *E. coli* (ETEC), with outbreaks reported in a number of countries.

General description

E. coli is a Gram-negative, rod-shaped bacterium that can be motile or non-motile. Commensal, non-pathogenic *E. coli* is present in large numbers in the normal intestinal microbiota of humans and animals, where it generally causes no harm. However, several diarrhoeagenic pathotypes of *E. coli* have been identified. These include enterohaemorrhagic *E. coli* (EHEC), ETEC, enteropathogenic *E. coli* (EPEC), enteroinvasive *E. coli* (EIEC), enteroaggregative *E. coli* (EAEC) and diffusely adherent *E. coli* (DAEC). ETEC, EPEC and EIEC are relatively well characterized compared to EAEC and DAEC. EPEC is classified into typical and atypical strains based on different genetic characteristics, serotypes and virulence properties. EIEC is generally considered to be part of the same pathotype as *Shigella*. The EHEC pathotype and *Shigella* are the subjects of separate fact sheets.

Human health effects

Diarrhoeagenic *E. coli* infections may be asymptomatic, or range from mild to severe diarrhoeal disease, especially in children. Infections can be acute or persistent. Diarrhoeagenic *E. coli* infections follow a similar pattern, leading to watery diarrhoea and other symptoms (abdominal cramps, malaise, nausea, vomiting fever and headache) depending on the pathotype. Complications following infection with ETEC and EAEC in endemic settings include malnourishment in children. DAEC infections can lead to chronic diarrhoea in young children. ETEC and EAEC have been reported to cause post-infectious irritable bowel syndrome. In most cases, illness is self-limiting and antibiotic treatment is not required, although treatment may be beneficial for severe or persistent infections. However, increasing prevalence of resistance to multiple antibiotics has been reported.

Transmission

Diarrhoeagenic *E. coli* are transmitted primarily by the faecal–oral route. The available evidence indicates that humans are the primary source of pathogenic ETEC, typical EPEC, EIEC and DAEC, but atypical EPEC may be a zoonotic pathogen. Infection is associated with ingestion of contaminated food or water, contact with recreational waters, person-to-person transmission and possibly contact with animals (for atypical EPEC). Person-to-person transmission is more common in settings where there is close contact among individuals, such as in nursing homes and day-care centres and under conditions of inadequate water, sanitation and hygiene.

Based on estimated doses associated with 50% infection rate (ID_{50}) values ranging from 10^5 to 10^8 organisms, the relative infectivity of diarrhoeagenic *E. coli* (other than EHEC) is low. The incubation period for all pathotypes is brief (<2 days). The duration of illness is usually 3–5 days. Based on studies with ETEC, the incidence of illness may decrease in older children due to acquired immunity.

Detection methods

Detection of diarrhoeagenic *E. coli* involves culture-based isolation methods followed by application of specific identification tests for pathotypes. Unlike EHEC, the other

diarrhoeagenic pathotypes produce β -glucuronidase and will be detected in media that use this activity as an identifying characteristic. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

There are limited data on the occurrence of diarrhoeagenic *E. coli*, other than EHEC, in the environment, but diarrhoeagenic *E. coli* have been detected in water and waste-water environments including surface water, and an intermittent drinking-water supply using molecular methods. Prevalence is influenced by the presence of faecal contamination including from human sewage (untreated or partially treated) and from animals. Most information on survival is based on testing for non-pathogenic *E. coli* as an indicator of faecal pollution. Based on limited data, the persistence of diarrhoeagenic *E. coli* in water is likely to be moderate (1 week to 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from diarrhoeagenic *E. coli* should include protection of source waters from contamination by human and animal faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes and protection of drinking-water quality during storage and distribution. Chlorine and UV light are effective drinking-water disinfectants against diarrhoeagenic *E. coli*. Non-pathogenic *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of diarrhoeagenic *E. coli* in drinking-water (see [section 11.6](#)).

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***Escherichia coli* – enterohaemorrhagic**

Significance in drinking-water

Enterohaemorrhagic *Escherichia coli* (EHEC) is an important zoonotic waterborne pathogen, with strong evidence linking transmission to drinking-water. Notable outbreaks have been caused by contamination of drinking-water supplies with sewage or cattle manure.

General description

E. coli is a Gram-negative, rod-shaped bacterium that can be motile or non-motile. Commensal, non-pathogenic *E. coli* is present in large numbers in the normal intestinal microbiota of humans and animals, where it generally causes no harm. However, several diarrhoeagenic pathotypes of *E. coli* associated with intestinal disease have been identified. These include EHEC, enterotoxigenic *E. coli* (ETEC), enteropathogenic *E. coli* (EPEC), enteroinvasive *E. coli* (EIEC), enteroaggregative *E. coli* (EAEC) and diffusely adherent *E. coli* (DAEC). EHEC is a subset of shiga toxin-producing *E. coli* that causes severe illness in humans. ETEC, EPEC, EIEC, EAEC and DAEC are the subject of a separate diarrhoeagenic *Escherichia coli* fact sheet.

Human health effects

Shiga toxin-producing EHEC serotypes, such as *E. coli* O157:H7 and *E. coli* O111 are the most potentially dangerous of the diarrhoeagenic *E. coli* pathotypes, causing diarrhoea that ranges from being mild and non-bloody to severe and extremely bloody. Other symptoms include stomach cramps, vomiting and mild fever. Disease severity can differ between different serotypes. It has been estimated (including as reported in a 2018 WHO fact sheet) that up to 10% of cases can develop into the potentially fatal haemolytic uraemic syndrome, characterized by acute renal failure and haemolytic anaemia. Further long-term sequelae from haemolytic uraemic syndrome include cardiac and gastrointestinal complications, neurological disorders, hypertension, chronic renal disease, cognition and behavioural changes, and diabetes mellitus. Antibiotics are not recommended for treating EHEC infections because they can exacerbate the complications of EHEC, leading to haemolytic uraemic syndrome. In addition, many strains are resistant to multiple antibiotics.

Epidemiology

E. coli (pathotypes not identified) are one of the most important causes of death due to bacterial infections, including those associated with antimicrobial resistance. *E. coli* O157:H7 has been an important cause of outbreaks in high-income regions of the world. Based on data from Australia, New Zealand and the United States of America (included in a 2014 systematic review), about half of cases occur in children up to 15 years of age. Children under 5 years of age are at most risk of developing haemolytic uraemic syndrome, with deaths most common in these children and people aged 60 years or older.

Transmission

EHEC is transmitted primarily by the faecal–oral route. EHEC strains are the only pathotype of diarrhoeagenic *E. coli* with a dominant zoonotic origin (primarily cattle, but sheep and goats as well). Humans are a secondary but still significant EHEC reservoir. Infection is associated with ingestion of contaminated food or water, person-to-person transmission, contact with animals and recreational waters, and use of contaminated irrigation water. Person-to-person transmission is particularly prevalent in communities where there is close contact among individuals, such as in nursing homes and day-care centres and under conditions of inadequate water, sanitation and hygiene.

The relative infectivity of *E. coli* O157:H7 is moderate, with a dose associated with a 50% infection rate (ID_{50}) of 210 organisms calculated from a dose–response model. This is substantially lower than the ID_{50} values of the other diarrhoeagenic *E. coli* pathotypes. The median incubation period is 3–4 days for most serotypes. Adults usually excrete the pathogen for about 1 week, although excretion may be up to 3 weeks in children; prolonged carriage is possible but uncommon. Studies suggest that previous exposure to EHEC results in a degree of acquired immunity against symptomatic infection from EHEC.

Detection methods

Detection of EHEC involves culture-based isolation methods followed by application of specific identification tests. Unlike most other strains of *E. coli*, EHEC does not

produce β -glucuronidase and will not be detected in media that use this activity as an identifying characteristic. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

EHEC has been detected in a variety of water and wastewater environments including surface water and drinking-water sourced from vulnerable groundwater supplies, with most studies focusing on *E. coli* O157. Prevalence is influenced primarily by the presence of animal faecal material, particularly from cattle. The presence of human faecal waste including sewage (untreated or partially treated) may also be a contributing factor. Persistence in water is considered to be moderate (up to 1 month at 20 °C), with survival influenced by environmental conditions.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from EHEC should include protection of source waters from contamination by human and animal faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Chlorine and UV light are effective drinking-water disinfectants against EHEC. Non-pathogenic *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of EHEC in drinking-water (see [section 11.6](#)). This applies even though standard tests for *E. coli* based on β -glucuronidase activity will not detect EHEC.

Bibliography

WHO (2025) *Enterohaemorrhagic Escherichia coli. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Francisella tularensis

Significance in drinking-water

There is strong evidence for waterborne transmission of *Francisella tularensis* from multiple outbreaks associated with drinking-water sources that were contaminated from animal carcasses and excreta with no or inadequate treatment.

General description

Francisella are Gram-negative, non-spore-forming, coccobacilli. There are at least 10 species of *Francisella*, but *F. tularensis* is the only species of concern for waterborne transmission. *F. tularensis* is further subdivided into four subspecies: *F. tularensis* subsp. *tularensis* (type A), *F. tularensis* subsp. *holarctica* (type B), *F. tularensis* subsp. *mediasiatica* and *F. tularensis* subsp. *novicida*. Subspecies type A is the most virulent and is predominantly found in North America; subspecies type B is less virulent, is widely distributed in the northern hemisphere and is the subspecies commonly linked to waterborne outbreaks.

Human health effects

Infection with *F. tularensis* can be asymptomatic or cause tularaemia, which is characterized by sudden onset of high fever, chills, general malaise, headache, anorexia

and fatigue. The clinical outcomes depend on the route of transmission. The oropharyngeal form is most commonly associated with waterborne transmission, although oculoglandular, ulceroglandular and pneumonic forms have been reported. Inhalation of *F. tularensis* in aerosols from dust and hay can result in acute and life-threatening pneumonia. A typhoidal form, probably associated with inhalation of contaminated aerosols, causes high fever and confusion, and mimics typhoid. Symptoms of oropharyngeal infections include a sore throat/tonsillitis, mouth ulcers, pharyngitis, swollen cervical lymph nodes with possible discharge of pus and occasionally gastrointestinal symptoms including diarrhoea. Treatment with antibiotics may be recommended for severe cases of tularaemia. Development of antibiotic resistance has not been identified, but is a concern.

Epidemiology

Tularaemia is generally restricted to the northern hemisphere, with isolated cases in Australia. The reported annual rates of disease are 0.1–0.3/100 000 people (based on an analysis published in 2021), while in the United States of America, the annual rate is <0.1/100 000 people (based on an analysis published in 2023). Generally, tularaemia is more common in males with prevalence highest between 45 and 64 years of age. However, for waterborne tularaemia, children and adults, as well as males and females, are equally affected. Endemic disease occurs primarily during warm weather and is primarily related to insect bites. In contrast, during colder seasons, infections usually occur through contaminated drinking-water consumption.

Transmission

There are multiple modes of transmission of *F. tularensis* to humans, including: direct contact with infected rodents, their carcasses or their excreta; bites from ticks, fleas and mosquitoes; consumption of contaminated food or water; contact with contaminated water or soils; and inhalation of aerosols from dust or hay. There is no evidence of person-to-person transmission.

The relative infectivity of *F. tularensis* can vary depending on the subspecies, the mode of transmission and the form of disease. Infectivity from oral exposure is low, with estimated doses of 10^6 – 10^8 organisms. The incubation period is typically 3–5 days.

Detection methods

Culture-based methods are available for *F. tularensis*, although isolation can be challenging because it is fastidious and slow growing. Molecular and immunological methods are also available (see [section 7.4](#)).

Source and occurrence

There are limited data on the occurrence of *Francisella*, but it has been detected in surface water and inadequately treated drinking-water with prevalence influenced by contamination by animal carcasses and faecal material, particularly rodents. *F. tularensis* can survive for at least moderate periods in water (1 week to 1 month at 20 °C). However, longer persistence can be supported by interactions with amoebae and in biofilms.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *F. tularensis* should include protection of source waters from access by rodents and contamination by animal carcasses or excreta, application of adequate water treatment processes, and protection of drinking-water quality from external contamination during storage and distribution. Control strategies to reduce biofilm formation can also reduce the potential for *F. tularensis* to survive in distribution systems. *F. tularensis* is moderately resistant to chlorine doses used in drinking-water treatment, but it can be effective provided adequate contact times are applied. UV light is an effective drinking-water disinfectant against *F. tularensis*. Due to longer survival of *F. tularensis* in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Francisella tularensis. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Helicobacter pylori

Significance in drinking-water

There is moderate evidence for *Helicobacter pylori* as a cause of waterborne disease. The presence of *H. pylori* in drinking-water and epidemiological evidence indicating higher rates of infection associated with drinking untreated water, as well as inadequate sanitation and hygiene, provide evidence of waterborne transmission, particularly in low-income countries.

General description

H. pylori is a Gram-negative, spiral-shaped, motile bacterium. There are over 40 species of *Helicobacter*, but only *H. pylori* has been identified as a major human pathogen.

Human health effects

Although most *H. pylori* infections are asymptomatic, the bacteria can cause acute and chronic gastritis. *H. pylori* is a significant risk factor for peptic ulcer disease, mucosa-associated lymphoid tissue, non-Hodgkin's lymphoma and gastric cancer in adults, and is epidemiologically associated with gastric cancer. The majority of *H. pylori* infections are initiated in childhood, and, without treatment, can be lifelong. Increasing rates of resistance of *H. pylori* to front-line antibiotics and declining rates of antimicrobial eradication have been reported.

Epidemiology

H. pylori is prevalent throughout the world. Most infections are acquired during childhood, but symptoms generally do not appear until adulthood. It has been estimated (in a review published in 2017) that about one in three adults in high-income countries are infected, while adults from low- and middle-income countries have higher prevalence ranging from 50% to 80%. A similar pattern of prevalence in children has been estimated. Gender does not seem to affect the risk of infection. Prevalence has

been decreasing over time, particularly in high-income countries. High incidence in indigenous communities has been reported globally.

Transmission

H. pylori is likely transmitted by ingestion of organisms in vomit and gastric juices through the oral and gastro–oral pathways. This is supported by the detection of high quantities of *H. pylori* in these fluids. Faecal–oral transmission is also considered possible, although few studies have successfully isolated the pathogen in faeces. Direct person-to-person contact within families has been identified as the most likely route of transmission, because the infection tends to cluster within households. Lower economic status, more siblings, crowding, infected mothers and not having access to sewage systems have been associated with higher *H. pylori* prevalence. Additional risk factors include drinking untreated water and eating in unhygienic conditions. Other transmission sources include contaminated food, and working in soil or water-related jobs or in health care. Primates, domestic cats, sheep and houseflies have also been identified as potential hosts, but zoonotic transmission has not been demonstrated.

The infectivity and incubation period are uncertain, but in a human volunteer study, inocula of 10^4 – 10^{10} organisms caused illness that occurred sporadically after 3 days and peaked between 9 and 12 days. Infection does not appear to lead to immunity.

Detection methods

Culture and molecular methods for *H. pylori* are available (see [section 7.4](#)). Culture methods are challenging as *H. pylori* is fastidious, microaerophilic and slow growing. Amoeba co-culture has been shown to enhance the growth of *H. pylori* before application of culture methods.

Source and occurrence

There are limited data on the occurrence of *H. pylori* in water and wastewater environments due to the challenges in detecting these organisms in environmental samples. However, they have been detected in surface water, groundwater and inadequately treated drinking-water, with prevalence influenced by the presence of sewage (untreated or partially treated). The persistence of *H. pylori* in surface water, based on culturability, is short (up to 1 week at 20 °C), but biofilms and free-living amoebae may increase the persistence of *H. pylori*.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *H. pylori* should include protection of source waters from contamination by faecal wastes, application of adequate water treatment processes and protection of drinking-water quality during storage and distribution. Control strategies to reduce biofilm formation can also reduce the potential for *H. pylori* to survive in distribution systems. Chlorine and UV light are effective disinfectants against *H. pylori*. As there is some evidence of the occurrence of *H. pylori* in faeces, *E. coli* (or thermotolerant coliforms) may be a suitable indicator of its possible presence in drinking-water, but since *H. pylori* may

persist in biofilms in drinking-water systems, the absence of *E. coli* is not a suitable indicator for its absence (see [section 11.6](#)).

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WHO (2025) *Helicobacter pylori. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Klebsiella

Significance in drinking-water

Klebsiella are of low health significance in drinking-water. Although *Klebsiella* can colonize drinking-water distribution systems, there is no evidence linking transmission with drinking-water for the general population. However, transmission of non-gastrointestinal infections by *Klebsiella* to susceptible patients in health-care settings may be possible.

General description

Klebsiella are Gram-negative, non-motile, rod-shaped bacteria. The genus consists of several species; many of these are natural members of the microbiome of most humans and many other animal hosts. However, *K. pneumoniae* and *K. oxytoca* are of potential concern to human health.

Human health effects

Klebsiella especially *K. pneumoniae* and *K. oxytoca*, are opportunistic pathogens that cause nosocomial infection, illness and death. They are major causes of pneumonia, infections of the urinary and gastrointestinal tracts, and infections associated with in-dwelling catheters, burns, wounds and surgery, as well as liver abscesses and septicaemia.

Epidemiology

Klebsiella are one of the most important causes of death due to bacterial infections including those associated with antimicrobial resistance. Patients with impaired immune systems, such as elderly people or very young children, and those with excessive burns, wounds, human immunodeficiency virus (HIV)/acquired immunodeficiency syndrome (AIDS) infection, respiratory deficiencies, diabetes, malignancies or undergoing immunosuppressive therapy are at the highest risk of infection.

Transmission

In health-care facilities, person-to-person contact or contact with contaminated fomites are the most common transmission routes. In particular, transmission of *K. pneumoniae* and *K. oxytoca* in health-care settings is associated with frequent contact with patients (e.g. in intensive care units). Factors that increase the likelihood of nosocomial infections include the presence of invasive devices such as respiratory support equipment and urinary catheters, post-surgical sites and wounds, and the use of antibiotics. Contaminated water, including biofilms and aerosols, are suspected sources of expo-

sure in health-care facilities, but this link is not confirmed. Sinks contaminated with *K. pneumoniae* and *K. oxytoca* have been implicated in hospital outbreaks of illness.

The infectivity of *K. pneumoniae* or other species through drinking-water is unknown. No human infectivity dose–response studies have been conducted for any of the plausible exposure routes, such as water or air. However, based on animal studies, the relative infectivity is probably low.

Detection methods

Klebsiella can be detected in drinking-water by standard culture-based methods for coliform and thermotolerant coliform bacteria. However, this must be followed by biochemical methods to distinguish them from the other bacteria in this group. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Klebsiella are common soil and water organisms. *Klebsiella* can grow in surface water receiving nutrient-rich industrial wastes (e.g. pulp and paper mills, textile plants, sugar cane processing plants). These species are typically not pathogenic. *Klebsiella* can also be detected in water environments impacted by faecal contamination, with *Klebsiella* commonly detected in the faeces of many healthy humans and animals. As a confounder, however, some species, that may not be faecal in origin, such as *K. oxytoca* can colonize drinking-water distribution systems and result in an indication of false-positive faecal contamination where thermotolerant coliform testing is used. *Klebsiella* can grow and persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies will control *Klebsiella*. These include protection of source waters from contamination by human and animal faecal wastes or industrial wastes (e.g. pulp and paper mills), application of adequate treatment processes and effective biofilm management in distribution systems and building water systems including health-care facilities. WSPs for health-care facilities need to also consider the susceptibilities of patients. Chlorine is an effective drinking-water disinfectant against *Klebsiella*. As *Klebsiella* include environmental strains that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

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WHO (2025) *Klebsiella. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Legionella

Significance in drinking-water

There is strong evidence for waterborne transmission of *Legionella*, with large numbers of outbreaks associated with drinking-water systems including hot water systems and devices such as cooling towers and spas. *Legionella* can be the most significant waterborne pathogen in high-income countries.

General description

Legionella are Gram-negative, rod-shaped, non-spore-forming, motile bacteria. They are found in a wide range of soil and water environments, primarily within free-living amoebae in biofilms, but also in sludges and sediments.

Thirty of 61 species of *Legionella* have been associated with disease (legionellosis). *L. pneumophila*, particularly serogroup 1, is the major waterborne pathogenic species, and is reported to cause most cases of legionellosis worldwide. A second species, *L. longbeachae*, associated with potting mixes and soils, causes about 5% of cases of legionellosis worldwide, but in Australia and New Zealand, it causes about 50% of cases. Other species associated with infrequent cases include *L. anisa*, *L. bozemanii*, *L. feelii*, *L. gormanii* and *L. micdadei*.

Human health effects

Legionella are opportunistic pathogens. Legionellosis can take one of two forms: Legionnaires' disease or Pontiac fever. The former is a pneumonic illness with initial symptoms including fever, loss of appetite, headache, malaise, dry cough and lethargy; other symptoms can include muscle pain, nausea, diarrhoea and confusion. The most frequent complications of Legionnaires' disease are respiratory failure, shock, and acute kidney and multi-organ failure. The death rate (including based on a review published in 2017) is generally within the range of 5% to more than 10%, with increased mortality in immunosuppressed and nosocomial cases. Pontiac fever is an acute but milder non-pneumonic and self-limiting disease with symptoms similar to those of influenza including fever, headache, nausea, vomiting, aching muscles, joint pain and coughing. Asymptomatic infections are common. Antibiotic resistance is rare.

Epidemiology

The worldwide incidence of legionellosis is unknown because of limitations in surveillance. Most reported data are from high-income countries, with national annual incidence rates ranging from 0.1 to 77 cases/million people (based on studies published between 2011 and 2021). In Europe and the United States of America, cases are generally classified as being community (60–70%), travel (20–25%) or hospital/nosocomial (5–10%) acquired. Host factors influence the likelihood of illness; males are more frequently affected than females, and most cases occur in people over 50 years of age. Risk factors associated with disease include smoking, diabetes, chronic respiratory disease, immunosuppression and excessive alcohol consumption. While disease is uncommon in the young, there have been cases in neonates associated with humidifiers and birthing pools. Legionellosis can occur at any time of year, but increased cases have been reported in the warmer months of summer and autumn. Most cases are sporadic, but outbreaks can occur, with the most common sources being cooling towers and hot and cold water systems in buildings and spas.

Transmission

The most common route of transmission of *Legionella* is the inhalation of contaminated aerosols produced by devices linked to drinking-water systems. Aspiration of contaminated water is also a potential route of transmission. There is little evidence of person-to-person transmission. The infectivity of *L. pneumophila* is not known and

is likely to vary depending on the susceptibility of those exposed and the virulence of particular strains. In some situations, the infective dose could be very low, as cases have been recorded following short exposure to *Legionella*. Growth within amoebae enhances the ability of *Legionella* to parasitize lung macrophages and cause disease. The incubation period for Legionnaires' disease is generally 2–10 days. The incubation period for Pontiac fever is 5 hours–3 days.

Detection methods

Standard culture-based methods for *Legionella* are available. Serological and molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Legionella occur naturally in relatively low numbers in many freshwater environments, such as groundwater, streams, rivers and reservoirs. *Legionella* may multiply in engineered water environments (e.g. drinking-water distribution systems) that provide conditions to support the survival and growth of *Legionella* in biofilms, particularly in association with free-living amoebae. Optimum growth temperatures for *Legionella* are 20–50 °C. Hot and cold water distribution systems in buildings, air-conditioning systems and spas are the most common sources of contaminated aerosols, but a range of devices or installations that produce mists or sprays can also be sources of *Legionella*.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Legionella* should focus on a combination of strategies to limit their survival during treatment and their growth in biofilms, including in free-living amoebae in distribution systems and building water systems. *Legionella* are moderately resistant to oxidizing disinfectants such as chlorine, but doses typically used in drinking-water treatment plants should be sufficient to inactivate planktonic *Legionella*. UV light is also effective against planktonic *Legionella*. Control strategies to reduce biofilm formation can reduce the potential for *Legionella* to survive and grow. Lastly, *Legionella* can be a particular concern in devices receiving drinking-water, such as cooling towers and hot water systems in large buildings; see [section 6.10](#) for more information on managing water systems in buildings including temperature regulation and minimizing dissemination of aerosols. As *Legionella* is a non-faecal, environmental organism that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water (see [section 11.6](#)).

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Leptospira

Significance in drinking-water

Leptospira are of low health significance in drinking-water. The evidence linking transmission with ingestion of drinking-water is weak, with reported drinking-water

cases and outbreaks having been linked only to storage reservoirs contaminated due to accidents or disasters. Water-based leptospirosis is normally caused by skin contact with contaminated surface water, especially after extreme rain events or flooding.

General description

Leptospira are tightly coiled, spiral-shaped bacteria. *Leptospira* include more than 35 species that are divided into three clusters. The pathogenic and intermediate clusters include the disease-causing species such as *L. interrogans*.

Human health effects

The severity of illness and the symptoms associated with leptospirosis vary widely. Infections are often subclinical. Symptoms may include fever, headache, muscle pain, chills, red eyes, abdominal pain, jaundice, haemorrhages of skin and mucous membranes, vomiting, diarrhoea and skin rash. The illness may progress to kidney or liver failure, aseptic meningitis or pulmonary bleeding, which are life-threatening complications. Long-lasting sequelae include depression, headaches, fatigue and joint pains. An acute and severe form of leptospirosis is known as Weil's disease, which is characterized by jaundice, renal failure and enlargement of the spleen.

Epidemiology

Leptospirosis occurs globally, affecting people living in temperate and tropical climates in rural and urban areas. Although the incidence of human infection is higher in tropical rather than temperate regions, transmission occurs in both high- and low-income countries. Seasonal peaks in disease incidence exist in most regions. In tropical countries, epidemiological clusters of leptospirosis are observed following floods, storms and hurricanes. In temperate climates, leptospirosis is commonly linked to occupational activities or recreational waters. Leptospirosis is recognized as a disease with epidemic potential in many parts of the world.

Transmission

The major transmission route for *Leptospira* is through contact with infected animals or contact with water and soil contaminated with the urine of infected animals. Pathogenic *Leptospira* persist in the renal tubules of mammalian hosts, which may have chronic asymptomatic infections with excretion persisting for a very long time, potentially even lifelong. Rodents, insectivores, dogs, cattle and swine are considered to be the main reservoirs. *Leptospira* can enter the body through cuts and abrasions in the skin or via the mucous membranes of the mouth, nose and eyes. Ingestion of contaminated food and water, or inhalation of aerosols, may occasionally cause infection, but it is not transmitted by the faecal–oral route. Person-to-person transmission is rarely observed. There is an increasing trend of outbreaks associated with recreational exposure to contaminated water. Outbreaks have also been associated with disasters involving flooding. Sexual contact, transplacental transmission and breast milk are potential routes of exposure. Infectivity of *Leptospira* varies depending on settings, exposure routes and human host status.

Detection methods

Culture-based and molecular methods are available for detection, quantification and typing of *Leptospira* in environmental samples (see [section 7.4](#)). Molecular methods are preferred since differentiation of pathogenic and non-pathogenic strains is more straightforward and reliable, and because biosafety concerns are lower, compared to culture methods.

Source and occurrence

There are limited data on occurrence of *Leptospira*, but they have been detected in water and wastewater environments including surface water with prevalence influenced by rainfall and contamination with urine and tissues of infected animals.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Leptospira* include protection of source waters and storage reservoirs from urine and tissues of infected animals, application of adequate treatment processes and protection of drinking-water quality during treated water distribution and storage from external contamination. Chlorine is an effective drinking-water disinfectant against *Leptospira*. As *Leptospira* is a non-faecal organism that can persist in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water.

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WHO (2025) *Leptospira. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Non-tuberculous *Mycobacterium*

Significance in drinking-water

There is moderate evidence for waterborne transmission of non-tuberculous *Mycobacterium* (NTM). In a number of NTM disease investigations, the detection of NTM in drinking-water and the identified routes of transmission suggested that drinking-water supplies were a plausible source of infection. In addition, a number of outbreaks have been reported in health-care centres associated with a single facility or a specific procedure.

General description

Mycobacterium are Gram-positive, non-spore-forming, rod-shaped bacteria, with a thick, lipid-rich outer membrane. The family includes up to 200 species and 13 sub-species. The “typical” species of *Mycobacterium*, such as *M. tuberculosis*, *M. bovis*, *M. africanum* and *M. leprae*, have only human or animal reservoirs and are not transmitted by water. In contrast, the atypical or non-tuberculous species of *Mycobacterium* (NTM) are natural inhabitants of a variety of water and soil environments. Among the NTM, the most common cause of disease is the *M. avium* complex (*M. avium*, *M. intracellulare* and *M. chimera*) followed by the *M. abscessus* complex, *M. kansasii* complex, *M. malmoense*, *M. simiae* and *M. xenopi*.

Human health effects

Mycobacterium are opportunistic pathogens. NTM infections are commonly asymptomatic, but can cause a broad range of diseases involving the skeleton, lymph nodes, skin and soft tissues, as well as the respiratory, gastrointestinal and genitourinary tracts. The most common presentation is pulmonary disease; other manifestations include disseminated disease, lymphadenitis, skin ulcers, post-traumatic wound infections and nosocomial disease. Disseminated infections are a common cause of death in HIV-positive persons. Due to their cell wall structure, NTM have a high resistance to antimicrobials, which complicates treatment.

Epidemiology

The worldwide incidence of NTM-mediated disease is unknown due to limited surveillance. Variation in disease presentation, lack of knowledge of diagnostic features and challenges with microbiological diagnosis contribute to uncertainty about incidence. However, there has been evidence (including from a 2021 study) of increasing prevalence of NTM-mediated disease in North America, Europe, Asia and Australia. Some species such as the *M. avium* complex and *M. abscessus* have worldwide significance, while others can be geographically limited. For example, *M. malmoense* is common in Europe, but rarely reported in the United States of America. However, compared to the widespread prevalence of the organism, disease is relatively infrequent. Estimated annual incidence rates reported range from 1.0 to 9.8 cases/100 000 persons in high-income countries (based on studies published in 2007 and 2016). Host factors play a significant role in susceptibility to disease including immunodeficiency (e.g. HIV infection), organ transplant, inflammatory disorders, genetic background, and presence of localized or generalized lung damage (e.g. chronic obstructive pulmonary disease, cystic fibrosis).

Transmission

Principal routes of waterborne infection are inhalation of aerosols, contact with contaminated materials or ingestion of contaminated water, primarily in health-care centres. Identifying sources of mycobacterial disease can be complicated by delays in diagnosis and microbial identification, but there is some evidence linking cases to organisms in drinking-water. Processed water used for dialysis and bioaerosols from heating-cooling units used during cardiothoracic surgery have been linked to outbreaks. Occurrence of pseudo-outbreaks (increase in positive cultures without evidence of disease) and pseudo-infections can complicate the identification of sources of clinically relevant NTM. These pseudo-outbreaks are often caused by sample or equipment contamination by NTM from health-care water supplies, leading to contamination of equipment or ice.

There is no evidence of person-to-person or zoonotic transmission of NTM, although indirect person-to-person transmission of *M. abscessus* between cystic fibrosis patients has been suggested. The infective dose is not known and is likely to vary depending on the susceptibility of those exposed and the virulence of individual species and strains. The incubation period is unknown and probably varies depending on the nature of the disease, but can be weeks to months.

Detection methods

There are culture-based methods for detection of NTM although isolation can be challenging due to their slow growth in the environment and in culture media. Molecular methods are also available for detection and speciation (see [section 7.4](#)).

Source and occurrence

NTM are ubiquitous inhabitants of natural water and soil environments and can thrive in engineered water environments (e.g. drinking-water distribution systems) that provide conditions to support the survival and growth of NTM in biofilms, including in association with free-living amoebae. *M. kansasii*, *M. xenopi* and *M. simiae* are recovered almost exclusively from drinking-water. Optimum growth temperatures for NTM species vary, but growth is reduced at temperatures outside the range of 20–55 °C.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from NTM should focus on a combination of strategies to limit their survival during treatment and their growth in biofilms including in free-living amoebae in distribution systems and building water systems. NTM are highly resistant to oxidizing disinfectants such as chlorine, but maintaining continuous levels of adequate disinfectant residuals in the distribution system and building water systems should reduce concentrations of planktonic NTM. Planktonic NTM are sensitive to UV light. Control strategies to reduce biofilm formation can reduce the potential for NTM to survive and grow. In a similar fashion to *Legionella*, NTM temperature regulation should be applied for building water systems to minimize growth of NTM; see [section 6.10](#) for further information. As NTM is a non-faecal, environmental organism that can survive and grow in water and is highly resistant to chlorine, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Non-tuberculous Mycobacterium. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Pseudomonas aeruginosa

Significance in drinking-water

Pseudomonas aeruginosa is of low health significance in drinking-water. Although *P. aeruginosa* can be detected in drinking-water, there is no evidence linking transmission with drinking-water for the general population. However, transmission of non-gastrointestinal infections by *P. aeruginosa* to susceptible patients in health-care facilities through drinking-water may be possible, with a limited number of hospital outbreaks having been reported due to use of contaminated water in patient care. The presence of high concentrations of *P. aeruginosa* can result in taste, odour and turbidity problems in drinking-water.

General description

P. aeruginosa is a Gram-negative, motile, rod-shaped bacteria. There are about 200 *Pseudomonas* species, of which *P. aeruginosa* is most prominent as a potential concern to human health, especially strains that are antimicrobial resistant.

Human health effects

P. aeruginosa is an opportunistic pathogen, commonly causing pneumonia, especially in health-care facilities. Cystic fibrosis and immunocompromised patients are prone to respiratory tract colonization with *P. aeruginosa*, which may lead to serious progressive pulmonary infections and death. *P. aeruginosa* may also cause chronic infections in individuals with chronic obstructive pulmonary disorder. In addition, *P. aeruginosa* infects sites such as burns, surgical wounds and physically damaged eyes. It can infect individuals who are using catheters or other in-dwelling medical devices. It is also a major cause of urinary tract infections. From these sites, it may invade the body, causing destructive lesions or septicaemia and meningitis. It rarely causes serious illness in healthy individuals without some predisposing factor. However, it can cause eye infections, especially among contact lens wearers, ear infections, skin infections and folliculitis associated with warm, recreational bathing waters.

Epidemiology

P. aeruginosa is one of the most important causes of illness and death due to bacterial infections, including those associated with antimicrobial resistance. Immunocompromised individuals and those with other predisposing factors are at highest risk of serious illness.

Transmission

Transmission of *P. aeruginosa* in the general population is principally through contact with recreational water. Ingestion of community drinking-water has not been identified as a source of infection. In health-care facilities, transmission can occur through aspiration of endogenous oral microbiota, aspiration from contaminated mechanical respiratory ventilators or other health-care devices, or through hematogenous spread. Physical contact with environmental sources such as water-associated fixtures, and cross-contamination in the health-care environment are also possible exposure routes. A limited number of hospital outbreaks have occurred due to contaminated water used for vulnerable patients including for enteral feeding, and to prepare medications. Two of three outbreaks were associated with bottled water/mineral water. The infectivity of *P. aeruginosa* for humans through ingestion is unknown, but based on a human volunteer study, it is expected to be low.

Detection methods

Culture-based and biochemical methods are available for *P. aeruginosa*. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

P. aeruginosa is a common water and soil organism and is found frequently in source water. *P. aeruginosa* has also been detected in drinking-water. It can persist for long

periods (over 1 month at 20 °C) and multiply in water environments and on the surface of suitable organic materials in contact with water in biofilms. *P. aeruginosa* are important bacteria in biofilms in premise plumbing, especially of health-care facilities. They have been isolated from a range of moist environments, such as sinks, hot water systems, showers, bathtubs, pools and spas in health-care and community settings.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies will control *P. aeruginosa*. These include application of adequate treatment processes and effective biofilm management in distribution systems and building water systems, including health-care facilities. WSPs for health-care facilities need to also consider the susceptibilities of patients. *P. aeruginosa* has low to moderate resistance to oxidizing disinfectants such as chlorine, but doses and contact times typically used in drinking-water treatment plants should be sufficient to inactivate planktonic *P. aeruginosa*. The presence of *Pseudomonas* in drinking-water supplies is generally considered a nuisance condition that could be indicative of inadequate management practices. As *P. aeruginosa* is an environmental organism and can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator of its possible presence or absence in drinking-water.

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WHO (2025) *Pseudomonas aeruginosa. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

***Salmonella* – non-typhoidal**

Significance in drinking-water

Non-typhoidal *Salmonella* (NTS) is an important zoonotic waterborne pathogen, with strong evidence from a number of outbreaks linking transmission to drinking-water particularly involving untreated or poorly treated groundwater and small rain-water systems.

General description

Salmonella are Gram-negative, non-spore-forming, rod-shaped bacteria that are predominantly motile. There are two species: *S. enterica* and *S. bongori*. *S. enterica* (subspecies enterica, salamae, arizonae, diarizonae, indica and houtenae) are associated with greater than 99% of *Salmonella*-caused diseases in humans and other warm-blooded animals. In addition to classification of subspecies, *Salmonella* are further subdivided into serovars. *S. enterica* subspecies enterica, serovar Typhi and Paratyphi are known as the typhoidal *Salmonella* serovars, whereas all other serovars are known as NTS. A large number of NTS serovars infect humans, with *S. enterica* subspecies enterica, serovars Typhimurium and Enteritidis being the most common causes of disease. Typhoidal *Salmonella* are the subject of a separate fact sheet.

Human health effects

In general, NTS presents less severe symptoms and lower mortality rates than infection with the typhoidal serovars *S. Typhi* and *S. Paratyphi*. Characteristic gastrointes-

tinal symptoms of NTS include diarrhoea (which occasionally may be bloody), often accompanied by fever, abdominal pain, nausea and vomiting. Symptoms are usually self-limiting but may persist for several days. Dehydration may be severe among vulnerable patients. Death is uncommon in otherwise healthy individuals, but is more likely in vulnerable age groups, as well as in immunocompromised individuals.

In addition to gastroenteritis, NTS has also been identified as a major cause of invasive disease (e.g. bacteraemia), particularly in Africa. As with some other gastrointestinal infections, a small subset of patients with non-typhoidal salmonellosis may also develop post-infectious disease syndromes such as reactive arthritis, irritable bowel syndrome and possibly inflammatory bowel disease. For severe cases of non-typhoidal salmonellosis, treatment with antibiotics may be recommended, but the prevalence of NTS resistant to multiple antibiotics is increasing. Prolonged carriers should not be treated with antibiotics.

Epidemiology

NTS is a common cause of diarrhoeal disease. However, in low-income, high-burden settings, NTS contributes only a small amount of the burden of childhood diarrhoeal disease compared with other enteric pathogens. Salmonellosis is most extensively reported in North America and Europe due to better surveillance systems. Invasive NTS is especially common in sub-Saharan Africa, where it has been identified as a leading cause of community-acquired blood stream infections.

Transmission

NTS is transmitted primarily by the faecal–oral (or urine–oral) route, with animals being the primary source. Food of animal origin is the most common source of salmonellosis. *S. Enteritidis* is commonly associated with poultry and poultry products, whereas *S. Typhimurium* has a wider species range, including pigs, cattle and poultry. An increasing number of illnesses have been associated with the consumption of fruits and vegetables. Transmission may also occur through ingestion of water contaminated with animal faeces, or by direct exposure to infected animals or their environments. Although documented cases are rare, person-to-person transmission may occur through direct contact or indirectly by pathogens from human faeces transferred on environmental surfaces or through unsafe handling of food. Direct person-to-person transmission is more likely to occur in infants and persons with certain underlying conditions.

There are several estimates of the infectivity of NTS based on ingestion of food, which might not be applicable to drinking-water. An analysis of data from human feeding trials determined a dose associated with a 50% infection rate (ID_{50}) of 23 600 organisms, while ID_{50} values ranging from 7 to 9610 organisms were estimated based on analysis of foodborne outbreaks. Overall, based on these estimates, the relative infectivity of NTS by ingestion is considered moderate. Incubation periods are typically 6–72 hours. Pathogen shedding occurs throughout the course of illness and may continue for varying periods afterwards. The median period of pathogen excretion is from 6 weeks to 3 months, depending on the serotype, but 1% of infected adults and 5% of infected children may excrete the pathogens for up to 1 year. Acquired immunity to NTS can develop to some extent, but invasive species may still cause infections.

Detection methods

Standard culture-based methods for NTS are available, as well as serological typing methods, immunoassays and molecular methods (see [section 7.4](#)).

Source and occurrence

NTS is found in a broad range of water and wastewater environments including surface water, vulnerable groundwater and drinking-water. Prevalence is influenced by the presence of faecal contamination including from human sewage (untreated or partially treated) and from animals. Persistence of NTS is expected to be moderate (1 week to 1 month at 20 °C), with survival influenced by sunlight, biotic factors and their ability to survive within amoebae.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from NTS should include protection of source waters from contamination by human and animal faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes and protection of drinking-water quality during distribution. Drinking-water should be protected from faecal contamination by birds and other animals during storage. Chlorine and UV light are effective drinking-water disinfectants against NTS. *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of NTS in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Non-typhoidal Salmonella. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Salmonella – typhoidal

Significance in drinking-water

Typhoidal *Salmonella* (*S. Typhi* and *S. Paratyphi*) is a major waterborne pathogen with strong evidence from multiple outbreaks and endemic disease linking transmission to contaminated drinking-water.

General description

Salmonella are Gram-negative, non-spore-forming, rod-shaped bacteria that are predominantly motile. There are two species: *S. enterica* and *S. bongori*. *S. enterica* (subspecies *enterica*, *salamae*, *arizonae*, *diarizonae*, *indica* and *houtenae*) are associated with greater than 99% of *Salmonella*-caused diseases in humans and other warm-blooded animals. In addition to classification of subspecies, *Salmonella* are further subdivided into serovars. *S. enterica* subspecies *enterica*, serovar Typhi (*S. Typhi*) and serovar Paratyphi (*S. Paratyphi*) are known as the typhoidal *Salmonella* for their respective abilities to cause typhoid and paratyphoid fevers in humans, whereas all other *Salmonella* are non-typhoidal (NTS). NTS are the subject of a separate fact sheet.

Human health effects

Severity of illness associated with *S. Typhi* varies from nonspecific illness to life-threatening typhoid fever. In endemic areas, many of those infected, particularly young children, may have a mild, nonspecific gastrointestinal disease that is not recognized clinically as typhoid fever. Initial symptoms of typhoid fever include headache, malaise, cough, diffuse abdominal pain, constipation and remittent fever, but patients may become progressively ill without treatment, and experience mental confusion, apathy with sustained fever and abdominal distension. Life-threatening complications include intestinal haemorrhage and perforation. Surviving patients may experience continuing weakness and persistent weight loss for months. Pneumonia, and joint and urinary tract infections are examples of atypical presentations of typhoid fever. Asymptomatic infection can also occur. Infection with *S. Paratyphi* causes similar, although generally milder, symptoms. Resistance of typhoidal *Salmonella* to multiple antibiotics emerged in the 1980s, and increasing resistance to fluoroquinolones has been reported.

Epidemiology

Typhoid fever is more prevalent than paratyphoid fever. The highest known rates of typhoid and paratyphoid fever are in south-east Asia and sub-Saharan Africa. The age distribution of illness from typhoid fever varies between endemic and low-prevalence settings. In endemic areas, illness is most common in children. In low-prevalence areas, the incidence has a more even age distribution, and a high proportion of cases are a result of travel to endemic disease areas.

Transmission

S. Typhi and *S. Paratyphi* are spread primarily by the faecal–oral (or urine–oral) route and are largely restricted to humans. Infection by typhoidal *Salmonella* is mostly associated with the consumption of contaminated water and food. There is a strong association of typhoid and paratyphoid fever with inadequate sanitation, unsafe drinking-water and poor hygiene. In endemic areas, there is evidence that children contract *S. Typhi* from heavily contaminated environments through physical contact with water sources. Typhoid fever can occasionally be transmitted by direct person-to-person contact, but this is a less important route than contaminated food or water.

The relative infectivity of *S. Typhi* is low, with an estimated dose associated with a 50% infection rate (ID_{50}) of 1.1×10^6 organisms. The incubation period is usually 8–14 days for typhoid fever and 1–10 days for paratyphoid fever. Patients excrete the pathogen throughout the period of symptomatic infections and for variable periods afterwards. For paratyphoid fever, shedding is common for 1–2 weeks after symptoms. For about 10% of *S. Typhi* patients with no treatment, shedding can continue for 3 months after. Between 1% and 5% of patients become chronic carriers without ongoing symptoms and excrete the pathogen for more than a year. A partial form of immunity may follow recovery from symptomatic and asymptomatic infection.

Detection methods

Standard culture-based methods for *S. Typhi* and *S. Paratyphi* are available, as well as serological typing methods, immunoassays and molecular methods (see [section 7.4](#)).

Source and occurrence

There are limited data on the occurrence of *S. Typhi* and *S. Paratyphi*, but they have been detected in water and wastewater environments including surface water and inadequately treated drinking-water. Prevalence is influenced by the presence of human faecal waste including from sewage (untreated or partially treated). Persistence in water is short (up to 1 week at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *S. Typhi* and *S. Paratyphi* should include protection of source waters from contamination by human faecal waste (including extra vigilance during heavy rain), application of adequate water treatment processes and protection of drinking-water quality during storage and distribution. Chlorine and UV light are effective drinking-water disinfectants against typhoidal *Salmonella*. *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of *S. Typhi* and *S. Paratyphi* in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Typhoidal Salmonella. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Shigella

Significance in drinking-water

Shigella are major waterborne pathogens, with strong evidence from multiple outbreaks linking transmission to contaminated drinking-water.

General description

Shigella are Gram-negative, non-spore-forming, non-motile, rod-like bacteria. There are four species of *Shigella*: *S. dysenteriae*, *S. flexneri*, *S. boydii* and *S. sonnei*, which are further divided into a number of serotypes. Although *Shigella* are classified in a separate genus, they are genetically very similar to pathogenic *E. coli*. It is generally considered that *Shigella* belong to the same pathotype as enteroinvasive *E. coli* (EIEC), but are more virulent. EIEC is the subject of the separate diarrhoeagenic *E. coli* fact sheet.

Human health effects

Shigella can cause mild to serious illness, including bacillary dysentery (shigellosis). The severity and case fatality vary by species and serotype, as well as host characteristics. Abdominal cramps, fever, watery diarrhoea and dehydration occur early in the infection. Illness due to *S. sonnei* is usually relatively mild and self-limiting. Shiga toxin-producing *S. dysenteriae* type 1 is most often associated with severe disease and associated complications, including toxic megacolon, intestinal perforations and haemolytic uraemic syndrome. Long-term complications in children infected with *S. dysenteriae* in low- and middle-income countries include persistent diarrhoea, with severe dehydration and malnutrition, while children hospitalized with shigellosis frequently experience seizures. *S. flexneri* has been associated with infectious arthritis.

Death from shigellosis is most likely in young children and elderly people. A high incidence of resistance of *Shigella* to all first-line antibiotics has been reported; therefore, treatment options for *Shigella*-mediated dysentery are becoming limited.

Epidemiology

Shigella are one of the most common causes of moderate to severe diarrhoea and death, including among young children in low- and middle-income countries. Seasonal fluctuations of shigellosis occur that vary between settings. *S. dysenteriae* type 1 pandemics have resulted in high rates of fatalities for all age groups in Central America, South Asia, and Central and East Africa. The four *Shigella* species have different distributions: *S. dysenteriae* is mainly found in South America, Africa and Asia, *S. flexneri* in low-income countries with endemic shigellosis, *S. boydii* in India and *S. sonnei* in high-income countries.

Transmission

Shigella are transmitted primarily by the faecal–oral route through person-to-person contact, and by ingestion of contaminated food or water. Humans and other higher primates appear to be the only natural hosts for *Shigella*. The risk of transmission is greatest in low-income settings where sanitation and hygiene conditions are generally poor, with sporadic cases and outbreaks of shigellosis often occurring in crowded communities or institutions, including day-care centres and health-care facilities. Cases and outbreaks have also been associated with recreational waters. Flies have also been identified as a transmission vector from contaminated faecal waste.

The relative infectivity varies among the species. *S. dysenteriae* is highly infective with an estimated dose associated with a 50% infection rate (ID_{50}) of 36 organisms. *S. flexneri* is moderately infective, with an estimated ID_{50} of 148 organisms. The incubation period ranges between 1 and 3 days, but may be up to 1 week, depending on the species and the host. The pathogen is usually shed for up to 4 weeks upon illness and following recovery; in rare cases, it can be shed for months. Repeated exposure to *Shigella* can provide protective immunity.

Detection methods

Standard culture-based methods for *Shigella* are available, but they are complex. In addition, commercial biochemical tests to type isolates exist. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

There are limited data on the occurrence of *Shigella* due to challenges in detecting these pathogens in environmental samples. Nevertheless, they have been detected in water and wastewater environments, including surface water, vulnerable groundwater and inadequately treated drinking-water. Prevalence is influenced by the presence of sewage (untreated or partially treated). Based on limited studies, the persistence of *Shigella* in water is likely to be moderate (1 week to 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage the potential risks from *Shigella* should include protection of source waters from contamination by human

faecal waste (including extra vigilance during heavy rain), application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Chlorine and UV light are effective drinking-water disinfectants against *Shigella*. *E. coli* (or thermotolerant coliforms) is a suitable indicator for the possible presence or absence of *Shigella* in drinking-water (see [section 11.6](#)).

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WHO (2025) *Shigella*. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health. Geneva, World Health Organization.

Staphylococcus aureus

Significance in drinking-water

Staphylococcus aureus is of low health significance in drinking-water. Although it has been detected in drinking-water, there is no evidence linking transmission with this source. Infections of susceptible individuals in health-care facilities through drinking-water may be possible, but have not been identified.

General description

S. aureus is a non-motile, non-spore-forming, Gram-positive round-shaped bacterium. The *Staphylococcus* genus contains at least 47 different species, of which only *S. aureus*, *S. epidermidis*, *S. saprophyticus*, *S. haemolyticus*, *S. lugdunensis* and *S. argenteus* are associated with disease in humans.

Human health effects

S. aureus is an opportunistic pathogen, and some strains are common human commensal organisms. It can affect healthy and vulnerable populations. Methicillin-resistant *S. aureus* causes more serious human health effects than methicillin-sensitive *S. aureus*, but both can be pathogenic. *S. aureus* skin and soft tissue infections are common and typically occur as pimples, furuncles, carbuncles and folliculitis. Multiplication in tissues can result in boils, skin sepsis, septicaemia, endocarditis, osteomyelitis, urinary tract infections and pneumonia. In health-care facilities, *S. aureus* can cause various invasive infections. The onset of clinical symptoms for these infections is relatively long, usually several days. *S. aureus* can also produce extracellular enzymes and toxins that commonly cause foodborne intoxications as well as clinical toxinoses such as toxic shock syndrome, septic shock and scalded skin syndrome.

Epidemiology

S. aureus, especially methicillin-resistant *S. aureus*, is an important cause of pneumonia and other serious and life-threatening diseases in community and health-care settings. It is one of the most important causes of death due to bacterial infections, including those associated with antimicrobial resistance.

Transmission

S. aureus is not transmitted by the faecal–oral route. Person-to-person contact (hand-to-hand and other skin-to-skin contact) or contact with contaminated fomites are the most common routes of transmission. Community-acquired exposures are more

likely when hygienic conditions are inadequate due to crowding, or where there is inadequate access to sanitation and poor personal hygiene, such as lack of handwashing access and practices, and sharing of towels, clothing, razors, unwashed bedding and intravenous drug use. Hospitals and other health-care facilities are also major reservoirs and exposure sources of methicillin-resistant *S. aureus*. Infected patients can transmit these bacteria person to person, from shed human skin and also release them to the environment via hospital wastewater discharges. Foodborne intoxication risks from *S. aureus* are also common, with outbreaks reported globally. Close contact with livestock is another source of exposure to methicillin-resistant and methicillin-sensitive *S. aureus*. The infectivity of *S. aureus* through ingestion is unknown.

Detection methods

Culture-based methods are available for *S. aureus*. Molecular methods are also available and used in clinical diagnostic analysis and epidemiological investigations (see [section 7.4](#)).

Source and occurrence

S. aureus is widespread in soil and water environments and has been detected in source waters. *S. aureus* is frequently detected in unchlorinated drinking-water at relatively low concentrations. It has also been detected in inadequately treated drinking-water. Concentrations can be higher in water from faucets with aerators, presumably due to human contact, colonization and biofilm formation. *S. aureus* can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies will control *S. aureus*. These include application of adequate treatment processes and effective biofilm management in distribution systems and building water systems, including health-care facilities. WSPs for health-care facilities need to also consider the susceptibilities of patients. Chlorine is an effective drinking-water disinfectant against *S. aureus*. As *S. aureus* is a non-faecal, environmental and commensal organism that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water.

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WHO (2025) *Staphylococcus aureus. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Tsukamurella

Significance in drinking-water

Tsukamurella are of low health significance in drinking-water. Although they have been detected in drinking-water, there is no evidence linking transmission with this source. Infection of susceptible individuals through drinking-water may be possible, but it has not been identified.

General description

Tsukamurella are Gram-positive, obligate bacteria with varying morphologies from straight to curved rods. There are about 14 species, with some associated with human infections including *T. tyrosinosolvens*, *T. hongkongensis*, *T. inchonensis*, *T. pseudospumae*, *T. paurometabola*, *T. pulmonis*, *T. sinensis*, *T. soli*, *T. spumae* and *T. strandjordae*.

Human health effects

Tsukamurella can cause respiratory infections and chronic lung diseases such as pneumonia and tuberculosis-like disease, bacteraemia, meningitis and brain abscesses, peritonitis, keratitis, conjunctivitis, cutaneous infections and acute otitis media.

Epidemiology

Most *Tsukamurella* infections are hospital associated. Risk factors for infection and illness include other predisposing health conditions, such as immunosuppression due to HIV/AIDS, cancer and immunosuppressive therapies, renal failure and exposure to contaminated medical equipment and materials. Community-acquired pneumonia has been reported, but is rare. Cases have been reported in Europe, Asia, America and Africa, although the disease burden from *Tsukamurella* is thought to be relatively low compared to other causes of the same diseases or syndromes in at-risk populations, usually in health-care facilities.

Transmission

Tsukamurella are not transmitted by the faecal–oral route. In health-care facilities, person-to-person contact through direct contact with wounds or contact with contaminated fomites are the likely transmission pathways for susceptible individuals. In community-acquired cases, sources of infection are typically not identified. In hospital and community-acquired cases, the original sources of infecting *Tsukamurella* are often not known, but may be environmental or humans and animal hosts. The infectivity of *Tsukamurella* is unknown. However, the widespread occurrence of *Tsukamurella* in the environment and their low disease burden in healthy people suggest relatively low infectivity.

Detection methods

Tsukamurella are recoverable by the heterotrophic plate count method. They are also recoverable by selective media for aerobic actinomycetes. Definitive identification in clinical diagnostic laboratories is based on molecular analysis (see [section 7.4](#)).

Source and occurrence

As actinomycetes, *Tsukamurella* are widespread in soil and water environments including surface water, and have been detected in drinking-water. As *Tsukamurella* are known to proliferate in the environment and survive in biofilms, they are expected to persist in water for long periods (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures that effectively manage the presence and growth of bacteria in drinking-water supplies and building water systems will control *Tsukamurella*. These include application of adequate treatment processes and

effective biofilm management in distribution systems and building water systems including health-care facilities. WSPs for health-care facilities need to also consider the susceptibilities of patients. Based on studies with other actinomycetes, *Tsukamurella* are likely to be moderately resistant to chlorine, but doses and contact times typically used in drinking-water treatment plants should provide adequate inactivation of planktonic *Tsukamurella*. As *Tsukamurella* are non-faecal environmental organisms that can persist and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

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Vibrio cholerae

Significance in drinking-water

Vibrio cholerae is a major waterborne pathogen, with strong evidence from epidemics and endemic areas linking transmission to drinking-water. Consumption of drinking-water contaminated with sewage is a primary source of *V. cholerae*.

General description

V. cholerae are small, motile, curved, Gram-negative rod-shaped bacteria. *V. cholerae* are classified into serogroups based on their O antigens with a limited number able to produce cholera enterotoxin. Only strains of serogroups O1 and O139 cause epidemics. *V. cholerae* serogroup O1 can be divided into “classical” and “El Tor” biotypes based on phenotypic or biochemical characteristics. Each O1 biotype is further classified into Ogawa, Inaba and Hikojima serotypes, with the Ogawa serotype being the most dominant serotype in cholera endemic areas.

Human health effects

Most infections caused by toxigenic *V. cholerae* are asymptomatic or result in mild symptoms. However, it can cause severe disease. There is no difference in the types of symptoms elicited by the different toxigenic strains of *V. cholerae*, but some El Tor O1 variants can cause more severe symptoms presumably due to the production of greater amounts of toxin. Non-toxin-producing strains of *V. cholerae* can cause self-limiting gastroenteritis, wound infections and bacteraemia. Host factor characteristics can also influence the severity of disease.

The initial symptoms of cholera are an increase in peristalses followed by loose, watery and mucus-flecked “rice-water” stools. Complications of cholera are mainly attributed to dehydration and electrolyte loss, which may be accompanied by muscle cramps and seizures. Severe complications include acute renal failure, stroke, miscarriages, premature deliveries and stillbirths. A substantially high percentage of untreated cholera patients may die from the disease (more than 50% according to a 2017 review), but well-established diarrhoeal disease control programmes can reduce case fatality rates to less than 1%. The prevalence of *V. cholerae* resistant to multiple antibiotics is steadily increasing. It is recommended that antibiotics should only be used in cholera cases with severe dehydration.

Epidemiology

V. cholerae is among the leading causes of diarrhoeal deaths globally. The majority of cases occur in Africa and south-east Asia. Cholera can be endemic, epidemic or pandemic. Epidemics are usually triggered by exogenous introduction of toxigenic *V. cholerae* and can be associated with disasters including floods, earthquakes, wars and other events that disrupt infrastructure and public health facilities. Climate change could increase the prevalence and severity of disrupting events. In endemic settings, the highest cholera incidence is among young children, with initial infection leading to immunity. When cholera epidemics occur in non-endemic populations, all ages lack pre-existing immunity, hence incidence rates of cholera are usually age independent. Prior infection by serotype-homogenous strains can provide partial immunity for at least 3 years. Exposure to El Tor Inaba cholera is partially protective against El Tor Ogawa cholera, but this cross-protection is not apparent in reverse, and no cross-protection has been detected between O1 and O139 strains. In cholera endemic countries, cases tend to occur seasonally, with one or two peaks per year depending on the country. The underlying basis for seasonality of cholera has not been determined, but suggested factors include temperature, rainfall and plankton blooms.

Transmission

Transmission of toxigenic *V. cholerae* occurs by the faecal–oral route through ingestion of contaminated water or food, with water the primary source of disease in endemic areas. Use of this water for drinking, cooking, bathing and washing can lead to infection. Person-to-person transmission can occur within families, communities, institutional settings and crowded camps, particularly where sanitation and hygiene are poor. Toxigenic *V. cholerae* has been detected in domestic and farm animals, but there is no evidence that these animals are a source of infection.

The relative infectivity is low in healthy people, with an estimated dose associated with a 50% infection rate (ID_{50}) of 6.4×10^8 organisms, but infectivity can be moderate to low in people with reduced gastric acidity (ID_{50} of 2.4×10^2 organisms). The incubation period is usually 18 hours–5 days. Symptomatic cases generally excrete the bacteria for only a few days after recovery, while asymptomatic cases may excrete the bacteria for 7–14 days. Chronic carriage is rare, although intermittent shedding can occur for several months.

Detection methods

Standard culture-based methods are available for *V. cholerae*. Species can be further characterized using biochemical and serological tests. Molecular methods are also available for detection of virulence factors including cholera toxin genes. Other methods, incorporating immunofluorescence have also been developed to detect toxigenic strains (see [section 7.4](#)).

Source and occurrence

The occurrence of toxigenic *V. cholerae* in wastewaters and source waters, particularly surface waters, is correlated with occurrence of disease, but there have only been a few reports of detection in drinking-water. There are also few reports on the occurrence of *V. cholerae* in groundwater. *V. cholerae* can persist for moderate to long periods in

water (1 week to over 1 month at 20 °C) due to a variety of adaptive responses that increase resistance to stressors including nutrient limitation, predation and changes in salinity, temperature and pH. *V. cholerae* can survive in biofilms, assume “persister” phenotypes and switch from a smooth to a more robust “rugose” (wrinkled) variant. *V. cholerae* can also enter a viable but non-culturable state, but the mechanism of re-suscitation following ingestion and the epidemiological importance of the viable but non-culturable state is uncertain.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risk from *V. cholerae* should include protection of source waters from human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Control strategies to reduce biofilm formation can also reduce the potential for *V. cholerae* to survive in distribution systems. Chlorine and UV light are effective drinking-water disinfectants against *V. cholerae*. *E. coli* (or thermotolerant coliforms) is not a suitable indicator for the absence of *V. cholerae* in drinking-water due to its long persistence. However, the presence of *E. coli* is a suitable indicator of the possible presence of *V. cholerae* (see [section 11.6](#)).

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Yersinia enterocolitica

Significance in drinking-water

There is moderate evidence for waterborne transmission of *Yersinia enterocolitica*. Limited outbreaks have been reported from inadequately or non-disinfected small-scale drinking-water supplies. Untreated drinking-water has also been identified as a risk factor for sporadic *Y. enterocolitica* infections.

General description

Yersinia are Gram-negative, rod-shaped bacteria that are motile at temperatures of 22–29 °C, but become non-motile at normal human body temperature. The species *Y. pestis*, *Y. pseudotuberculosis* and certain serotypes of *Y. enterocolitica* are pathogenic for humans. While *Y. enterocolitica* is the dominant cause of enteric yersiniosis, *Y. pseudotuberculosis* can be responsible for extra-intestinal yersiniosis. *Y. pestis* is the cause of bubonic plague through contact with rodents and their fleas.

Human health effects

Infection with *Y. enterocolitica* can produce a range of clinical outcomes, although yersiniosis generally presents as a self-limiting gastrointestinal infection. Severe yersiniosis generally presents as acute gastroenteritis with or without diarrhoea (which may be bloody), fever and abdominal pain. Clinical symptoms may also include acute inflammation of the mesenteric lymph nodes mimicking appendicitis, inflammation of the pharynx and bacteraemia. More rarely, a range of non-gastrointestinal infections

such as abscesses, pyomyositis, pneumonia, meningitis, urinary tract infection, septic arthritis and conjunctivitis may occur. Post-infection complications include skin rash and reactive arthritis. Antibiotics may be used in severe cases involving systemic infections or invasive complications. Resistance to antibiotics in *Y. enterocolitica* varies among the different biotypes, but generally remains low with some exceptions.

Epidemiology

Detection of yersiniosis has primarily been limited to high-income countries, where it is an important cause of foodborne bacterial gastrointestinal illness. However, *Y. enterocolitica* has also been detected in humans and animals in low-income settings, although their significance is less well understood in these regions because of a lack of foodborne disease surveillance. Children appear to be more susceptible to infection; diarrhoea and abdominal pain seem to be more acute in children than in adults.

Transmission

Y. enterocolitica is transmitted primarily by the faecal–oral route. Wild and domestic animals, pigs in particular, are the principal reservoirs for infectious *Y. enterocolitica*. The major source of infection is considered to be contaminated food, particularly meat and meat products from pigs, but outbreaks from contaminated vegetables have been reported. Ingestion of contaminated drinking-water and contact with infected animals are also considered as potential risk factors for infection.

The relative infectivity of *Y. enterocolitica* is unknown, but it is considered to be low with an estimated dose associated with a 50% infection rate (ID_{50}) of 10^9 organisms. Direct person-to-person transmission is therefore unlikely. The incubation period for yersiniosis in humans is usually under 10 days. Symptoms usually last for 2–3 weeks, and pathogens are shed in faeces for this period. Untreated cases may shed pathogens for 2–3 months, and long-term asymptomatic carriage can occur.

Detection methods

Culture-based methods are available, but they lack sensitivity and selectivity, which is likely to contribute to a substantial underestimation of the presence of pathogenic *Y. enterocolitica* in the environment. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Y. enterocolitica is a common soil and water organism, and has been detected in surface water, groundwater and inadequately treated drinking-water. Prevalence is influenced primarily by the presence of animal faecal material, particularly from pigs. The presence of human faecal waste including sewage (untreated or partially treated) may also be a contributing factor. Limited studies suggest that *Y. enterocolitica* can survive for long periods in surface water (over 1 month at 20 °C). The presence of free-living amoeba and formation of biofilms have been shown to enhance their survival.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Y. enterocolitica* should include protection of source waters from contamination by animal (particularly from pigs) and human faecal wastes, application of adequate

water treatment processes, and protection of drinking-water quality during storage and distribution. Control strategies to reduce biofilm formation can also reduce the potential for *Y. enterocolitica* to survive and grow in distribution systems. Chlorine and UV light are effective drinking-water disinfectants against *Y. enterocolitica*. Since some strains of *Y. enterocolitica* are environmental organisms and can persist and grow in water, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its absence in drinking-water. However, the presence of *E. coli* is a suitable indicator for the possible presence of *Y. enterocolitica* (see [section 11.6](#)).

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WHO (2025) *Yersinia enterocolitica. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

11.2 Viral pathogens

Viruses with moderate to strong evidence of waterborne transmission are predominantly enteric pathogens. With the exception of hepatitis E virus, humans are the only established source of human infectious species. However, zoonotic transmission may be possible for adenoviruses, noroviruses, rotaviruses and sapoviruses. Enteric viruses are typically transmitted by ingestion of contaminated drinking-water. Although infection is typically initiated by replication in the gastrointestinal tract, the illnesses caused by enteric viruses are more diverse than those caused by enteric bacteria and protozoa. For example, enteroviruses can cause mild illness characterized by cough, rash and fever through to a broad spectrum of diseases such as aseptic meningitis, encephalitis and poliomyelitis, while hepatitis A and E viruses can cause infectious hepatitis. For further information, including an overview of treatment considerations for viruses, see [section 7.2.2](#).

Adenoviruses

Significance in drinking-water

Human adenoviruses (HAdVs) are important waterborne pathogens, with outbreaks primarily associated with recreational waters. HAdVs have been detected in substantial numbers in drinking-water and represent a likely but unconfirmed source of infection. HAdVs are also considered important because they are among the most resistant waterborne pathogens to UV light.

General description

HAdVs consist of a double-stranded deoxyribonucleic acid (DNA) genome in a non-enveloped icosahedral capsid that is about 90–100 nm in diameter. The family Adenoviridae is classified into five recognized genera: *Mastadenovirus* with mammalian hosts, *Aviadenovirus* with avian hosts, and *Atadenovirus*, *Ichadenovirus* and *Siadenovirus* with a broad range of non-human hosts. HAdVs have been classified into seven species (A to G) and are further classified into 68 distinct types.

Human health effects

HAdVs cause a wide range of infections with a spectrum of clinical manifestations. Adenovirus types 40 and 41 (species F) are the most common associated with infections of the gastrointestinal tract (gastroenteritis), especially in infants and children. Other types are associated with infections of the respiratory tract, the urinary tract and the eyes. HAdV infection can occasionally be fatal. Many infections are asymptomatic.

Epidemiology

Adenovirus infections have been detected worldwide. There have been numerous outbreaks associated with hospitals, military establishments, child-care centres and schools. In low-income settings, adenovirus types 40 and 41 have been identified as an important cause of diarrhoeal disease among young children. In addition to young children, immunocompromised individuals and elderly people are at risk of more severe disease. Infections occur year-round, with no marked seasonality.

Transmission

HAdV infections typically spread via person-to-person contact; this can include faecal–oral, oral–oral and hand–eye contact transmission. Inhalation of airborne viruses, ingesting contaminated recreational water and contact with contaminated fomites can also play a role in transmission. The consumption of contaminated food or drinking-water may be an important source of enteric illness, although there is no substantial evidence supporting these routes of transmission. Adenoviruses are generally only pathogenic to the species of origin, but there is some evidence of zoonotic transmission.

HAdVs are highly infective, with an estimated dose associated with a 50% infection rate (ID_{50}) of 0.74 virus particles. The incubation period for HAdV-associated disease is 2–14 days. Individuals infected with enteric adenoviruses shed higher concentrations of virus particles in faeces than those infected with non-enteric adenoviruses. Shedding can continue for months after symptoms resolve. Serotype-specific immunity is conferred following infection with enteric adenoviruses, but this wanes over time.

Detection methods

Cell culture and molecular methods for HAdVs are available (see [section 7.4](#)). However, some serotypes, including serotypes 40 and 41 are not readily isolated by cell culture methods. Combining cell culture infectivity and molecular methods may improve the detection of infectious HAdVs.

Source and occurrence

HAdVs are found in a broad range of water and wastewater environments, and have been frequently detected in surface waters and treated drinking-water supplies. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). HAdVs can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from HAdVs should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HAdVs by drinking-water treatment varies depending on the process. Chlorine is an effective drinking-water disinfectant, but UV light at commonly used doses is ineffective. Due to the smaller size of HAdVs relative to *E.coli* resulting in lower removal by treatment processes such as filtration, and their greater environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their absence in drinking-water. However, the presence of *E.coli* is a suitable indicator for the possible presence of HAdVs (see [section 11.6](#)).

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Aichiviruses

Significance in drinking-water

Human aichiviruses (HAiVs) are of low health significance in drinking-water. Although HAiVs are transmitted by the faecal–oral route and have been detected in drinking-water, the evidence linking transmission with drinking-water is weak.

General description

Aichiviruses are small (30 nm in diameter) and consist of a single-stranded ribonucleic acid (RNA) genome in a non-enveloped capsid. There are six species of aichiviruses (A to F) that can infect a broad range of animals. Aichivirus A is further divided into six types, one of which infects humans. Within HAiVs, three genotypes have been described (A, B and C). HAiV infection was first reported in 1989 in Japan. A retrospective study in archival samples dated its presence to July 1987 in sewage in the Netherlands.

Human health effects

Clinical symptoms of HAiV infection are similar to gastroenteritis, and include diarrhoea, abdominal pain, nausea, vomiting and fever. The majority of individuals infected with HAiVs remain asymptomatic.

Epidemiology

HAiVs have been detected in human faecal samples in Africa, Asia, Europe and South America, suggesting that they are distributed worldwide. However, the epidemiology of gastroenteritis caused by HAiVs remains largely unknown.

Transmission

HAiVs can be transmitted by the faecal–oral route through ingestion of contaminated food, particularly shellfish. Transmission through ingestion of drinking-water is plau-

sible, but has not been confirmed. The relative infectivity of HAIvS is unknown, but given their similarities to other enteric viruses, it is expected to be high.

Detection methods

Cell culture and molecular methods are available for HAIv (see [section 7.4](#)).

Source and occurrence

Aichiviruses have been detected in a broad range of water and wastewater environments. They have been found in surface water, groundwater and in one study of drinking-water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). The persistence of HAIvS in water is unknown, but as they are resistant to environmental stress and based on similarities with other non-enveloped viruses, it is expected to be long (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against HAIv. These include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HAIvS by drinking-water treatment varies depending on the process. Based on their known resistance to environmental stress, HAIvS are likely to be moderately resistant to disinfection with chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of HAIvS relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their expected environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of HAIvS (see [section 11.6](#)).

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Astroviruses

Significance in drinking-water

Human astroviruses (HAsTVs) are of low significance in drinking-water. Although HAsTVs are transmitted by the faecal–oral route and have been detected in drinking-water, there is only weak evidence linking transmission with drinking-water.

General description

Astroviruses are small (28–35 nm in diameter) and consist of a single-stranded RNA genome with a non-enveloped icosahedral capsid. A proportion of astroviruses have a distinct star-shaped surface structure. There are human and animal strains with eight different serotypes of human astroviruses. Human astrovirus serotype 1 (HAsTV-1) is the most commonly identified of these serotypes. Two novel groups of highly divergent HAsTVs have also been identified.

Human health effects

HAsTVs cause short-term, self-limiting diarrhoea with fever, abdominal pain and dehydration. Asymptomatic infection also occurs. Symptoms are generally mild and resolve without treatment within 2–3 days. In rare cases, severe central nervous system disease develops with high mortality rates in severely immunocompromised human hosts.

Epidemiology

The global burden of HAsTVs is unknown. HAsTVs affect children worldwide, with limited reports of disease in normally healthy adults. However, immunocompromised individuals and elderly people are at high risk. Geographically, astroviruses are widespread. Although HAsTV-1 is the most common, other serotypes are also widely distributed with some geographical differences. In temperate regions, HAsTV infections are higher in winter, while in tropical regions, infections peak in the rainy season.

Transmission

Human astroviruses are transmitted by the faecal–oral route, with person-to-person spread as the most common route of exposure. Contaminated food and fomites have also been identified as exposure sources. Transmission through ingestion of drinking-water is plausible but has not been confirmed. The relative infectivity of HAsTVs is unknown, but given their similarities to other enteric viruses, it is probably high.

Detection methods

Cell culture and molecular methods are available for HAsTVs (see [section 7.4](#)).

Source and occurrence

Astroviruses have been detected in a broad range of water and wastewater environments, and have been often found in surface water. Astroviruses have also been detected in groundwater and drinking-water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). HAsTVs can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against HAsTVs. These include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HAsTVs by drinking-water treatment varies depending on the process. Although HAsTVs are moderately resistant to disinfection with chlorine, doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of HAsTVs relative to *E. coli*, resulting in lower removal by treatment processes such as filtration and their longer environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of HAsTVs.

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Coronaviruses

Significance in drinking-water

Coronaviruses are of low health significance in drinking-water. There is currently no evidence linking transmission with drinking-water. Furthermore, evidence suggests that drinking-water is an unlikely route of transmission.

General description

Coronaviruses have a single-stranded RNA genome in an enveloped nucleocapsid with a diameter of 50–200 nm. Coronaviridae is a family of mammalian and avian viruses with two subfamilies: Torovirinae and Coronavirinae. Most human coronaviruses are in the Coronavirinae subfamily, which has four genera: *Alphacoronavirus* (human coronaviruses 229E and NL63); *Betacoronavirus* (human coronaviruses HKU1 and OC43; Middle East respiratory syndrome coronavirus (MERS-CoV); severe acute respiratory syndrome coronavirus (SARS-CoV-1) and (SARS-CoV-2)); *Deltacoronavirus* (porcine coronavirus HKU15); and *Gammacoronavirus* (no human viruses). Principal hosts are birds and mammals, especially bats, but also other animals such as swine, camels, civets and minks.

Human health effects

Coronavirus respiratory illnesses range from mild to fatal, including the common cold (caused by human coronaviruses OC43, 229E, HKU1 and NL63), pneumonia and respiratory distress syndromes. The most serious illnesses are SARS (caused by SARS-CoV-1), COVID-19 (caused by SARS-CoV-2) and MERS (caused by MERS-CoV). Asymptomatic or subclinical infections also occur. Long-term effects can occur for some coronaviruses, particularly from COVID-19, with common symptoms of fatigue, attention disorder and laboured breathing.

Epidemiology

The epidemiology differs significantly by virus, with human coronaviruses 229E and OC43 being largely endemic in communities and hospital patients with respiratory illness. Coronaviruses causing SARS, MERS and COVID-19 have been characterized by epidemics and global pandemics of severe disease, with substantially higher case fatality rates than the common cold coronaviruses. Risk groups for more severe disease associated with the common cold coronaviruses, MERS-CoV and SARS-CoV-2 include immunocompromised individuals and elderly people. Those with underlying diseases are also at increased risk for severe illness from coronaviruses causing MERS and COVID-19. There are seasonal peaks in winter for the common cold coronaviruses, and co-infections with other respiratory viruses are common.

Transmission

SARS-CoV-1, SARS-CoV-2 and MERS-CoV coronaviruses are zoonotic in origin, followed primarily by person-to-person respiratory spread. Hospitals and close contact settings with poor ventilation have been important in propagating outbreaks. Health-care workers are especially at risk of secondary infection. The relative infectivity of SARS-CoV-1 and SARS-CoV-2 is considered high.

Inhalation of the virus from aerosolized faecal droplets has been linked to one outbreak of SARS and another of COVID-19. The droplets were generated from untreated sewage in faulty plumbing systems of a residential building where an infected individual was present. These outbreaks were unique, with design characteristics of the building sewage and air handling systems contributing to possible but unproven transmission. Outbreaks transmitted through food or water have not been documented. There is greater potential for environmental transmission via surfaces and fomites as potential sources of hand contamination, particularly where there is inadequate hygiene and poor sanitation.

Detection methods

Culture methods for coronaviruses are available; however, due to the difficulty of culturing coronaviruses, molecular methods have been more widely used for detection in water (see [section 7.4](#)).

Source and occurrence

Nucleic acids from SARS-CoV-1, SARS-CoV-2 and MERS-CoV have been detected in urine and faeces. Despite widespread detection of SARS-CoV-2 RNA in sewage, there is no evidence that infectious virus is present in faecal wastes and environmental waters or causes infection by this route. The persistence of infectious coronaviruses in water is considered to be short to moderate (up to 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential but unidentified risks from coronaviruses should include application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Evidence suggests chlorine would be an effective drinking-water disinfectant to inactivate coronaviruses. As coronaviruses are not regarded as being waterborne, indicator organisms are not relevant.

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Enteroviruses

Significance in drinking-water

There is moderate evidence linking transmission of enteroviruses with drinking-water based on a limited number of enterovirus outbreaks where unsafe drinking-water supplies were identified as risk factors.

General description

Enteroviruses are among the smallest known viruses (20–30 nm in diameter), and consist of a single-stranded RNA genome in a non-enveloped icosahedral capsid. There are 12 species of enteroviruses (A to L), which include over 100 serotypes that infect humans. Among these are poliovirus, coxsackievirus and echovirus, which retain their old taxonomic designations as common names. Most enterovirus strains are human specific, but some species infect animals other than humans. None of these non-human-specific forms are known to infect humans.

Human health effects

Enterovirus infection originates in the gastrointestinal tract. It can cause a broad spectrum of diseases ranging from mild illness characterized by cough, rash and fever to severe effects ranging from neurological illness, including aseptic meningitis, encephalitis and poliomyelitis, and respiratory illnesses. Aseptic meningitis appears to be the most common severe clinical manifestation associated with enterovirus infections in children and adults, especially in high-income countries. It has been estimated (including in a 2017 publication) that mortality rates from severe clinical infections can be as high as 10%. The prevalence of asymptomatic infections can be extremely high for enteroviruses. Reinfection with the same virus often results in milder symptoms or subclinical or asymptomatic infection.

Epidemiology

Enteroviruses are a common but underreported cause of human infections globally. With the exception of poliomyelitis, the disease burden is poorly estimated. Young children are at most risk for infections. Enteroviruses exhibit epidemic and endemic patterns of incidence. Seasonally, infections from enteroviruses have tended to occur in summer in countries in the temperate areas of the northern hemisphere and year-round in tropical areas. Worldwide, *Enterovirus* B, led by Echovirus 30, was the most detected species with *Enterovirus* C more often detected in Africa, and *Enterovirus* A in Asia. The global prevalence of poliovirus (*Enterovirus* C) has decreased due to worldwide eradication efforts through vaccination, but wild polioviruses continue to be detected in some parts of Africa and the eastern Mediterranean. Novel enteroviruses (e.g. D68, enterovirus 71) continue to emerge and may become new public health threats.

Transmission

Enteroviruses are transmitted by the faecal–oral route and by inhalation of airborne viruses. The relative importance of the different modes of transmission varies with the particular enterovirus and environmental settings. The faecal–oral pathway through person-to-person contact predominates in areas with poor sanitary conditions, while transmission through inhalation predominates in higher-income settings. Oral–oral spread by contact with contaminated saliva may be an exposure pathway. Transmission through contaminated fomites and soil is considered important. Waterborne transmission has been confirmed from two recreational water outbreaks. Drinking-water represents a likely source of enterovirus infection based on three outbreaks associated with drinking-water.

The infective doses have not been determined or vary for most species of enteroviruses, but they may be as low as one infectious particle. Most enteroviruses have incubation periods of 3–10 days on average. Viruses tend to be shed in stools for 10 days to 2 months after exposure. Some immunity is conferred following infection, with symptoms possibly reduced or subclinical post reinfection.

Detection methods

A standard method using culture and molecular approaches is available for enteroviruses (see [section 7.4](#)). However, some species are not readily isolated by cell culture methods (e.g. enterovirus 5). Combining cell culture infectivity and molecular methods may improve the detection of infectious enteroviruses.

Source and occurrence

Enteroviruses are found in a broad range of water and wastewater environments. They have been often detected in drinking-water sources and occasionally in inadequately treated drinking-water. Prevalence is influenced by the presence of human faeces, particularly sewage (untreated or partially treated). Persistence in water is likely to vary among species, but in general can be considered long (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from enteroviruses should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of enteroviruses by drinking-water treatment varies depending on the process. Enteroviruses have low to moderate resistance to disinfection with chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of enteroviruses relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their greater environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of human enteroviruses (see [section 11.6](#)).

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Filoviruses (Ebola and Marburg viruses)

Significance in drinking-water

Filoviruses are of low health significance in drinking-water. There is currently no evidence linking transmission with drinking-water. Furthermore, evidence suggests that drinking-water is an unlikely route of transmission.

General description

Filoviruses are filamentous (50 by 800–1000 nm), and consist of a single-stranded RNA genome in an enveloped nucleocapsid. The Filoviridae family has five genera: Cuevavirus (species Lloviu cuevavirus, not causing human disease); Ebolavirus (species Bundibugyo, Reston, Sudan, Tai Forest and Zaire); Marburgvirus (species Marburg Marburgvirus); Striavirus (species Xilang striavirus); and Thamnovirus (species Huangjiao thamnovirus). Hosts are mammals, including humans, bats and non-human primates.

Human health effects

Ebola and Marburg viruses cause severe haemorrhagic fevers. Initial symptoms are nonspecific, including fever, chills, rash, fatigue, vomiting and diarrhoea, progressing to respiratory, renal and hepatic involvement, massive fluid and electrolyte loss through high-volume diarrhoea, multi-organ failure and mucosal haemorrhage. Case fatality rates in outbreaks caused by filoviruses vary, but can be as high as approximately 90% (according to a number of publications including a 2016 report). Asymptomatic infections of Ebola virus have been documented.

Epidemiology

Filovirus disease occurs mostly in sporadic outbreaks, predominantly in Africa, with cases exported to other regions. Transmission within hospitals has been important in outbreaks, and health-care workers are at a demonstrated risk of infection. Marburg virus was identified from two large laboratory outbreaks in Europe associated with transmission from non-human African primates.

Transmission

Filoviruses are zoonotic, acquired by humans from animal reservoirs, potentially through bushmeat consumption or other contact with infected animal tissues or body fluids. It then spreads from person to person through mucosal surface contact or through broken skin with infected body fluids, particularly blood and other body fluids contaminated with blood. The significance of environmental sources in the potential spread of filoviruses is unclear; outbreaks spread through food, water, surfaces or fomites have not been documented. The relative infectivity of filoviruses in humans is considered high.

Detection methods

Ebola and Marburg viruses can be detected by cell culture, but this should be undertaken only in high-containment laboratories. Antibody and molecular methods are also available (see [section 7.4](#)).

Source and occurrence

Filoviruses have not been detected in drinking-water sources or wastewater, although one study detected Ebola genetic material in wastewater. Faecal contamination of drinking-water sources is not expected since Ebola viruses are not likely to be in faeces. Their persistence in water is expected to be short (up to 1 week at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential but unidentified risks from filoviruses should include application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Evidence suggests chlorine would be an effective drinking-water disinfectant to inactivate filoviruses. As filoviruses are not regarded as being waterborne, indicator organisms are not relevant.

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Hepatitis A virus

Significance in drinking-water

Hepatitis A virus (HAV), also known as *Hepatovirus A*, is an important waterborne pathogen, with strong evidence from many outbreaks linking transmission to drinking-water. A common risk associated with outbreaks is consumption of untreated and vulnerable groundwater.

General description

HAV consists of a single-stranded RNA genome in a non-enveloped, icosahedral capsid that is 27–35 nm in diameter. HAV includes six genotypes with genotypes I–III including sub-genotypes A and B associated with human infection. Genotypes IV–VI are associated with infections in simians. There is no evidence of cross-species transmission.

Human health effects

HAV causes the liver disease hepatitis A, commonly known as infectious hepatitis. Hepatitis A is clinically indistinguishable from other forms of hepatitis. It is characterized by a sudden onset of fever, malaise, nausea, anorexia, abdominal discomfort and eventually jaundice (yellowing of the skin and white of the eyes). Itchiness, dark urine and pale stools may also occur. Symptoms of hepatitis A develop, on average, around 4 weeks after infection. In most cases, hepatitis A resolves without treatment, but it can sometimes progress to life-threatening acute liver failure. The severity of illness increases with age and those with pre-existing liver problems. The majority of young children experience asymptomatic infection or mild symptoms, while older children and adults often experience symptomatic disease. Mortality is rare and is higher in those over 40 years of age.

Epidemiology

The highest rates of acute infections and deaths occur in low- and lower-middle-income countries. In such settings, endemicity is high, and children are often infected at a very early age. In high-income countries, childhood infections are rare, so older children and adults remain susceptible unless vaccinated. Circulation of HAV genotypes and subtypes varies geographically, with genotype I being the most prevalent

worldwide, subgenotype IA being more common than IB, and subgenotype IIIA increasingly circulating in some Asian countries and causing outbreaks in Europe.

Transmission

Transmission is primarily by the faecal–oral route, but can also occur by sexual, (drug) injection or parenteral routes including transfusions. Person-to-person spread is probably the most common route of transmission, but contaminated food and water are important sources of infection. Travel from low to intermediate endemic regions (areas with good sanitation and water supplies) to high endemic areas without immunization provides a high risk of infection. HAV is considered to be highly infective, with estimated doses associated with 50% infection rates of tissue culture cells (TCID₅₀) of 10–100. The incubation period for HAV infection is 28–30 days on average. Shedding in faeces can begin several weeks before symptoms occur, reaching its peak concurrently with gastrointestinal symptoms and tapering off as the patient shows symptoms of jaundice. Infection with HAV confers lifelong immunity.

Detection methods

HAV cannot be readily detected by cell culture, but there are molecular methods (see [section 7.4](#)). Combining cell culture infectivity and molecular methods may improve the detection of infectious HAV.

Source and occurrence

HAV is found in a broad range of water and wastewater environments. It has often been found in surface water and occasionally in groundwater and inadequately treated drinking-water. HAV is reported to be more prevalent in water in low-income countries, in particular, in Africa and the eastern Mediterranean, compared to in high-income countries. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). HAV has long persistence in water (over 1 month at 20 °C), with survival likely influenced by the composition of other organisms.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential risk from HAV should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HAV by drinking-water treatment varies depending on the process. HAV is moderately resistant to disinfection with chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of HAV relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their greater environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of its absence in drinking-water. However, the presence of *E. coli* is a suitable indicator for the possible presence of HAV (see [section 11.6](#)).

Bibliography

WHO (2025) *Hepatitis A virus. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Hepatitis E virus

Significance in drinking-water

Hepatitis E virus (HEV) is a significant waterborne pathogen with strong evidence from large outbreaks in low- and middle-income areas linking transmission to drinking-water. These outbreaks occurred in areas with poor sanitation, indicating faecal contamination of drinking-water.

General description

HEV consists of a single-stranded RNA genome in a non-enveloped icosahedral capsid with a diameter of 27–34 nm. HEV including the strains associated with waterborne disease and previously known as *Orthohepevirus A*, is now named *Paslahepevirus balayani*. HEV is subdivided into eight genotypes. Genotypes 1 and 2 have been found only in humans, genotypes 3, 4 and 7 in humans and animal species such as pigs, wild boars, deer, rats and camels, and potential zoonotic transmission of genotypes 5 and 8.

Human health effects

HEV causes symptoms that are clinically indistinguishable from hepatitis from other causes. Symptoms usually resolve, varying among individuals, but often include anorexia, jaundice, liver enlargement, abdominal pain, nausea and fever. More severe outcomes associated with HEV infection, especially in high-risk populations (e.g. pre-existing liver disease, transplant patients), include fulminant hepatic failure, a range of neurological syndromes, renal injury, pancreatitis and haematological disorders, but also death. Mortality rates are usually low, but it has been reported (including in a 2023 WHO fact sheet), that they can reach 20–25% in pregnant women infected in the third trimester. There are also higher mortality rates in high-risk populations. Asymptomatic infection is common.

Epidemiology

HEV has a characteristic distribution. HEV is endemic and causes clinical diseases in low- and middle-income countries, where genotypes 1 and 2 are more prevalent. Outbreaks in such settings have been associated with flooding and monsoon seasons. First infections are typically seen in young adults. Large outbreaks have also occurred in areas of conflict and in displaced populations. Although seroprevalence can be high, clinical cases and outbreaks are rare in higher-income countries. In these settings, no prominent HEV seasonality has been observed, and genotypes 3 and 4 are more common in those over 50 years of age and males are most affected.

Transmission

Transmission is primarily by the faecal–oral route, with very large waterborne outbreaks involving thousands of cases due to genotypes 1 and 2 documented. Food is an important source of exposure of genotypes 3 and 4. Direct contact with infected animals may also serve as a route of exposure for some genotypes but not genotypes 1 and 2. Person-to-person transmission is not common, but transmission associated with the vertical (mother-to-child) route (particularly genotypes 1 and 2), transfusions and organ transplants (particularly genotypes 3 and 4) is possible. The relative infectivity for HEV is unknown, but similar to other enteric viruses, it is expected to be

high (infective doses of $1-10^2$ organisms). The average incubation period is 40 days. HEV is excreted in faeces from approximately 1 week before the onset of symptoms and usually persists for 2–4 weeks. Immunity following infection was initially thought to be lifetime, but may wane with time.

Detection methods

HEV cannot be readily detected by cell culture, but there are molecular methods (see [section 7.4](#)). Combining cell culture infectivity and molecular methods may improve the detection of infectious HEV.

Source and occurrence

There are limited data on the occurrence of HEV, but it has been detected in water and wastewater environments including surface water, groundwater and inadequately treated drinking-water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated) and possibly animal faecal material. There are no data on the persistence of the virus in water, but similar to other non-enveloped viruses, it is likely to be long (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential risk from HEV should include protection of source waters from contamination by human and animal faecal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HEV by drinking-water treatment varies depending on the process. HEV has a low to moderate resistance to disinfection with chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of HEV relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and its likely environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of its absence in drinking-water. However, the presence of *E. coli* is a suitable indicator for the possible presence of HEV (see [section 11.6](#)).

Bibliography

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Noroviruses

Significance in drinking-water

Noroviruses are major waterborne pathogens, with strong evidence from numerous outbreaks linking transmission to drinking-water. Common risk factors include sewage contamination and inadequate or failed disinfection.

General description

Noroviruses (previously known as Norwalk viruses) consist of a single-stranded RNA genome in a non-enveloped icosahedral capsid, and are 30–35 nm in size. There are

at least seven known norovirus genogroups, GI to GVII, further subdivided into 32 genotypes. Human infection is most commonly caused by viruses belonging to GII, especially genotype 4 (GII.4), as well as GI, and to a very limited extent, GIV. Humans are the only established reservoir for human infective norovirus strains. Human norovirus genogroups have also been detected in animal hosts, such as mice, cows, dogs and pigs, suggesting a potential for zoonotic transmission.

Human health effects

Symptoms of norovirus infection typically include an abrupt onset of nausea, vomiting (often projectile) and diarrhoea. In healthy individuals, the illness is usually self-limiting, with symptoms lasting for 24–72 hours. Dehydration is the most common complication. Some cases may become severe and require prompt medical treatment. Asymptomatic norovirus prevalence varies depending on settings and countries, but can be common. Asymptomatic individuals may play an important role in norovirus outbreaks.

Epidemiology

Noroviruses are a frequent cause of diarrhoeal disease, contributing to significant disease burdens in all income settings. Severe cases and deaths have been reported primarily in people over 65 years in high-income countries, and in young children in low- and middle-income countries. Immunocompromised individuals are also more susceptible to infection, with more prolonged and severe symptoms. The majority of reported outbreaks are associated with semi-closed and crowded settings such as long-term health-care facilities, schools, hospitals, child day-care centres, cruise ships and hotels or other lodging facilities. In temperate regions, norovirus infection is a seasonal illness, with the peak of cases and outbreaks occurring in winter months. In tropical climates and in low-income settings, there is some evidence of year-round circulation, but patterns are less clear. GII.4 is the predominant outbreak strain globally, with new clusters or variants of the strain emerging every 2–5 years, often resulting in pandemics such as the 2012 pandemic caused by the GII.4 Sydney strain. Since 2012, non-GII.4 strains have emerged and become predominant in several countries.

Transmission

Noroviruses are transmitted primarily by the faecal–oral route. Person-to-person contact is the most important means of transmission, with other sources being contaminated food, water and fomites. Drinking-water such as tap water, well and spring water have been identified as sources of infection. Vomitus–oral transmission can also occur through aerosolization, followed by ingestion or environmental contamination.

The relative infectivity of noroviruses is considered to be high. The dose associated with a 50% infection rate (ID_{50}) is estimated to be as low as 10–20 virus particles. The median incubation period for symptomatic GI and GII infections is estimated to be as low as 1.2 days. Norovirus spreads most easily during the acute stage of disease, and may be shed for 2–3 weeks after symptoms resolve. Asymptomatically infected individuals have also been shown to shed pathogens in similar concentrations as symptomatic individuals, although based on studies in health-care settings, asymptomatic individuals do not seem to transmit noroviruses as commonly. Immu-

nity to noroviruses post infection is relatively short, and reinfections are common. A subpopulation of some individuals is resistant to infection by several norovirus genotypes including GII.4.

Detection methods

Noroviruses are not readily detected by cell culture, but standard molecular methods are available (see [section 7.4](#)). Advances in culturing noroviruses in human intestinal enteroids may provide a useful detection method. Combining cell culture infectivity and molecular methods may improve the detection of infectious noroviruses.

Source and occurrence

Noroviruses are found in a broad range of water and wastewater environments. They have been found often in surface water and occasionally in groundwater and drinking-water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). Noroviruses can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential risk from noroviruses should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of noroviruses by drinking-water treatment varies depending on the process. Based on studies with non-human noroviruses, it is likely that disinfection processes, including chlorination, will be effective for inactivating human strains of norovirus. Due to the smaller size of noroviruses relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their greater environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of noroviruses (see [section 11.6](#)).

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WHO (2025) *Noroviruses. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Orthomyxoviruses (Influenza viruses)

Significance in drinking-water

Influenza viruses are of low health significance in drinking-water. There is no evidence linking transmission with drinking-water.

General description

Influenza viruses have a single-stranded RNA genome in an enveloped nucleocapsid with a diameter of 80–120 nm. The Orthomyxoviridae family has seven genera, three containing species of human health importance: influenza A, influenza B and influenza C viruses. Influenza A viruses infect humans, birds and a range of other

mammals. Birds and swine are important to human transmission. Influenza B viruses infect mostly humans, and influenza C viruses infect humans, swine and dogs. Viral subtypes are designated by their hemagglutinin (H) and neuraminidase (N) types (e.g. H1N1). Influenza B viruses are not divided into subtypes, but two distinct lineages circulate among humans.

Human health effects

Influenza viruses cause a range of respiratory illnesses in humans, from mild to fatal. Seasonal influenza caused by influenza A and influenza B is characterized by a sudden onset of fever, cough, headache, muscle and joint pain, severe malaise, sore throat and a runny nose. Gastrointestinal symptoms can also occur, particularly in children, but there is no conclusive evidence that these symptoms are due to intestinal infection and they may be due to inflammatory responses in the gut. The prevalence of asymptomatic influenza infections (including based on a 2022 review) can be greater than 50%. Secondary bacterial pneumonia is a frequent complication following influenza infection.

Epidemiology

The burden of seasonal influenza is substantial. Children, adults greater than 65 years of age, people with chronic illness and immunocompromised people are at higher risk for complicated illness or death. The epidemiology of human influenza viruses can be broadly divided into epidemic and pandemic spread. Epidemics typically occur in annual seasonal cycles that peak in winter months in temperate climates. Influenza can occur throughout the year in tropical regions, causing more irregular outbreaks. Although reported sporadically, the transmission of avian and swine influenza viruses from animals to humans is uncommon.

Transmission

Seasonal influenza viruses are primarily transmitted by the respiratory route, spreading from person to person; droplets and droplet nuclei play a role in transmission. Transmission can also take place by hand-to-mucous membrane contact, which may be either direct hand-to-hand contact or possibly by indirect hand-to-hand contact via surfaces and fomites. Outbreak spread through food or water has not been documented. Zoonotic transmission can be either animal-to-human followed by limited person-to-person transmission, or by re-assortment of animal and human viruses (antigenic shift), producing new strains that subsequently spread from person to person. The incubation period is short, usually 1–5 days. The relative infectivity of influenza viruses is considered high, although data are limited.

Detection methods

Influenza viruses can be detected by mammalian cell culture or in embryonated chicken eggs; however, molecular methods have been more widely used for detection in water (see [section 7.4](#)).

Source and occurrence

Influenza viruses or nucleic acids most likely from avian and swine sources have been detected in surface waters, groundwaters or farm waters. Influenza virus nucleic acids

have been detected in surface water during human outbreaks. Prevalence is influenced by the presence of faecal waste. Influenza viruses are likely to persist for long periods in surface water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against influenza viruses. These include protection of source waters from contamination by human and animal faecal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of influenza viruses by drinking-water treatment varies depending on the process. Studies of susceptibility of influenza viruses to drinking-water disinfectants are limited, but suggest chlorine is effective in inactivating influenza viruses. Due to the smaller size of influenza viruses relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their expected longer environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of influenza viruses (see [section 11.6](#)).

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WHO (2025) *Orthomyxoviridae (Influenza viruses). Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Parechoviruses

Significance in drinking-water

Human parechoviruses (HPeVs) are of low health significance in drinking-water. Although HPeVs can be transmitted by the faecal–oral route and have been detected in sewage and surface water, there is no evidence linking transmission with drinking-water.

General description

Human parechoviruses (HPeVs) are small (28 nm in diameter), and consist of a single-stranded RNA genome in a non-enveloped capsid. There are four species of Parechovirus: Parechovirus A (formerly named Human Parechovirus), Parechovirus B (formerly named Ljungan virus), Parechovirus C (Sebokele virus) and Parechovirus D (Ferret parechovirus). Parechovirus A consists of 19 types, HPeV-1 to HPeV-19, based on phylogenetic analysis of amino acid sequences of the viral capsid protein. Humans are the only known reservoir for the human disease-causing strains of HPeVs.

Human health effects

Infection can cause mild to severe illness, with symptoms including diarrhoea and flu-like disease as well as severe diseases such as meningoencephalitis, sepsis and flaccid paralysis. Clinical manifestation of HPeV infection varies between HPeV genotypes. HPeV-3 causes severe disease in neonates and infants younger than 3 months of age. Most individuals infected with HPeVs are asymptomatic.

Epidemiology

HPeV infections are globally endemic throughout the year. In temperate climates, HPeV infections peak during the summer and autumn. Biannual cycles of infection have also been observed. Children under 2 years of age are at the greatest risk of developing symptomatic HPeV infection.

Transmission

Transmission is through the faecal–oral and respiratory routes. HPeVs can be spread by person-to-person contact, airborne transmission, fomites and possibly contaminated food and water. The infective dose of HPeV is unknown.

Detection methods

Cell culture and molecular methods are available for HPeVs (see [section 7.4](#)).

Source and occurrence

HPeVs have been commonly detected in surface water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). There are no data on persistence, but similar to other non-enveloped viruses, it is likely to be long.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against HPeVs. These include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of HPeVs by drinking-water treatment varies depending on the process. HPeVs are likely to be moderately resistant to disinfection with chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of HPeVs relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their expected environmental persistence, *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of HPeVs (see [section 11.6](#)).

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Rotaviruses

Significance in drinking-water

Rotaviruses are major waterborne pathogens, with strong evidence from several waterborne outbreaks linking transmission to drinking-water. A common risk factor is inadequate disinfection of drinking-water supplies.

General description

Rotaviruses consist of a double-stranded RNA genome in a non-enveloped icosahedral capsid that is 50–65 nm in diameter. This capsid is surrounded by a double-

layered shell, giving the virus the appearance of a wheel – hence the name rotavirus. The diameter of the entire virus is about 100 nm. There are nine species of rotavirus designated A to D and F to J, each of which consists of a number of subgroups. Species A, B, C and H can infect humans, with species A being the most commonly diagnosed cause of rotavirus infections. Rotavirus A is further divided into serotypes using characteristics of the outer capsid proteins VP4 and VP7. Several unique types of VP4 (P types) and VP7 (G types) are currently known, demonstrating great antigenic diversity. Humans are the only established reservoir for human infective rotavirus strains. There is some evidence that animal rotavirus strains can infect humans, suggesting a potential for zoonotic transmission.

Human health effects

Acute human rotavirus infection has an abrupt onset of severe watery diarrhoea, with fever, abdominal pain and vomiting. Dehydration and metabolic acidosis may develop, and the outcome may be fatal if the infection is not treated appropriately, especially in infants and underweight children. Adults tend to have similar but milder symptoms compared to children. Infections may also be asymptomatic, especially in older children and adults.

Epidemiology

The burden of diarrhoeal disease caused by rotavirus among infants and young children is extremely high, with nearly every child infected by 5 years of age. It is also the cause of a substantial proportion of diarrhoeal disease mortality. Differences in transmission patterns, health-care access, childhood co-infections and co-morbidities drive substantial differences in the rotavirus disease burden between low- and high-income settings. The rotavirus disease burden is greater in low-income than in high-income settings, and most rotavirus deaths occur in low-income settings. The relative importance of rotavirus in contributing to diarrhoeal disease is expected to decrease due to the use of rotavirus vaccines. In low-income, typically tropical settings, there is usually a background of year-round circulation punctuated by episodes of intense transmission, but some studies suggest there is some seasonality with peaks during cooler and drier months. In higher-income, temperate regions, illnesses peak in winter and spring months.

Transmission

Human rotaviruses are transmitted primarily by the faecal–oral route. Person-to-person contact is likely the most important source of transmission, with other sources being contaminated food, drinking-water or fomites. Airborne transmission of aerosolized virus is also possible. Rotaviruses are highly infective, with an estimated dose associated 50% infection rate (ID_{50}) of 6 focus-forming units. The incubation period can range from 24 to 48 hours. Rotaviruses are excreted from 2 days before and up to 10 days after symptomatic illness. Rotavirus reinfections are quite common throughout life, although with each reinfection, the illness tends to become milder.

Detection methods

Wild-type strains of rotavirus A are not readily detected by cell culture, but there are molecular methods to determine the presence and type of rotavirus (see [section 7.4](#)).

A combined cell culture and molecular method has been used to detect infectious rotaviruses.

Source and occurrence

Rotaviruses are found in a broad range of water and wastewater environments, and have been often found in surface water. Rotaviruses have also been detected in ground-water and drinking-water. Prevalence is influenced by the presence of human faeces including sewage (untreated or partially treated). They can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential risk from rotaviruses should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Reduction of rotaviruses by drinking-water treatment varies depending on the process. Rotaviruses are moderately resistant to free chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Due to the smaller size of rotaviruses relative to *E. coli*, resulting in lower removal by treatment processes such as filtration, and their greater environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of human rotaviruses (see [section 11.6](#)).

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Sapoviruses

Significance in drinking-water

There is moderate evidence linking transmission of sapoviruses with drinking-water from a limited number of outbreaks associated with sapoviruses, in conjunction with other pathogens. Risk factors include lack of treatment and sewage contamination through failures in drinking-water distribution systems.

General description

Sapoviruses consist of a single-stranded RNA genome in a non-enveloped capsid that is 30–38 nm in diameter. Sapoviruses include 19 different genogroups, with viruses infecting humans belonging to genogroups I, II, IV and V. These genogroups are further subdivided into 19 genotypes: GI and GII each comprise 8 genotypes, GIV has only 1 genotype and GV has 2 genotypes. Humans are the only established reservoir for human infective strains of *Sapovirus*. Other genogroups have been shown to infect animals such as pigs, mink, dogs, sea lions and bats. Phylogenetic analysis showing that animals have contributed to sapovirus epidemiology suggests a potential for zoonotic transmission.

Human health effects

Sapoviruses cause gastrointestinal illness that cannot be clinically distinguished from illness caused by noroviruses. Symptoms are typically mild to moderate, with abrupt onset of primary symptoms including nausea, vomiting (often projectile) and diarrhoea. Dehydration is the most common complication. In most cases, symptoms resolve within 1 week. Deaths from *Sapovirus* infections are rare, but have been reported in elderly patients. A number of small-scale studies suggest there is also a high level of asymptomatic carriage among healthy individuals.

Epidemiology

Sapoviruses affect people of all ages worldwide, but sporadic cases occur more frequently in young children. Outbreaks are most common among elderly people living in long-term care facilities, and children at day-care centres. Research into other susceptible groups such as immunocompromised individuals is scarce, but such groups may be at higher risk of severe infection and prolonged viral shedding. Sapovirus infections show a marked seasonality in temperate climates, with increased activity usually peaking in colder months.

Transmission

Sapoviruses are transmitted primarily by the faecal–oral route. Person-to-person contact is the most important means of transmission, with other sources being contaminated food and fomites and possibly water. Vomitus–oral transmission can also occur through aerosolization, followed by ingestion or environmental contamination.

The relative infectivity of sapoviruses is unknown, but similar to other enteric viruses, it is likely to be high (infective doses of $1\text{--}10^2$ organisms). The average incubation period for symptomatic infection is 1.7 days with viral shedding decreasing gradually after the onset of symptoms. Reinfection and recurrence of illness can occur, suggesting that immunity is genotype specific, with little cross-protection even within genogroups.

Detection methods

Cell culture methods for detecting sapoviruses are not available, but molecular methods are available (see [section 7.4](#)). Combining cell culture infectivity and molecular methods may improve the detection of infectious sapoviruses.

Source and occurrence

Sapoviruses are found in a broad range of water and wastewater environments. However, there are limited data on the presence of sapoviruses in surface water and drinking-water. They have not been detected in groundwater, although data are limited. There are no data on persistence of sapoviruses, but it is likely to be similar to noroviruses, which can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential risk from sapoviruses should include protection of source waters from contamination by human faecal waste, application of adequate water treatment processes, and protection of drinking-

water quality during storage and distribution. Reduction of sapoviruses by drinking-water treatment varies depending on the process. Similar to noroviruses, it is likely that disinfection processes, including chlorination will be effective for inactivating sapoviruses. Due to the smaller size of sapoviruses relative to *E.coli*, resulting in lower removal by treatment processes such as filtration, and their likely environmental persistence, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the potential presence of human sapoviruses (see [section 11.6](#)).

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11.3 Protozoan pathogens

Most pathogenic protozoa with moderate to strong evidence of waterborne transmission are enteric pathogens. *Cryptosporidium*, *Giardia* and *Toxoplasma* are zoonotic with important animal sources. In the case of *Toxoplasma*, humans are intermediate hosts with infection not leading to excretion of oocysts. There are also two free-living protozoa – *Naegleria fowleri* and *Acanthamoeba* – that can grow in water environments, including in biofilms in drinking-water distribution systems. Enteric protozoa are transmitted by ingestion of contaminated drinking-water, while *Acanthamoeba* is transmitted primarily by contact where contaminated drinking-water is used for cleaning or storing contact lenses and *N. fowleri* when contaminated water enters nasal passages. For further information, including an overview of treatment considerations for protozoa, see [section 7.2.2](#).

Acanthamoeba

Significance in drinking-water

Acanthamoeba are not transmitted through the ingestion of drinking-water. However, there is strong evidence for water-based transmission of *Acanthamoeba* leading to eye infections through the use of non-sterile water including drinking-water to clean or store contact lenses. In addition, *Acanthamoeba* can harbour bacterial pathogens such as *Legionella* and potentially enteric viruses, and may play a role in their survival and the pathogenicity of *Legionella* in drinking-water.

General description

Acanthamoeba are free-living amoebae commonly found in fresh water and soil. Their life cycle includes two stages: an actively growing trophozoite (12–35 µm) and a dormant cyst (5–20 µm) that forms and survives in response to adverse environmental conditions. There are over 20 species of *Acanthamoeba*. Historically, they were categorized into three groups based on size and cyst morphology. More recently, *Acanthamoeba* have been divided into 20 genotypes based on DNA sequencing. Genotype T4, which includes *A. castellani*, is the most common disease-causing genotype.

Human health effects

Acanthamoeba infections are commonly asymptomatic, but *Acanthamoeba* can cause three types of disease. In immunocompetent people, the most common disease is an eye infection known as acanthamoebic keratitis (AK). Symptoms of AK include inflammation of the cornea, blurred vision, ulceration and blindness. Infections of the central nervous system known as granulomatous amoebic encephalitis (GAE) and cutaneous (skin) infections can occur in immune-deficient individuals. Clinical symptoms of GAE are similar to meningitis and include headache, fever, lethargy, stiff neck, confusion and irritability. GAE can lead to death in some instances. The lack of distinguishing features of GAE makes diagnosis difficult. Symptoms of cutaneous infections include skin lesions and nodules, or sinus lesions and sinusitis.

Epidemiology

Although there is no global estimate for AK, it is likely widespread due to the common and increasing use of contact lenses and the ubiquity of *Acanthamoeba* in the environment. The majority of cases of AK occur in wearers of soft contact lenses. GAE and cutaneous infection are predominantly identified in immunocompromised or debilitated individuals and are extremely rare in immunocompetent people.

Transmission

The most common route of transmission for AK in wearers of contact lenses is the use of contaminated water for washing or storing contact lenses. Swimming, using a spa or showering while wearing contact lenses also present a risk of infection. Infections can also occur following eye injury or surgery. Ingestion of drinking-water is not a route of transmission. Person-to-person and zoonotic transmissions of *Acanthamoeba* have not been reported. GAE and cutaneous infections are not considered to be waterborne. It is likely that *Acanthamoeba* enter the bloodstream from other sites of colonization such as the nose, lungs or skin lesions following injury or trauma. The human infective doses associated with any forms of disease caused by *Acanthamoeba* are unknown. The incubation period for AK is unknown, but is considered to be several days.

Detection methods

Acanthamoeba can be detected using culture-based methods and subsequently classified using molecular methods (see [section 7.4](#)).

Source and occurrence

Acanthamoeba are very common soil and water organisms. They have been detected in drinking-water supplies, swimming pools, bottled water, seawater and fresh water. *Acanthamoeba* trophozoites can multiply in water and soil; cysts can form and persist for long periods (over 1 month at 20 °C). *Acanthamoeba* can survive and grow in biofilms in drinking-water distribution systems, including premise plumbing.

Prevention and management

The most effective control measure to remove the risk of AK arising from drinking-water is to prevent exposure. Drinking-water should not be used for cleaning or storing contact lenses. Within a WSP (see [chapter 4](#)), control measures to manage

potential risks from *Acanthamoeba* should include application of adequate water treatment processes, protection of drinking-water quality during storage and distribution, and minimizing their growth in biofilms in distribution systems and building water systems. Filtration systems designed to remove other protozoa such as *Cryptosporidium* are likely to be effective in removing the larger *Acanthamoeba* cysts and trophozoites. *Acanthamoeba* trophozoites are moderately resistant to oxidizing disinfectants such as chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. Cysts are highly resistant to chlorine and relatively resistant to UV light. As *Acanthamoeba* are non-faecal, environmental organisms that can survive and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water (see [section 11.6](#)).

Bibliography

WHO (2025) *Acanthamoeba*. Background document for the WHO Guidelines for drinking-water quality. Geneva, World Health Organization.

Balantioides coli

Significance in drinking-water

Balantioides coli is of low health significance in drinking-water. Although *B. coli* is transmitted by the faecal–oral route and has been detected in drinking-water, the evidence linking transmission with drinking-water is weak. Only one waterborne outbreak has been reported, occurring in 1971.

General description

B. coli is a protozoan parasite. It exists in two forms: trophozoites of about 50 µm by 150 µm that are covered with cilia for motility, and environmentally stable cysts of approximately 40–70 µm. Following ingestion of cysts, excystation occurs in the small intestine and the trophozoites colonize the large intestine, where they replicate and form new cysts that are excreted in the faeces. It is the only ciliate that infects humans.

Human health effects

Human infections of *B. coli* are relatively rare and most are asymptomatic, but gastrointestinal illness can occur. Trophozoites can destroy host cells as they multiply. Clinical symptoms may include diarrhoea, nausea, vomiting, headache and anorexia. Illness is typically limited to several days with complete recovery, but some illnesses develop into chronic infections with long duration. Infrequently, and especially in immunosuppressed hosts or those with other health conditions, illness may progress to dysentery, with possible further spread to other organs.

Epidemiology

B. coli infection and illness in humans occurs worldwide, with higher rates in immunocompromised individuals. Global prevalence was estimated (in a 2017 review) to be about 1%, but could have been as high as 30% in tropical regions where there was high occupational exposures to pigs, especially where sanitation and hygiene were inadequate.

Transmission

B. coli transmission is by the faecal–oral route, from contact with infectious persons or animals, primarily swine or by consumption of contaminated water or food. The only reported waterborne outbreak of balantidiasis occurred from ingestion of water contaminated with swine faeces after a typhoon. Infectivity in humans is unknown.

Detection methods

Microscopic and molecular methods are available for *B. coli*. Using microscopy, there is a possibility of confusion with other ciliates of the *Balantidium* genus that could be present in the water sample.

Source and occurrence

There are limited data on the occurrence of *B. coli* cysts, but they have been detected in water and wastewater environments including surface water and inadequately managed drinking-water. Prevalence is influenced by the presence of faecal material from humans including sewage (untreated or partially treated) and other animals, particularly pigs. *B. coli* can persist in water for long periods (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against *B. coli*. These include protection of source water from contamination by human and animal faecal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. *B. coli* cysts are likely removed by conventional filtration methods because of their relatively large size (larger than *Cryptosporidium* oocysts). Based on data available for other protozoa, they are likely highly resistant to chlorine. Due to the environmental persistence of *B. coli* cysts and their probable resistance to chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *B. coli*.

Bibliography

WHO (2025) *Balantoides coli*. Background document for the WHO Guidelines for drinking-water quality. Geneva, World Health Organization.

Blastocystis

Significance in drinking-water

Blastocystis is of low health significance in drinking-water. Although *Blastocystis* is transmitted by the faecal–oral route and has been detected in drinking-water, the evidence linking transmission with drinking-water is weak. In addition, there is uncertainty about the pathogenicity of *Blastocystis*.

General description

Blastocystis are protozoan parasites known since the early 1900s, but knowledge gaps remain about their human pathogenicity. There are at least 17 different subtypes, of which 10 are zoonotic. Subtypes 1–4 are the most common subtypes reported in

humans. The parasite varies in size between 2 and 40 µm and lives in the host colon in several morphological forms, including the infectious and environmentally resistant faecal cyst.

Human health effects

Blastocystis pathogenicity remains controversial due to nonspecific symptoms and the prevalence of asymptomatic infections. Symptoms include watery or loose stools, diarrhoea, flatulence, nausea, abdominal pain, anal itching and weight loss. Infections can last for weeks, months or years. In some patients, symptoms resolve, even when *Blastocystis* remain detectable in stools. In children, *Blastocystis* infection has been associated with reduced growth outcomes, including stunting, wasting and underweight status.

Epidemiology

Blastocystis infections occur globally, and prevalence differs geographically, seasonally and demographically. Prevalence is higher in areas where sanitary and hygienic conditions are inadequate and the climate is warm and humid, such as tropical areas with inadequate safe drinking-water, sanitation and hygiene. *Blastocystis* may be commensal organisms that become pathogenic with host changes (e.g. immunosuppression, malnourishment and other infections), although infection occurs in immunocompetent and immunocompromised individuals. Infection appears to be widespread among both adults and children, but some studies report peak infections occur between 10 and 14 years of age.

Transmission

Transmission is mainly faecal–oral, with humans and other animal hosts including insects, reptiles, birds, livestock, domestic pets, rodents and non-human primates all being natural reservoirs for the organism. There is some evidence for transmission from water and food, with not washing hands after toileting, drinking faecally contaminated water and repeatedly eating outside identified as risk factors. Other risk factors that have been identified include exposure to infected animals and crowded settings such as child-care facilities and schools. Direct contact with infected humans or fomites have also been suggested as exposure sources. The infectious dose is unknown.

Detection methods

Culture-based methods, in vitro infectivity and molecular methods are available for *Blastocystis*.

Source and occurrence

Blastocystis cysts have been detected in water and wastewater environments including surface water, vulnerable groundwater and inadequately treated drinking-water during investigations of illness associated with water. Prevalence is influenced by the presence of faecal material from humans including sewage (untreated or partially treated) and other animals. *Blastocystis* cysts can persist in water for long periods (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against *Blastocystis*. These include protection of source water from contamination by human and animal faecal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. *Blastocystis* cysts are likely removed by conventional filtration methods because of their size (similar or larger than *Cryptosporidium* oocysts). They have relatively high resistance to chlorine and ozone, but can be inactivated by UV light. Due to the environmental persistence of *Blastocystis* and their high resistance to chemical disinfectants, *E. coli* (or thermotolerant coliforms) is not a suitable indicator of its absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *Blastocystis*.

Bibliography

WHO (2025) *Blastocystis. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Cryptosporidium

Significance in drinking-water

Cryptosporidium are major zoonotic waterborne pathogens with strong evidence of transmission through drinking-water from a large number of outbreaks, including those where sewage or livestock faeces were the source of contamination. Outbreaks of cryptosporidiosis are commonly associated with insufficient treatment or treatment malfunction, recent rainfall in water catchments, untreated and vulnerable ground-water sources, and distribution system deficiencies.

General description

Cryptosporidium are coccidian protozoan parasites with a complex life cycle including sexual and asexual reproduction. Thick-walled *Cryptosporidium* oocysts found in the environment are typically 4–6 µm. When ingested, the oocysts excyst and complete their life cycle in the small intestine, resulting in production of thin- and thick-walled oocysts. Thin-walled oocysts can excyst within the same host and start a new life cycle leading to heavy infections, while thick-walled oocysts are excreted in faeces. Of the 44 *Cryptosporidium* species, 19 have been reported in humans worldwide, with *C. hominis* and *C. parvum* the most frequently reported.

Human health effects

Cryptosporidium infection (cryptosporidiosis) can result in watery diarrhoea of sudden onset, abdominal pain, nausea, vomiting and low-grade fever. Asymptomatic infection can also occur. Symptoms are usually self-limiting and last for up to 2 weeks, but can lead to profound weight loss, prostration, and increased morbidity and mortality. Cryptosporidiosis in malnourished children under 2 years of age has been associated with moderate to severe diarrhoea that can be persistent and result in growth faltering, cognitive deficit disorders and mortality. Severely immunocompromised individuals may suffer chronic, severe diarrhoea with significant loss of fluid,

as infections spread throughout the gastrointestinal tract or to other sites. Seronegative reactive arthritis has been reported in adults and children. There is no effective therapy or vaccine for cryptosporidiosis.

Epidemiology

Cryptosporidiosis is a very common gastrointestinal disease. It is one of the most important causes of diarrhoeal disease and deaths among young children, especially in low-income settings. While both *C. hominis* and *C. parvum* cause disease worldwide, there are geographical and seasonal differences with *C. hominis* dominant in most regions, but *C. parvum* is more common in selected countries or regions such as the Middle East and New Zealand. There are also differences in distribution between urban and rural areas, with *C. hominis* more common in urban areas and *C. parvum* more common in rural areas. In addition to young children, caregivers of infected individuals, travellers to countries where infection is endemic and individuals who handle infected livestock are more likely to become infected.

Transmission

Cryptosporidium are transmitted by the faecal–oral route. Outbreaks have been reported after ingestion of contaminated drinking-water, recreational waters and food, and after contact with infected humans and animals. Livestock (predominantly calves and lambs) are important sources of human infectious *C. parvum*.

The relative infectivities of *C. hominis* and *C. parvum* are considered to be high. For *C. hominis*, the dose associated with a 50% infection rate (ID_{50}) is estimated at 83 oocysts, while for *C. parvum*, the ID_{50} values differ between isolates, ranging from 12 to 455 oocysts. The incubation period is typically 5–7 days. Symptomatic and asymptomatic individuals can shed infective oocysts for weeks to months after infection. While previous infections do not confer immunity, they may protect against diarrhoea and severe disease.

Detection methods

Standard methods for *Cryptosporidium* based on microscopy are available, but cannot determine the viability or infectivity of oocysts. Cell cultures and animal models can be used to determine infectivity. Molecular methods are also available to identify *C. hominis*, *C. parvum* and other *Cryptosporidium* species (see [section 7.4](#)).

Source and occurrence

Cryptosporidium oocysts are found in a broad range of water and wastewater environments. *Cryptosporidium* have been frequently detected in vulnerable groundwater and inadequately treated drinking-water, with prevalence influenced by rainfall and the presence of faecal material including from human sewage (untreated or partially treated) and animals, particularly cattle. Infective *Cryptosporidium* oocysts can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Cryptosporidium* oocysts should include protection of source waters from contamination

by human and animal faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Filtration processes are important barriers for *Cryptosporidium* in drinking-water treatment. *Cryptosporidium* oocysts are highly resistant to chlorine, but UV light and ozone are effective. Due to the environmental persistence of *Cryptosporidium* oocysts and their resistance to some chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *Cryptosporidium* (see [section 11.6](#)).

Bibliography

WHO (2025) *Cryptosporidium*. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health. Geneva, World Health Organization.

Cyclospora cayetanensis

Significance in drinking-water

There is moderate evidence linking transmission of *Cyclospora cayetanensis* with drinking-water, based on a small number of outbreaks possibly associated with undisinfected or inadequately disinfected drinking-water supplies.

General description

Cyclospora are coccidian protozoan parasites with a complex life cycle including sexual and asexual reproduction. Environmentally stable oocysts typically measure 8–10 µm. When ingested, the oocysts excyst and complete their life cycle in the small intestine, leading to production of immature oocysts. Once excreted, immature oocysts must undergo sporulation over days to weeks to mature infectious oocysts. Nineteen species of *Cyclospora* have been described, of which *C. cayetanensis* is the only species in the genus known to infect humans.

Human health effects

Infection by *C. cayetanensis* (cyclosporiasis) is characterized by watery diarrhoea, abdominal cramping, weight loss, anorexia, myalgia, fatigue, and occasionally vomiting and fever. Serious sequelae such as Guillain–Barré syndrome and Reiter’s syndrome have been reported. Symptoms are usually self-limiting, but relapsing illness can last weeks or months. Increased susceptibility, prolonged diarrhoea, high rates of reoccurrence and severe intestinal injury have been observed in immunocompromised individuals. Asymptomatic infection is common in endemic areas. Early and persistent exposure is associated with decreased likelihood of symptomatic infection. Treatment is generally not required, but antibiotics can be used to manage cyclosporiasis.

Epidemiology

C. cayetanensis has a global distribution with predominance in low-income, tropical and subtropical countries where it is often endemic. Community-based studies in endemic settings (included in a 2019 review) have found widely varying infection prevalence rates ranging from 5% to 40%. Children frequently have the highest preva-

lence rates, although the age range of increased prevalence (e.g. 2–4 years or older) has varied among studies. The moderate to severe childhood diarrhoeal disease burden attributable to *C. cayetanensis* in low-income, high-burden settings is low compared to other gastrointestinal pathogens. However, attribution in endemic settings is often complicated by the high proportion of asymptomatic cases. In endemic countries, the prevalence of *C. cayetanensis* has a marked seasonality, but this varies, with prevalence high in some countries during wet seasons and in dry seasons in other countries. This suggests that the life cycle of *C. cayetanensis* in the environment is likely to be influenced by a mixture of factors, such as rainfall, temperature and humidity. In non-endemic settings, infections are observed in the general population regardless of age and immune status, and are almost always symptomatic. Illness is linked mostly to travel to endemic areas or consumption of food from endemic areas.

Transmission

C. cayetanensis is transmitted by the faecal–oral route. Contaminated food, particularly raw fruit and vegetables, appears to be the primary source of transmission in low-income countries. Contact with surface and recreational waters and soils have also been documented as sources. In addition, there is some evidence that ingestion of contaminated water may be a source of exposure. Person-to-person transmission is virtually impossible because oocysts excreted in faeces require a maturation period in the environment in order to sporulate and become infective. Humans are the only confirmed reservoir for *C. cayetanensis*.

The relative infectivity of *C. cayetanensis* is unknown, but based on the high infectivity of oocysts of similar enteric protozoa (e.g. *Cryptosporidium*), it is probably high (infective doses of 1–10² organisms). The incubation period for cyclosporiasis is typically 1 week. Shedding can be intermittent and can also occur when the host is asymptomatic or for many weeks after symptoms resolve in untreated patients. Although the likelihood of diarrhoeal symptoms decreases with each subsequent infection, outbreaks have been reported in endemic areas, which could suggest that any acquired immunity is not long lasting.

Detection methods

Microscopic and molecular methods are available for *Cyclospora* oocysts (see [section 7.4](#)).

Source and occurrence

There are limited data on occurrence, but *C. cayetanensis* oocysts have been detected in water and wastewater environments including surface water, groundwater and inadequately treated drinking-water. Prevalence is influenced by faecal contamination from humans including from sewage (untreated or partially treated). They can persist in water for long periods (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *C. cayetanensis* should include protection of source waters from contamination by human faecal waste (including extra vigilance during heavy rain), application of adequate

water treatment processes, and protection of drinking-water quality during storage and distribution. Filtration processes that are effective against *Cryptosporidium* oocysts should be effective at removing the slightly larger *Cyclospora* oocysts. Based on their cell wall structure, *Cyclospora* oocysts are probably highly resistant to chlorine and are expected to be as sensitive to UV light as other protozoan cysts and oocysts. Due to the environmental persistence of *Cyclospora* oocysts and their probable resistance to chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *C. cayetanensis* (see [section 11.6](#)).

Bibliography

WHO (2025) *Cyclospora cayetanensis*. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health. Geneva, World Health Organization.

Cytoisospora belli

Significance in drinking-water

Cytoisospora belli is of low health significance in drinking-water. The evidence linking transmission of *C. belli* with drinking-water is weak. Although three waterborne outbreaks have been identified, none were epidemiologically associated with drinking-water, but rather with consumption of vegetables irrigated with water from a sewage treatment plant.

General description

C. belli is a coccidian protozoan parasite. There are many species of *Cytoisospora* that infect animals, but only *C. belli* infects humans. *C. belli* has a complex life cycle including asexual and sexual replication. Environmentally stable oocysts typically measure 25–35 µm by 18–20 µm. When ingested, the oocysts excyst and complete their life cycle in the small intestine, leading to production of immature oocysts. Once excreted in faeces, immature oocysts require a period of 1–2 days in the environment to mature into infectious oocysts.

Human health effects

Infection can be symptomatic or asymptomatic. Symptoms include a low-grade fever, lassitude and malaise, followed soon by mild, watery diarrhoea, abdominal pain, cramp and nausea. Eosinophilia is a distinctive feature of this pathogen and is not caused by other coccidian parasites. The infection is usually self-limiting and lasts 1–4 weeks. Immunocompromised individuals experience more severe symptoms in which diarrhoea may become intense and prolonged, and lead to weight loss and malabsorption. In rare cases, they may also develop extra-intestinal disease that may spread.

Epidemiology

Infections with *C. belli* are widespread in humans, with higher prevalence of infection in immunocompromised people and children. Infections are more common in the tropics and subtropics, although they may occur elsewhere, including in high-income countries. Prevalence also depends on sanitation and hygiene conditions.

The burden of disease from *C. belli* is variable, and it is poorly quantified on a global basis. Outbreaks have been reported, especially among immunocompromised hosts and in child-care and health-care facilities where hygiene and sanitation conditions are inadequate.

Transmission

C. belli is transmitted via the faecal–oral route. Therefore, contaminated water, food and fomites are considered likely sources of transmission. However, transmission from fomites and drinking-water has not been confirmed. Direct person-to-person transmission is unlikely. The infectious dose is unknown.

Detection methods

C. belli oocysts can be detected by light microscopy. Molecular methods are also available (see [section 7.4](#)).

Source and occurrence

There are limited data on the occurrence of *C. belli* oocysts, but they have been detected in water and wastewater environments including surface waters and inadequately treated drinking-water, with prevalence influenced by faecal contamination from humans including sewage (untreated or partially treated). The persistence of *C. belli* is expected to be long (over 1 month at 20 °C), based on studies with other coccidian parasites.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against *C. belli*. These include protection of source water from contamination by human faecal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. *C. belli* oocysts are likely removed by conventional filtration methods because of their relatively large size (larger than *Cryptosporidium* oocysts). Based on data available for other protozoa, they are likely to be resistant to chlorine. Due to the likely environmental persistence of *C. belli* oocysts and their probable resistance to chemical disinfectants, *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *C. belli*.

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WHO (2025) *Cytoisospora belli*. *Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Entamoeba histolytica

Significance in drinking-water

Entamoeba histolytica is a significant waterborne pathogen, with strong evidence from several outbreaks linking transmission to drinking-water. A common risk factor associated with outbreaks is contamination of drinking-water by sewage.

General description

Entamoeba are protozoan parasites. *Entamoeba* cysts found in the environment are typically 10–20 µm. Following ingestion, excystation occurs in the intestine to become feeding, replicative trophozoites (10–50 µm). Some trophozoites may invade human cells throughout the body, while others will develop into cysts that are excreted in the faeces. There are many species of *Entamoeba*, of which four (*E. histolytica*, *E. polecki*, *E. coli* and *E. hartmanni*) are considered human pathogens. The majority of symptomatic disease in humans is caused by *E. histolytica*. Humans are the only known reservoir of *E. histolytica*.

Human health effects

There are two main forms of human disease due to *E. histolytica*: acute intestinal invasive amoebiasis and extra-intestinal invasive amoebiasis. Infection can be asymptomatic or range from mild to life-threatening colitis. Most symptomatic cases present as amoebic dysentery. Symptom onset is gradual, with diarrhoea initially being watery but then becoming mucoid and bloody. Severe abdominal pain is common. Invasive amoebiasis may then progress to ulcerative colitis and perforation of the intestine. Intestinal perforation is the most common cause of death. In uncomplicated cases, recovery from intestinal amoebiasis usually takes several weeks, but in some cases, illness lasts longer. Amoebic liver abscesses are the most common form of extra-intestinal amoebiasis. In rare cases, *E. histolytica* may invade other parts of the body. The main clinical symptoms are the abrupt onset of pain and local tenderness over the liver. Diarrhoea is usually not present. Several drugs are available to treat amoebiasis; the choice depends on the disease manifestations.

Epidemiology

Amoebiasis occurs worldwide, with higher incidence in tropical and subtropical regions. Within these regions, there is substantial variation in the prevalence and type of infection, which could be influenced by gender, type of work, ecological and physiological factors. Amoebic liver abscesses mainly affect men between the ages of 18 and 50 years. *E. histolytica* gastrointestinal infection affects all ages, but young children in low-income settings are particularly susceptible.

Transmission

E. histolytica is transmitted primarily by the faecal–oral route. Person-to-person contact and ingestion of faecally contaminated food or water appear to be the most significant sources of transmission. Sexual transmission through anal–oral contact has also been documented. Risk of transmission is greatest in areas with inadequate sanitation and poor drinking-water quality and crowded living conditions.

The relative infectivity of *E. histolytica* cysts is unknown, but based on the high infectivity of cysts of other similar enteric protozoa (e.g. *Giardia duodenalis*), it is also assumed to be high (infective doses of 1–10² organisms). The incubation period for acute intestinal amoebiasis is typically 2–4 weeks. Pathogen shedding can be months or years, and is usually through asymptomatic carriers and those with chronic infections. Immunity to amoebiasis can develop after infection, but may be short lived.

Detection methods

Methods used for *E. histolytica* detection include microscopy, serology, molecular analysis, antigen detection and amoeba culture with isoenzyme analysis. Molecular methods can also be used for the speciation of *Entamoeba* (see [section 7.4](#)).

Source and occurrence

There have been limited studies on occurrence of *E. histolytica*, but it has been detected in water and wastewater environments including surface water, groundwater and inadequately treated drinking-water. Prevalence is influenced by faecal contamination from humans including sewage (untreated or partially treated). The persistence of *E. histolytica* cysts in water is unknown, but based on the persistence of similar enteric protozoan cysts (e.g. *Giardia duodenalis*), it is expected to be moderate to long (at least 1 week at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *E. histolytica* should include prevention of source water contamination by human faecal waste (including extra vigilance during heavy rain), application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Filtration is an effective barrier to *E. histolytica* cysts, which are larger than *Cryptosporidium* oocysts. Although *E. histolytica* cysts are moderately resistant to chlorine, doses and contact times typically used in drinking-water treatment should provide adequate inactivation. *E. histolytica* cysts are expected to be as sensitive to UV light as other protozoan cysts and oocysts. Due to the probably long environmental persistence of *E. histolytica* cysts and their moderate resistance to chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of its absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *E. histolytica* (see [section 11.6](#)).

Bibliography

WHO (2025) *Entamoeba histolytica*. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health. Geneva, World Health Organization.

Giardia duodenalis

Significance in drinking-water

Giardia duodenalis is a significant zoonotic waterborne pathogen, with strong evidence from a large number of outbreaks linking transmission to drinking-water. Outbreaks of *Giardia* infection are commonly associated with insufficient treatment or treatment malfunction, untreated and vulnerable groundwater sources, and distribution system deficiencies.

General description

G. duodenalis (synonyms *G. lamblia* and *G. intestinalis*) are motile, protozoan parasites. *G. duodenalis* are separated into eight genetic assemblages (A to H) that have dif-

ferent hosts. Assemblages A and B are found in humans and other primates, cats, dogs and some wild mammals, and are generally regarded as zoonotic. Assemblage A is also prevalent in livestock. Other assemblages are not pathogenic to humans. *Giardia* cysts found in the environment measure 11–14 µm by 7–10 µm. Following ingestion, excystation occurs to become trophozoites, which replicate in the duodenum. Further down the intestinal tract, trophozoites develop into cysts, which are excreted in the faeces.

Human health effects

Giardia infection (giardiasis) can result in a watery diarrhoea that comes with a broad spectrum of gastrointestinal symptoms, including flatulence, bloating, weight loss, abdominal cramping, nausea, malabsorption, greasy and foul-smelling stools, fatigue and chills. These symptoms may last for weeks, and can be chronic and recurring. Failure to thrive can occur, particularly in young children, with weight loss. Asymptomatic infections can occur, especially in children, in highly endemic areas and in persons with prior infections. Giardiasis is often regarded as a benign gastrointestinal illness, although chronic or debilitating giardiasis has been reported. Giardiasis is usually a self-limiting infection, but anti-parasitic drugs can be used to treat serious, chronic or recurring infections.

Epidemiology

Giardiasis is a very common intestinal infection worldwide. Giardiasis is most frequently seen in young children in both low- and high-income countries. Others more likely to become infected include caregivers of infected individuals and travellers to countries where infection is endemic. The current understanding of giardiasis is limited by the lack of monitoring and reporting systems in many countries.

Transmission

G. duodenalis is transmitted by the faecal–oral route. Transmission occurs frequently in environments where personal hygiene levels are challenged, such as child day-care centres or in disadvantaged communities. Most giardiasis cases occur sporadically, but common source outbreaks can occur. Transmission sources for giardiasis include water, food, and person-to-person and animal contact. Most waterborne outbreaks are associated with drinking-water and recreational water.

The relative infectivity of *G. duodenalis* is considered high. The dose associated with a 50% infection rate (ID₅₀) for *G. duodenalis* was calculated at 35 cysts from a series of challenge studies in adult males. The incubation period ranges from 3 days post infection to up to 3 weeks, depending upon the assemblage, dose and host factors such as immune status. Cysts are infectious for as long as they are being excreted. Two studies of waterborne outbreaks suggested that previous infection with *Giardia* may produce some level of protection against reinfection.

Detection methods

Standard methods for *Giardia* based on light microscopy are available, but cannot determine the viability or infectivity of oocysts. Molecular methods are available to identify *Giardia* species and subtypes including assemblages A and B (see [section 7.4](#)).

Source and occurrence

Giardia cysts are found in a broad range of water and wastewater environments. *Giardia* have been frequently detected in surface water and have been detected in vulnerable groundwater and inadequately treated drinking-water. Prevalence is influenced by rainfall and the presence of faecal material, including from human sewage (untreated or partially treated) and animals. *Giardia* cysts can persist for moderate to long periods in water (1 week to over 1 month at 20 °C), with survival influenced by temperature and other environmental factors.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risk from *Giardia* should include protection of source waters from contamination by animal and human faecal wastes (including extra vigilance during heavy rain), application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Filtration processes are important barriers for *Giardia* in drinking-water treatment. Although *Giardia* cysts are highly resistant to chlorine, it can be effective where drinking-water storage facilities provide adequate contact times. *Giardia* cysts are sensitive to UV light. Due to the environmental persistence of *Giardia* cysts and their resistance to chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator of the absence of *G. duodenalis* in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *G. duodenalis* (see [section 11.6](#)).

Bibliography

WHO (2025) *Giardia duodenalis. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Microsporidia

Significance in drinking-water

Microsporidia are of low health significance in drinking-water. Although microsporidia are transmitted by the faecal–oral route and have been detected in drinking-water, the evidence linking transmission with drinking-water is weak. Epidemiological evidence from a single outbreak is inconclusive.

General description

Microsporidia are spore-forming parasites. Although previously classified as protozoa, evidence indicates that they could be fungi or closely related to fungi. At least 14 genera and 17 species (out of more than 2000 species) have been implicated in human infections, including *Enterocytozoon*, *Encephalitozoon*, *Nosema*, *Pleistophora*, *Vittaforma* and *Trachipleistophora*, as well as a collective group of unclassified microsporidia. Of these, some are zoonotic and others appear to be human specific. There are two main clades of microsporidia: typical (or advanced) and atypical (or primitive). Spores of the typical microsporidia are generally 2–8 µm in size, but can vary from 1 µm to greater than 30 µm in length. Within an infected cell, a complex process

of multiplication takes place, and new spores are produced and released in faeces, urine, respiratory secretions or other body fluids.

Human health effects

Microsporidia cause illness, predominantly in persons with AIDS and others who are immunocompromised, immunosuppressed or immune deficient. The most common symptom in AIDS patients is severe enteritis involving chronic diarrhoea, dehydration, weight loss and wasting. However, a broad range of other illnesses can also occur. Prolonged illness for up to years has been reported. In immunocompetent individuals, infections can be asymptomatic and where there is illness, symptoms are less pronounced with the typical illness being gastroenteritis.

Epidemiology

Microsporidia infections occur worldwide, with documented illness in persons from all continents. The burden of disease with microsporidia is variable and uncertain depending on population segment, age, immunocompetence, HIV-infection status and antiretroviral therapy access, geographical region, occupation, extent of contact with animals and sanitation conditions. The microsporidia most frequently found in human infections are *Enterocytozoon bieneusi*, followed by the *Encephalitozoon* species, particularly *E. intestinalis*, followed by *E. cuniculi* and *E. hellem*. The overall prevalence rate of microsporidial infections (based on a 2021 meta-analysis) is estimated to be 10%. *Enterocytozoon* infection is generally localized and limited to intestinal enterocytes and biliary epithelium. *Encephalitozoon* spp. can cause systemic infection involving multiple sites, with complications including keratoconjunctivitis, myositis and hepatitis. Several other species are also capable of causing disseminated disease.

Transmission

Microsporidia can be transmitted by the faecal–oral route through contact with infected people, animals or fomites, or ingestion of faecally contaminated water, food or soil. Although spores have been detected in faecally contaminated water and food, waterborne and foodborne outbreaks are rarely reported. Drinking-water is an unconfirmed exposure source. However, microsporidia have been associated with eye infections from exposures to soil, mud and recreational waters. Urine may also be an important source of exposure. Transmission by the inhalation of airborne spores is plausible, but documented evidence is lacking. Epidemiological and experimental studies in mammals suggest that *Encephalitozoon* spp. can be transmitted horizontally and possibly transplacentally from mother to offspring, but documented evidence is lacking for humans. Although the role of animals in microsporidia transmission to humans remains unclear, zoonotic transmission of some strains is likely. The human infectious dose is unknown.

Detection methods

Microsporidia can be detected by microscopy, immunoassays or molecular methods. Cell culture and fluorescent in situ hybridization can be used to detect viability (see [section 7.4](#)).

Source and occurrence

Microsporidia are found in a broad range of water and wastewater environments, and have been detected in surface water, groundwater and drinking-water. Prevalence is influenced by the presence of human and animal wastes including from sewage (untreated or partially treated). Microsporidia spores can persist for long periods in water (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to reduce potential microbial risks should be effective against microsporidia. These include protection of source water from contamination by human and animal wastes, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Microsporidia are likely removed by conventional filtration methods. They are moderately resistant to chlorine, although typical doses and contact times used for drinking-water treatment should be sufficient to inactivate microsporidia. Microsporidia are sensitive to ozone and UV light. Due to the environmental persistence of microsporidia spores and their moderate resistance to chemical disinfectants, *E. coli* (or thermotolerant coliforms) is not a suitable indicator of their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of microsporidia.

Bibliography

WHO (2025) *Microsporidia. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Naegleria fowleri

Significance in drinking-water

Naegleria fowleri is not transmitted through the ingestion of drinking-water. However, there is strong evidence for water-based transmission from water entering nasal passages during recreational activities involving drinking-water (e.g. in swimming pools, playing with garden hoses) and through nasal rinsing and ablution. *Naegleria* can also harbour bacterial pathogens such as *Legionella* and may play a role in their survival and pathogenicity in drinking-water.

General description

Naegleria are free-living amoebae commonly found in warm fresh water and soil. The life cycle of *Naegleria* consists of three stages. The trophozoite (10–35 µm) is the feeding and infective stage, while the cyst (7–15 µm) is the environmentally persistent stage. The amoeba-flagellate (10–16 µm) is a temporary motile stage that develops in response to environmental changes. There are more than 30 species of *Naegleria*, but only one, *N. fowleri*, has been isolated from human cases of disease. *N. fowleri* is thermophilic and feeds on bacteria in water and soil.

Human health effects

N. fowleri causes primary amoebic meningoencephalitis. Infection occurs when trophozoites in water are forced up the nasal cavity, migrate through the olfactory nerve

and cause extensive damage to the frontal lobes of the brain. Symptoms include headache, fever, nausea and vomiting, rapidly progressing to a stiff neck, seizures, altered mental status, hallucinations and coma. The survival rate (including based on a 2021 review) is less than 5%.

Epidemiology

Although *N. fowleri* is widespread in fresh water and soil, primary amoebic meningoencephalitis is infrequent; only a few hundred cases have been reported worldwide, with reporting in all continents except Antarctica. Exposure is associated with age, gender and activities. Disease associated with exposure to contaminated drinking-water used in recreational activities is highest in male children, while disease associated with nasal ablution or irrigation is highest in adult males. Most cases of disease are associated with recreational exposure to warm fresh water and contaminated drinking-water in warm climates.

Transmission

N. fowleri is transmitted by contaminated water entering the nasal passages. Transmission does not occur through consumption of drinking-water, and there is no person-to-person transmission. Cases associated with drinking-water supplies have typically been associated with conditions that favoured the growth of the organism (e.g. hot weather, inadequate treatment and poor management of drinking-water distribution systems) followed by exposures via swimming pools, playing with garden hoses, nasal ablution and irrigating sinuses. The human infective dose is unknown. The median incubation period for primary amoebic meningoencephalitis is 5 days. The development of illness is rapid, progressing through coma to death within a median of 5 days.

Detection methods

Naegleria can be detected using culture-based methods, with molecular methods available to confirm the presence of *N. fowleri* (see [section 7.4](#)).

Source and occurrence

N. fowleri is a natural inhabitant of warm fresh water and soil, and is distributed worldwide with habitats including lakes, streams, groundwater, spas, swimming pools, hydrotherapy pools and hot springs. The amoebae grow in biofilms in drinking-water distribution systems and storage tanks in the absence of disinfectant residuals provided that the water temperature is consistently 25–42 °C. Increased water temperatures (greater than 25 °C) can also increase colonization by *N. fowleri*. Cysts form under stressful environmental conditions and can persist for long periods in water.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *N. fowleri* should include application of adequate water treatment processes, protection of drinking-water quality during storage and distribution, and minimizing their growth in biofilms in distribution systems and building water systems. Filtration systems designed to remove other protozoa, such as *Cryptosporidium* are likely to be effective in removing the larger *Naegleria* cysts and trophozoites. Cysts and tropho-

zoites are moderately resistant to oxidizing disinfectants such as chlorine, but doses and contact times typically used in drinking-water treatment should provide adequate inactivation. *N. fowleri* is sensitive to UV light. In distribution systems, maintaining a disinfectant residual provides an effective barrier to *N. fowleri*. As *N. fowleri* is a non-faecal, environmental organism that can survive and grow in water, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for its possible presence or absence in drinking-water.

Bibliography

WHO (2025) *Naegleria fowleri*. Background document for the WHO Guidelines for drinking-water quality. Geneva, World Health Organization.

Toxoplasma gondii

Significance in drinking-water

Toxoplasma gondii is a significant zoonotic waterborne pathogen, with strong evidence from a number of outbreaks linking transmission to drinking-water. A common risk factor associated with outbreaks is use of unfiltered surface water.

General description

T. gondii is a coccidian protozoan parasite with a complex life cycle including a sexual component that occurs only in felids and an asexual component that occurs in all warm-blooded animals. Domestic cats and other members of the Felidae family are the definitive host. In the natural life cycle of the parasite, rodents and birds containing infective tissue cysts are eaten by felids. Sexual reproduction of the parasite occurs in the intestines of these animals and leads to the formation of oocysts (10–12 µm). Excreted oocysts become infectious in the environment after sporulation. Following ingestion, asexual reproduction leads to oocysts transforming into tachyzoites (3–6 µm), which localize in host tissue and transform into bradyzoites that form cysts. In humans, tissue cysts form most commonly in skeletal muscle, myocardium, brain and eyes.

Human health effects

Toxoplasmosis is usually asymptomatic. In about 10% of cases, symptoms develop and include acute flu-like symptoms and enlargement of the lymph nodes, liver and spleen. The clinical course is usually self-limiting. During chronic infection, dormant tissue cysts persist in the host. In immunocompetent individuals, there is a rare risk of developing ocular toxoplasmosis later in life. In immunodeficient individuals, reactivation of cysts can result in disseminated disease involving the central nervous system and lungs, and lead to severe neurological disorders or pneumonia that can be fatal. If a woman is infected for the first time during pregnancy, the parasite is transmitted to the unborn child in about one third of cases. Infection during the first trimester can lead to severe outcomes including spontaneous abortion, stillbirth or fetal abnormality. Congenital toxoplasmosis from first trimester infections can result in chorioretinitis, cerebral calcifications, hydrocephalus, severe thrombocytopenia and convulsions. Infections later in pregnancy can be asymptomatic at birth, but ocular toxoplasmosis can occur years later.

Epidemiology

T. gondii is found throughout the world, but infection is more common in warm and humid climates. The prevalence of human *T. gondii* infection varies among countries and regions within the same country, with rates (including based on a 2019 systematic review), up to 90% reported. Poor socioeconomic status and the ingestion of raw or undercooked meat increase the risk of infection. The estimated seroprevalence of *T. gondii* infection in pregnant women varies widely in different parts of the world. Case-control studies in Europe have shown that consuming meat is the main risk factor for infection in pregnant women. In Ethiopia, anti-*T. gondii* seropositivity among pregnant women was associated with having an unsafe water source as well as age, abortion history, contact with cats, cat ownership, lack of knowledge about toxoplasmosis and being a housewife.

Transmission

T. gondii is transmitted via the consumption of raw or undercooked meat containing tissue cysts or through ingestion of oocysts in food such as contaminated vegetables and fruit and raw or undercooked shellfish, in untreated or inadequately treated drinking-water or in surface waters during recreational bathing. *T. gondii* can be transmitted from cats and through ingestion of soil. Humans are an intermediate host and do not excrete oocysts. Person-to-person transmission does not occur except through vertical transmission from mother to child, by organ transplantation or through blood transfusion.

The dose–response relationship for ingestion of *T. gondii* oocysts or tissue cysts has not been established in humans. However, based on animal dose–response data, the relative infectivity of *T. gondii* is expected to be high (infective doses of 1–10² organisms). The incubation period for humans varies depending on the *T. gondii* life stage initiating infection. Reports from human outbreaks suggest incubation periods of 10–23 days after ingestion of infected tissue cysts and 5–20 days after ingesting oocysts from contaminated environmental matrices.

Detection methods

Methods for *T. gondii* include microscopy, molecular techniques, loop-mediated isothermal amplification and bioassays.

Source and occurrence

There are limited data on the occurrence of *T. gondii*, but they have been detected in water and wastewater environments including surface water, groundwater and inadequately treated drinking-water. Prevalence is influenced by contamination from wild and domestic cats. Oocysts of *T. gondii* can persist in water for long periods (over 1 month at 20 °C).

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *T. gondii* oocysts should include prevention of source water contamination by wild and domestic cats, application of adequate water treatment processes, and protection of drinking-water quality during storage and distribution. Since *T. gondii* oocysts are

larger than *Cryptosporidium* oocysts, filtration processes are important barriers for *T. gondii* in drinking-water treatment. *T. gondii* oocysts are highly resistant to chlorine, but are sensitive to UV light. Due to the environmental persistence of *T. gondii* oocysts and their resistance to chemical disinfectants, the absence of *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their absence in drinking-water. However, the presence of *E. coli* is a suitable indicator of the possible presence of *T. gondii* (see [section 11.6](#)).

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WHO (2025) *Toxoplasma gondii. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

11.4 Helminth pathogens

The word “helminth” comes from the Greek word meaning “worm”, and refers to all types of worms – free living and parasitic. The major parasitic worms are classified primarily in the phylum Nematoda (roundworms including *Dracunculus*) and the phylum Platyhelminthes (flatworms including *Fasciola* and *Schistosoma*). For most helminths, drinking-water is not a significant route of transmission. There are two exceptions: *Dracunculus medinensis* (guinea worm) and *Fasciola* (*F. hepatica* and *F. gigantica*) (liver flukes). However, for both these exceptions, transmission has been linked to consumption of surface water rather than drinking-water supplies. *Dracunculus* and *Fasciola* both require intermediate hosts to complete their life cycles but are transmitted through drinking-water by different mechanisms. Other helminthiasis can be transmitted through water contact (schistosomiasis) or are associated with the use of untreated wastewater in agriculture (ascariasis, trichuriasis, hookworm infections and strongyloidiasis), but are not usually transmitted through drinking-water (for further information, see the background documents for these pathogens prepared for the *WHO Guidelines on sanitation and health* (WHO, 2025)). Although other free-living nematodes aside from *D. medinensis* do not cause disease, there is a potential for them to harbour pathogens and provide protection from disinfection.

Dracunculus

Significance in drinking-water

There is strong evidence for waterborne transmission of *Dracunculus medinensis*. While the geographical spread of dracunculiasis has been reduced, in the limited number of countries where *D. medinensis* still occurs, ingestion of surface water contaminated with infected copepods remains the primary source of infection. Formal drinking-water supplies have not been linked to the transmission of *D. medinensis*.

General description

D. medinensis, also called guinea worm, is a nematode (roundworm) that infects people, dogs and possibly other mammalian hosts. The worm has a complex life cycle involving human (or dog) hosts, copepods (e.g. water fleas, usually cyclops), fish and frogs. Fertilized female worms measure 60–100 cm in length.

Human health effects

Dracunculiasis (guinea worm disease) becomes apparent about 1 year post ingestion of *D. medinensis* larvae. The infecting female worm emerges from the skin, typically in the lower leg to produce a painful, debilitating wound that persists until the entire worm exits the body, which is a process that can take many weeks. Slight fever, itchy rash, nausea, vomiting, diarrhoea and dizziness may be associated with emergence. Over time, cellulitis, abscesses, sepsis and septic arthritis near the exit wound can occur, causing the joints to lock and deform; secondary tetanus infection can also occur. The wound at the site of worm emergence must be managed hygienically to minimize the risks of secondary bacterial infection. Mortality rates are very low, but permanent disability can result from contractures of tendons.

Epidemiology

A global eradication campaign that began in the 1980s was responsible for the mass reduction in the number of cases of dracunculiasis and continues to eliminate the pathogen in a few remaining endemic countries in Africa. People of all ages are susceptible to infection and illness. Particular risk groups are the poorest of rural dwellers in low-income settings who use contaminated surface water as a source of drinking-water, often with little or no treatment. A complication in the prevention and control of dracunculiasis is the presence of the same worm in domestic dogs in these under-served settings.

Transmission

Humans typically become infected with *D. medinensis* by drinking surface water containing copepods harbouring *D. medinensis* larvae. Humans may also become infected by eating raw or undercooked fish or frogs harbouring copepods. Larvae ingested by humans penetrate the stomach and intestinal wall, mature and migrate throughout the body. The emergence of the fertilized female worm from the skin causes severe pain that often leads infected people to immerse the emerging worm wound in water, resulting in the release of thousands of larvae. These larvae can live for a few days in water. Copepods present in stagnant surface water accumulate immature larvae and support the development of mature, infectious larvae in about 2 weeks. Person-to-person transmission has not been reported. There is no immunity to *D. medinensis* infection, and reinfection is possible. The human infectious dose of *D. medinensis* is unknown.

Detection methods

Analytical methods for *D. medinensis* in water are difficult. The copepods infected with *D. medinensis* larvae need to be recovered, and there is no culture system for the worm, except for experimental animals. Detection is currently limited to microscopy, although molecular methods to detect *D. medinensis* larvae in copepods are being developed (see [section 7.4](#)).

Source and occurrence

D. medinensis is found typically in stagnant surface waters that contain its copepod hosts. Such waters are the natural aquatic reservoirs and human exposure sources for

this pathogen. In such waters, the copepods containing *D. medinensis* can persist for long time periods and are endemically present. Hence, the persistence of *D. medinensis* is long (over 1 month at 20 °C). Other *Dracunculus* species that do not infect humans may also be present in water, further complicating analytical efforts. Accordingly, there are no data on occurrence of *D. medinensis* in water.

Prevention and management

The provision of a safe water supply to communities relying on surface water sources is a key component to eradicating dracunculiasis. Within a WSP (see [chapter 4](#)), control measures that can be applied to prevent transmission of *Dracunculus* require a comprehensive approach. These include encouraging infected people to avoid entering surface water sources, eradicating copepods from surface water sources with an approved larvicide, preventing entry of infected dogs into water bodies and filtration to remove copepods. Copepods harbouring *D. medinensis* larvae are sensitive to disinfectants such as chlorine. This may result in the release of larvae that have low to moderate resistance to chlorine, although filtration is the recommended treatment barrier.

Since *Dracunculus* are non-faecal organisms and due to their complex life cycle and environmental survival, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

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WHO (2025) *Dracunculus. Background document for the WHO Guidelines for drinking-water quality*. Geneva, World Health Organization.

Fasciola

Significance in drinking-water

There is moderate evidence for waterborne transmission of *Fasciola* from surface water used as sources of drinking-water in hyperendemic areas.

General description

Fasciola hepatica (known as the common liver fluke) and *F. gigantica* (known as the giant liver fluke) are trematodes of the family Fasciolidae. *Fasciola* are vector-borne zoonotic organisms with complex life cycles involving flukes, eggs, miracidia, cercariae and infective metacercariae. Development involves passage through human or animal hosts and an intermediate freshwater snail vector. Their natural hosts are mainly mid- and large-sized ruminants and equines. Pigs can be important reservoirs in human endemic areas. Humans can also be an important source of infectious *Fasciola*. Adult *F. hepatica* and *F. gigantica* flukes can reach 29 mm and 52 mm in size, respectively, while metacercariae are about 0.2 mm in diameter.

Human health effects

Infection with *Fasciola* in humans can cause severe symptoms with acute and chronic stages or can be asymptomatic. The acute stage lasts from 3 to 4 months after ingestion of metacercariae, and is mainly related to the destruction of the abdominal peritoneum and liver tissue as the flukes migrate from the intestines to the liver and bile

ducts. Major symptoms include fever, abdominal pain, gastrointestinal disturbances, urticaria and respiratory symptoms. The chronic stage may last from 9 to 13 years after ingestion, and involves inflammation or blockage of the bile ducts or the gall bladder. Clinical manifestations include epigastric pain, fatty food intolerance, nausea, jaundice, pruritus, right upper-quadrant abdominal tenderness and biliary colic. Further, more severe, life-threatening complications may also manifest, including anaemia, cirrhosis and extrahepatic venous thrombosis. Juvenile flukes sometimes deviate during migration and may enter other organs and cause ectopic fascioliasis, which may be highly pathogenic or even fatal when affecting the central nervous system or eye.

Epidemiology

Estimates of global *Fasciola* infections (based on a 2020 review) range from 2.4 million to 17 million people or higher. Human infection varies largely in relation to the distribution of snail vector populations in freshwater bodies and general physiographic and climatic conditions, including seasonal patterns influenced by temperature and rainfall. Geographically, *F. hepatica* is distributed throughout Europe, northern and eastern Africa, Asia, Oceania, and the Americas (generally colder countries and higher altitudes). However, *F. gigantica* is restricted to Africa and southern Asia (generally warmer countries and lowlands). Other factors influencing fascioliasis are climate change, water resource development projects and movement of host animals. In endemic areas, prevalence and intensity of infection peaks in 9–11 year-olds, but infection persists in adults, with females appearing to be more affected. In high-income countries, the peak appears in 30–40 year-olds. However, infection is uncommon in North America, where most cases are associated with travellers and immigrants. Endemic areas are known to occur in several countries in the Americas, Africa, Asia and the Middle East, with the highest prevalence detected in South America and Africa.

Transmission

Consumption of infective metacercariae attached to uncooked freshwater plants (e.g. watercress) is the most common route of transmission. Fresh water becomes contaminated when eggs are excreted by infected human and animal hosts, miracidia hatch from the eggs and penetrate an intermediate snail host where cercariae are developed (larvae), which are released into water. Cercariae swim for a short period until they reach a suitable attachment site (aquatic plants), where they encyst to form metacercariae, which become infective within 24 hours. In addition to consumption of aquatic plants, drinking-water contaminated with metacercariae and consumption of contaminated beverages, juices and soups are receiving increasing attention as sources of fascioliasis. Washing of vegetables, fruits and kitchen utensils with contaminated water can also be sources of disease. The high diversity of infection sources and their heterogeneity in different countries underlie the large epidemiological heterogeneity of human fascioliasis. Person-to-person transmission is not possible due to the complex life cycle. Evidence indicates that metacercariae are highly infective (infective doses of $1-10^2$ organisms), but the infective dose has not been determined.

Detection methods

Metacercariae in water may be identified by microscopic observation following filtration. Molecular methods are useful to confirm the classification of metacercariae as *Fasciola* (see [section 7.4](#)).

Source and occurrence

Fasciola occur in different types of fresh water, linked to excretion of eggs from infected human and animal hosts and the distribution of the snail vector populations. Lymnaeids prefer stagnant water, and their high colonization capacity explains their presence in waterbodies including springs, artificial fountains and dams. Temporary water bodies inhabited by lymnaeids explain links between seasonal transmission and rainy seasons, such as the monsoon period in south central Asia. Permanent water bodies inhabited by lymnaeids in high-altitude and high-evapotranspiration areas where monthly temperature variations are small allow for disease transmission throughout the year. *Fasciola* are not found in groundwater since lymnaeids need light to survive. Reports on the prevalence of *Fasciola* in drinking-water are limited. Metacercariae have a hard outer wall and can survive for long periods in water (over 1 month at 20 °C), but are susceptible to high temperatures and sunlight.

Prevention and management

Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Fasciola* should include protection of source waters from human and animal faeces, application of adequate water treatment processes, and protection of storage and distribution systems from external contamination. Based on morphology and resistance to environmental pressures, metacercariae are likely to be relatively resistant to disinfection. However, based on their size, they should be removed by physical processes. Due to the complex life cycle and environmental survival of *Fasciola*, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

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WHO (2025) *Fasciola. Background document for the WHO Guidelines for drinking-water quality and the WHO Guidelines on sanitation and health*. Geneva, World Health Organization.

Free-living nematodes

Significance in drinking-water

Free-living nematodes (excluding *Dracunculus medinensis*, which is the subject of a separate fact sheet) do not cause disease. Potential risks have been identified due to free-living nematodes harbouring pathogens and providing protection from disinfection. However, there have been no outbreaks of waterborne diseases caused by pathogens carried by free-living nematodes. In poorly maintained drinking-water systems, free-living nematodes may survive and grow in biofilms, but there is little evidence that they cause a deterioration in the aesthetic quality of drinking-water.

General description

Nematoda (roundworms) are a diverse animal phylum; they are structurally simple, with the digestive tract running from the mouth to the opening near the tail. The majority of species are not parasitic and are described as free living, i.e. they live entirely in the natural environment (water and soil). Food sources for different nematodes include algae, fungi, protozoa, dead matter and meiofauna. At optimal temperatures, free-living nematodes range in size from 0.5 to 1.6 mm. A limited number of nematodes have a complete or partial parasitic life cycle and are pathogenic. Only one (*D. medinensis*) is a human pathogen and can be transmitted through drinking-water; it is the subject of a separate fact sheet.

Human health effects

Free-living nematodes are not infectious and do not present a direct threat to human health. However, they may harbour pathogenic microorganisms and potentially protect them from disinfection processes used in drinking-water treatment. For example, studies have indicated their potential to be transport hosts of enteric pathogenic bacteria and protozoa. Gram-negative bacteria ingested by free-living nematodes have been shown to survive the presence of chlorine and UV light.

Epidemiology

Since free-living nematodes are not responsible for infectious diseases in humans, this section is not applicable.

Transmission

Although humans can be exposed to free-living nematodes through ingestion of water and consumption of fresh vegetables, there is no indication that they cause infections.

Detection methods

Free-living nematodes may be identified by microscopic examination of particulate matter concentrated by filtration of water. Distinguishing pathogenic larvae of hookworm and threadworm from free-living, non-pathogenic nematodes in water is difficult and requires special knowledge of nematology.

Source and occurrence

Free-living nematodes are ubiquitous in soil and water environments, and have been detected in drinking-water supplies. They may enter the drinking-water supply chain at any stage, although investigations on occurrence in drinking-water have been limited. Higher numbers have been found at higher water temperatures and when turbidity and nutrient levels of source waters are high.

Prevention and management

Although free-living nematodes are of low concern in drinking-water supplies, control measures within a WSP (see [chapter 4](#)) to protect source waters from the input of nutrients, ensure adequate water treatment processes, and protect drinking-water quality during storage and distribution will reduce the presence of free-living nematodes. There is little quantitative information on the removal and/or inactivation of

free-living nematodes by drinking-water treatment processes. However, based on their size, they should be removed by filtration, but some may colonize granular and biological filters and may subsequently be released into drinking-water. Regular backwashing and maintenance of filters should reduce the accumulation of nematodes. Free-living nematodes are highly resistant to chlorine and UV light, and will not be inactivated by doses typically used to disinfect drinking-water. Maintenance procedures to safeguard the cleanliness of distribution systems, such as scheduled flushing of pipes, should reduce the presence of free-living nematodes in distribution systems. As free-living nematodes are non-faecal environmental organisms, *E. coli* (or thermo-tolerant coliforms) is not a suitable indicator for their possible presence or absence in drinking-water.

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Schistosoma

Significance in drinking-water

There is no evidence linking transmission of schistosomiasis with ingestion of contaminated drinking-water. However, there is strong evidence linking transmission to poor sanitation and lack of safe drinking-water. Readily available safe drinking-water contributes to disease prevention by reducing use of infested water for domestic purposes (e.g. washing clothes, bathing).

General description

Schistosoma (commonly known as blood flukes) is a genus of trematodes. There are 23 species of *Schistosoma*. Most human infections are caused by *S. haematobium*, *S. mansoni* and *S. japonicum*, but other species can infect humans, namely *S. mekongi*, *S. guineensis* and *S. intercalatum*. Humans are the definitive hosts of human schistosomes, but animal species have been reported to be infected with human schistosomes. *Schistosoma* have a complex life cycle involving flukes, eggs and infective cercaria and involves passage through an intermediate snail vector and a definitive vertebrate host, which may be human and/or animal, depending on the species. *Schistosoma* flukes can be 7–20 mm long and infective cercariae 0.4–0.6 µm long. *Schistosoma* flukes can live and produce eggs in their human hosts for several years, with initial egg production taking 3–4 months after exposure.

Human health effects

Infection with *Schistosoma* can lead to two main forms of disease. Symptoms of schistosomiasis typically result from the immune response to eggs in tissues, and not from the flukes themselves. Therefore, the symptoms depend on the amount and location of eggs in the body. Urogenital schistosomiasis (caused by *S. haematobium*) typically manifests as presence of blood in urine. Long-term urogenital schistosomiasis can cause bladder cancer and renal failure. Urogenital schistosomiasis can also cause genital lesions in men and women. In women, potential complications include miscarriage-

es, ectopic pregnancy and infertility. Intestinal schistosomiasis (caused by *S. mansoni*, *S. japonicum*, *S. intercalatum*, *S. guineensis* and *S. mekongi*) produces abdominal pains, diarrhoea, blood in stools, and enlargement of the liver and spleen. Life-threatening complications arising from chronic infections include liver fibrosis and portal hypertension. Anaemia and malnutrition can be associated with the disease, with potential long-term effects on growth and cognitive development in children. In some people, acute schistosomiasis (Katayama fever), including fever, rash, chills, enlarged liver and spleen, muscle pain and cough, can begin within 1–2 months post infection, particularly with *S. japonicum*. If cercariae of avian schistosomes penetrate the human skin, they can cause an inflammatory response at the point of penetration known as schistosome cercarial dermatitis (or more commonly swimmer's itch).

Epidemiology

It has been estimated (including as reported in a 2023 WHO fact sheet) that schistosomiasis affects over 250 million people worldwide. Global distribution of schistosomiasis varies by causative species, since each has a specific range of suitable snail hosts. For example, *S. haematobium* and *S. mansoni* are found in Africa, while the latter is also found in the Middle East, Caribbean and South America and *S. japonicum* is found in China, Indonesia and the Philippines.

Most infections occur in poor communities with lack of access to safe drinking-water and inadequate sanitation. In endemic areas, incidence varies spatially and seasonally with distance to water bodies, rainfall, temperature and irrigation practices that influence exposure and the snail host. Prevalence of infection is linked to the proportion of a population having contact with contaminated surface water, as well as frequency and duration of contact. Exposure occurs through domestic, occupational or recreational activities involving water contact. Use of untreated faecal sludge and wastewater in agriculture has been associated with increased risk of schistosome infections. People of all ages are susceptible to infection, but school-age children are particularly vulnerable due to increased swimming and fishing in contaminated water and inadequate hygiene.

Transmission

Transmission of *Schistosoma* is caused primarily by contact with water contaminated with cercariae, which can penetrate skin. Infection does not occur through ingestion of drinking-water. Fresh water becomes contaminated by *Schistosoma* eggs when infected people (or animals) defecate or urinate in the water. The eggs hatch, invade intermediate snail hosts and develop into infective cercariae (larvae), which are released into water after several weeks. Once released from the host snail, cercariae can remain infective to humans for 1–3 days in fresh water. Shedding of eggs can be years, if not decades. There is some evidence of increasing age-associated immunity in endemic populations.

Detection methods

Schistosome eggs in faeces and urine are detected via microscopic examination and molecular methods (see [section 7.4](#)). Qualitative methods for the detection of cercariae in standing water are also available.

Source and occurrence

The distribution of schistosomes is largely dependent on the availability of freshwater habitats for the snail host, such as ponds, lakes and slow-flowing streams. Water resource development projects have been identified as potential sources of elevated and shifting rates of schistosomiasis as a result of the creation of new habitats for freshwater snails. In endemic areas, surface waters containing host snails are susceptible to contamination by schistosome eggs in faeces and urine. Schistosome eggs can survive days to weeks outside water, but require contact with freshwater and snail vectors to complete their life cycle. Eggs deposited near water bodies through open defecation and urination may still hatch if washed into the water bodies by rain. There is limited data on the occurrence of schistosome eggs and cercariae in surface water. Schistosome eggs cannot survive and complete their life cycle in groundwater; similarly, it is unlikely that cercariae could contaminate groundwater.

Prevention and management

Providing safe drinking-water is a key pillar for controlling schistosomiasis. Within a WSP (see [chapter 4](#)), control measures to manage potential risks from *Schistosoma* should include protection of source waters from human and animal faeces and urine, snail control programmes, application of adequate water treatment processes, and protection of storage and distribution systems from external contamination. Schistosome cercariae can be removed from water by filtration. Although schistosome cercariae are moderately resistant to chlorine, doses and contact times typically used in drinking-water treatment are effective for inactivating cercariae. UV light is also an effective disinfectant for schistosome cercariae. Due to the complex life cycle and environmental survival of *Schistosoma*, *E. coli* (or thermotolerant coliforms) is not a suitable indicator for the possible presence or absence of *Schistosoma* in drinking-water.

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11.5 Toxic cyanobacteria

Detailed information on cyanobacteria and their toxins is available in the supporting document *Toxic cyanobacteria in water* ([Annex 1](#)). See also *Management of cyanobacteria in drinking-water supplies: information for regulators and water suppliers, second edition* ([Annex 1](#)). For more information on the four key cyanobacterial toxin groups – microcystins (MCs), cylindrospermopsins (CYNs), anatoxin-a variants (ATXs) and saxitoxins (STXs), including guideline values and their derivation – see the [chapter 12](#) fact sheets.

Significance in drinking-water

Cyanobacteria occur at low cell density in most surface water bodies. Where conditions promote their growth, they may attain high biomass concentrations termed “blooms”. Such biomass accumulations cause high organic loads, challenging treatment; some also have aesthetic impacts, particularly taste and odour, challenging

acceptability (see [section 10.1](#)). The most notable feature of cyanobacteria in terms of public health impact is that many species may produce toxins (“cyanotoxins”).

Cases of liver damage have been reported where cyanotoxins occurred in poorly treated drinking-water and where scum-forming cyanobacteria were ingested during recreational activities. Further, human fatalities attributable to cyanotoxins (MCs, with a possible contribution of CYNs) have occurred through renal dialysis using inadequately treated water (including drinking-water) containing high cyanotoxin levels. Although mats of filamentous cyanobacteria growing on surfaces can contain high concentrations of neurotoxic ATXs and/or STXs within their cells and have caused numerous cases of rapid deaths of dogs that ingest such material, hazardous concentrations at sites of drinking-water abstraction from these mats have not been reported. As toxins released from such mats are rapidly diluted, hazardous concentrations in drinking-water would be expected only if substantial amounts of cell material from such mats entered the supply system and if treatment was inadequate.

General description

Cyanobacteria are photosynthetic bacteria that share some properties with algae. Notably, they possess chlorophyll *a* and liberate oxygen during photosynthesis. They also contain a blue pigment specific to cyanobacteria that can give some species a bluish-green appearance; hence, a common term for these organisms is blue-green algae. However, because they produce different pigments, a large number of cyanobacteria are not blue-green; they can range in colour from blue-green to yellow-brown to red. Most cyanobacteria are phototrophs, but some also exhibit heterotrophic growth, and some species can fix elemental nitrogen dissolved in water.

Cyanobacteria can be identified by their morphology under a microscope to genus or even to species level. They may grow as separate cells or in multicellular filaments or colonies, which are sometimes large enough to be visible without a microscope. Some species float in the water, (i.e. “planktonic”), whereas other species may grow on surfaces as mats of filamentous cyanobacteria (i.e. “benthic”). Some planktonic species can regulate their buoyancy via intracellular gas vacuoles and thus can form surface scums or accumulate at the thermocline of thermally stratified reservoirs. In contrast, other planktonic species are evenly dispersed throughout the mixed layers of the water body. A number of species encompass genotypes that can produce a range of toxins, as shown in [Table 11.1](#).

Human health effects

Each cyanotoxin group has specific properties, with distinct concerns, including liver and kidney damage, neurotoxicity and tumour promotion. Repeated or chronic exposure is the primary concern for two groups of cyanotoxins: MCs and CYNs. For other groups, concern chiefly pertains to acute toxicity (in particular, the neurotoxic STXs and possibly ATXs). Acute symptoms reported after recreational contact with cyanobacterial blooms in fresh water include gastrointestinal disorders, fever and irritations of the skin, ears, eyes, throat and respiratory tract, but these effects are rare and usually mild. As they do not correspond to the mode of action of the known cyanotoxins, it is uncertain whether they are caused by yet unknown cyanobacterial

Table 11.1 Main bloom-forming planktonic freshwater cyanobacteria that produce toxins^a

Current genus (previous genus) and species ^b	Cyanotoxins ^c
<i>Aphanizomenon</i> spp.	CYNs, STXs, ATXs
<i>Chrysosporum</i> (<i>Aphanizomenon</i>) <i>ovalisporum</i>	CYNs
<i>Cuspidothrix</i> (<i>Aphanizomenon</i>) <i>issatschenkoi</i>	ATXs, STXs
<i>Cuspidothrix</i> (<i>Raphidiopsis</i>) <i>mediterranea</i>	
<i>Dolichospermum</i> (<i>Anabaena</i>) spp.	MCs, CYNs, ATXs, STXs
<i>Microcystis</i> spp.	MCs
<i>Planktothrix</i> spp.	MCs
<i>Raphidiopsis</i> spp., in particular, <i>Raphidiopsis</i> (<i>Cylindrospermopsis</i>) <i>raciborskii</i>	CYNs, STXs

^a Confirmed in isolated strains using reliable analysis (high-performance liquid chromatography, liquid chromatography–mass spectrometry, genetics).

^b Note that substantial taxonomic reorganization has moved some species from one genus to another; species name is given where the toxin(s) has been found only within that species or to highlight changes of the taxonomical classification, where the latter is indicated in parentheses.

^c Cyanotoxins are given roughly in the order of their frequency of reporting.

Sources: Bernard et al. (2016); chapter 2 in the supporting document *Toxic cyanobacteria in water* (Annex 1)

components or by other agents in the water, possibly associated with the bloom (such as other organisms growing in the mucilage of cyanobacterial colonies). Cyanobacteria do not multiply in the human body and hence are not infectious.

Source and occurrence

Cyanobacteria are found in a diverse range of environments, including soils, sea-water and, most notably, freshwater environments. They can occur widely in lakes, reservoirs, ponds and slow-flowing rivers. Blooms may rapidly form scums of very variable duration (hours to weeks). Blooms can occur seasonally, for several months on end, and in some climates perennially. They tend to recur in a given water body. Depending on the species, blooms may result in pronounced turbidity and greenish discoloration of water due to a high density of suspended cells. Planktonic species proliferate if concentrations of nutrients (phosphorus and nitrogen) in water bodies are sufficiently high, and thus blooms typically occur under eutrophic conditions. Other environmental conditions that can promote their growth include low turbulence and warm weather. Benthic species need clear water and thus typically occur where nutrient concentrations are too low for blooms. Where cyanobacterial blooms occur, toxins can be expected, at variable concentrations, depending on the toxin content of the cyanobacterial cells.

Route of exposure

The oral route is usually the most significant source of exposure to cyanotoxins. Exposure can occur through ingestion of drinking-water, during recreation, and in some situations potentially through consumption of food, particularly fish, molluscs and shellfish from bloom-ridden water bodies. Where surface water is used as the source for drinking-water, this may be the most relevant source of chronic exposure to cyanotoxins. Exposure to cyanotoxins via inhalation may also be a relevant pathway for specific recreational activities such as waterskiing or jet-skiing, and for specific occupational activities involving spraying with bloom-contaminated water.

Although dermal exposure to toxins in marine cyanobacteria through recreational activities can be relevant, the compounds causing dermal effects in fresh waters are uncertain (see “Human health effects”, above). For specific population groups, exposure to cyanotoxins may occur by the parenteral route—for example, associated with the use of contaminated water for renal dialysis or infusions.

Prevention and management

Cyanobacteria are most effectively controlled in the context of developing a WSP (see [chapter 4](#)). Control measures to manage potential risks from cyanobacteria and their toxins in drinking-water should include adequate treatment, as well as measures to control development of cyanobacterial blooms. A variety of resource protection and source management actions are available to decrease the probability of bloom occurrence. The most sustainable and effective measure is to reduce nutrient (particularly phosphorus) inputs to water bodies to limit the amount of cyanobacterial biomass that can grow. This is achieved by controlling nutrient loads from sewage effluents, erosion and the amount of manure and fertilizers spread in the catchment. Furthermore, some measures within water bodies can render hydro-physical conditions less suitable for cyanobacteria; some can address phosphorus release from sediments; and some (“biomanipulation”) can support other biota in outcompeting cyanobacteria. Situating the site for raw water abstraction away from bays or water layers in which blooms accumulate may substantially reduce concentrations, and abstraction via bank filtration may remove both cells and toxins effectively. Drinking-water treatment that removes particles, such as conventional water treatment (coagulation, flocculation and filtration), can effectively remove cyanobacteria, including cell-bound cyanotoxins, if the process is optimized to target their removal. Care needs to be taken to avoid cell rupture and toxin release. Other water treatment techniques can remove extracellular cyanotoxins; see the cyanotoxin fact sheets in [chapter 12](#) for more information.

Measures to control cyanobacterial blooms may require years to take effect. While blooms occur, it is therefore important to implement short-term responses to prevent exposure to cyanotoxins. These need to be based on effective monitoring and inspection programmes that prioritize water bodies with conditions rendering blooms likely. Measures also need to be adaptive, increasing sampling and analyses when there is evidence of increasing amounts of cyanobacteria. Early-warning and short-term management responses are most effectively organized within an alert level framework (ALF) ([Figure 11.1](#)). An ALF primarily uses levels of cyanobacterial biomass to trigger responses when biomass reaches levels at which concentrations exceeding cyanotoxin alert values can no longer be excluded. Measures of biomass used in these Guidelines ([Table 11.2](#)) are biovolume or chlorophyll *a*. However, any other indicator of biomass that is locally more useful may be used (e.g. fluorometry, cell counts, satellite imaging or even turbidity) if periodically “calibrated” to ensure that it adequately indicates cyanotoxin concentrations. Wherever possible, toxin analysis should be performed when blooms occur in the source water, particularly when biomass indicators approach Alert Level 2. Toxin concentrations in blooms can vary substantially, and periodically establishing local toxin:biomass ratios improves hazard analysis. Since the alert values for these biomass indicators reflect the

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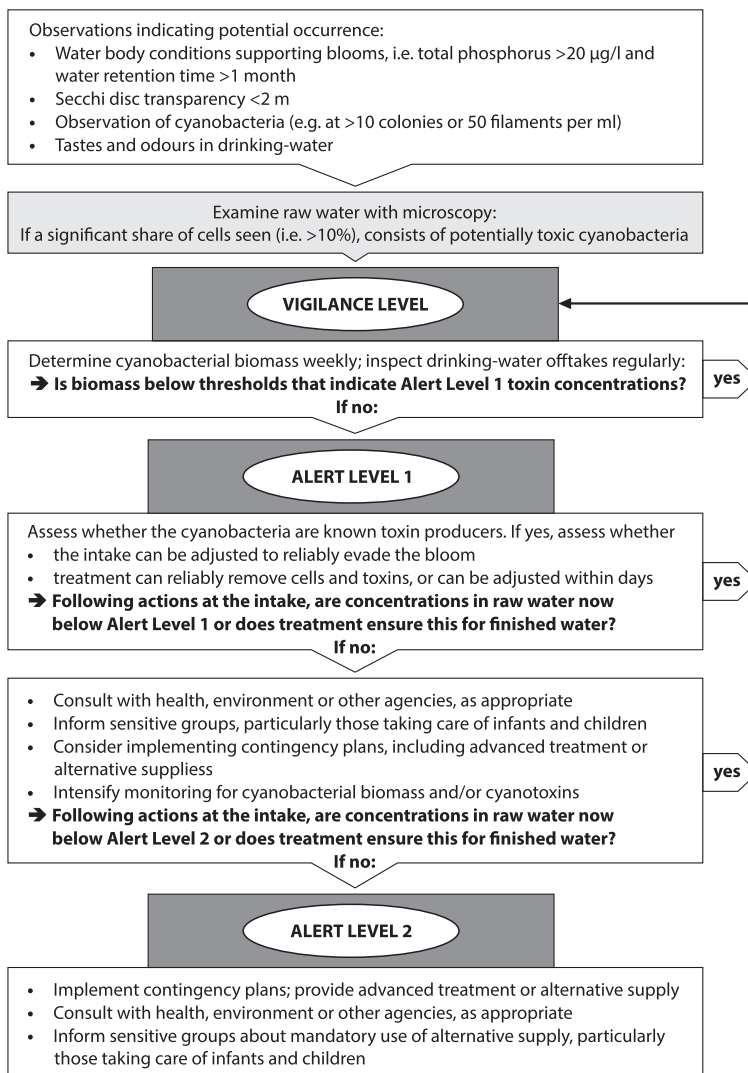


Figure 11.1 Simplified scheme of an alert level framework for cyanotoxins in drinking-water

Note: For the alert levels mentioned, see [Table 11.2](#); for a more detailed alert level framework scheme, see the drinking-water section in chapter 5 of the supporting document *Toxic cyanobacteria in water* ([Annex 1](#)).

upper range of MCs:biomass ratios found in field samples, toxin content in many blooms is much lower and calibrating toxin:biomass ratios through occasional toxin analysis from the local blooms may allow higher biomass thresholds to be used. Also, specifically for CYNs, the fraction dissolved in water may persist well after the CYN producing cyanobacteria have disappeared, and thus alert levels based on cyanobacterial biomass may not reflect the hazard from this toxin. For finished drinking-water, toxin analysis is important to assess the efficacy of removal by treatment.

Table 11.2 Alert levels for cyanobacterial biomass indicators that should trigger management responses

Alert level (AL) thresholds	Alert values for indicators of cyanobacterial biomass		Cyanotoxin alert values ^a			
	Biovolume	Chlorophyll <i>a</i> (with cyanobacteria dominant)	MCs	CYNs	ATXs	STXs
	(mm ³ /l)	(µg/l)			(µg/l)	
AL 1	0.3	1	1 (lifetime pGV)	0.7 (lifetime pGV)	(3 ^b) (1/10 of AL 2)	(0.3 ^b) (1/10 of AL 2)
AL 2	4	12	12 (short-term pGV)	3 (short-term pGV)	30 (short-term provisional reference value; also applicable for acute exposure)	3 (acute GV)

GV, guideline value; pGV, provisional guideline value

^a See the corresponding [chapter 12](#) fact sheets for information on how these values should be interpreted.

^b Note that the AL 1 thresholds for ATXs and STXs are not formal WHO values for lifetime exposure, but merely concentrations 10-fold below those for acute exposure.

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11.6 Indicator organisms

Indicator organisms are used for a range of purposes, including as indicators of:

- faecal pollution in verification and surveillance monitoring;
- the effectiveness of processes such as filtration or disinfection in validation;
- integrity and cleanliness of distribution systems in operational monitoring.

Further discussion on indicator organisms is contained in [section 7.4](#) and the supporting document *Assessing microbial safety of drinking water* ([Annex 1](#)).

Total coliform bacteria

General description

Total coliform bacteria include a wide range of aerobic and facultatively anaerobic, Gram-negative, non-spore-forming bacilli capable of growing in the presence of

relatively high concentrations of bile salts with the fermentation of lactose and production of acid or aldehyde within 24 hours at 35–37 °C. *Escherichia coli* and thermotolerant coliforms are a subset of the total coliform group that can ferment lactose at higher temperatures (see below). As part of lactose fermentation, total coliforms produce the enzyme β -galactosidase. Traditionally, coliform bacteria were regarded as belonging to the genera *Escherichia*, *Citrobacter*, *Klebsiella* and *Enterobacter*, but the group is more heterogeneous and includes a wider range of genera, such as *Serratia* and *Hafnia*. The total coliform group includes both faecal and environmental species.

Indicator value

Total coliforms include organisms that can survive and grow in water. Hence, they are not useful as an indicator of faecal pathogens, but they can be used to assess the cleanliness and integrity of distribution systems and the potential presence of biofilms. However, there are better indicators for these purposes. It has been proposed that total coliforms could be used as a disinfection indicator. However, the test for total coliforms is far slower and less reliable than direct measurement of disinfectant residual. In addition, total coliforms are far more sensitive to disinfection than are enteric viruses and protozoa. Heterotrophic plate count (HPC) measurements detect a wider range of microorganisms and are generally considered a better indicator of distribution system integrity and cleanliness.

Source and occurrence

Total coliform bacteria (excluding *E. coli*) occur in both sewage and natural waters. Some of these bacteria are excreted in the faeces of humans and animals, but many coliforms are heterotrophic and able to multiply in water and soil environments. Total coliforms can also survive and grow in water distribution systems, particularly in the presence of biofilms.

Application in practice

Total coliforms are generally measured in 100 ml samples of water. A variety of relatively simple procedures are available based on the production of acid from lactose or the production of the enzyme β -galactosidase. The procedures include membrane filtration followed by incubation of the membranes on selective media at 35–37 °C and counting of colonies after 24 hours. Alternative methods include most probable number procedures using tubes or microtitre plates and presence/absence tests. Field test kits are available.

Significance in drinking-water

Total coliforms should be absent immediately after disinfection, and the presence of these organisms indicates inadequate treatment. The presence of total coliforms in distribution systems and stored water supplies can reveal regrowth and possible biofilm formation or contamination through ingress of foreign material, including soil or plants.

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***Escherichia coli* and thermotolerant coliform bacteria**

General description

Total coliform bacteria that are able to ferment lactose at 44–45 °C are known as thermotolerant coliforms. In most waters, the predominant genus is *Escherichia*, but some types of *Citrobacter*, *Klebsiella* and *Enterobacter* are also thermotolerant. *Escherichia coli* can be differentiated from the other thermotolerant coliforms by the ability to produce indole from tryptophan or by the production of the enzyme β -glucuronidase. *Escherichia coli* is present in very high numbers in human and animal faeces and is rarely found in the absence of faecal pollution, although there is some evidence for growth in tropical soils. Thermotolerant coliform species other than *E. coli* can include environmental organisms.

Indicator value

Escherichia coli is considered the most suitable indicator of faecal contamination. In most circumstances, populations of thermotolerant coliforms are composed predominantly of *E. coli*; as a result, this group is regarded as a less reliable but acceptable indicator of faecal pollution. *Escherichia coli* (or thermotolerant coliforms) is the first organism of choice in monitoring programmes for verification, including surveillance of drinking-water quality. These organisms are also used as disinfection indicators, but testing is far slower and less reliable than direct measurement of disinfectant residual. In addition, *E. coli* is far more sensitive to disinfection than are enteric viruses and protozoa.

Source and occurrence

Escherichia coli occurs in high numbers in human and animal faeces, sewage and water subject to recent faecal pollution. Water temperatures and nutrient conditions present in drinking-water distribution systems are highly unlikely to support the growth of these organisms.

Application in practice

Escherichia coli (or thermotolerant coliforms) are generally measured in 100 ml samples of water. A variety of relatively simple procedures are available based on the production of acid and gas from lactose or the production of the enzyme β -glucuronidase. The procedures include membrane filtration followed by incubation of the membranes on selective media at 44–45 °C and counting of colonies after 24 hours. Alternative

methods include most probable number procedures using tubes or microtitre plates and presence/absence tests, some for volumes of water larger than 100 ml. Field test kits are available.

Significance in drinking-water

The presence of *E. coli* (or thermotolerant coliforms) provides evidence of recent faecal contamination, and detection should lead to consideration of further action, which could include further sampling and investigation of potential sources such as inadequate treatment or breaches in distribution system integrity.

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Heterotrophic plate counts

A substantial review of the use of HPC is available (see the supporting document *Heterotrophic plate counts and drinking-water safety*; [Annex 1](#)).

General description

HPC measurement detects a wide spectrum of heterotrophic microorganisms, including bacteria and fungi, based on the ability of the organisms to grow on rich growth media, without inhibitory or selective agents, over a specified incubation period and at a defined temperature. The spectrum of organisms detected by HPC testing includes organisms sensitive to disinfection processes, such as coliform bacteria; organisms resistant to disinfection, such as spore formers; and organisms that rapidly proliferate in treated water in the absence of residual disinfectants. The tests detect only a small proportion of the microorganisms that are present in water. The population recovered will differ according to the method and conditions applied. Although standard methods have been developed, there is no single universal HPC measurement. A range of media is available, incubation temperatures used vary from 20 °C to 37 °C and incubation periods range from a few hours to 7 days or more.

Indicator value

The test has little value as an indicator of pathogen presence but can be useful in operational monitoring as a treatment and disinfectant indicator, where the objective is to keep numbers as low as possible. In addition, HPC measurement can be used in assessing the cleanliness and integrity of distribution systems and the presence of biofilms.

Source and occurrence

Heterotrophic microorganisms include both members of the natural (typically non-hazardous) microbial flora of water environments and organisms present in a range of pollution sources. They occur in large numbers in raw water sources. The actual organisms detected by HPC tests vary widely between locations and between consecutive samples. Some drinking-water treatment processes, such as coagulation and sedimentation, reduce the number of HPC organisms in water. However, the organisms proliferate in other treatment processes, such as biologically active carbon and sand filtration. Numbers of HPC organisms are reduced significantly by disinfection practices, such as chlorination, ozonation and UV light irradiation. However, in practice, none of the disinfection processes sterilizes water; under suitable conditions, such as the absence of disinfectant residuals, HPC organisms can grow rapidly. HPC organisms can grow in water and on surfaces in contact with water as biofilms. The principal determinants of growth or “regrowth” are temperature, availability of nutrients, including assimilable organic carbon, lack of disinfectant residual and stagnation.

Application in practice

No sophisticated laboratory facilities or highly trained staff are required. Results on simple aerobically incubated agar plates are available within hours to days, depending on the characteristics of the procedure used.

Significance in drinking-water

After disinfection, numbers would be expected to be low; for most uses of HPC test results, however, actual numbers are of less value than changes in numbers at particular locations. In distribution systems, increasing numbers can indicate a deterioration in cleanliness, possibly stagnation and the potential development of biofilms. HPC can include potentially “opportunistic” pathogens such as *Acinetobacter*, *Aeromonas*, *Flavobacterium*, *Klebsiella*, *Moraxella*, *Serratia*, *Pseudomonas* and *Xanthomonas*. However, there is no evidence of an association of any of these organisms with gastrointestinal infection through ingestion of drinking-water in the general population.

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Intestinal enterococci

General description

Intestinal enterococci are a subgroup of the larger group of organisms defined as faecal streptococci, comprising species of the genus *Streptococcus*. These bacteria are

Gram-positive and relatively tolerant of sodium chloride and alkaline pH levels. They are facultatively anaerobic and occur singly, in pairs or as short chains. Faecal streptococci including intestinal enterococci all give a positive reaction with Lancefield's Group D antisera and have been isolated from the faeces of warm-blooded animals. The subgroup intestinal enterococci consists of the species *Enterococcus faecalis*, *E. faecium*, *E. durans* and *E. hirae*. This group was separated from the rest of the faecal streptococci because they are relatively specific for faecal pollution. However, some intestinal enterococci isolated from water may occasionally also originate from other habitats, including soil, in the absence of faecal pollution.

Indicator value

The intestinal enterococci group can be used as an indicator of faecal pollution. Most species do not multiply in water environments. The numbers of intestinal enterococci in human faeces are generally about an order of magnitude lower than those of *E. coli*. Important advantages of this group are that they tend to survive longer in water environments than *E. coli* (or thermotolerant coliforms), are more resistant to drying and are more resistant to chlorination. Intestinal enterococci have been used in testing of raw water as an indicator of faecal pathogens that survive longer than *E. coli* and in drinking-water to augment testing for *E. coli*. In addition, they have been used to test water quality after repairs to distribution systems or after new mains have been laid.

Source and occurrence

Intestinal enterococci are typically excreted in the faeces of humans and other warm-blooded animals. Some members of the group have also been detected in soil in the absence of faecal contamination. Intestinal enterococci are present in large numbers in sewage and water environments polluted by sewage or wastes from humans and animals.

Application in practice

Enterococci are detectable by simple, inexpensive cultural methods that require basic bacteriology laboratory facilities. Commonly used methods include membrane filtration with incubation of membranes on selective media and counting of colonies after incubation at 35–37 °C for 48 hours. Other methods include a most probable number technique using microtitre plates where detection is based on the ability of intestinal enterococci to hydrolyse 4-methyl-umbelliferyl- β -D-glucoside in the presence of thallium acetate and nalidixic acid within 36 hours at 41 °C.

Significance in drinking-water

The presence of intestinal enterococci provides evidence of recent faecal contamination, and detection should lead to consideration of further action, which could include further sampling and investigation of potential sources such as inadequate treatment or breaches in distribution system integrity.

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Clostridium perfringens

General description

Clostridium spp. are Gram-positive, anaerobic, sulfite-reducing bacilli. They produce spores that are exceptionally resistant to unfavourable conditions in water environments, including UV irradiation, temperature and pH extremes, and disinfection processes, such as chlorination. The characteristic species of the genus, *C. perfringens*, is a member of the normal intestinal flora of 13–35% of humans and other warm-blooded animals. Other species are not exclusively of faecal origin. Like *E. coli*, *C. perfringens* does not multiply in most water environments and is a highly specific indicator of faecal pollution.

Indicator value

In view of the exceptional resistance of *C. perfringens* spores to disinfection processes and other unfavourable environmental conditions, *C. perfringens* has been proposed as an indicator of protozoa in treated drinking-water supplies. In addition, *C. perfringens* can serve as an indicator of faecal pollution that took place previously and hence can indicate sources liable to intermittent contamination. The evidence that *Clostridium* is a reliable indicator for enteric viruses is limited and inconsistent, largely based on one study of reductions by drinking-water treatment. Results should be treated with some caution, as the exceptionally long survival times of its spores are likely to far exceed those of enteric pathogens. *Clostridium perfringens* spores are smaller than protozoan (oo)cysts and may be useful indicators of the effectiveness of filtration processes.

Source and occurrence

Clostridium perfringens and its spores are virtually always present in sewage. The organism does not multiply in water environments. *Clostridium perfringens* is present more often and in higher numbers in the faeces of some animals, such as dogs, than in the faeces of humans and less often in the faeces of many other warm-blooded animals. The numbers excreted in faeces are normally substantially lower than those of *E. coli*.

Application in practice

Vegetative cells and spores of *C. perfringens* are usually detected by membrane filtration techniques in which membranes are incubated on selective media under strict anaerobic conditions. These detection techniques are not as simple and inexpensive as those for other indicators, such as *E. coli* and intestinal enterococci.

Significance in drinking-water

The presence of *C. perfringens* in drinking-water can be an indicator of intermittent faecal contamination. Potential sources of contamination should be investigated. Filtration processes designed to remove enteric viruses or protozoa should also remove *C. perfringens*. Detection in water immediately after treatment should lead to investigation of filtration plant performance.

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Coliphages

General description

Bacteriophages (phages) are viruses that use only bacteria as hosts for replication. Coliphages use *E. coli* and closely related species as hosts and hence can be released by these bacterial hosts into the faeces of humans and other warm-blooded animals. Coliphages used in water quality assessment are divided into the major groups of somatic coliphages and F-RNA coliphages. Differences between the two groups include the route of infection.

Somatic coliphages initiate infection by attaching to receptors permanently located on the cell wall of hosts. They replicate more frequently in the gastrointestinal tract of warm-blooded animals but can also replicate in water environments. Somatic coliphages consist of a wide range of phages (members of the phage families Myoviridae, Siphoviridae, Podoviridae and Microviridae) with a spectrum of morphological types.

F-RNA coliphages initiate infection by attaching to fertility (F-, sex) fimbriae on *E. coli* hosts. These F-fimbriae are produced only by bacteria carrying the fertility (F-) plasmid. As F-fimbriae are produced only in the logarithmic growth phase at temperatures above 30 °C, F-RNA phages are not likely to replicate in environments other than the gastrointestinal tract of warm-blooded animals. F-RNA coliphages comprise a restricted group of closely related phages, which belong to the family Leviviridae, and consist of a single-stranded RNA genome and an icosahedral capsid that is morphologically similar to that of picornaviruses. F-RNA coliphages have been divided into serological types I–IV, which can be identified as genotypes by molecular techniques such as gene probe hybridization. Members of groups I and IV have to date been found exclusively in (non-human) animal faeces, and group III in human faeces. Group II phages have been detected in human faeces and no animal faeces other than

about 28% of porcine faeces. This specificity, which is not fully understood, offers a potential tool to distinguish between faecal pollution of human and animal origin under certain conditions and limitations.

Indicator value

Phages share many properties with human viruses, notably composition, morphology, structure and mode of replication. As a result, coliphages are useful models or surrogates to assess the behaviour of enteric viruses in water environments and the sensitivity to treatment and disinfection processes. In this regard, they are superior to faecal bacteria and could be considered for inclusion in verification and surveillance monitoring where source waters are known to be affected by human faecal waste. However, there is no direct correlation between numbers of coliphages and numbers of enteric viruses. In addition, coliphages cannot be absolutely relied upon as an indicator for enteric viruses. This has been confirmed by the isolation of enteric viruses from treated and disinfected drinking-water supplies that yielded negative results in conventional tests for coliphages.

F-RNA coliphages provide a more specific indicator of faecal pollution than somatic phages. In addition, F-RNA coliphages are better indicators of the behaviour of enteric viruses in water environments and their response to treatment and disinfection processes than are somatic coliphages. This has been confirmed by studies in which the behaviour and survival of F-RNA coliphages, somatic phages, faecal bacteria and enteric viruses have been compared. Available data indicate that the specificity of F-RNA serogroups (genotypes) for human and animal excreta may prove useful in the distinction between faecal pollution of human and animal origin. However, there are shortcomings and conflicting data that need to be resolved, and the extent to which this tool can be applied in practice remains to be elucidated. Owing to the limitations of coliphages, they are best used in laboratory investigations, pilot trials and possibly validation testing. They are not suitable for operational or verification (including surveillance) monitoring.

Source and occurrence

Coliphages are excreted by humans and animals in relatively low numbers. As a result of their respective modes of replication and host specificity, somatic coliphages are generally excreted by most humans and animals, whereas F-RNA coliphages are excreted by a variable and generally lower percentage of humans and animals. Available data indicate that in some communities, F-RNA phages are detectable in 10% of human, 45% of bovine, 60% of porcine and 70% of poultry faecal specimens. Somatic coliphages have been found to generally outnumber F-RNA phages in water environments by a factor of about 5 and cytopathogenic human viruses by a factor of about 500, although these ratios vary considerably. Sewage contains somatic coliphages in numbers of the order of 10^6 – 10^8 per litre; in one study, slaughterhouse wastewater was found to contain somatic coliphages in numbers up to 10^{10} per litre. There are indications that they may multiply in sewage, and somatic coliphages may multiply in natural water environments using saprophytic hosts. Somatic phages and F-RNA phages have been detected in numbers up to 10^5 per litre in lake and river water.

Application in practice

Somatic coliphages are detectable by relatively simple and inexpensive plaque assays, which yield results within 24 hours. Plaque assays for F-RNA coliphages are not quite as simple, because the culture of host bacteria has to be in the logarithmic growth phase at a temperature above 30 °C to ensure that F-fimbriae are present. Plaque assays using large petri dishes have been designed for the quantitative enumeration of plaques in 100 ml samples, and presence/absence tests have been developed for volumes of water of 500 ml or more.

Significance in drinking-water

As coliphages typically replicate in the gastrointestinal tract of humans and warm-blooded animals, their presence in drinking-water provides an indicator of faecal pollution and hence the potential presence of enteric viruses and possibly also other pathogens. The presence of coliphages in drinking-water also indicates shortcomings in treatment and disinfection processes designed to remove enteric viruses. F-RNA coliphages provide a more specific indicator for faecal pollution. The absence of coliphages from treated drinking-water supplies does not confirm the absence of pathogens such as enteric viruses and protozoan parasites.

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***Bacteroides fragilis* phages**

General description

The bacterial genus *Bacteroides* inhabits the human gastrointestinal tract in greater numbers than *E. coli*. Faeces can contain 10^9 – 10^{10} *Bacteroides* per gram compared with 10^6 – 10^8 *E. coli* per gram. *Bacteroides* are rapidly inactivated by environmental oxygen levels, but *Bacteroides* bacteriophages are resistant to unfavourable conditions. Two groups of *B. fragilis* phages are used as indicators in water quality assessment. One is a restricted group of phages that specifically uses *B. fragilis* strain HSP40 as host. This group of phages appears unique, because it is found only in human faeces and not in faeces of animals. The numbers of these phages in sewage appear to be relatively low, and they are almost absent in some geographical areas. The *B. fragilis* HSP40 phages

belong to the family Siphoviridae, with flexible non-contractile tails, double-stranded DNA and capsids with a diameter of up to 60 nm. The second group of *Bacteroides* phages used as indicators is those that use *B. fragilis* strain RYC2056 as a host. This group includes a substantially wider spectrum of phages, occurring in the faeces of humans and many animals. The numbers of these phages in sewage are generally substantially higher than those of *B. fragilis* HSP40 phages.

Indicator value

Bacteroides bacteriophages have been proposed as a possible indicator of faecal pollution as a result of their specific association with faecal material and exceptional resistance to environmental conditions. In particular, *B. fragilis* HSP40 phages are found only in human faeces. *Bacteroides fragilis* phage B40-8, a typical member of the group of *B. fragilis* HSP40 phages, has been found to be more resistant to inactivation by chlorine than poliovirus type 1, simian rotavirus SA11, coliphage f2, *E. coli* and *Streptococcus faecalis*. *Bacteroides fragilis* strain RYC2056 phages seem to be likewise relatively resistant to disinfection. Indicator shortcomings of *B. fragilis* phages include relatively low numbers in sewage and polluted water environments. This applies in particular to *B. fragilis* HSP40 phages. Human enteric viruses have been detected in drinking-water supplies that yielded negative results in conventional tests for *B. fragilis* HSP40 phages. Owing to the limitations of *Bacteroides* bacteriophages, they are best used in laboratory investigations, pilot trials and possibly validation testing.

Source and occurrence

Bacteroides fragilis HSP40 phages are excreted by about 10–20% of humans in certain parts of the world; consequently, their numbers in sewage are substantially lower than those of somatic and even F-RNA coliphages. A mean count of 67 *B. fragilis* HSP40 phages per litre in a sewage-polluted river has been reported. In some parts of the world, *B. fragilis* HSP40 phages would appear not to be detectable in sewage at all. Phages using *B. fragilis* RYC2056 as host are excreted in larger numbers and seem to occur more universally. On average, these phages are excreted by more than 25% of humans. In a survey of water environments, *B. fragilis* HSP40 phages have been found to outnumber cytopathogenic enteric viruses on average by only about 5-fold. Theoretically, wastewaters could be expected to contain higher levels of *B. fragilis* phages than those detected. The reason for the discrepancy may be due to failure in maintaining sufficiently anaerobic conditions during the performance of plaque assays. Improvement of detection methods may result in the recording of higher numbers of *B. fragilis* phages in sewage and polluted water environments.

Application in practice

Disadvantages of *B. fragilis* phages are that the detection methods are more complex and expensive than those for coliphages. Costs are increased by the need to use antibiotics for purposes of selection and to incubate cultures and plaque assays under absolute anaerobic conditions. Results of plaque assays are usually available after about 24 hours compared with about 8 hours for coliphages.

Significance in drinking-water

The presence of *B. fragilis* phages in drinking-water is sound evidence of faecal pollution as well as shortcomings in water treatment and disinfection processes. In addition, the presence of *B. fragilis* HSP40 phages strongly indicates faecal pollution of human origin. However, *B. fragilis* phages occur in relatively low numbers in sewage, polluted water environments and drinking-water supplies. This implies that the absence of *B. fragilis* phages from treated drinking-water supplies does not confirm the absence of pathogens such as enteric viruses and protozoan parasites.

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Enteric viruses

General description

The viruses referred to here are a combined group of those that infect the human gastrointestinal tract and are predominantly transmitted by the faecal–oral route. Well-known members of this group include the enteroviruses, astroviruses, enteric adenoviruses, orthoreoviruses, rotaviruses, caliciviruses and hepatitis A and E viruses. The enteric viruses cover a wide spectrum of viruses, members of which are a major cause of morbidity and mortality worldwide. Members of the group of enteric viruses differ with regard to structure, composition, nucleic acid and morphology. There are also differences in the numbers and frequency of excretion, survival in the environment and resistance to water treatment processes. Enteric viruses have robust capsids that enable them to survive unfavourable conditions in the environment as well as allowing passage through the acidic and proteolytic conditions in the stomach on their way to the duodenum, where they infect susceptible epithelial cells.

Indicator value

The use of enteric viruses as indicator organisms is based on the shortcomings of the existing choices. The survival of faecal bacteria in water environments and the sensitivity to treatment and disinfection processes differ substantially from those of enteric viruses. Monitoring based on one or more representatives of the large group of enteric

viruses themselves would therefore be more valuable for assessment of the presence of any of the enteric viruses in water and the response to control measures.

Source and occurrence

Enteric viruses are excreted by individuals worldwide at a frequency and in numbers that result in many of these viruses being universally present in substantial numbers in wastewater. However, the prevalence of individual members may vary to a large extent as a result of variations in rates of infection and excretion. Much higher numbers would be present during outbreaks.

Application in practice

Practical methods are not yet available for the routine monitoring of water supplies for a broad spectrum of enteric viruses. Viruses that are more readily detectable include members of the enterovirus, adenovirus and orthoreovirus groups. These viruses occur in polluted environments in relatively high numbers and can be detected by reasonably practical and moderate-cost techniques based on cytopathogenic effect in cell culture that yield results within 3–12 days (depending on the type of virus). In addition, progress in technology and expertise is decreasing costs. The cost for the recovery of enteric viruses from large volumes of drinking-water has been reduced extensively. Some techniques—for instance, those based on glass wool adsorption–elution—are inexpensive. The cost of cell culture procedures has also been reduced. Consequently, the cost of testing drinking-water supplies for cytopathogenic viruses has become acceptable for certain purposes. Testing could be used to validate the effectiveness of treatment processes and, in certain circumstances, as part of specific investigations to verify the performance of processes. The incubation times, cost and relative complexity of testing mean that enteric virus testing is not suitable for operational or verification (including surveillance) monitoring. Orthoreoviruses, and at least the vaccine strains of polioviruses detected in many water environments, also have the advantage of not constituting a health risk to laboratory workers.

Significance in drinking-water

The presence of any enteric viruses in drinking-water should be regarded as an indicator for the potential presence of other enteric viruses, is conclusive evidence of faecal pollution and also provides evidence of shortcomings in water treatment and disinfection processes.

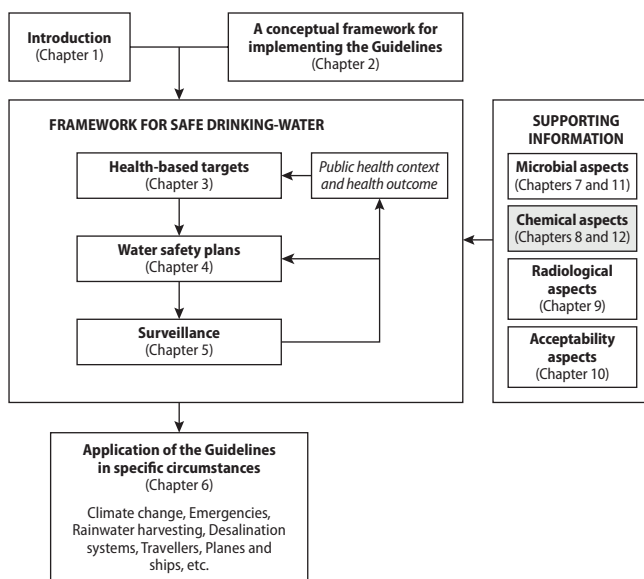
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12

Chemical fact sheets

The background documents referred to in this chapter (as the principal reference for each fact sheet) may be found on the Water, Sanitation, Hygiene and Health web site at <https://www.who.int/teams/environment-climate-change-and-health/water-sanitation-and-health/chemical-hazards-in-drinking-water>. A complete list of references cited in this chapter, including the background documents for each chemical, is provided in Annex 2.



12.1 Chemical contaminants in drinking-water

Acrylamide

Residual acrylamide monomer occurs in polyacrylamide coagulants used in the treatment of drinking-water. In general, the maximum authorized dose of polymer is 1 mg/l. At a monomer content of 0.05%, this corresponds to a maximum theoretical concentration of 0.5 µg/l of the monomer in water. Practical concentrations may be lower by a factor of 2–3. This applies to the anionic and non-ionic polyacrylamides, but residual levels from cationic polyacrylamides may be higher. Polyacrylamides are also used as grouting agents in the construction of drinking-water reservoirs and wells. Human ex-

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Guideline value	0.0005 mg/l (0.5 µg/l)
Occurrence	Concentrations up to a few micrograms per litre occasionally detected in tap water
Basis of guideline value derivation	Combined mammary, thyroid and uterine tumours observed in female rats in a drinking-water study, and using the linearized multistage model
Limit of detection	0.032 µg/l by gas chromatography (GC); 0.2 µg/l by high-performance liquid chromatography (HPLC); 10 µg/l by HPLC with ultraviolet (UV) detection
Treatment performance	Conventional treatment processes do not remove acrylamide. Acrylamide concentrations in drinking-water are usually controlled by limiting either the acrylamide content of polyacrylamide flocculants or the dose used, or both. Advances in analytical techniques are also beginning to allow control by direct measurement (see background document).
Additional comments	Every effort should be made to limit free acrylamide monomer in polyacrylamide used for water treatment, and water suppliers should also make every effort to ensure that residual acrylamide in drinking-water is kept as low as is technically feasible. In particular, if acrylamide is controlled by limiting the amount dosed, overdosing should always be avoided.
Assessment date	2011
Principal references	FAO/WHO (2011) <i>Evaluation of certain contaminants in food</i> WHO (2011) <i>Acrylamide in drinking-water</i>

posure is much greater from food than from drinking-water, owing to the formation of acrylamide in foods (e.g. breads, fried and roasted foods) cooked at high temperatures.

Following ingestion, acrylamide is readily absorbed from the gastrointestinal tract and widely distributed in body fluids. Acrylamide can cross the placenta. It is neurotoxic, affects germ cells and impairs reproductive function. In mutagenicity assays, acrylamide was negative in the Ames test but induced gene mutations in mammalian cells and chromosomal aberrations in vitro and in vivo. In a long-term carcinogenicity study in rats exposed via drinking-water, acrylamide induced scrotal, thyroid and adrenal tumours in males and mammary, thyroid and uterine tumours in females. The International Agency for Research on Cancer (IARC) has placed acrylamide in Group 2A (probably carcinogenic to humans). The Joint Food and Agriculture Organization of the United Nations (FAO)/World Health Organization (WHO) Expert Committee on Food Additives (JECFA) has recently noted concerns regarding the carcinogenicity and neurotoxicity of acrylamide and concluded that dietary exposure should be reduced to as low a level as technically achievable. Recent data have shown that exposure to acrylamide from cooked food is much higher than previously thought. As it is difficult to control the intake of acrylamide from food, it is very important that the acrylamide content of polyacrylamide used as a coagulant aid in water treatment, the most important source of drinking-water contamination by acrylamide, be as low as possible and that polyacrylamide not be overdosed in an attempt to take a shortcut to improving coagulation.

Alachlor

Alachlor (Chemical Abstracts Service [CAS] No. 15972-60-8) is a pre-emergence and post-emergence herbicide used to control annual grasses and many broad-leaved weeds in maize and a number of other crops. It is lost from soil mainly through volatilization, photodegradation and biodegradation. Many alachlor degradation products have been identified in soil. Alachlor was included in the Prior Informed Consent procedure of the Rotterdam Convention on the basis of the final regulatory actions taken by the European Community and by Canada to ban alachlor as a pesticide.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Has been detected in groundwater and surface water; has also been detected in drinking-water at levels below 0.002 mg/l
Basis of guideline value derivation	Calculated by applying the linearized multistage model to data on the incidence of nasal tumours in rats
Limit of detection	0.1 µg/l by gas-liquid chromatography with electrolytic conductivity detection in the nitrogen mode or by capillary column GC with a nitrogen-phosphorus detector
Treatment performance	0.001 mg/l should be achievable using granular activated carbon (GAC)
Assessment date	1993
Principal reference	WHO (2003) <i>Alachlor in drinking-water</i>

On the basis of available experimental data, evidence for the genotoxicity of alachlor is considered to be equivocal. However, a metabolite of alachlor, 2,6-diethylaniline, has been shown to be mutagenic. Available data from two studies in rats clearly indicate that alachlor is carcinogenic, causing benign and malignant tumours of the nasal turbinate, malignant stomach tumours and benign thyroid tumours.

Aldicarb and its metabolites

Aldicarb (CAS No. 116-06-3) is a systemic pesticide used to control nematodes in soil and insects and mites on a variety of crops. It is very soluble in water and highly mobile in soil. It degrades mainly by biodegradation and hydrolysis, persisting for weeks to months.

Guideline value	Aldicarb, aldicarb sulfoxide and aldicarb sulfone: 0.01 mg/l (10 µg/l)
Occurrence	Aldicarb has been frequently found as a contaminant in groundwater in the vicinity of application areas, particularly when associated with sandy soil; concentrations in well water as high as 500 µg/l have been measured; aldicarb sulfoxide and aldicarb sulfone residues are found in an approximately 1:1 ratio in groundwater
Acceptable daily intake (ADI)	0–0.003 mg/kg body weight based on cholinesterase depression in a single oral dose study in human volunteers
Limit of detection	0.001 mg/l by reversed-phase HPLC with fluorescence detection
Treatment performance	0.001 mg/l should be achievable using GAC or ozonation

Guideline value derivation	
• allocation to water	10% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value derived from the 1992 assessment of the Joint FAO/WHO Meeting on Pesticide Residues (JMPR) was very similar to the guideline value derived in the second edition, which was therefore retained.
Assessment date	2003
Principal references	FAO/WHO (1993) <i>Pesticide residues in food—1992 evaluations</i> WHO (2003) <i>Aldicarb in drinking-water</i>

Aldicarb is one of the most acutely toxic pesticides in use, although the only consistently observed toxic effect with both long-term and single-dose administration is acetylcholinesterase inhibition. It is converted to the sulfoxide and sulfone. Aldicarb sulfoxide is a more potent inhibitor of acetylcholinesterase than aldicarb itself, whereas aldicarb sulfone is considerably less toxic than either aldicarb or the sulfoxide. The weight of evidence indicates that aldicarb, aldicarb sulfoxide and aldicarb sulfone are not genotoxic or carcinogenic. IARC has concluded that aldicarb is not classifiable as to its carcinogenicity (Group 3).

Aldrin and dieldrin

Aldrin (CAS No. 309-00-2) and dieldrin (CAS No. 60-57-1) are chlorinated pesticides that are used against soil-dwelling pests, for wood protection and, in the case of dieldrin, against insects of public health importance. Since the early 1970s, many countries have either severely restricted or banned the use of both compounds, particularly in agriculture. The two compounds are closely related with respect to their toxicology and mode of action. Aldrin is rapidly converted to dieldrin under most environmental conditions and in the body. Dieldrin is a highly persistent organochlorine compound that has low mobility in soil, can be lost to the atmosphere and bioaccumulates. Dietary exposure to aldrin/dieldrin is very low and decreasing.

Guideline value	<i>Aldrin and dieldrin (combined)</i> : 0.000 03 mg/l (0.03 µg/l)
Occurrence	Seldom detected in drinking water; concentrations of aldrin and dieldrin in drinking-water normally less than 0.01 µg/l; rarely present in groundwater
Provisional tolerable daily intake (PTDI)	0.1 µg/kg body weight (combined total for aldrin and dieldrin), based on no-observed-adverse-effect levels (NOAELs) of 1 mg/kg diet in the dog and 0.5 mg/kg diet in the rat, which are equivalent to 0.025 mg/kg body weight per day in both species, and applying an uncertainty factor of 250 based on concern about carcinogenicity observed in mice
Limit of detection	0.003 µg/l for aldrin and 0.002 µg/l for dieldrin by GC with electron capture detector (ECD)
Treatment performance	0.02 µg/l should be achievable using coagulation, GAC or ozonation

Guideline value derivation	
• allocation to water	1% of PTDI (In view of the reduction in exposure from food this value is probably very conservative.)
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Aldrin and dieldrin are listed under the Stockholm Convention on Persistent Organic Pollutants. Hence, monitoring may occur in addition to that required by drinking-water guidelines.
Assessment date	2003
Principal references	FAO/WHO (1995) <i>Pesticide residues in food—1994 evaluations</i> WHO (2003) <i>Aldrin and dieldrin in drinking-water</i>

Both compounds are highly toxic in experimental animals, and cases of poisoning in humans have occurred. Aldrin and dieldrin have more than one mechanism of toxicity. The target organs are the central nervous system and the liver. In long-term studies, dieldrin was shown to produce liver tumours in both sexes of two strains of mice. It did not produce an increase in tumours in rats and does not appear to be genotoxic. IARC has classified aldrin and dieldrin in Group 3 (not classifiable as to its carcinogenicity to humans). Exposure through food has decreased significantly with the dramatic reduction in use.

Aluminium

Aluminium is the most abundant metallic element and constitutes about 8% of Earth's crust. Aluminium salts are widely used in water treatment as coagulants to reduce organic matter, colour, turbidity and microorganism levels. Such use may lead to increased concentrations of aluminium in finished water. Where residual concentrations are high, undesirable colour and turbidity may ensue. Concentrations of aluminium at which such problems may occur are highly dependent on a number of water quality parameters and operational factors at the water treatment plant. Aluminium intake from foods, particularly those containing aluminium compounds used as food additives, represents the major route of aluminium exposure for the general public. The contribution of drinking-water to the total oral exposure to aluminium is usually less than 5% of the total intake.

Reason for not establishing a guideline value	A health-based value of 0.9 mg/l could be derived from the JECFA provisional tolerable weekly intake (PTWI), but this value exceeds practicable levels based on optimization of the coagulation process in drinking-water plants using aluminium-based coagulants: 0.1 mg/l or less in large drinking-water supplies and 0.2 mg/l or less in small supplies
Assessment date	2009
Principal references	FAO/WHO (2007) <i>Aluminium</i> (from all sources, including food additives) IPCS (1997) <i>Aluminium</i> WHO (2010) <i>Aluminium in drinking-water</i>

There is little indication that orally ingested aluminium is acutely toxic to humans despite the widespread occurrence of the element in foods, drinking-water and many antacid preparations. It has been hypothesized that aluminium exposure is a risk factor for the development or acceleration of onset of Alzheimer disease in humans. The 1997 WHO Environmental Health Criteria document for aluminium concludes that:

On the whole, the positive relationship between aluminium in drinking-water and AD [Alzheimer disease], which was demonstrated in several epidemiological studies, cannot be totally dismissed. However, strong reservations about inferring a causal relationship are warranted in view of the failure of these studies to account for demonstrated confounding factors and for total aluminium intake from all sources.

Taken together, the relative risks for AD from exposure to aluminium in drinking-water above 100 µg/l, as determined in these studies, are low (less than 2.0). But, because the risk estimates are imprecise for a variety of methodological reasons, a population-attributable risk cannot be calculated with precision. Such imprecise predictions may, however, be useful in making decisions about the need to control exposures to aluminium in the general population.

In 2007, JECFA developed a PTWI for aluminium from all sources of 1 mg/kg body weight. JECFA concluded the following:

... the available studies have many limitations and are not adequate for defining the dose-response relationships. The Committee therefore based its evaluation on the combined evidence from several studies. The relevance of studies involving administration of aluminium compounds by gavage was unclear because the toxicokinetics after gavage were expected to differ from toxicokinetics after dietary administration, and the gavage studies generally did not report total aluminium exposure including basal levels in the feed. The studies conducted with dietary administration of aluminium compounds were considered most appropriate for the evaluation. The lowest LOELs [lowest-observed-effect levels] for aluminium in a range of different dietary studies in mice, rats and dogs were in the region of 50–75 mg/kg bw [body weight] per day expressed as Al.

The Committee applied an uncertainty factor of 100 to the lower end of this range of LOELs (50 mg/kg bw per day expressed as Al) to allow for inter- and intraspecies differences. There are deficiencies in the database, notably the absence of NOELs [no-observed-effect levels] in the majority of the studies evaluated and the absence of long-term studies on the relevant toxicological end-points. The deficiencies are counterbalanced by the probable lower bioavailability of the less soluble aluminium species present in food. Overall, an additional uncertainty factor of three was considered to be appropriate. The Committee confirmed that the resulting health-based guidance value should be expressed as a PTWI, because of the potential for bioaccumulation. The Committee established a PTWI of 1 mg/kg bw for Al, which applies to all aluminium compounds in food, including additives.

A health-based value derived from the JECFA PTWI would be 0.9 mg/l (rounded value), based on an allocation of 20% of the PTWI to drinking-water and assuming a 60 kg adult drinking 2 litres of water per day. However, there remain uncertainties as to the extent of aluminium absorption from drinking-water, which depends on a number of parameters, such as the aluminium salt administered, pH (for aluminium speciation and solubility), bioavailability and dietary factors.

The beneficial effects of the use of aluminium as a coagulant in water treatment are recognized. Taking this into account, and considering the health concerns about aluminium (i.e. its potential neurotoxicity), a practicable level is derived, based on optimization of the coagulation process in drinking-water plants using aluminium-based coagulants, to minimize aluminium levels in finished water.

Several approaches are available for minimizing residual aluminium concentrations in treated water. These include use of optimum pH in the coagulation process, avoiding excessive aluminium dosage, good mixing at the point of application of the coagulant, optimum paddle speeds for flocculation and efficient filtration of the aluminium floc. Under good operating conditions, concentrations of aluminium of 0.1 mg/l or less are achievable in large drinking-water supplies. Small supplies (e.g. those serving fewer than 10 000 people) might experience some difficulties in attaining this level, because the small size of the plant provides little buffering for fluctuation in operation; moreover, such supplies often have limited resources and limited access to the expertise needed to solve specific operational problems. For these small supplies, 0.2 mg/l or less is a practicable level for aluminium in finished water.

As indicated above, a health-based value derived from the JECFA PTWI would be 0.9 mg/l (rounded value) based on an allocation of 20% of the PTWI to drinking-water and assuming a 60 kg adult drinking 2 litres of water per day. However, as also noted above, practicable levels based on optimization of the coagulation process in drinking-water plants using aluminium-based coagulants are less than 0.1 mg/l in large supplies and less than 0.2 mg/l in small supplies. In view of the importance of optimizing coagulation to prevent microbial contamination and the need to minimize deposition of aluminium floc in distribution systems, it is important to ensure that average residuals do not exceed these values.

Ammonia

The term ammonia includes the non-ionized (NH_3) and ionized (NH_4^+) species. Ammonia in the environment originates from metabolic, agricultural and industrial processes and from disinfection with chloramine. Natural levels in groundwater and surface water are usually below 0.2 mg/l. Anaerobic groundwaters may contain up to 3 mg/l. Intensive rearing of farm animals can give rise to much higher levels in surface water. Ammonia contamination can also arise from cement mortar pipe linings. Ammonia in water is an indicator of possible bacterial, sewage and animal waste pollution.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	1993
Principal reference	WHO (2003) <i>Ammonia in drinking-water</i>

Ammonia is a major component of the metabolism of mammals. Exposure from environmental sources is insignificant in comparison with endogenous synthesis of ammonia. Toxicological effects are observed only at exposures above about 200 mg/kg body weight.

Ammonia in drinking-water is not of immediate health relevance, and therefore no health-based guideline value is proposed. However, ammonia can compromise disinfection efficiency, result in nitrite formation in distribution systems, cause the failure of filters for the removal of manganese and cause taste and odour problems (see also [chapter 10](#)).

Anatoxins (cyanobacterial toxins)¹

Anatoxin-a, homoanatoxin-a and their dihydro derivatives (ATXs) are naturally occurring alkaloids produced by strains of various species of cyanobacteria, primarily in freshwater environments.² ATXs have been found in many countries but generally have been reported less often than MCs or CYNs. They have been reported from a number of cyanobacterial genera, including *Anabaena*, *Dolichospermum*, *Aphanizomenon* and *Cuspidothrix*, many of which are primarily benthic (i.e. grow on sediments or other submerged surfaces) (see also [section 11.5](#)). ATXs, like MCs and STXs, usually occur bound to cyanobacterial cells.

Drinking-water is the most likely route of exposure to ATXs where surface water with cyanobacterial blooms is the drinking-water source. Recreational activities in lakes with cyanobacterial blooms may also be a relevant exposure pathway, potentially to high concentrations (see WHO *Guidelines on recreational water quality*, 2021).

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value
Provisional reference value (short-term)*	<i>Total ATXs (sum of all congeners, free plus cell-bound):</i> 0.03 mg/l The reference value is based on data for anatoxin-a only
Occurrence	Concentrations reported usually range well below 1 mg/l; outside of scum areas, they rarely exceed several µg/l. ATXs largely occur cell-bound unless cell damage causes release.
TDI	A formal TDI could not be derived because of database limitations. However, 98 µg/kg bw per day, based on a 28-day study in mice, was selected as a NOAEL. An uncertainty factor of 100 (10 each for inter- and intra-species variability) was applied. An uncertainty factor for database limitations was not applied because of the conservative assumptions used to select the NOAEL (see text following the table for further detail).
Limit of detection	0.05 µg/L by LC-MS/MS and <30 µg/L by HPLC coupled with post-derivatization fluorescence. LC-MS/MS requires quantitative reference standards, which are available for anatoxin-a and for dihydro-anatoxin-a. A receptor-binding assay that is commercially available circumvents this problem and provides more reliable results than HPLC. Prior extraction of cells with freeze-thaw cycles and acidified water or acidified mixtures of methanol/water is necessary for cell-bound ATX; neglecting extraction from cells will lead to dramatic underestimation of concentrations.

¹ As cyanobacteria and their toxins are a concern in many areas and considering the complexities in their management, this chemical fact sheet has been expanded.

² ATXs do not include anatoxin-a(S), a naturally occurring organophosphate with a different mode of neurotoxicity.

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0.15 µg/L for anatoxin-a and 10 µg/L for homoanatoxin-a by commercially available immunoassay kits (ELISA); although these are less precise than LC with the above-mentioned detection methods, they probably capture all anatoxin-a congeners and thus are useful for most monitoring purposes.

Monitoring	The likelihood of blooms can be assessed by understanding water body conditions (in particular, nutrient concentrations, water body depth, water retention time, patterns of mixing and stratification; see section 11.5). Where conditions render blooms likely, visual monitoring of source water (including microscopy for potentially ATX-containing genera) for evidence of increasing cyanobacterial biomass (blooms) is important because biomass can increase rapidly. Exceeding alert values of biomass indicators or ATX concentrations should trigger management responses to prevent exposure to elevated toxin concentrations (see the alert level framework in section 11.5). Analysis of cyanotoxins is particularly useful for validating and optimizing the efficacy of control measures such as riverbank filtration or treatment.
Prevention and treatment	Actions to decrease the probability of bloom occurrence include catchment and source water management, such as reducing nutrient loading or changing reservoir stratification and mixing. Filtration is effective for removing intact cyanobacterial cells, but some dissolved ATX may be present. For this dissolved fraction, oxidation with ozone at sufficient concentrations and contact times, as well as GAC and some PAC applications, are effective. Chlorination is not reliably effective for oxidation of ATXs (see chapters 7–10 of <i>Toxic cyanobacteria in water</i> ; Annex 1).
Reference value derivation	• allocation to water 100% • weight 60 kg adult • consumption 2 litres/day
Additional comments	Total ATXs as gravimetric or molar equivalents should be evaluated against the reference value since these toxins can occur as mixtures. Although the reference value is based on anatoxin-a, limited evidence suggests that all ATXs are similarly toxic. Because ATX is acutely toxic, avoiding any exposure above the reference value is recommended. It is recommended, as a precautionary measure, that bottle-fed infants and small children be provided with an alternative safe drinking-water source (e.g. bottled water that is certified by the responsible authorities) if concentrations are greater than 6 µg/L, even for short periods.
Assessment date	2020
Principal references	WHO (2020) <i>Cyanobacterial toxins: anatoxin-a and analogues</i> Chorus & Welker (2021) <i>Toxic cyanobacteria in water</i>

* Although a formal guideline value cannot be established, the provisional reference value was derived based on limited available studies, recognizing that a “bounding value” may be useful, to provide guidance to Member States in the event of need. Reference values are too uncertain to be used for developing regulations or standards.

ATXs have been much investigated because of the rapid animal deaths that have been observed upon ingestion of cyanobacterial cells in beached scum material or dislodged benthic mats—in particular, dogs that ingest significant amounts. In contrast, high concentrations of dissolved ATXs near drinking-water abstraction sites have not been reported, presumably because ATX released from the cells in benthic mats or scums is usually rapidly diluted.

The (+) enantiomer of anatoxin-a binds with high affinity to nicotinic acetylcholine receptors of nerve cells, causing chronic overstimulation. This can lead to increases in heart rate and blood pressure, as well as fatigue and eventual paralysis of muscles, which can cause death when it occurs in respiratory muscles. Although anatoxin-a is the best studied analogue, limited evidence suggests that homoanatoxin-a and the dihydro derivatives of these compounds bind to the same receptor and may have similar potency to anatoxin-a when administered orally.

The toxicological database on ATXs is not adequate to support derivation of a formal guideline value since a nonlethal dose that caused significant adverse effects was not identified in the available repeated-dose studies. Nevertheless, a “bounding value” may be useful to risk assessors. Based on the limited available studies of acute and subchronic anatoxin-a toxicity, a health-based reference value is provided that is unlikely to cause adverse effects in exposed adults. The reference value is based on a 28-day mouse study that found no effects that were clearly ATX related in any of the doses tested. However, as a mouse died from unexplained causes at each of two lower doses, the dose below these two has been selected as the NOAEL. Two other studies indicate that this selection is very conservative.

Practical considerations

Where nutrient (phosphorus and nitrogen) concentrations are elevated in lakes, reservoirs or slowly flowing rivers, cyanobacteria occur widely. Where their excessive growth leads to high biomass, sometimes termed “bloom” events, ATXs can reach concentrations in raw water that are potentially hazardous to human health. Such blooms tend to recur in the same water bodies. Cells of some cyanobacterial species (e.g. *Anabaena*, *Cuspidothrix*, *Dolichospermum*) may accumulate at the surface as scums. Such accumulations may develop rapidly and may be of very variable duration (hours to weeks). In many circumstances, blooms and accumulations are seasonal, whereas others occur perennially.

Cyanobacteria are most effectively controlled in the context of developing a WSP (see [chapter 4](#)). Control measures to manage potential risks from cyanobacteria, and in particular from their toxins, in drinking-water should include not only adequate treatment, but also measures to control cyanobacterial bloom development. See [section 11.5](#) for more information on cyanobacteria, including further details on monitoring cyanobacterial blooms, the alert level framework, and prevention and management of cyanobacteria in source waters. Effectively minimizing the formation of blooms and locating the raw water intake away from blooms reduce the treatment steps required to remove cyanotoxins.

Drinking-water treatment that removes particles—that is, soil, slow sand or riverbank filtration, conventional water treatment (coagulation, flocculation and

filtration) or dissolved air flotation—can remove cell-bound ATXs effectively. Soil, slow sand and riverbank filtration can also remove dissolved cyanotoxins. For all these processes, it is important that they are optimized to target the removal of cells and dissolved toxins. Chlorination is not reliably effective, whereas ozonation at sufficiently high doses and contact times is effective for degrading dissolved ATXs; however, elevated organic carbon in bloom situations will substantially increase the disinfectant demand. Chlorine dioxide and chloramine are ineffective for degrading ATXs. Both for pre-oxidation and conventional treatment, cell rupture and toxin release should be avoided. GAC and PAC can be effective for removing dissolved ATXs, with efficacy dependent on several factors, including the type of activated carbon, contact times (PAC), flow rates (GAC) and water quality. As the challenges that blooms present for treatment are complex, periodic validation of efficacy during bloom situations and under the specific local conditions is particularly important. Avoiding bloom occurrence and intake is therefore the preferred option.

Antimony

Elemental antimony forms very hard alloys with copper, lead and tin. Antimony compounds have various therapeutic uses. Antimony is used in solders as a replacement for lead, but there is little evidence of any significant contribution to drinking-water concentrations from this source. Total exposure from environmental sources, food and drinking-water is very low compared with occupational exposure.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Concentrations in groundwater less than 0.001 µg/l; concentrations in surface water less than 0.2 µg/l; concentrations in drinking-water appear to be less than 5 µg/l
Tolerable daily intake (TDI)	6 µg/kg body weight, based on a NOAEL of 6.0 mg/kg body weight per day for decreased body weight gain and reduced food and water intake in a 90-day study in which rats were administered potassium antimony tartrate in drinking-water, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation, 10 for the short duration of the study)
Limit of detection	0.01 µg/l by electrothermal atomic absorption spectrometry (AAS); 0.1–1 µg/l by inductively coupled plasma mass spectrometry (ICP-MS); 0.8 µg/l by graphite furnace AAS; 5 µg/l by hydride generation AAS
Treatment performance	Conventional treatment processes do not remove antimony. However, antimony is not normally a raw water contaminant. As the most common source of antimony in drinking-water appears to be dissolution from metal plumbing and fittings, control of antimony from such sources would be by product control.
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	2003
Principal reference	WHO (2003) <i>Antimony in drinking-water</i>

There has been a significant increase in the toxicity data available since the previous review, although much of it pertains to the intraperitoneal route of exposure. The form of antimony in drinking-water is a key determinant of the toxicity, and it would appear that antimony leached from antimony-containing materials would be in the form of the antimony(V) oxo-anion, which is the less toxic form. The subchronic toxicity of antimony trioxide is lower than that of potassium antimony tartrate, which is the most soluble form. Antimony trioxide, owing to its low bioavailability, is genotoxic only in some in vitro tests, but not in vivo, whereas soluble antimony(III) salts exert genotoxic effects in vitro and in vivo. Animal experiments from which the carcinogenic potential of soluble or insoluble antimony compounds may be quantified are not available. IARC has concluded that antimony trioxide is possibly carcinogenic to humans (Group 2B) on the basis of an inhalation study in rats, but that antimony trisulfide was not classifiable as to its carcinogenicity to humans (Group 3). However, chronic oral uptake of potassium antimony tartrate may not be associated with an additional carcinogenic risk, as antimony after inhalation exposure was carcinogenic only in the lung but not in other organs and is known to cause direct lung damage following chronic inhalation as a consequence of overload with insoluble particulates. Although there is some evidence for the carcinogenicity of certain antimony compounds by inhalation, there are no data to indicate carcinogenicity by the oral route.

Arsenic¹

Arsenic is found widely in Earth's crust in oxidation states of -3, 0, +3 and +5, often as sulfides or metal arsenides or arsenates. In water, it is mostly present as arsenate (+5), but in anaerobic conditions, it is likely to be present as arsenite (+3). It is usually present in natural waters at concentrations of less than 1–2 µg/l. However, in waters, particularly groundwaters, where there are sulfide mineral deposits and sedimentary deposits deriving from volcanic rocks, the concentrations can be significantly elevated.

Arsenic is found in the diet, particularly in fish and shellfish, in which it is found mainly in the less toxic organic form. There are only limited data on the proportion of inorganic arsenic in food, but these indicate that approximately 25% is present in the inorganic form, depending on the type of food. Apart from occupational exposure, the most important routes of exposure are through food and drinking-water, including beverages that are made from drinking-water. Where the concentration of arsenic in drinking-water is 10 µg/l or greater, this will be the dominant source of intake. In circumstances where soups or similar dishes are a staple part of the diet, the drinking-water contribution through preparation of food will be even greater.

Provisional guideline value	0.01 mg/l (10 µg/l)

The guideline value is designated as provisional on the basis of treatment performance and analytical achievability.

¹ As arsenic is one of the chemicals of greatest health concern in some natural waters, its chemical fact sheet has been expanded.

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Occurrence	Levels in natural waters generally range between 1 and 2 µg/l, although concentrations may be elevated (up to 12 mg/l) in areas containing natural sources
Basis of guideline value derivation	There remains considerable uncertainty over the actual risks at low concentrations, and available data on mode of action do not provide a biological basis for using either linear or non-linear extrapolation. In view of the practical difficulties in removing arsenic from drinking-water, as well as the practical quantification limit in the region of 1–10 µg/l, the guideline value of 10 µg/l is retained and designated as provisional.
Limit of detection	0.1 µg/l by ICP-MS; 2 µg/l by hydride generation AAS or flame AAS
Treatment performance	It is technically feasible to achieve arsenic concentrations of 5 µg/l or lower using any of several possible treatment methods. However, this requires careful process optimization and control, and a more reasonable expectation is that 10 µg/l should be achievable by conventional treatment (e.g. coagulation).
Assessment date	2011
Principal references	FAO/WHO (2011) <i>Evaluation of certain contaminants in food</i> IARC (1987) <i>Overall evaluations of carcinogenicity</i> IPCS (2001) <i>Arsenic and arsenic compounds</i> ISO (1982) <i>Water quality—determination of total arsenic</i> USNRC (2001) <i>Arsenic in drinking water, 2001 update</i> WHO (2011) <i>Arsenic in drinking-water</i>

Both pentavalent and trivalent soluble arsenic compounds are rapidly and extensively absorbed from the gastrointestinal tract. Metabolism is characterized by 1) reduction of pentavalent to trivalent arsenic and 2) oxidative methylation of trivalent arsenic to form monomethylated, dimethylated and trimethylated products. Methylation of inorganic arsenic facilitates the excretion of inorganic arsenic from the body, as the end-products monomethylarsonic acid and dimethylarsinic acid are readily excreted in urine. There are major qualitative and quantitative interspecies differences in methylation, but in humans and most common laboratory animals, inorganic arsenic is extensively methylated, and the metabolites are excreted primarily in the urine. There is large interindividual variation in arsenic methylation in humans, probably due to a wide difference in the activity of methyltransferases and possible polymorphism. Ingested organoarsenicals are much less extensively metabolized and more rapidly eliminated in urine than inorganic arsenic.

Arsenic has not been demonstrated to be essential in humans. The acute toxicity of arsenic compounds in humans is predominantly a function of their rate of removal from the body. Arsine is considered to be the most toxic form, followed by the arsenites, the arsenates and organic arsenic compounds. Acute arsenic intoxication associated with the ingestion of well water containing very high concentrations (21.0 mg/l) of arsenic has been reported.

Signs of chronic arsenicism, including dermal lesions such as hyperpigmentation and hypo-pigmentation, peripheral neuropathy, skin cancer, bladder and lung cancers and peripheral vascular disease, have been observed in populations ingesting arsenic-contaminated drinking-water. Dermal lesions were the most commonly

observed symptom, occurring after minimum exposure periods of approximately 5 years. Effects on the cardiovascular system were observed in children consuming arsenic-contaminated water (mean concentration 0.6 mg/l) for an average of 7 years.

Numerous epidemiological studies have examined the risk of cancers associated with arsenic ingestion through drinking-water. Many are ecological-type studies, and many suffer from methodological flaws, particularly in the measurement of exposure. However, there is overwhelming evidence that consumption of elevated levels of arsenic through drinking-water is causally related to the development of cancer at several sites. Nevertheless, there remain considerable uncertainty and controversy over both the mechanism of carcinogenicity and the shape of the dose–response curve at low intakes. The International Programme on Chemical Safety (IPCS) concluded that long-term exposure to arsenic in drinking-water is causally related to increased risks of cancer in the skin, lungs, bladder and kidney, as well as other skin changes, such as hyperkeratosis and pigmentation changes. These effects have been demonstrated in many studies using different study designs. Exposure–response relationships and high risks have been observed for each of these end-points. The effects have been most thoroughly studied in Taiwan, China, but there is considerable evidence from studies on populations in other countries as well. Increased risks of lung and bladder cancer and of arsenic-associated skin lesions have been reported to be associated with ingestion of drinking-water at concentrations below 50 µg of arsenic per litre. There is a need for more analytical epidemiological studies to determine the dose–time response for skin lesions, as well as cancer, in order to assist in developing suitable interventions and determining practical intervention policies.

Inorganic arsenic compounds are classified by IARC in Group 1 (carcinogenic to humans) on the basis of sufficient evidence for carcinogenicity in humans and limited evidence for carcinogenicity in animals. Although there is a substantial database on the association between both internal and skin cancers and the consumption of arsenic in drinking-water, there remains considerable uncertainty over the actual risks at low concentrations. In its updated evaluation, the United States National Research Council concluded that “the available mode-of-action data on arsenic do not provide a biological basis for using either a linear or nonlinear extrapolation”. The maximum likelihood estimates, using a linear extrapolation, for bladder and lung cancer for populations in the United States of America (USA) exposed to arsenic at concentrations of 10 µg/l in drinking-water are, respectively, 12 and 18 per 10 000 population for females and 23 and 14 per 10 000 population for males. The actual numbers indicated by these estimated risks would be very difficult to detect by current epidemiological methods. There is also uncertainty over the contribution of arsenic in food—a higher intake of inorganic arsenic from food would lead to a lower risk estimate for water—and the impact of factors such as variation in the metabolism of arsenic and nutritional status. Some studies in areas with arsenic concentrations somewhat above 50 µg/l have not detected arsenic-related adverse effects in the residents. It remains possible that the estimates of cancer risk associated with various arsenic intakes are overestimates. The concentration of arsenic in drinking-water below which no effects can be observed remains to be determined, and there is an urgent need for identifica-

tion of the mechanism by which arsenic causes cancer, which appears to be the most sensitive toxicity end-point.

The practical quantification limit for arsenic is in the region of 1–10 µg/l, and removal of arsenic to concentrations below 10 µg/l is difficult in many circumstances. In view of the practical difficulties in removing arsenic from drinking-water, particularly from small supplies, and the practical quantification limit for arsenic, the guideline value of 10 µg/l is retained as a goal and designated as provisional.

The provisional guideline value of 10 µg/l was previously supported by a JECFA PTWI of 15 µg/kg body weight, assuming an allocation of 20% to drinking-water. However, JECFA recently re-evaluated arsenic and concluded that the existing PTWI was very close to the lower confidence limit on the benchmark dose for a 0.5% response (BMDL_{0.5}) calculated from epidemiological studies and was therefore no longer appropriate. The PTWI was therefore withdrawn. Nevertheless, given that, in many countries, even the provisional guideline value may not be attainable, it is retained on the basis of treatment performance and analytical achievability with the proviso that every effort should be made to keep concentrations as low as reasonably possible.

Practical considerations

A silver diethyldithiocarbamate spectrophotometric method (ISO 6595:1982) is available for the determination of arsenic; the detection limit is about 1 µg/l. Graphite furnace AAS, hydride generation AAS and ICP-MS are more sensitive. HPLC in combination with ICP-MS can also be used to determine various arsenic species.

It is technically feasible to achieve arsenic concentrations of 5 µg/l or lower using any of several possible treatment methods. However, this requires careful process optimization and control, and a more reasonable expectation is that 10 µg/l should be achievable by conventional treatment (e.g. coagulation). For local non-piped water supplies, the first option is often substitution by, or dilution with, microbially safe low-arsenic sources. It may also be appropriate to use alternative sources for drinking and cooking but to use the contaminated sources for purposes such as washing and laundry. There are also an increasing number of effective small-scale treatment techniques, usually based around coagulation and precipitation or adsorption, available at relatively low cost for removal of arsenic from small supplies.

For further guidance on identifying and managing arsenic contamination of drinking-water, see *Arsenic primer* ([Annex 1](#)).

Asbestos

Asbestos is introduced into water by the dissolution of asbestos-containing minerals and ores, as well as from industrial effluents and atmospheric pollution. However, the main source of asbestos in drinking-water is asbestos–cement pipes in the distribution system. Rainwater harvesting can contribute to exposure where water runoff is collected from asbestos–cement roofing that is inadequately sealed. Exfoliation of asbestos fibres from asbestos–cement pipes is related to the aggressiveness of the water supply, although, where pipes are already degraded, actions to control water

corrosivity will not prevent asbestos fibre release from the pipes. Exposure to airborne asbestos released from tap water during showers or humidification is unlikely to be significant for human health.

Reason for not establishing a guideline value	No consistent evidence that ingested asbestos is hazardous to health. Further, epidemiological studies have a number of limitations that would preclude their use for deriving a guideline value.
Occurrence	Concentrations in drinking-water are typically less than 1 million fibres per litre (MFL) and rarely exceed 10 MFL. However, much higher concentrations have been reported in drinking-water supplies where asbestos–cement piping is used, although occurrence data are limited.
Limit of detection	Less than 0.1 MFL in water by transmission electron microscopy
Treatment performance	Conventional treatment, including coagulation and filtration, can effectively remove asbestos fibres (>99%) if operation is optimized.
Additional comments	Where existing asbestos–cement pipes are still in active use, suppliers should map and record their location, assess their condition (including related to water aggressiveness) and determine the most appropriate risk reduction strategies.
	Degradation of asbestos–cement materials in drinking-water supplies should be minimized. As these materials fail or deteriorate significantly, the asbestos–cement materials should be replaced with non-asbestos-containing materials. No new sources of asbestos fibres in drinking-water should be introduced.
	Investigative monitoring of asbestos–cement pipes should be considered to provide additional information on the contribution of older asbestos–cement pipes to numbers, types, sizes and shape of fibres in drinking-water.
Assessment date	2021
Principal reference	WHO (2021) <i>Asbestos in drinking-water</i>

Although asbestos is a known human carcinogen by the inhalation route, findings from epidemiological studies that evaluated the correlation between asbestos exposure via drinking-water and incidence of cancers of the stomach and gastrointestinal tract have been inconsistent, with some studies suggesting a weak positive correlation and others finding no evidence of a correlation. However, in extensive studies in experimental animal species, asbestos has not consistently increased the incidence of tumours of the gastrointestinal tract. Thus, the overall weight of evidence from epidemiological and animal studies does not support the hypothesis that oral exposure to asbestos in drinking-water is associated with an increased risk of developing cancer. Further, uncertainties associated with the epidemiological data preclude the establishment of a health-based guideline value for asbestos in drinking-water.

The primary issue surrounding asbestos–cement in drinking-water is for people working on the outside of the pipes (e.g. cutting pipe) or those repairing roof tiles, because of the risk of inhalation of asbestos dust. Where replacement or repair of

pipes is required, it is essential that appropriate measures are taken to prevent any worker exposure to asbestos dust, including during transport and disposal. Similar to asbestos-cement pipes, it is important that appropriate measures are taken to prevent worker and public exposure to asbestos dust generated during work on asbestos-cement roof tiles.

Atrazine and its metabolites

Atrazine is a selective systemic herbicide of the chlorotriazine class, used for the control of annual broadleaf and grassy weeds. Atrazine and its chloro-s-triazine metabolites—deethyl-atrazine, deisopropyl-atrazine and diaminochlorotriazine—have been found in surface water and groundwater as a result of the use of atrazine as a pre-emergent or early post-emergent herbicide. The metabolite hydroxyatrazine is more commonly detected in groundwater than in surface water.

Guideline values	<i>Atrazine and its chloro-s-triazine metabolites:</i> 0.1 mg/l (100 µg/l) <i>Hydroxyatrazine:</i> 0.2 mg/l (200 µg/l)
Occurrence	Concentrations rarely exceed 2 µg/l and are commonly well below 0.1 µg/l
Group ADI for atrazine and its chloro-s-triazine metabolites	0–0.02 mg/kg body weight based on the NOAEL for atrazine of 1.8 mg/kg body weight per day identified on the basis of luteinizing hormone surge suppression and subsequent disruption of the estrous cycle seen at 3.6 mg/kg body weight per day in a 6-month study in rats, using a safety factor of 100
ADI for hydroxyatrazine	0–0.04 mg/kg body weight based on the NOAEL of 1.0 mg/kg body weight per day identified on the basis of kidney toxicity at 7.8 mg/kg body weight per day in a 24-month study in rats, using a safety factor of 25, based on kinetic considerations
Limit of detection	<i>Atrazine:</i> 1 ng/l, isotope dilution MS with solid-phase extraction; 10 ng/l, GC-MS with solid-phase extraction; 50 ng/l, liquid chromatography (LC)–MS with solid-phase extraction; 100 ng/l, GC with nitrogen–phosphorus detection <i>Metabolites:</i> 5 ng/l, capillary GC with nitrogen thermionic specific detection and HPLC with photodiode array absorption detection following extraction with styrene-divinylbenzene sorbents and elution with acetone
Treatment performance	0.1 µg/l can be achieved using GAC or powdered activated carbon (PAC); bankside filtration and nanofiltration are also effective
Guideline value derivation	• allocation to water 20% of upper limit of ADI • body weight 60 kg adult • consumption 2 litres/day
Additional comments	JMPR considered that the NOAEL for atrazine is protective for the consequences of neuroendocrine and other adverse effects caused by prolonged exposure to atrazine and its chloro-s-triazine metabolites.

JMPR was not able to assess the source allocation of atrazine to drinking-water. As such, the default 20% allocation was chosen, as it will be very conservative in most countries; in addition, it is expected that exposure of the public will be primarily through drinking-water.

Assessment date	2011
Principal references	FAO/WHO (2009) <i>Pesticide residues in food—2007 evaluations</i> WHO (2011) <i>Atrazine and its metabolites in drinking-water</i>

JMPR agreed that it is unlikely that atrazine is genotoxic and concluded that atrazine is not likely to pose a carcinogenic risk to humans, as the mode of carcinogenic action in certain susceptible rat strains is not relevant for human risk assessment. The weight of evidence from the epidemiological studies also did not support a causal association between exposure to atrazine and the occurrence of cancer in humans.

In special studies of reproductive toxicity, exposure of rats during early pregnancy (i.e. the luteinizing hormone-dependent period) caused increased pre-implantation or post-implantation losses, including full-litter resorptions. Attenuation of the luteinizing hormone surge and subsequent disruption of the estrous cycle (characterized by an increase in days in estrus) were observed at and above 3.65 mg/kg body weight per day, with a NOAEL of 1.8 mg/kg body weight per day. The effects on the luteinizing hormone surge and disruption of the estrous cycle were further supported by a number of short-term mechanistic studies. Additional experiments suggested that the effects of atrazine on luteinizing hormone and prolactin secretion are mediated via a hypothalamic site of action. JMPR concluded that atrazine was not teratogenic.

Studies using a variety of test systems in vitro and in vivo indicated that modulation of the immune system occurs after exposure to atrazine. However, effects suggestive of impaired function of the immune system were observed only at doses greater than those shown to affect neuroendocrine function, leading to disruption of the estrous cycle or developmental effects.

The toxicity profiles and mode of action of the chloro-*s*-triazine metabolites are similar to those of atrazine; the potency of these metabolites with regard to their neuroendocrine-disrupting properties appeared to be similar to that of the parent compound.

The metabolite hydroxyatrazine does not have the same mode of action or toxicity profile as atrazine and its chloro-*s*-triazine metabolites. The main effect of hydroxyatrazine was kidney toxicity (owing to its low solubility in water, resulting in crystal formation and a subsequent inflammatory response), and there was no evidence that hydroxyatrazine has neuroendocrine-disrupting properties. There was no evidence of carcinogenicity, and hydroxyatrazine did not show genotoxicity in an adequate range of tests in vitro and in vivo.

Barium

Barium compounds are present in nature as ore deposits and in igneous and sedimentary rocks, and are used in a variety of industrial applications. Barium in water comes primarily from natural sources, although barium also enters the environment from

industrial emissions and anthropogenic uses. Food is the primary source of intake for the non-occupationally exposed population. However, where barium concentrations in water are high, drinking-water may contribute significantly to total intake.

Guideline value	1.3 mg/l (1300 µg/l)
Occurrence	Concentrations in drinking-water are generally below 100 µg/l, although concentrations above 1 mg/l have been measured in drinking-water derived from groundwater
TDI	0.21 mg/kg bw per day, derived by applying an uncertainty factor of 300 to account for intraspecies variation (10), interspecies variation (10) and database deficiencies (3 for the lack of a developmental toxicity study) to a BMDL ₀₅ of 63 mg/kg bw per day for nephropathy in mice in a 2-year study
Limit of detection	0.004–0.8 µg/l by ICP-MS; 1.0 µg/l by ICP-AES
Treatment performance	Ion exchange, lime softening or direct filtration with chemical precipitation may be able to remove barium to below 1 mg/l
Guideline value derivation	
• allocation to water	20% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	As rounding can have significant practical implications at milligram per litre levels, it was concluded that a guideline value with two significant figures was reasonable in this case
	The guideline value derived based on the long-term mouse study is not inconsistent with health-based values that could be derived from limited human studies
Assessment date	2016
Principal references	IPCS (2001) <i>Barium and barium compounds</i> USEPA (2005) Toxicological review of barium and compounds. In support of summary information on the Integrated Risk Information System (IRIS). WHO (2016) <i>Barium in drinking-water</i>

There is no evidence that barium is carcinogenic or genotoxic. Acute hypertension has been observed in case reports, but the effects may be secondary to hypokalaemia. The critical study that had been identified previously for deriving the guideline value has several limitations (e.g. no effect observed at the single dose evaluated, limitations in the exposure methodology and design, no control for important risk factors for hypertension). Another human study that reported no effects on hypertension at 10 mg/l is limited by the small study size and short exposure duration. Barium has been shown to cause nephropathy in laboratory animals, and this was selected as the toxicological end-point of concern for the current guideline.

Bentazone

Bentazone (CAS No. 25057-89-0) is a post-emergence herbicide used for selective control of broadleaf weeds and sedges occurring among a variety of crops. It is highly soluble in water and very resistant to hydrolysis; it is also very mobile in soil. However,

photodegradation occurs in both soil and water. Bentazone may leach from soil into groundwater, particularly during heavy rainfall, and may contaminate surface water through effluents from production plants, drainage waters and actual use in the water (rice fields). Exposure from food is likely to be low.

Reason for not establishing a guideline value	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Health-based value*	0.5 mg/l
Acute health-based value**	20 mg/l
Occurrence	Concentrations up to 120 µg/l in groundwater and up to 14 µg/l in surface water have been measured
ADI	0–0.09 mg/kg bw, based on a NOAEL of 9 mg/kg bw per day for prolonged blood coagulation and clinical chemistry changes indicative of effects on liver and kidney from a 2-year toxicity and carcinogenicity study in rats and application of a safety factor of 100
ARfD	0.5 mg/kg bw, based on a NOAEL of 50 mg/kg bw for decreased motor activity observed in male rats on day 0 in an acute neurotoxicity study and application of a safety factor of 100
Limit of detection	0.1 µg/l by GC with ECD after liquid–liquid extraction; limit of quantification of 0.01 µg/l by LC-MS/MS
Treatment performance	Conventional treatment, including coagulation and filtration, not effective; activated carbon may be effective under certain circumstances
Health-based value derivation	
• allocation to water	20% of upper bound of ADI
• weight	60 kg adult
• consumption	2 litres/day
Acute health-based value derivation	
• allocation to water	100% of ARfD (0.5 mg/kg bw)
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The default allocation factor of 20% has been used to account for the fact that the available food exposure data, which suggest that exposure via this route is low, do not generally include information from developing countries, where exposure via this route may be higher
	As a general principle, the concentration of pesticides in water, including bentazone, should be kept as low as possible and concentrations should not be allowed to increase up to the health-based value
	Further guidance on interpreting the health-based value and deciding when to monitor can be found in section 8.5.3
Assessment date	2016 and 2020

Principal references	WHO (2013) <i>Pesticide residues in food – 2012 evaluations</i> FAO/WHO (2016) <i>Pesticide residues in food – 2016 evaluations</i> WHO (2020) <i>Bentazone in drinking-water</i>

* When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary.

** For more information on acute health-based values, see [section 8.7.5](#).

Bentazone is not carcinogenic in rats or mice, and showed no evidence of genotoxicity in a range of in vitro and in vivo assays. Consistent observations in repeated-dose toxicity studies in mice, rats and dogs are effects on haematology and blood coagulation (e.g. prolongation of prothrombin time and partial thromboplastin time).

Benzene

Benzene is used principally in the production of other organic chemicals. It is present in petrol, and vehicular emissions constitute the main source of benzene in the environment. Benzene may be introduced into water by industrial effluents and atmospheric pollution.

Guideline value	0.01 mg/l (10 µg/l)
Occurrence	Concentrations in drinking-water, when present, generally much less than 5 µg/l
Basis of guideline value derivation	Robust linear extrapolation model (because of statistical lack of fit of some of the data with the linearized multistage model) applied to leukaemia and lymphomas in female mice and oral cavity squamous cell carcinomas in male rats in a 2-year gavage study
Limit of detection	0.2 µg/l by GC with photoionization detection and confirmation by MS
Treatment performance	0.01 mg/l should be achievable using GAC or air stripping
Additional comments	Lower end of estimated range of concentrations in drinking-water corresponding to an upper-bound excess lifetime cancer risk of 10^{-5} (10–80 µg/l) corresponds to the estimate derived from data on leukaemia from epidemiological studies involving inhalation exposure, which formed the basis for the previous guideline value. The previous guideline value is therefore retained.
Assessment date	1993
Principal reference	WHO (2003) <i>Benzene in drinking-water</i>

Acute exposure of humans to high concentrations of benzene primarily affects the central nervous system. At lower concentrations, benzene is toxic to the haematopoietic system, causing a continuum of haematological changes, including leukaemia. Because benzene is carcinogenic to humans, IARC has classified it in Group 1. Haematological abnormalities similar to those observed in humans have been observed in experimental animal species exposed to benzene. In animal studies, benzene was shown to be carcinogenic following both inhalation and ingestion. It induced several types of tumours in both rats and mice in a 2-year carcinogenesis bioassay by gavage in corn

oil. Benzene has not been found to be mutagenic in bacterial assays, but it has been shown to cause chromosomal aberrations in vivo in a number of species, including humans, and to be positive in the mouse micronucleus test.

Beryllium

The primary source of beryllium compounds in water appears to be release from coal burning and other industries using beryllium. Other sources of beryllium in surface water include deposition of atmospheric beryllium and weathering of rocks and soils containing beryllium. Beryllium is not likely to be found in natural water above trace levels as a result of the insolubility of beryllium oxides and hydroxides in the normal pH range.

Reason for not establishing a guideline value	Rarely found in drinking-water at concentrations of health concern
Assessment date	2009
Principal references	IPCS (2001) <i>Beryllium and beryllium compounds</i> WHO (2009) <i>Beryllium in drinking-water</i>

As beryllium is rarely, if ever, found in drinking-water at concentrations of concern, it is not considered necessary to set a formal guideline value.

A health-based value for beryllium in drinking-water of 12 µg/l can be calculated based on an allocation of 20% of the TDI of 2 µg/kg body weight, derived from a long-term study in which dogs exhibited lesions of the small intestine, to drinking-water and assuming a 60 kg adult drinking 2 litres of water per day. This allocation is probably conservative, as the limited data on food indicate that exposure from this source is likely to be well below the TDI.

Although beryllium appears to be found in drinking-water sources and drinking-water at low concentrations, the database on occurrence is limited, and there may be specific circumstances in which concentrations can be elevated due to natural sources where the pH is either below 5 or above 8 or there is high turbidity.

Boron

Boron compounds are used in the manufacture of glass, soaps and detergents and as flame retardants. Naturally occurring boron is present in groundwater primarily as a result of leaching from rocks and soils containing borates and borosilicates. The borate content of surface water can be increased as a result of wastewater discharges, but this use has decreased significantly, and levels of boron in wastewater discharges continue to fall.

Guideline value	2.4 mg/l (2400 µg/l)
Occurrence	Concentrations vary widely and depend on the surrounding geology and wastewater discharges; for most of the world, the concentration of boron in drinking-water is judged to be below 0.5 mg/l

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TDI	0.17 mg/kg body weight, based on a BMDL ₀₅ of 10.3 mg/kg body weight per day for developmental toxicity (decreased fetal body weight in rats) and an uncertainty factor of 60 (10 for interspecies variation and 6 for intraspecies variation)
Limit of detection	0.15 µg/l by ICP-MS; 6–10 µg/l by ICP–atomic emission spectrometry (AES)
Treatment performance	Conventional water treatment (coagulation, sedimentation, filtration) does not significantly remove boron, and special methods need to be used in order to remove boron from waters with high boron concentrations. Ion exchange and reverse osmosis processes may enable substantial reduction but are likely to be prohibitively expensive. Blending with low-boron supplies may be the only economical method to reduce boron concentrations in waters where these concentrations are high.
Guideline value derivation	
• allocation to water	40% of TDI (because intake from other sources is low)
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Because it will be difficult to achieve the guideline value of 2.4 mg/l in some desalinated supplies and in areas with high natural boron levels, local regulatory and health authorities should consider a value in excess of 2.4 mg/l by assessing exposure from other sources.
Assessment date	2009
Principal reference	WHO (2009) <i>Boron in drinking-water</i>

Short- and long-term oral exposures to boric acid or borax in laboratory animals have demonstrated that the male reproductive tract is a consistent target of toxicity. Testicular lesions have been observed in rats, mice and dogs given boric acid or borax in food or drinking-water. Developmental toxicity has been demonstrated experimentally in rats, mice and rabbits. Negative results in a large number of mutagenicity assays indicate that boric acid and borax are not genotoxic. In long-term studies in mice and rats, boric acid and borax caused no increase in tumour incidence.

Bromate

Sodium and potassium bromate are powerful oxidizers used mainly in permanent wave neutralizing solutions and the dyeing of textiles using sulfur dyes. Potassium bromate has also been used as an oxidizer to mature flour during milling, in treating barley in beer making and in fish paste products, although JECFA has concluded that the use of potassium bromate in food processing is not appropriate. Bromate is not normally found in water, but can occur as a result of pollution from industrial sources, sometimes as a consequence of its presence in contaminated soil. However, the main source in drinking-water is its formation during ozonation when the bromide ion is present in water. Bromate may also be formed in hypochlorite solutions produced by electrolysis of bromide-containing salt.

Provisional guideline value	0.01 mg/l (10 µg/l)
	The guideline value is provisional because of limitations in available analytical and treatment methods.

Occurrence	Has been reported in drinking-water with a variety of source water characteristics after ozonation at concentrations ranging from less than 2 to 293 µg/l, depending on bromide ion concentration, ozone dosage, pH, alkalinity and dissolved organic carbon; can also be formed in the electrolytic generation of chlorine and hypochlorite from brine with a high level of bromide contamination
Basis of guideline value derivation	Upper-bound estimate of cancer potency for bromate is 0.19 per mg/kg body weight per day, based on low-dose linear extrapolation (a one-stage Weibull time-to-tumour model was applied to the incidence of mesotheliomas, renal tubule tumours and thyroid follicular tumours in male rats given potassium bromate in drinking-water, using the 12-, 26-, 52- and 77-week interim kill data). A health-based value of 2 µg/l is associated with the upper-bound excess cancer risk of 10 ⁻⁵ . A similar conclusion may be reached through several other methods of extrapolation, leading to values in the range 2–6 µg/l.
Limit of detection	0.2 µg/l by ion chromatography with UV/visible absorbance detection; 0.3 µg/l by ion chromatography with detection by ICP-MS; 1.5 µg/l by ion chromatography with suppressed conductivity detection
Treatment performance	Bromate is difficult to remove once formed. By appropriate control of disinfection conditions, it is possible to achieve bromate concentrations below 0.01 mg/l.
Assessment date	2003
Principal reference	WHO (2003) <i>Bromate in drinking-water</i>

IARC has concluded that although there is inadequate evidence of carcinogenicity in humans, there is sufficient evidence for the carcinogenicity of bromate from high-dose studies in experimental animals; IARC has classified bromate in Group 2B (possibly carcinogenic to humans). Bromate is mutagenic both in vitro and in vivo. At this time, there is not sufficient evidence to conclude as to the mode of carcinogenic action for bromate. Observation of tumours at a relatively early time and the positive response of bromate in a variety of genotoxicity assays suggest that the predominant mode of action at low doses is due to oxidative deoxyribonucleic acid (DNA) damage. Although there is evidence to suggest that the DNA reactivity in kidney tumours may have a non-linear dose–response relationship, there is no evidence to suggest that this same dose–response relationship operates in the development of mesotheliomas or thyroid tumours. Oxidative stress may play a role in the formation of kidney tumours, but the evidence is insufficient to establish lipid peroxidation and free radical production as key events responsible for the induction of kidney tumours. However, emerging evidence points to rapid decomposition of bromate in the gastrointestinal tract, blood and liver, which supports a non-linear dose–response relationship at low doses.

Bromide

Bromide is commonly found in nature along with sodium chloride, owing to their similar physical and chemical properties, but in smaller quantities. Bromide concentrations in seawater range from 65 mg/l to well over 80 mg/l, in fresh water from trace amounts to about 0.5 mg/l and in desalinated waters up to 1 mg/l.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2009
Principal reference	WHO (2009) <i>Bromide in drinking-water</i>

Inorganic bromide was evaluated in 1966 by JMPR, which recommended an ADI of 0–1 mg/kg body weight, based on a minimum pharmacologically effective dosage in humans of about 900 mg of potassium bromide, equivalent to 600 mg of bromide ion. The JMPR ADI was reaffirmed with new data in 1988.

The results of human studies suggest a conservative no-observed-effect level (NOEL) (for marginal effect within normal limits of electroencephalograms in females) of 4 mg/kg body weight per day, giving an ADI of 0–0.4 mg/kg body weight, including a safety factor of 10 for population diversity.

The upper limit of the ADI of 0–0.4 mg/kg body weight yields an acceptable total daily intake of 24 mg/person for a 60 kg person. Assuming a relative source contribution of 50%, the drinking-water value for a 60 kg adult consuming 2 litres/day would be up to 6 mg/l; for a 10 kg child consuming 1 litre/day, the value would be up to 2 mg/l. However, the dietary bromide contribution for a 10 kg child would probably be less than that for an adult. These are reasonably conservative values, and they are unlikely to be encountered in drinking-water supplies.

Bromide can be involved in the reaction between chlorine and naturally occurring organic matter in drinking-water, forming brominated and mixed chloro-bromo by-products, such as trihalomethanes (THMs) and halogenated acetic acids (HAAs), or it can react with ozone to form bromate. The levels of bromide that can result in the formation of these substances are well below the health-based values suggested above. This guidance applies specifically to inorganic bromide ion and not to bromate or organohalogen compounds, for which individual health-based guideline values have been developed.

Brominated acetic acids

Brominated acetic acids are formed during disinfection of water that contains bromide ions and organic matter. Bromide ions occur naturally in surface water and groundwater and exhibit seasonal fluctuations in levels. Bromide ion levels can increase as a result of either saltwater intrusion resulting from drought conditions or pollution. Brominated acetates are generally present in surface water and groundwater distribution systems at mean concentrations below 5 µg/l.

Reason for not establishing guideline values	Available data inadequate to permit derivation of health-based guideline values
Assessment date	2003
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2004) <i>Brominated acetic acids in drinking-water</i>

The database for dibromoacetic acid is considered inadequate for the derivation of a guideline value. There are no systemic toxicity studies of subchronic duration or longer. The database also lacks suitable toxicokinetic studies, a carcinogenicity study, a developmental study in a second species and a multigeneration reproductive toxicity study. Available mutagenicity data suggest that dibromoacetate is genotoxic.

Data are also limited on the oral toxicity of monobromoacetic acid and bromochloroacetic acid. Limited mutagenicity and genotoxicity data give mixed results for monobromoacetic acid and generally positive results for bromochloroacetic acid. Data gaps include subchronic or chronic toxicity studies, multigeneration reproductive toxicity studies, standard developmental toxicity studies and carcinogenicity studies. The available data are considered inadequate to establish guideline values for these chemicals.

Cadmium

Cadmium metal is used in the steel industry and in plastics. Cadmium compounds are widely used in batteries. Cadmium is released to the environment in wastewater, and diffuse pollution is caused by contamination from fertilizers and local air pollution. Contamination in drinking-water may also be caused by impurities in the zinc of galvanized pipes and solders and some metal fittings. Food is the main source of daily exposure to cadmium. The daily oral intake is 10–35 µg. Smoking is a significant additional source of cadmium exposure.

Guideline value	0.003 mg/l (3 µg/l)
Occurrence	Levels in drinking-water usually less than 1 µg/l
PTMI	25 µg/kg body weight, based on the relationship between β_2 -microglobulin excretion in urine and cadmium excretion in urine for individuals who are 50 years of age and older
Limit of detection	0.01 µg/l by ICP-MS; 2 µg/l by flame AAS
Treatment performance	0.002 mg/l should be achievable using coagulation or precipitation softening
Guideline value derivation	
• allocation to water	10% of provisional tolerable monthly intake (PTMI) because of high intake from food
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Although new information indicates that a proportion of the general population may be at increased risk for tubular dysfunction when exposed at the current PTMI, the risk estimates that can be made at present are imprecise.
	It is recognized that the margin between the PTMI and the actual monthly intake of cadmium by the general population is small and that this margin may be even smaller in smokers.
Assessment date	2011
Principal references	FAO/WHO (2011) <i>Evaluation of certain food additives and contaminants</i> WHO (2003) <i>Cadmium in drinking-water</i>

Absorption of cadmium compounds is dependent on the solubility of the compounds. Cadmium accumulates primarily in the kidneys and has a long biological half-life in humans of 10–35 years. There is evidence that cadmium is carcinogenic by the inhalation route, and IARC has classified cadmium and cadmium compounds in Group 2A (probably carcinogenic to humans). However, there is no evidence of carcinogenicity by the oral route and no clear evidence for the genotoxicity of cadmium. The kidney is the main target organ for cadmium toxicity.

In its recent evaluation of cadmium, JECFA found that data relating excretion of the biomarker β_2 -microglobulin in urine to cadmium excretion in urine for individuals who are 50 years of age and older provided the most reliable basis on which to determine a critical concentration of cadmium in the urine. Urinary excretion of less than 5.24 μg of cadmium per gram creatinine was not associated with an increased excretion of β_2 -microglobulin, and the dietary exposure that would result in a urinary cadmium concentration at the breakpoint of 5.24 $\mu\text{g/g}$ creatinine was estimated to be 0.8 $\mu\text{g/kg}$ body weight per day or about 25 $\mu\text{g/kg}$ body weight per month. Because of cadmium's exceptionally long half-life, the previous PTWI of 7 $\mu\text{g/kg}$ body weight was withdrawn, and a PTMI of 25 $\mu\text{g/kg}$ body weight was established.

Carbaryl

Carbaryl (CAS No. 63-25-2) is a broad-spectrum carbamate insecticide that is used to control insect pests in crops, trees and ornamental plants. It also has some uses in public health and veterinary practice. Carbaryl has not been reported in drinking-water; however, it could occur following overspraying or spillage into surface water. Exposure through drinking-water is therefore considered to be low unless in exceptional circumstances. The major route of carbaryl intake for the general population is food, but residues are considered to be relatively low.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2006
Principal references	FAO/WHO (2002) <i>Pesticide residues in food—2001 evaluations</i> WHO (2008) <i>Carbaryl in drinking-water</i>

Carbaryl acts through inhibition of brain cholinesterase, and this is also its primary mode of toxicity. However, carbaryl is also considered to be a non-genotoxic carcinogen in mice, in which it causes vascular tumours in males. On this basis, JMPR established an ADI of 0–0.008 mg/kg body weight. This was based on a lowest-observed-adverse-effect level (LOAEL) of 15 mg/kg body weight per day and application of a safety factor of 2000 (10 for interspecies variation, 10 for intraspecies variation and 20 to reflect the occurrence of the rare and malignant tumour for which a no-effect level could not be identified).

A health-based value of 50 $\mu\text{g/l}$ (rounded value) can be determined from the JMPR ADI of 0–0.008 mg/kg body weight, assuming a 60 kg adult drinking 2 litres of water per day and allowing 20% of the upper limit of the ADI from drinking-

water. However, carbaryl does not appear to be found in drinking-water at significant concentrations, and so it is not considered necessary to propose a formal guideline value.

Carbofuran

Carbofuran (CAS No. 1563-66-2) is used worldwide as a pesticide for many crops. Residues in treated crops are generally very low or not detectable. The physicochemical properties of carbofuran and the few data on occurrence indicate that drinking-water from both groundwater and surface water sources is potentially the major route of exposure.

Guideline value	0.007 mg/l (7 µg/l)
Occurrence	Has been detected in surface water, groundwater and drinking-water, generally at levels of a few micrograms per litre or lower; highest concentration (30 µg/l) measured in groundwater
ADI	0–0.002 mg/kg body weight based on a NOAEL of 0.22 mg/kg body weight per day for acute (reversible) effects in dogs in a short-term (4-week) study conducted as an adjunct to a 13-week study in which inhibition of erythrocyte acetylcholinesterase activity was observed, and using an uncertainty factor of 100
Limit of detection	0.1 µg/l by GC with a nitrogen–phosphorus detector; 0.9 µg/l by reversed-phase HPLC with a fluorescence detector
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Use of a 4-week study was considered appropriate because the NOAEL is based on a reversible acute effect; the NOAEL will also be protective for chronic effects.
Assessment date	1998
Principal references	FAO/WHO (1997) <i>Pesticide residues in food—1996 evaluations</i> WHO (2004) <i>Carbofuran in drinking-water</i>

Carbofuran is highly toxic after acute oral administration. The main systemic effect of carbofuran poisoning in short-term and long-term toxicity studies appears to be cholinesterase inhibition. No evidence of teratogenicity has been found in reproductive toxicity studies. On the basis of available studies, carbofuran does not appear to be carcinogenic or genotoxic.

Carbon tetrachloride

Carbon tetrachloride is used mainly in the production of chlorofluorocarbon refrigerants, foam-blowing agents and solvents. However, since the Montreal Protocol on Substances that Deplete the Ozone Layer (1987) and its amendments (1990 and 1992) established a timetable for the phase-out of the production and consumption of

carbon tetrachloride, manufacture and use have dropped and will continue to drop. Carbon tetrachloride is released mostly into the atmosphere but also into industrial wastewater. Although it readily migrates from surface water to the atmosphere, levels in anaerobic groundwater may remain elevated for months or even years. Although available data on concentrations in food are limited, the intake from air is expected to be much greater than that from food or drinking-water.

Guideline value	0.004 mg/l (4 µg/l)
Occurrence	Concentrations in drinking-water generally less than 5 µg/l
TDI	1.4 µg/kg body weight, based on a NOAEL of 1 mg/kg body weight per day for hepatotoxic effects in a 12-week oral gavage study in rats, adjusting for daily dosing and applying an uncertainty factor of 500 (100 for interspecies and intraspecies variation, 10 for the duration of the study and a modifying factor of 0.5 because it was a bolus study)
Limit of detection	0.1–0.3 µg/l by GC-ECD or GC-MS
Treatment performance	0.001 mg/l should be achievable using air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value is lower than the range of values associated with upper-bound lifetime excess cancer risks of 10^{-4} , 10^{-5} and 10^{-6} calculated by linear extrapolation.
Assessment date	2003
Principal references	IPCS (1999) <i>Carbon tetrachloride</i> WHO (2004) <i>Carbon tetrachloride in drinking-water</i>

The primary targets for carbon tetrachloride toxicity are liver and kidney. In experiments with mice and rats, carbon tetrachloride proved to be capable of inducing hepatomas and hepatocellular carcinomas. The doses inducing hepatic tumours were higher than those inducing cell toxicity. It is likely that the carcinogenicity of carbon tetrachloride is secondary to its hepatotoxic effects. On the basis of available data, carbon tetrachloride can be considered to be a non-genotoxic compound. Carbon tetrachloride is classified by IARC as being possibly carcinogenic to humans (Group 2B): there is sufficient evidence that carbon tetrachloride is carcinogenic in laboratory animals, but inadequate evidence in humans.

Chloral hydrate

Chloral hydrate, or trichloroacetaldehyde, can be formed as a by-product of the chlorination of water containing organic precursor material, such as fulvic and humic acids. It has been found in drinking-water at concentrations of up to 100 µg/l, but concentrations are usually below 10 µg/l. Concentrations are generally higher in surface water than in groundwater, and concentrations appear to increase during distribution.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2004
Principal references	IPCS (2000) <i>Chloral hydrate</i> IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2005) <i>Chloral hydrate in drinking-water</i>

Chloral hydrate is used as an intermediate in the production of insecticides, herbicides and hypnotic drugs. It has also been widely used as a sedative or hypnotic drug in humans at oral doses of up to about 750–1000 mg/day. Although intake from clinical use is considerably higher than intake from drinking-water, clinical exposure is of shorter-term duration.

No epidemiological or carcinogenic studies were found in humans that associated exposure to chloral hydrate with cancer, despite the fact that chloral hydrate has been used for many decades (and still is used) as a sedative and hypnotic drug in adults and children (specifically for dental procedures). IARC classified chloral hydrate as not classifiable as to its carcinogenicity to humans (Group 3), based on inadequate evidence in humans and limited evidence in experimental animals. There is equivocal evidence for the genotoxicity of chloral hydrate.

A health-based value of 0.1 mg/l (rounded figure) can be calculated on the basis of a TDI of 0.0045 mg/kg body weight derived based on an increased incidence of liver histopathology observed in mice in a 2-year drinking-water study, allocating 80% of the TDI to drinking-water (because most exposure to chloral hydrate is from drinking-water) and assuming a 60 kg adult consuming 2 litres of water per day. However, because chloral hydrate usually occurs in drinking-water at concentrations well below those of health concern, it is not considered necessary to derive a guideline value.

Chloral hydrate levels in drinking-water can be controlled by changes to disinfection practice (e.g. enhanced coagulation and softening to remove organic precursor compounds, moving the point of disinfection to reduce the reaction between chlorine and precursor compounds and using chloramines for residual disinfection instead of chlorine) and by GAC treatment.

Chloramines (monochloramine, dichloramine, trichloramine)

Monochloramine, dichloramines and trichloramines are considered by-products of drinking-water chlorination, being formed when chlorine and ammonia are added to water. Monochloramine may also be added to maintain residual disinfection activity in potable water distribution systems. Because higher chloramines are formed only occasionally and cause taste and odour problems at concentrations lower than those at which monochloramine causes taste and odour problems, only monochloramine has been considered for development of a health-based guideline value. Chloramine is rapidly decomposed in the stomach by gastric juice. The use of chloramines for disinfection instead of chlorine reduces the formation of THMs in drinking-water supplies. However, formation of other by-products, such as halo ketones, chloropicrin, cyanogen chloride, HAAs, haloacetonitriles, aldehydes and chlorophenols, has been

reported. Monochloramine, the most abundant chloramine, is recognized as a less effective disinfectant than chlorine and is used as a secondary disinfectant to maintain a residual in distribution systems.

Guideline value	<i>Monochloramine</i> : 3 mg/l (3000 µg/l)
Occurrence	Typical chloramine concentrations of 0.5–2 mg/l are found in drinking-water supplies where chloramine is used as a primary disinfectant or to provide a chlorine residual in the distribution system
TDI	94 µg/kg body weight, based on a NOAEL of 9.4 mg/kg body weight per day, the highest dose administered to male rats in a 2-year United States National Toxicology Program (NTP) drinking-water study (although mean body weights of rats given the highest dose were lower than those of their respective control groups, it is probable that the lower body weights were caused by the unpalatability of the drinking-water)
Limit of detection	10 µg/l by colorimetric methods
Treatment performance	It is possible to reduce the concentration of chloramine effectively to zero (< 0.1 mg/l) by reduction; however, it is normal practice to supply water with a chloramine residual of a few tenths of a milligram per litre to provide some protection against microbial recontamination and limit microbial growth problems during distribution.
Guideline value derivation	
• allocation to water	100% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	An additional uncertainty factor for possible carcinogenicity was not applied because equivocal cancer effects reported in the NTP study in only one species and in only one sex were within the range observed in historical controls.
	Most individuals are able to taste chloramines at concentrations below 5 mg/l, and some at levels as low as 0.3 mg/l.
Assessment date	2003
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2004) <i>Monochloramine in drinking-water</i>
Reason for not establishing guideline values	Available data inadequate to permit derivation of health-based guideline values for <i>dichloramine</i> and <i>trichloramine</i>
Assessment date	1993
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i>

Monochloramine

Although monochloramine has been shown to be mutagenic in some in vitro studies, it has not been found to be genotoxic in vivo. IARC has classified chloramine in Group 3 (not classifiable as to its carcinogenicity to humans). In the NTP bioassay in two spe-

cies, the incidence of mononuclear cell leukaemias in female rats was increased, but no other increases in tumour incidence were observed. IPCS did not consider the increase in mononuclear cell leukaemia to be treatment related.

Dichloramine and trichloramine

Dichloramine and trichloramine have not been extensively studied, and available data are inadequate to permit derivation of health-based guideline values for either of these chemicals. However, these substances can cause taste and odour problems (see [chapter 10](#)) if formation of monochloramine is not controlled adequately.

Chlordane

Chlordane (CAS No. 57-47-9) is a broad-spectrum insecticide that has been used since 1947. Its use has recently been increasingly restricted in many countries, and it is now used mainly to destroy termites by subsurface injection into soil. Chlordane may be a low-level source of contamination of groundwater when applied by subsurface injection. Technical chlordane is a mixture of compounds, with the *cis* and *trans* forms of chlordane predominating. It is very resistant to degradation, highly immobile in soil and unlikely to migrate to groundwater, where it has only rarely been found. It is readily lost to the atmosphere. Although levels of chlordane in food have been decreasing, it is highly persistent and has a high bioaccumulation potential.

Guideline value	0.0002 mg/l (0.2 µg/l)
Occurrence	Has been detected in both drinking-water and groundwater, usually at levels below 0.1 µg/l
PTDI	0.5 µg/kg body weight based on a NOAEL of 50 µg/kg body weight per day for increased liver weights, serum bilirubin levels and incidence of hepatocellular swelling, derived from a long-term dietary study in rats, and using an uncertainty factor of 100 (10 each for interspecies and intraspecies variation)
Limit of detection	0.014 µg/l by GC with ECD
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	1% of PTDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Chlordane is listed under the Stockholm Convention on Persistent Organic Pollutants. Hence, monitoring may occur in addition to that required by drinking-water guidelines.
Assessment date	2003
Principal references	FAO/WHO (1995) <i>Pesticide residues in food—1994 evaluations</i> WHO (2003) <i>Chlordane in drinking-water</i>

In experimental animals, prolonged exposure in the diet causes liver damage. Chlordane produces liver tumours in mice, but the weight of evidence indicates that it is not genotoxic. Chlordane can interfere with cell communication in vitro, a characteristic of many tumour promoters. IARC re-evaluated chlordane in 1991 and concluded that there is inadequate evidence for its carcinogenicity in humans and sufficient evidence for its carcinogenicity in animals, classifying it in Group 2B.

Chloride

Chloride in drinking-water originates from natural sources, sewage and industrial effluents, urban runoff containing de-icing salt and saline intrusion.

The main source of human exposure to chloride is the addition of salt to food, and the intake from this source is usually greatly in excess of that from drinking-water.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Chloride in drinking-water</i>

Excessive chloride concentrations increase rates of corrosion of metals in the distribution system, depending on the alkalinity of the water. This can lead to increased concentrations of metals in the supply.

No health-based guideline value is proposed for chloride in drinking-water. However, chloride concentrations in excess of about 250 mg/l can give rise to detectable taste in water (see [chapter 10](#)).

Chlorine

Chlorine is produced in large amounts and widely used both industrially and domestically as an important disinfectant and bleach. In particular, it is widely used in the disinfection of swimming pools and is the most commonly used disinfectant and oxidant in drinking-water treatment. In water, chlorine reacts to form hypochlorous acid and hypochlorites. Concentrations of chlorate and some perchlorates increase in hypochlorite solutions upon storage at high ambient temperatures or when new hypochlorite is added to old hypochlorite.

Guideline value	5 mg/l (5000 µg/l) for free chlorine
Occurrence	Present in most disinfected drinking-water at concentrations of 0.2–1 mg/l
TDI	150 µg/kg body weight, derived from a NOAEL for the absence of toxicity in rodents ingesting chlorine in drinking-water for 2 years
Limit of detection	0.01 µg/l following pre-column derivatization to 4-bromoacetanilide by HPLC; 10 µg/l as free chlorine by colorimetry; 200 µg/l by ion chromatography

Treatment performance	It is possible to reduce the concentration of chlorine effectively to zero (< 0.1 mg/l) by reduction. However, it is normal practice to supply water with a chlorine residual of a few tenths of a milligram per litre to provide some protection against microbial recontamination and limit microbial growth problems during distribution.
Guideline value derivation	• allocation to water 100% of TDI • weight 60 kg adult • consumption 2 litres/day
Additional comments	The guideline value is conservative, as no adverse effect level was identified in the critical study. Most individuals are able to taste chlorine at the guideline value.
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorine in drinking-water</i>

In humans and experimental animals exposed to chlorine in drinking-water, no specific adverse treatment-related effects have been observed. IARC has classified hypochlorite in Group 3 (not classifiable as to its carcinogenicity to humans).

Chlorine dioxide, chlorite and chlorate

Chlorite and chlorate are DBPs resulting from the use of chlorine dioxide as a disinfectant and for odour and taste control in water. Sodium chlorite and sodium chlorate are both used in the production of chlorine dioxide as well as for other commercial purposes. Chlorite and chlorate are also formed during the decomposition of hypochlorite solutions that are stored for long periods, particularly at warm temperatures. Where hypochlorite or chlorine dioxide is used as a disinfectant, the major route of environmental exposure to chlorite and chlorate is expected to be through drinking-water.

Provisional guideline values	<i>Chlorite</i>: 0.7 mg/l (700 µg/l) <i>Chlorate</i>: 0.7 mg/l (700 µg/l) The guideline values for chlorite and chlorate are designated as provisional because use of aged hypochlorite or of chlorine dioxide as disinfectants may result in the chlorite and chlorate guideline values being exceeded, and difficulties in meeting the guideline values must never be a reason for compromising adequate disinfection
Occurrence	When chlorine dioxide is used as the final disinfectant at typical doses, the resulting chlorite concentration would normally be less than 0.2 mg/l. Chlorate concentrations above 1 mg/l have been reported when hypochlorite was used, but such high concentrations would be unusual unless hypochlorite is stored under adverse conditions.
ADIs	<i>Chlorite</i> : 0–0.03 mg/kg bw based on a NOAEL of 3 mg/kg bw per day for reduced liver weight of F ₀ females and F ₁ males and females in a two-generation reproductive toxicity study in rats and using a safety factor of 100 (10 each for interspecies and intraspecies variability)

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Chlorate: 0–0.01 mg/kg bw based on a BMDL₁₀ of 1.1 mg/kg bw per day for non-neoplastic effects on the thyroid of male rats in a carcinogenicity study and using a safety factor of 100 (10 to allow for intraspecies variability and an additional factor of 10 to allow for the deficiencies in the database; a safety factor for interspecies variation was not considered necessary because humans are likely to be less sensitive than rats to these effects)

Limit of detection	MDLs as low as 0.45 µg/l for chlorite and 0.78 µg/l for chlorate (IC with conductivity detection) and 78 µg/l for chlorine dioxide (UV/visible spectrophotometric method)
Prevention and treatment	When using hypochlorite, the following control approach is recommended to minimize formation of chlorite and chlorate: purchase fresh solutions that are of an appropriate quality, store them in a cool place and out of direct sunlight, and use the hypochlorite as soon as possible after purchase (e.g. within a month, if possible). Further, new hypochlorite solutions should not be added to containers containing old hypochlorite solutions, as this will accelerate chlorate formation. It is possible to reduce the concentration of chlorine dioxide and chlorite effectively to zero (<0.1 mg/l) by reduction; however, it is normal practice to supply water with a chlorine dioxide residual of a few tenths of a milligram per litre to provide some protection against microbial regrowth and limit microbial growth problems during distribution. With chlorine dioxide disinfection, the concentrations of chlorate and chlorite depend on process conditions (in both the chlorine dioxide generator and the water treatment plant) and applied dose of chlorine dioxide. As there is no low-cost option for reducing concentrations of chlorate once it is formed, control of chlorate concentration must rely on preventing its addition (from sodium hypochlorite) or formation (from chlorine dioxide). If chlorine dioxide is used as a pre-oxidant, the resulting chlorite concentration may need to be reduced using ferrous iron, sulfur reducing agents or activated carbon.
Guideline value derivation	• allocation to water 80% of ADI • weight 60 kg adult • consumption 2 litres/day
Additional comments	Concentrations should be maintained as low as reasonably practical, without compromising adequate disinfection. Although a health-based value of 0.3 mg/l could be derived from the ADI for chlorate, in some circumstances, it may not be possible to adequately disinfect potable water and maintain chlorate concentrations at or below the health-based value as chlorate is a byproduct of hypochlorite. Therefore, the previous provisional guideline value is retained. Moreover, even this provisional guideline value may be exceeded when aged hypochlorite is used and difficulties in meeting the guideline value must never be a reason for compromising adequate disinfection.
Assessment date	2016
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2008) <i>Acidified sodium chlorite</i> WHO (2016) <i>Chlorine dioxide, chlorate and chlorite in drinking-water</i>

Chlorine dioxide

Any chlorine dioxide remaining at the consumer’s tap will be reduced to chlorite and chloride upon ingestion. Consequently, a guideline value for chlorine dioxide has not been established. The provisional guideline values for chlorite and chlorate are adequately protective for potential toxicity from chlorine dioxide. The taste and odour threshold for chlorine dioxide is 0.2–0.4 mg/l.

Chlorite

IARC has concluded that chlorite is not classifiable as to its carcinogenicity to humans. The primary and most consistent finding arising from exposure to chlorite in a number of species was oxidative stress resulting in changes in the red blood cells. This observation was supported by a number of biochemical studies conducted in vitro. Studies with human volunteers for up to 12 weeks did not identify any effect on blood parameters at the highest dose tested, 36 µg/kg bw per day.

Chlorate

Although chlorate has also been reported to have effects on red blood cells, the most sensitive effects observed in rats administered sodium chlorate in drinking-water for 21 or 90 days were changes in thyroid histology (e.g. colloid depletion, hypertrophy, incidence and severity of hyperplasia) and in thyroid hormones. As with chlorite, a chlorate dose of 36 µg/kg bw per day for 12 weeks did not result in any adverse effects in human volunteers.

Chloroacetones

1,1-Dichloroacetone is formed from the reaction between chlorine and organic precursors and has been detected in chlorinated drinking-water. Concentrations are estimated to be less than 10 µg/l and usually less than 1 µg/l.

Reason for not establishing guideline values	Available data inadequate to permit derivation of health-based guideline values for any of the chloroacetones
Assessment date	1993
Principal reference	WHO (2003) <i>Chloroacetones in drinking-water</i>

The toxicological data on 1,1-dichloroacetone are very limited, although studies with single doses indicate that it affects the liver.

There are insufficient data at present to permit the setting of guideline values for 1,1-dichloroacetone or any of the other chloroacetones.

Chlorophenols (2-chlorophenol, 2,4-dichlorophenol, 2,4,6-trichlorophenol)

Chlorophenols are present in drinking-water as a result of the chlorination of phenols, as by-products of the reaction of hypochlorite with phenolic acids, as biocides or as degradation products of phenoxy herbicides. Those most likely to occur in drinking-water as by-products of chlorination are 2-chlorophenol, 2,4-dichlorophenol and 2,4,6-trichlorophenol. The taste thresholds for chlorophenols in drinking-water are low.

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Guideline value	2,4,6-Trichlorophenol: 0.2 mg/l (200 µg/l)
Occurrence	Concentrations of chlorophenols in drinking-water usually less than 1 µg/l
Basis of guideline value derivation	Applying the linearized multistage model to leukaemias in male rats observed in a 2-year feeding study (hepatic tumours found in this study were not used for risk estimation because of the possible role of contaminants in their induction)
Limit of detection	0.5–5 µg/l by formation of pentafluorobenzyl ether derivatives; 0.01 µg/l using GC with ECD
Treatment performance	2,4,6-Trichlorophenol concentrations can be reduced using GAC
Additional comments	The guideline value for 2,4,6-trichlorophenol exceeds its lowest reported taste threshold.
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenols in drinking-water</i>
Reason for not establishing guideline values	Available data inadequate to permit derivation of health-based guideline values for 2-chlorophenol and 2,4-dichlorophenol
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenols in drinking-water</i>

2-Chlorophenol

Data on the toxicity of 2-chlorophenol are limited. Therefore, no health-based guideline value has been derived.

2,4-Dichlorophenol

Data on the toxicity of 2,4-dichlorophenol are limited. Therefore, no health-based guideline value has been derived.

2,4,6-Trichlorophenol

2,4,6-Trichlorophenol has been reported to induce lymphomas and leukaemias in male rats and hepatic tumours in male and female mice. The compound has not been shown to be mutagenic in the Ames test but has shown weak mutagenic activity in other in vitro and in vivo studies. IARC has classified 2,4,6-trichlorophenol in Group 2B (possibly carcinogenic to humans).

Chloropicrin

Chloropicrin, or trichloronitromethane, is formed by the reaction of chlorine with humic and amino acids and with nitrophenols. Its formation is increased in the presence of nitrates. Limited data from the USA indicate that concentrations in drinking-water are usually less than 5 µg/l.

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value

Assessment date	1993
Principal reference	WHO (2003) <i>Chloropicrin in drinking-water</i>

Decreased survival and body weights have been reported following long-term oral exposure in laboratory animals. Chloropicrin has been shown to be mutagenic in bacterial tests and in in vitro assays in lymphocytes. Because of the high mortality in a carcinogenesis bioassay and the limited number of end-points examined in the 78-week toxicity study, the available data were considered inadequate to permit the establishment of a guideline value for chloropicrin.

Chlorotoluron

Chlorotoluron (CAS No. 15545-48-9) is a pre emergence or early post emergence herbicide that is slowly biodegradable and mobile in soil. There is only very limited exposure to this compound from food.

Guideline value	0.03 mg/l (30 µg/l)
Occurrence	Detected in drinking-water at concentrations of less than 1 µg/l
TDI	11.3 µg/kg body weight, derived from a NOAEL of 11.3 mg/kg body weight per day for systemic effects in a 2-year feeding study in mice using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for evidence of carcinogenicity)
Limit of detection	0.1 µg/l by separation by reversed-phase HPLC followed by UV and electrochemical detection
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorotoluron in drinking-water</i>

Chlorotoluron is of low toxicity following single, short-term and long-term exposures in experimental animals, but it has been shown to cause an increase in adenomas and carcinomas of the kidneys of male mice given high doses for 2 years. As no carcinogenic effects were reported in a 2-year study in rats, it has been suggested that chlorotoluron has a carcinogenic potential that is both species and sex specific. Chlorotoluron and its metabolites have shown no evidence of genotoxicity.

Chlorpyrifos

Chlorpyrifos (CAS No. 2921-88-2) is a broad-spectrum organophosphorus insecticide used for the control of mosquitoes, flies, various crop pests in soil and on foliage, household pests and aquatic larvae. Although it is not recommended for addition to water for

public health purposes by the WHO Pesticide Evaluation Scheme (WHOPES), it may be used in some countries as an aquatic larvicide for the control of mosquito larvae. Chlorpyrifos is strongly absorbed by soil and does not readily leach from it, degrading slowly by microbial action. It has a low solubility in water and great tendency to partition from aqueous phases into organic phases in the environment.

Guideline value	0.03 mg/l (30 µg/l)
Occurrence	Detected in surface waters in the USA, usually at concentrations below 0.1 µg/l; also detected in groundwater in less than 1% of the wells tested, usually at concentrations below 0.01 µg/l
ADI	0–0.01 mg/kg body weight on the basis of a NOAEL of 1 mg/kg body weight per day for inhibition of brain acetylcholinesterase activity in studies in mice, rats and dogs, using a 100-fold uncertainty factor, and on the basis of a NOAEL of 0.1 mg/kg body weight per day for inhibition of erythrocyte acetylcholinesterase activity in a study of human subjects exposed for 9 days, using a 10-fold uncertainty factor
Limit of detection	1 µg/l by GC using ECD or flame photometric detection
Treatment performance	No data available; should be amenable to treatment by coagulation (10–20% removal), activated carbon adsorption and ozonation
Guideline value derivation	
• allocation to water	10% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	2003
Principal references	FAO/WHO (2000) <i>Pesticide residues in food—1999 evaluations</i> WHO (2003) <i>Chlorpyrifos in drinking-water</i>

JMPR concluded that chlorpyrifos is unlikely to pose a carcinogenic risk to humans. Chlorpyrifos was not genotoxic in an adequate range of studies in vitro and in vivo. In long-term studies, inhibition of cholinesterase activity was the main toxicological finding in all species.

Chromium

Chromium is widely distributed in Earth's crust. It can exist in valences of –2 to +6 but is primarily in its trivalent (III) and hexavalent (VI) forms. In general, with the exception of individuals living close to a point source of contamination, food appears to be the major source of intake.

Guideline value	<i>Total chromium</i> : 0.05 mg/l (50 µg/l)
Occurrence	Total chromium concentrations in drinking-water are usually low (<5 µg/l), although there may be elevated concentrations in areas containing natural sources or as a result of anthropogenic contamination. Few data are available on the speciation of chromium in drinking-water.

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Basis of guideline value derivation	The guideline value is based on achievability by available treatment technologies, measurability by analytical methods, and toxicology. The guideline value is for total chromium because of limitations in currently available analytical methods that preclude reliable speciation of chromium in water. Hyperplasia in the small intestine was considered the most sensitive end-point of tumour formation, and is protective of both noncancer (in the case of chromium(III) and chromium(VI)) and cancer (in the case of chromium(VI)) effects.
Limit of detection	0.08–7 µg/l for total chromium by ICP-AES, ICP-MS, AES or graphite furnace AAS; 0.5 mg/ for total chromium by FAAS. FAAS and electrothermal atomization AAS methods are recommended when concentrations are 0.5–20 mg/l and <0.1 mg/ml, respectively. 0.0044–0.015 µg/l for chromium(VI) by IC with post-column derivatization and UV-visible spectroscopy; 0.5 µg/l for chromium (III) and chromium(VI) using ion chromatography followed by ICP-MS. Determination of chromium species remains difficult; reliable and validated methods to separate analysis of chromium(III) and chromium(VI) are still required.
Treatment performance	Effective central treatment technologies for chromium(III) and chromium(VI) include conventional water treatment (coagulation, sedimentation, filtration—requires chromium(VI) reduction to chromium(III)), adsorption by iron oxides, ion exchange, reverse osmosis and nanofiltration.
Additional comments	As chromium is usually found in drinking-water below the guideline value, in general, monitoring and inclusion in drinking-water standards would only be necessary if there were indications that a problem might exist. Monitoring can usually be limited to treatment works; however, specific pollution events may need to be considered on a case-by-case basis.
Assessment date	2020
Principal reference	WHO (2020) <i>Chromium in drinking-water</i>

Chromium toxicity is dependent on its valence state; chromium(VI) is known to be more toxic than chromium(III). Following oral exposure, chromium(VI) is rapidly and efficiently reduced in the gastrointestinal tract to chromium(III), although a proportion of chromium(VI) may remain available for absorption. IARC has classified chromium(VI) compounds as “carcinogenic to humans” (Group 1) by the inhalation route of exposure; however, data on human carcinogenicity via the oral route are lacking. In a 2-year chronic drinking-water study, increased incidence of tumours of the oral cavity squamous epithelium and of the small intestinal epithelium were reported in rats and mice (respectively) exposed to chromium(VI), at doses of 0.77 mg/kg bw per day in rats and 0.38 mg/kg bw/day in mice. These tumours have been attributed to a threshold mode of action, recognizing an absence of mutagenicity in highly proliferative intestinal tissue following drinking-water exposure, lack of concordance of mutagenicity and tumour development, and early onset of crypt cell proliferation, which is unlikely to result from a fixed mutation. Concentrations in drinking-water rarely approach the guideline value, and evaluation of drinking-water carcinogenicity studies indicate that most environmental concentrations of chromium(VI) are orders

of magnitude below the lowest concentrations tested in the rodent cancer bioassays; therefore, it was determined that the existing guideline value continues to be adequately health-protective.

Copper

Copper is both an essential nutrient and a drinking-water contaminant. It is used to make pipes, valves and fittings and is present in alloys and coatings. Copper sulfate pentahydrate is sometimes added to surface water for the control of algae. Copper concentrations in drinking-water vary widely, with the primary source most often being the corrosion of interior copper plumbing. Levels in running or fully flushed water tend to be low, whereas those in standing or partially flushed water samples are more variable and can be substantially higher (frequently above 1 mg/l). Copper concentrations in treated water often increase during distribution, especially in systems with an acid pH or high-carbonate waters with an alkaline pH. Food and water are the primary sources of copper exposure in developed countries. Consumption of standing or partially flushed water from a distribution system that includes copper pipes or fittings can considerably increase total daily copper exposure, especially for infants fed formula reconstituted with tap water.

Guideline value	2 mg/l (2000 µg/l)
Occurrence	Concentrations in drinking-water range from ≤ 0.005 to > 30 mg/l, primarily as a result of the corrosion of interior copper plumbing
Basis of guideline value derivation	To be protective against acute gastrointestinal effects of copper and provide an adequate margin of safety in populations with normal copper homeostasis
Limit of detection	0.02–0.1 µg/l by ICP-MS; 0.3 µg/l by ICP–optical emission spectroscopy; 0.5 µg/l by flame AAS
Treatment performance	Copper is not removed by conventional treatment processes. However, copper is not normally a raw water contaminant.
Additional comments	For adults with normal copper homeostasis, the guideline value should permit consumption of 2 or 3 litres of water per day, use of a nutritional supplement and copper from foods without exceeding the tolerable upper intake level of 10 mg/day or eliciting an adverse gastrointestinal response. Staining of laundry and sanitary ware occurs at copper concentrations above 1 mg/l. At levels above 2.5 mg/l, copper imparts an undesirable bitter taste to water; at higher levels, the colour of water is also impacted. In most instances where copper tubing is used as a plumbing material, concentrations of copper will be below the guideline value. However, there are some conditions, such as highly acidic or aggressive waters, that will give rise to much higher copper concentrations, and the use of copper tubing may not be appropriate in such circumstances.
Assessment date	2003
Principal references	IPCS (1998) <i>Copper</i> WHO (2004) <i>Copper in drinking-water</i>

IPCS concluded that the upper limit of the acceptable range of oral intake in adults is uncertain but is most likely in the range of several (more than 2 or 3 mg/day), but not many, milligrams per day in adults. This evaluation was based solely on studies of gastrointestinal effects of copper-contaminated drinking-water. The available data on toxicity in experimental animals were not considered helpful in establishing the upper limit of the acceptable range of oral intake owing to uncertainty about an appropriate model for humans, but they help to establish a mode of action for the response. The data on the gastrointestinal effects of copper must be used with caution, as the effects observed are influenced by the concentration of ingested copper to a greater extent than the total mass or dose ingested in a 24-hour period. Recent studies have delineated the threshold for the effects of copper in drinking-water on the gastrointestinal tract, but there is still some uncertainty regarding the long-term effects of copper on sensitive populations, such as carriers of the gene for Wilson disease and other metabolic disorders of copper homeostasis.

Cyanazine

Cyanazine (CAS No. 21725-46-2) is a member of the triazine family of herbicides. It is used as a pre-emergence and post-emergence herbicide for the control of annual grasses and broadleaf weeds. It can be degraded in soil and water by microorganisms and by hydrolysis.

Guideline value	0.0006 mg/l (0.6 µg/l)
Occurrence	Has been detected in surface water and groundwater, usually at concentrations of a few micrograms per litre, although levels as high as 1.3 and 3.5 mg/l have been measured in surface water and groundwater, respectively
TDI	0.198 µg/kg body weight based on a NOAEL of 0.198 mg/kg body weight for hyperactivity in male rats in a 2-year toxicity/carcinogenicity study, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for limited evidence of carcinogenicity)
Limit of detection	0.01 µg/l by GC-MS
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1998
Principal reference	WHO (2003) <i>Cyanazine in drinking-water</i>

On the basis of the available mutagenicity data on cyanazine, evidence for genotoxicity is equivocal. Cyanazine causes mammary gland tumours in rats but not in mice. The mechanism of mammary gland tumour development in rats is currently under investigation and may prove to be hormonal. Cyanazine is also teratogenic in rats at dose levels of 25 mg/kg body weight per day and higher.

Cyanide

Cyanides can be found in some foods, particularly in some developing countries, and they are occasionally found in drinking-water, but usually only at very low concentrations. However, there are occasions on which large spills of cyanide, associated with industry, occur, and these can give rise to very high concentrations in drinking-water source waters, particularly surface waters.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern, except in emergency situations following a spill to a water source
Assessment date	2009
Principal references	IPCS (2004) <i>Hydrogen cyanide and cyanides</i> WHO (2009) <i>Cyanide in drinking-water</i>

Cyanide is highly acutely toxic. It is detoxified in the liver by first-pass metabolism following oral exposure. As a consequence, exposure to a dose spread over a longer period, through a day, for example, will result in lower toxicity, or higher tolerance, than the same dose given in a single bolus dose. Exposure to high doses can give rise to thyroid toxicity as a secondary effect of exposure due to the inhibition of iodine uptake from the thiocyanate generated through the detoxifying action of rhodanese. It is difficult to interpret human data in view of the difficulty in assessing the actual absorbed dose in humans following acute fatal intoxication and the lack of well-conducted studies on sublethal toxicity.

There is a need for guidance regarding concentrations that would not be of concern for public health following short-term exposure to cyanide. However, because cyanide is unlikely to occur in drinking-water at concentrations of health concern, it is considered unnecessary to derive a formal guideline value for short-term exposure to cyanide.

The data on acute exposure to cyanide are unsuitable for use in deriving a health-based value for short-term exposure because of the high uncertainty surrounding the data. Using the NOAEL for effects on the reproductive organs of male rats in a 13-week study and an uncertainty factor of 100, a TDI of 0.045 mg/kg body weight can be derived. Because this health-based value is intended for short-term use and exposure would not exceed 5 days, it is considered to be acceptable to allocate 40% of the TDI to drinking-water to allow for exposure to cyanogenic glycosides in food. Therefore, assuming a 60 kg adult drinking 2 litres of water per day with an allocation of 40% of the TDI to drinking-water, a health-based value of 0.5 mg/l (rounded value) for short-term exposure can be calculated.

This health-based value is well below the level that is normally considered to be of health concern for humans. Cyanide is rapidly detoxified, and exposure spread throughout the day will further reduce the potential for effects. This health-based value would be suitable for use for a limited period of up to 5 days, which is the longest period likely to be required under the circumstances of such an emergency. However, it is probable that, in most circumstances, this value will be highly conservative for short-term exposure.

It should be noted that the lowest reported odour threshold for cyanide in drinking-water is 0.17 mg/l, which is below the short-term health-based value. It is therefore possible that a small number of individuals will detect cyanide by odour at concentrations below the health-based value.

The health-based value relates to total cyanide concentration at the tap, including cyanide from cyanogen chloride in drinking-water as a by-product of disinfection with chlorine. Cyanogen chloride rapidly breaks down to cyanide in the distribution system or when ingested. As the low levels of cyanide normally found in drinking-water are mostly a consequence of the presence of cyanogen chloride, it is not considered necessary to develop a guideline value for long-term exposure to cyanide.

Cyanogen chloride

Cyanogen chloride may be formed as a by-product of chloramination or chlorination of water. It is also formed by the chlorination of cyanide ion present in raw water.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2009
Principal references	IPCS (2004) <i>Hydrogen cyanide and cyanides</i> WHO (2009) <i>Cyanogen chloride in drinking-water</i>

Cyanogen chloride is rapidly metabolized to cyanide in the body. There are few data available on the oral toxicity of cyanogen chloride.

As cyanogen chloride is unlikely to be found in drinking-water at concentrations that are of health concern, it is considered unnecessary to develop a formal guideline value for cyanogen chloride. Instead, for guidance purposes, a health-based value is derived based on cyanide.

Using a NOAEL for cyanide of 4.5 mg/kg body weight per day for minor changes in the testis in a subchronic study in which rats were exposed through their drinking-water and an uncertainty factor of 100, a TDI for cyanide of 0.045 mg/kg body weight (corresponding to a cyanogen chloride dose of 0.11 mg/kg body weight) can be derived. In view of the minor nature of the changes observed and the NOAEL in a previous chronic study, it is not considered necessary to include an additional uncertainty factor to allow for the length of the study. Further, it appears that a dose that may be toxic in acute poisoning would certainly be tolerated under chronic conditions, owing to efficient detoxification. Assuming a 60 kg adult drinking 2 litres of water per day and allowing 20% of the TDI to come from water because of the potential for exposure to cyanogenic glycosides in food, the health-based value for long-term exposure is 0.3 mg/l for cyanide or 0.6 mg/l for cyanogen chloride (rounded values).

Although low concentrations of cyanide in raw waters will be converted to cyanogen chloride by chlorination, cyanogen chloride may also be formed during the production of chloramines in situ as a residual disinfectant to maintain the hygienic condition of the distribution system. It is important that treatment be optimized to minimize the formation of cyanogen chloride while maintaining adequate chloramine residuals where chloramination is practised.

Cylindrospermopsins (cyanobacterial toxins)¹

Cylindrospermopsin (CYN) and its four variants are naturally occurring alkaloids produced by strains of various species of cyanobacteria, primarily in freshwater environments. They have been found in cyanobacteria around the globe and have most frequently been reported from the cyanobacterial genera *Raphidiopsis* (formerly *Cylindrospermopsis*), *Aphanizomenon* and *Chrysosporum* (see also [section 11.5](#)). Unlike MCs, a major fraction of CYNs is often found dissolved in water; particularly at lower temperatures, these toxins may persist even after the producing cyanobacteria are no longer present.

Drinking-water is the most likely route of exposure to CYNs where surface water with cyanobacterial blooms is the drinking-water source. Recreational activities in lakes with cyanobacterial blooms may also be a relevant exposure pathway, potentially to high, usually intermittent concentrations (see WHO *Guidelines on recreational water quality*, 2021). Limited data suggest that CYNs may also accumulate in some food items.

Provisional guideline value (lifetime)	<i>Total CYNs (sum of all congeners, free plus cell-bound): 0.0007 mg/l (0.7 µg/l)</i> The guideline value is provisional because of the high level of uncertainty—it is based on data for only CYN and the database is limited, as reflected in the composite uncertainty factor of 1000
Provisional short-term guideline value	<i>Total CYNs (sum of all congeners, free plus cell-bound): 0.003 mg/l (3 µg/l)</i>
Occurrence	Concentrations reported usually range well below 1 mg/l; outside of scum areas, they rarely exceed several µg/l. Major fractions (up to 90%) can occur dissolved in water
TDI	0.03 µg/kg bw, based on a NOAEL of 30 µg/kg bw per day for renal pathology observed in an 11-week study in mice and applying an uncertainty factor of 1000 (10 each for inter- and intra-species variability and 10 for database deficiencies,* taking into consideration limitations in the database—in particular, limited data on chronic toxicity, reproductive toxicity and carcinogenicity)
Limit of detection	0.065 µg/L by LC-MS/MS or LC (including HPLC) followed by UV/PDA detection. LC-MS/MS has the highest specificity and sensitivity but requires quantitative reference standards for each CYN in the sample. For UV/PDA detection, the signal from one can be used to estimate the concentrations of each. Prior extraction of cells with freeze–thaw cycles and water or methanol/water is necessary for cell-bound CYNs; neglecting extraction from cells will lead to dramatic underestimation of concentrations. 0.05 µg/l by commercially available immunoassay kits (ELISA); although these are less precise than LC with the above-mentioned detection methods, they capture all CYNs and thus are useful for most monitoring purposes.

¹ As cyanobacteria and their toxins are a concern in many areas and considering the complexities in their management, this chemical fact sheet has been expanded.

Monitoring	The likelihood of blooms can be assessed by understanding water body conditions (in particular, nutrient concentrations, water body depth, water retention time, patterns of mixing and stratification; see section 11.5). Where conditions render blooms likely, visual monitoring of source water (including microscopy for potentially CYN-containing genera) for evidence of increasing cyanobacterial biomass (blooms) is important because biomass can increase rapidly. Exceeding alert values for biomass indicators or CYN concentrations should trigger responses to prevent exposure to elevated toxin concentrations (see the alert level framework in section 11.5). As a major fraction of CYN may occur dissolved in water and be persistent, if blooms of potentially CYN-producing genera have been observed, monitoring should also include toxin analysis for CYNs, if possible. Analysis of cyanotoxins is particularly useful for validating and optimizing the efficacy of control measures such as riverbank filtration or treatment.
Prevention and treatment	Actions to decrease the probability of bloom occurrence include catchment and source water management, such as reducing nutrient loading or changing reservoir stratification and mixing. Filtration is effective for removing intact cyanobacterial cells. For the often large dissolved fraction of CYNs, oxidation with chlorine or ozone at sufficient concentrations and contact times, as well as GAC and some PAC applications, are effective (see chapters 7–10 of <i>Toxic cyanobacteria in water</i> ; Annex 1).
Guideline value derivation	• allocation to water 80% of TDI; for short-term exposure, 100% of TDI • weight 60 kg adult • consumption 2 litres/day
Additional comments	Total CYNs as gravimetric or molar equivalents should be evaluated against the guideline values since CYNs usually occur as mixtures. Although the guideline values are based on CYN, limited evidence suggests that other CYN congeners have similar toxicity to CYN. The provisional short-term drinking-water guideline value is intended to indicate the extent to which the lifetime value can be exceeded for periods of up to about 2 weeks until water treatment can be augmented to bring the concentration of CYNs back under control. It is not intended to allow repeated seasonal exceedances of the lifetime value. It is recommended, as a precautionary measure, that bottle-fed infants and small children be provided with an alternative safe drinking-water source (e.g. bottled water that is certified by the responsible authorities) if concentrations are greater than 0.7 µg/L, even for short periods.
Assessment date	2020
Principal references	WHO (2020) <i>Cyanobacterial toxins: cylindrospermopsins</i> Chorus & Welker (2021) <i>Toxic cyanobacteria in water</i>

* For the short-term guideline value, a database uncertainty factor was applied due to the limited database, including lack of data on reproductive effects after oral exposure, and evidence of potential in vivo genotoxicity of CYNs. An uncertainty factor of 3 was applied because lack of chronic toxicity data does not affect derivation of a guideline value for short-term exposures.

CYN is a potent inhibitor of protein synthesis, and also has cytochrome P450-dependent effects on other processes—for example, DNA damage and induction of cellular stress responses. CYN was the likely cause of a mass human poisoning incident in Australia in 1979.

The provisional guideline values for CYNs are based on a study in male mice using CYN, corroborated by two other studies in both sexes of different strains of mice. The studies demonstrated that a range of organs were adversely affected. The kidneys were identified as the most sensitive organs in these studies, and one study found gender-based differences in the sensitivity of the liver and kidneys. Other studies in mice have demonstrated CYN-induced DNA damage in various organs.

Practical considerations

Where nutrient (phosphorus and nitrogen) concentrations are elevated in lakes, reservoirs or slowly flowing rivers, cyanobacteria occur widely. Where their excessive growth leads to high biomass, sometimes termed “bloom” events, CYNs can reach concentrations in raw water that are potentially hazardous to human health. Such blooms tend to recur in the same water bodies and to be seasonal, whereas others occur perennially. Although some CYN-producing cyanobacteria form scums or accumulate at the thermocline of thermally stratified reservoirs, they tend to not be as pronounced as the scums and accumulations of MC-producing cyanobacteria.

Cyanobacteria are most effectively controlled in the context of developing a WSP (see [chapter 4](#)). Control measures to manage potential risks from cyanobacteria, and in particular from their toxins, in drinking-water should include not only adequate treatment, but also measures to control cyanobacterial bloom development. See [section 11.5](#) for more information on cyanobacteria, including further details on monitoring cyanobacterial blooms, the alert level framework, and prevention and management of cyanobacteria in source waters. Effectively minimizing the formation of blooms and locating the raw water intake away from blooms reduce the treatment steps required to remove cyanotoxins.

Drinking-water treatment that removes particles—that is, soil, slow sand or river-bank filtration, conventional water treatment (coagulation, flocculation and filtration or dissolved air flotation) or dissolved air flotation—can remove cell-bound CYNs effectively. Soil, slow sand and riverbank filtration can also remove dissolved cyanotoxins. For all these processes it is important that they are optimized to target the removal of cells and dissolved toxins. While for both pre-oxidation and conventional treatment, cell rupture and toxin release should be avoided, treatment also needs to target the typically large fraction of dissolved CYN. Chlorination and ozonation at sufficiently high doses and contact times are effective for degrading dissolved CYNs; however, elevated organic carbon in bloom situations will substantially increase the disinfectant demand. Chlorine dioxide and chloramine are ineffective for degrading CYNs. GAC and PAC can be effective for removing dissolved CYNs, with efficacy dependent on several factors, including the type of activated carbon, contact times (PAC), flow rates (GAC) and water quality. As the challenges that blooms present for treatment are complex, periodic validation of efficacy during bloom situations and under the specific local conditions is particularly important. Avoiding bloom occurrence and intake is therefore the preferred option.

2,4-D

The term 2,4-D is used here to refer to the free acid, 2,4-dichlorophenoxyacetic acid (CAS No. 94-75-7). Commercial 2,4-D products are marketed as the free acid, alkali and amine salts and ester formulations. 2,4-D itself is chemically stable, but its esters are rapidly hydrolysed to the free acid. 2,4-D is a systemic herbicide used for control of broad-leaved weeds, including aquatic weeds. 2,4-D is rapidly biodegraded in the environment. Residues of 2,4-D in food rarely exceed a few tens of micrograms per kilogram.

Guideline value	0.03 mg/l (30 µg/l)
Occurrence	Levels in water usually below 0.5 µg/l, although concentrations as high as 30 µg/l have been measured
ADI	0–0.01 mg/kg body weight for the sum of 2,4-D and its salts and esters, expressed as 2,4-D, on the basis of a NOAEL of 1 mg/kg body weight per day in a 1-year study of toxicity in dogs (for a variety of effects, including histopathological lesions in kidneys and liver) and a 2-year study of toxicity and carcinogenicity in rats (for renal lesions)
Limit of detection	0.1 µg/l by gas–liquid chromatography with electrolytic conductivity detection
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value applies to 2,4-D, as salts and esters of 2,4-D are rapidly hydrolysed to the free acid in water.
Assessment date	1998
Principal references	FAO/WHO (1997) <i>Pesticide residues in food—1996 evaluations</i> WHO (2003) <i>2,4-D in drinking-water</i>

Epidemiological studies have suggested an association between exposure to chlorophenoxy herbicides, including 2,4-D, and two forms of cancer in humans: soft tissue sarcomas and non-Hodgkin lymphoma. The results of these studies, however, are inconsistent; the associations found are weak, and conflicting conclusions have been reached by the investigators. Most of the studies did not provide information on exposure specifically to 2,4-D, and the risk was related to the general category of chlorophenoxy herbicides, a group that includes 2,4,5-trichlorophenoxyacetic acid (2,4,5-T), which was potentially contaminated with dioxins. JMPR concluded that it was not possible to evaluate the carcinogenic potential of 2,4-D on the basis of the available epidemiological studies. JMPR also concluded that 2,4-D and its salts and esters are not genotoxic. The toxicity of the salts and esters of 2,4-D is comparable to that of the acid.

2,4-DB

The half-lives for degradation of chlorophenoxy herbicides, including 2,4-DB, or 2,4-dichlorophenoxybutyric acid (CAS No. 94-82-6), in the environment are in the order of several days. Chlorophenoxy herbicides are not often found in food.

Guideline value	0.09 mg/l (90 µg/l)
Occurrence	Chlorophenoxy herbicides not frequently found in drinking-water; when detected, concentrations usually no greater than a few micrograms per litre
TDI	30 µg/kg body weight, based on a NOAEL of 3 mg/kg body weight per day for effects on body and organ weights, blood chemistry and haematological parameters in a 2-year study in rats, with an uncertainty factor of 100 (for interspecies and intraspecies variation)
Limit of detection	1 µg/l to 1 mg/l for various methods commonly used for the determination of chlorophenoxy herbicides in water, including solvent extraction, separation by GC, gas-liquid chromatography, thin-layer chromatography or HPLC, with ECD or UV detection
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The NOAEL used in the guideline value derivation is similar to the NOAEL of 2.5 mg/kg body weight per day obtained in a short-term study in dogs and the NOAEL for hepatocyte hypertrophy of 5 mg/kg body weight per day obtained in a 3-month study in rats.
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenoxy herbicides (excluding 2,4-D and MCPA) in drinking-water</i>

Chlorophenoxy herbicides, as a group, have been classified in Group 2B (possibly carcinogenic to humans) by IARC. However, the available data from studies in exposed populations and experimental animals do not permit assessment of the carcinogenic potential to humans of any specific chlorophenoxy herbicide. Therefore, drinking-water guidelines for these compounds are based on a threshold approach for other toxic effects.

DDT and metabolites

The structure of dichlorodiphenyltrichloroethane, or DDT (CAS No. 107917-42-0), permits several different isomeric forms; commercial products consist predominantly of *p,p'*-DDT. Its use has been restricted or banned in several countries, although DDT is still used in some countries for the control of vectors that transmit yellow fever, sleeping sickness, typhus, malaria and other insect-transmitted diseases. DDT and its metabolites are persistent in the environment and resistant to complete degrada-

tion by microorganisms. Food is the major source of intake of DDT and related compounds for the general population, although exposure has significantly decreased as a consequence of the greatly reduced use of DDT for all except specialist applications.

Guideline value	0.001 mg/l (1 µg/l)
Occurrence	Detected in surface water at concentrations below 1 µg/l; also detected in drinking-water at 100-fold lower concentrations
PTDI	0.01 mg/kg body weight based on a NOAEL of 1 mg/kg body weight per day for developmental toxicity in rats, applying an uncertainty factor of 100 (for interspecies and intraspecies variation)
Limit of detection	0.011 µg/l by GC using ECD
Treatment performance	0.1 µg/l should be achievable using coagulation or GAC
Guideline value derivation	
• allocation to water	1% of PTDI
• weight	10 kg child
• consumption	1 litre/day
Additional comments	DDT is listed under the Stockholm Convention on Persistent Organic Pollutants. Hence, monitoring may occur in addition to that required by drinking-water guidelines.
	It should be noted that the level of DDT and its metabolites in food has been falling steadily, and the allocation of 1% of the PTDI may be very conservative.
	The guideline value is derived on the basis of a 10 kg child consuming 1 litre of drinking-water per day, because infants and children may be exposed to greater amounts of chemicals in relation to their body weight and because of concern over the bioaccumulation of DDT.
	It should be emphasized that the benefits of DDT use in malaria and other vector control programmes outweigh any health risk from the presence of DDT in drinking-water.
Assessment date	2003
Principal references	FAO/WHO (2001) <i>Pesticide residues in food—2000 evaluations</i> WHO (2004) <i>DDT and its derivatives in drinking-water</i>

A working group convened by IARC classified the DDT complex (the mixture of the various isomers of DDT and associated compounds) as a non-genotoxic carcinogen in rodents and a potent promoter of liver tumours. IARC has concluded that there is insufficient evidence in humans and sufficient evidence in experimental animals for the carcinogenicity of DDT (Group 2B) based upon liver tumours observed in rats and mice. The results of epidemiological studies of pancreatic cancer, multiple myeloma, non-Hodgkin lymphoma and uterine cancer did not support the hypothesis of an association with environmental exposure to the DDT complex. Conflicting data were obtained with regard to some genotoxic end-points. In most studies, DDT did not induce genotoxic effects in rodent or human cell systems, nor

was it mutagenic to fungi or bacteria. The United States Agency for Toxic Substances and Disease Registry concluded that the DDT complex could impair reproduction and development in several species. Hepatic effects of DDT in rats include increased liver weights, hypertrophy, hyperplasia, induction of microsomal enzymes, including cytochrome P450, cell necrosis, increased activity of serum liver enzymes and mitogenic effects, which might be related to a regenerative liver response to high doses of DDT.

1,2-Dibromo-3-chloropropane

1,2-Dibromo-3-chloropropane (CAS No. 96-12-8), or DBCP, is a soil fumigant that is highly soluble in water. It has a taste and odour threshold in water of 10 µg/l. DBCP was detected in vegetables grown in treated soils, and low levels have been detected in air.

Guideline value	0.001 mg/l (1 µg/l)
Occurrence	Limited survey found levels of up to a few micrograms per litre in drinking-water
Basis of guideline value derivation	Linearized multistage model was applied to the data on the incidence of stomach, kidney and liver tumours in the male rat in a 104-week dietary study
Limit of detection	0.02 µg/l by GC with ECD
Treatment performance	1 µg/l should be achievable using air stripping followed by GAC
Additional comments	The guideline value of 1 µg/l should be protective for the reproductive toxicity of DBCP.
Assessment date	1993
Principal reference	WHO (2003) <i>1,2-Dibromo-3-chloropropane in drinking-water</i>

On the basis of data from different strains of rats and mice, DBCP was determined to be carcinogenic in both sexes by the oral, inhalation and dermal routes. DBCP was also determined to be a reproductive toxicant in humans and several species of laboratory animals. DBCP was found to be genotoxic in a majority of in vitro and in vivo assays. IARC has classified DBCP in Group 2B based upon sufficient evidence of carcinogenicity in animals. Recent epidemiological evidence suggests an increase in cancer mortality in individuals exposed to high levels of DBCP.

1,2-Dibromoethane

1,2-Dibromoethane (CAS No. 106-93-4), or ethylene dibromide, is used as a lead scavenger in tetraalkyl lead petrol and antiknock preparations and as a fumigant for soils, grains and fruits. However, with the phasing out of leaded petrol and of the use of 1,2-dibromoethane in agricultural applications in many countries, use of this substance has declined significantly. In addition to its continued use as a petrol additive in some countries, 1,2-dibromoethane is currently used principally as a solvent and as an intermediate in the chemical industry.

Provisional guideline value	0.0004 mg/l (0.4 µg/l)
	The guideline value is provisional owing to serious limitations of the critical studies.
Occurrence	Detected in groundwater following its use as a soil fumigant at concentrations as high as 100 µg/l
Basis of guideline value derivation	Lower end of the range (and thus more conservative estimate) of lifetime low-dose cancer risks calculated by linearized multistage modelling of the incidences of haemangiosarcomas and tumours in the stomach, liver, lung and adrenal cortex (adjusted for the observed high early mortality, where appropriate, and corrected for the expected rate of increase in tumour formation in rodents in a standard bioassay of 104 weeks) of rats and mice exposed by gavage
Limit of detection	0.01 µg/l by microextraction GC-MS; 0.03 µg/l by purge-and-trap GC with halogen-specific detector; 0.8 µg/l by purge-and-trap capillary column GC with photoionization and electrolytic conductivity detectors in series
Treatment performance	0.1 µg/l should be achievable using GAC
Assessment date	2003
Principal references	IPCS (1995) <i>Report of the 1994 meeting of the Core Assessment Group</i> IPCS (1996) <i>1,2-Dibromoethane</i> WHO (2003) <i>1,2-Dibromoethane in drinking-water</i>

1,2-Dibromoethane has induced an increased incidence of tumours at several sites in all carcinogenicity bioassays identified in which rats or mice were exposed to the compound by gavage, ingestion in drinking-water, dermal application and inhalation. However, many of these studies were characterized by high early mortality, limited histopathological examination, small group sizes or use of only one exposure level. The substance acted as an initiator of liver foci in an initiation/promotion assay but did not initiate skin tumour development. 1,2-Dibromoethane was consistently genotoxic in in vitro assays, although results of in vivo assays were mixed. Biotransformation to active metabolites, which have been demonstrated to bind to DNA, is probably involved in the induction of tumours. Available data do not support the existence of a non-genotoxic mechanism of tumour induction. The available data thus indicate that 1,2-dibromoethane is a genotoxic carcinogen in rodents. Data on the potential carcinogenicity in humans are inadequate; however, it is likely that 1,2-dibromoethane is metabolized similarly in rodent species and in humans (although there may be varying potential for the production of active metabolites in humans, owing to genetic polymorphism). IARC classified 1,2-dibromoethane in Group 2A (probably carcinogenic to humans).

Dichloroacetic acid

Chlorinated acetic acids, including dichloroacetic acid (DCA), are formed from organic material during water chlorination. DCA has been used as a therapeutic agent to treat lactic acidosis, diabetes and familial hyperlipidaemia in humans.

Provisional guideline value	0.05 mg/l (50 µg/l)
	The guideline value is designated as provisional on the basis of technical achievability.
Occurrence	Found in groundwater and surface water distribution systems at concentrations up to about 100 µg/l, with mean concentrations below 20 µg/l
Basis of guideline value derivation	Linear multistage model applied to combined data for carcinomas and adenomas in male mice exposed to doses up to 429 mg/kg body weight per day for up to 2 years
Limit of detection	< 0.1–0.4 µg/l by GC with ECD; practical quantification limit 1 µg/l
Treatment performance	Concentrations may be reduced by installing or optimizing coagulation to remove precursors or by controlling the pH during chlorination.
Additional comments	The concentration associated with a 10^{-5} upper-bound excess lifetime cancer risk is 40 µg/l. In some circumstances, however, it may not be possible to adequately disinfect potable water and maintain DCA levels below 40 µg/l, so the provisional guideline value of 50 µg/l is retained.
Assessment date	2004
Principal reference	WHO (2005) <i>Dichloroacetic acid in drinking-water</i>

IARC reclassified DCA as Group 2B (possibly carcinogenic to humans) in 2002, based on the absence of data on human carcinogenicity and sufficient evidence of its carcinogenicity in experimental animals. This classification was based primarily on findings of liver tumours in rats and mice. Genotoxicity data are considered to be inconclusive, particularly at lower doses. Glycogen deposition, peroxisome proliferation, changes in signal transduction pathways and DNA hypomethylation have all been observed following DCA exposure and have been hypothesized to be involved in its carcinogenicity. However, the available data are not sufficient to establish a cancer mode of action with reasonable certainty, especially at the very low exposure levels expected to apply to humans ingesting chlorinated drinking-water. Recent data suggest that there may be more than one mechanism leading to tumours, as altered hepatic foci from treated mice were found to have three different types of cellular characteristics.

Dichlorobenzenes (1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene)

The dichlorobenzenes (DCBs) are widely used in industry and in domestic products such as odour-masking agents, chemical dyestuffs and pesticides. Sources of human exposure are predominantly air and food.

Guideline values	<i>1,2-Dichlorobenzene</i> : 1 mg/l (1000 µg/l)
	<i>1,4-Dichlorobenzene</i> : 0.3 mg/l (300 µg/l)
Occurrence	Have been found in raw water sources at levels as high as 10 µg/l and in drinking-water at concentrations up to 3 µg/l; much higher concentrations (up to 7 mg/l) present in contaminated groundwater

TDIs	<i>1,2-Dichlorobenzene</i>: 429 µg/kg body weight, based on a NOAEL of 60 mg/kg body weight per day for tubular degeneration of the kidney identified in a 2-year mouse gavage study, adjusting for daily dosing and using an uncertainty factor of 100 (for interspecies and intraspecies variation) <i>1,4-Dichlorobenzene</i>: 107 µg/kg body weight, based on a LOAEL of 150 mg/kg body weight per day for kidney effects identified in a 2-year rat study, adjusting for daily dosing and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the use of a LOAEL instead of a NOAEL and the carcinogenicity end-point)
Limit of detection	0.01–0.25 µg/l by gas–liquid chromatography with ECD; 3.5 µg/l by GC using a photoionization detector
Treatment performance	0.01 mg/l should be achievable using air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Guideline values for both 1,2- and 1,4-DCB far exceed their lowest reported taste thresholds in water of 1 and 6 µg/l, respectively.
Assessment date	1993
Principal reference	WHO (2003) <i>Dichlorobenzenes in drinking-water</i>
Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value for <i>1,3-dichlorobenzene</i>
Assessment date	1993
Principal reference	WHO (2003) <i>Dichlorobenzenes in drinking-water</i>

1,2-Dichlorobenzene

1,2-DCB is of low acute toxicity by the oral route of exposure. Oral exposure to high doses of 1,2-DCB affects mainly the liver and kidneys. The balance of evidence suggests that 1,2-DCB is not genotoxic, and there is no evidence for its carcinogenicity in rodents.

1,3-Dichlorobenzene

There are insufficient toxicological data on this compound to permit a guideline value to be proposed, but it should be noted that it is rarely found in drinking-water.

1,4-Dichlorobenzene

1,4-DCB is of low acute toxicity, but there is evidence that it increases the incidence of renal tumours in rats and of hepatocellular adenomas and carcinomas in mice after long-term exposure. IARC has placed 1,4-DCB in Group 2B (possibly carcinogenic to humans). 1,4-DCB is not considered to be genotoxic, and the relevance for humans of the tumours observed in experimental animals is doubtful.

1,1-Dichloroethane

1,1-Dichloroethane is used as a chemical intermediate and solvent. There are limited data showing that it can be present at concentrations of up to 10 µg/l in drinking-water. It is primarily of concern for groundwater.

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value
Assessment date	1993
Principal reference	WHO (2003) <i>1,1-Dichloroethane in drinking-water</i>

1,1-Dichloroethane is rapidly metabolized by mammals to acetic acid and a variety of chlorinated compounds. It is of relatively low acute toxicity, and limited data are available on its toxicity from short-term and long-term studies. There is limited in vitro evidence of genotoxicity. One carcinogenicity study by gavage in mice and rats provided no conclusive evidence of carcinogenicity, although there was some evidence of an increased incidence of haemangiosarcomas in treated animals.

In view of the very limited database on toxicity and carcinogenicity, it was concluded that no guideline value should be proposed.

1,2-Dichloroethane

1,2-Dichloroethane is used mainly as an intermediate in the production of vinyl chloride and other chemicals and to a lesser extent as a solvent. It was used as a scavenger for tetraethyl lead in gasoline. It may enter surface waters via effluents from industries that manufacture or use the substance. It may also enter groundwater, where it may persist for long periods, following disposal in waste sites. It is found in urban air.

Guideline value	0.03 mg/l (30 µg/l)
Occurrence	Has been found in drinking-water at levels of up to a few micrograms per litre
Basis of guideline value derivation	Applying the linearized multistage model to haemangiosarcomas observed in male rats in a 78-week gavage study
Limit of detection	0.03 µg/l by GC with photoionization detection; 0.03–0.2 µg/l by GC with electrolytic conductivity detector; 0.06–2.8 µg/l by GC-MS; 5 µg/l by GC with flame ionization detection (FID)
Treatment performance	0.0001 mg/l should be achievable using GAC
Additional comments	The guideline value of 0.03 mg/l is consistent with the value derived from IPCS (1998), based on a 10^{-5} risk level.
Assessment date	2003
Principal references	IPCS (1995) <i>1,2-Dichloroethane, 2nd ed.</i> IPCS (1998) <i>1,2-Dichloroethane</i> WHO (2003) <i>1,2-Dichloroethane in drinking-water</i>

IARC has classified 1,2-dichloroethane in Group 2B (possible human carcinogen). It has been shown to produce statistically significant increases in a number of tumour types in laboratory animals, including the relatively rare haemangiosarcoma, and the balance of evidence indicates that it is potentially genotoxic. Targets of 1,2-dichloroethane toxicity in orally exposed animals included the immune system, central nervous system, liver and kidney. Data indicate that 1,2-dichloroethane is less potent when inhaled.

1,1-Dichloroethene

1,1-Dichloroethene, or vinylidene chloride, is used mainly as a monomer in the production of polyvinylidene chloride co-polymers and as an intermediate in the synthesis of other organic chemicals. It is an occasional contaminant of drinking-water, usually being found together with other chlorinated hydrocarbons. There are no data on levels in food, but levels in air are generally less than 40 ng/m³ except at some manufacturing sites. 1,1-Dichloroethene is detected in finished drinking-water taken from groundwater sources at median concentrations of 0.28–1.2 µg/l and in public drinking-water supplies at concentrations up to 0.5 µg/l.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2004
Principal references	IPCS (2003) <i>1,1-Dichloroethene (vinylidene chloride)</i> WHO (2005) <i>1,1-Dichloroethene in drinking-water</i>

1,1-Dichloroethene is a central nervous system depressant and may cause liver and kidney toxicity in occupationally exposed humans. It causes liver and kidney damage in laboratory animals. IARC has placed 1,1-dichloroethene in Group 3 (not classifiable as to its carcinogenicity to humans). It was found to be genotoxic in a number of test systems in vitro but was not active in the dominant lethal and micronucleus assays in vivo. It induced kidney tumours in mice in one inhalation study but was reported not to be carcinogenic in a number of other studies, including several in which it was given in drinking-water.

A health-based value of 140 µg/l (rounded value) can be calculated on the basis of a TDI of 0.046 mg/kg body weight, derived using the benchmark dose (BMD) approach from a study in which the critical effect was minimal hepatocellular mid-zonal fatty change in female rats. However, this value is significantly higher than the concentrations of 1,1-dichloroethene normally found in drinking-water. It is therefore considered unnecessary to set a formal guideline value for 1,1-dichloroethene in drinking-water.

1,2-Dichloroethene

1,2-Dichloroethene exists in a *cis* and a *trans* form. The *cis* form is more frequently found as a water contaminant. The presence of these two isomers, which are metabolites of other unsaturated halogenated hydrocarbons in wastewater and anaerobic

groundwater, may indicate the simultaneous presence of other organochlorine chemicals, such as vinyl chloride. Accordingly, their presence indicates that more intensive monitoring should be conducted. There are no data on exposure from food. Concentrations in air are low, with higher concentrations, in the microgram per cubic metre range, near production sites. The *cis* isomer was previously used as an anaesthetic.

Guideline value	0.05 mg/l (50 µg/l)
Occurrence	Has been found in drinking-water supplies derived from groundwater at levels up to 120 µg/l
TDI	17 µg/kg body weight, based on a NOAEL (for increases in serum alkaline phosphatase levels and increased thymus weight) of 17 mg/kg body weight from a 90-day study in mice administered <i>trans</i> -1,2-dichloroethene in drinking-water, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the short duration of the study)
Limit of detection	0.17 µg/l by GC-MS
Treatment performance	0.01 mg/l should be achievable using GAC or air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Data on the <i>trans</i> isomer were used to calculate a joint guideline value for both isomers because toxicity for the <i>trans</i> isomer occurred at a lower dose than for the <i>cis</i> isomer and because data suggest that the mouse is a more sensitive species than the rat.
Assessment date	1993
Principal reference	WHO (2003) <i>1,2-Dichloroethene in drinking-water</i>

There is little information on the absorption, distribution or excretion of 1,2-dichloroethene. However, by analogy with 1,1-dichloroethene, 1,2-dichloroethene would be expected to be readily absorbed, distributed mainly to the liver, kidneys and lungs and rapidly excreted. The *cis* isomer is more rapidly metabolized than the *trans* isomer in in vitro systems. Both isomers have been reported to cause increased serum alkaline phosphatase levels in rodents. In a 3-month study in mice given the *trans* isomer in drinking-water, there was a reported increase in serum alkaline phosphatase and reduced thymus and lung weights. Transient immunological effects were also reported, the toxicological significance of which is unclear. *Trans*-1,2-dichloroethene also caused reduced kidney weights in rats, but at higher doses. Only one rat toxicity study is available for the *cis* isomer, which produced toxic effects in rats similar in magnitude to those induced by the *trans* isomer in mice, but at higher doses. There are limited data to suggest that both isomers may possess some genotoxic activity. There is no information on carcinogenicity.

Dichloromethane

Dichloromethane, or methylene chloride, is widely used as a solvent for many purposes, including coffee decaffeination and paint stripping. Exposure from drinking-water is likely to be insignificant compared with that from other sources.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Has been found in surface water samples at concentrations ranging from 0.1 to 743 µg/l; levels usually higher in groundwater because volatilization is restricted, with concentrations as high as 3600 µg/l reported; mean concentrations in drinking-water less than 1 µg/l
TDI	6 µg/kg body weight, derived from a NOAEL of 6 mg/kg body weight per day for hepatotoxic effects in a 2-year drinking-water study in rats, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for concern about carcinogenic potential)
Limit of detection	0.3 µg/l by purge-and-trap GC with MS detection (note that dichloromethane vapour readily penetrates tubing during the procedure)
Treatment performance	20 µg/l should be achievable using air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Dichloromethane in drinking-water</i>

Dichloromethane is of low acute toxicity. An inhalation study in mice provided conclusive evidence of carcinogenicity, whereas drinking-water studies in rats and mice provided only suggestive evidence. IARC has placed dichloromethane in Group 2B (possible human carcinogen); however, the balance of evidence suggests that it is not a genotoxic carcinogen and that genotoxic metabolites are not formed in relevant amounts in vivo.

1,2-Dichloropropane

1,2-Dichloropropane (CAS No. 78-87-5), or 1,2-DCP, is used as an insecticide fumigant on grain and soil and to control peach tree borers. It is also used as an intermediate in the production of tetrachloroethene and other chlorinated products and as a solvent. 1,2-DCP is relatively resistant to hydrolysis, is poorly adsorbed onto soil and can migrate into groundwater.

Provisional guideline value	0.04 mg/l (40 µg/l)
	The guideline value is provisional owing to limitations of the toxicological database.

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Occurrence	Detected in groundwater and drinking-water, usually at concentrations below 20 µg/l, although levels as high as 440 µg/l have been measured in well water
TDI	14 µg/kg body weight based on a LOAEL of 71.4 mg/kg body weight per day (100 mg/kg body weight per day adjusted for daily dosing) for changes in haematological parameters in a 13-week study in male rats, with an uncertainty factor of 5000 (100 for interspecies and intraspecies variation, 10 for use of a LOAEL and 5 to reflect limitations of the database, including the limited data on in vivo genotoxicity and use of a subchronic study)
Limit of detection	0.02 µg/l by purge-and-trap GC with an electrolytic conductivity detector or GC-MS
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1998
Principal reference	WHO (2003) <i>1,2-Dichloropropane (1,2-DCP) in drinking-water</i>

1,2-DCP was evaluated by IARC in 1986 and 1987. The substance was classified in Group 3 (not classifiable as to its carcinogenicity to humans) on the basis of limited evidence for its carcinogenicity in experimental animals and insufficient data with which to evaluate its carcinogenicity in humans. Results from in vitro assays for mutagenicity were mixed. The in vivo studies, which were limited in number and design, were negative. In accordance with the IARC evaluation, the evidence from the long-term carcinogenicity studies in mice and rats was considered limited, and it was concluded that the use of a threshold approach for the toxicological evaluation of 1,2-DCP was appropriate.

1,3-Dichloropropane

1,3-Dichloropropane (CAS No. 142-28-9) has several industrial uses and may be found as a contaminant of soil fumigants containing 1,3-dichloropropene. It is rarely found in water.

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value
Assessment date	1993
Principal reference	WHO (2003) <i>1,3-Dichloropropane in drinking-water</i>

1,3-Dichloropropane is of low acute toxicity. There is some indication that it may be genotoxic in bacterial systems. No short-term, long-term, reproductive or developmental toxicity data pertinent to exposure via drinking-water could be located in the literature. The available data are considered insufficient to permit recommendation of a guideline value.

1,3-Dichloropropene

1,3-Dichloropropene (CAS Nos. 542-75-6 isomer mixture; 10061-01-5 *cis* isomer; 10061-02-6 *trans* isomer) is a soil fumigant, the commercial product being a mixture of *cis* and *trans* isomers. It is used to control a wide variety of soil pests, particularly nematodes in sandy soils. Notwithstanding its high vapour pressure, it is soluble in water at the gram per litre level and can be considered a potential water contaminant.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Has been found in surface water and groundwater at concentrations of a few micrograms per litre
Basis of guideline value derivation	Calculated by applying the linearized multistage model to the observation of lung and bladder tumours in female mice in a 2-year gavage study
Limit of detection	0.34 and 0.20 µg/l by purge-and-trap packed column GC using an electrolytic conductivity detector or microcoulometric detector for the <i>cis</i> and <i>trans</i> isomers, respectively
Treatment performance	No information found on removal from water
Assessment date	1993
Principal reference	WHO (2003) <i>1,3-Dichloropropene in drinking-water</i>

1,3 Dichloropropene is a direct-acting mutagen that has been shown to produce forestomach tumours following long term oral gavage exposure in rats and mice. Tumours have also been found in the bladder and lungs of female mice and the liver of male rats. Long term inhalation studies in the rat have proved negative, whereas some benign lung tumours have been reported in inhalation studies in mice. IARC has classified 1,3 dichloropropene in Group 2B (possible human carcinogen).

Dichlorprop

The half-lives for degradation of chlorophenoxy herbicides, including dichlorprop (CAS No. 120-36-5), or 2,4-DP, in the environment are in the order of several days. Chlorophenoxy herbicides are not often found in food.

Guideline value	0.1 mg/l (100 µg/l)
Occurrence	Chlorophenoxy herbicides not frequently found in drinking-water; when detected, concentrations usually no greater than a few micrograms per litre
TDI	36.4 µg/kg body weight, based on a NOAEL of 3.64 mg/kg body weight per day for renal toxicity in a 2-year dietary study in rats, applying an uncertainty factor of 100 (for intraspecies and interspecies variation)
Limit of detection	1 µg/l to 1 mg/l for various methods commonly used for the determination of chlorophenoxy herbicides in water, including solvent extraction, separation by GC, gas-liquid chromatography, thin-layer chromatography or HPLC, with ECD or UV detection

Treatment performance	No information found on removal from water
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenoxy herbicides (excluding 2,4-D and MCPA) in drinking-water</i>

Chlorophenoxy herbicides, as a group, have been classified in Group 2B (possible human carcinogen) by IARC. However, the available data from studies in exposed populations and experimental animals do not permit assessment of the carcinogenic potential to humans of any specific chlorophenoxy herbicide. Therefore, drinking-water guidelines for these compounds are based on a threshold approach for other toxic effects. In dietary studies in rats, slight liver hypertrophy was observed in a 3-month study, and effects in a 2-year study included hepatocellular swelling, mild anaemia, increased incidence of brown pigment in the kidneys (possibly indicative of slight degeneration of the tubular epithelium) and decreased urinary specific gravity and protein.

Dichlorvos

Dichlorvos (CAS No. 62-73-7) is a broad-spectrum organophosphorus insecticide used primarily for controlling household pests and for protecting stored products from insects. It is no longer approved for use in some jurisdictions because of concerns over its acute toxicity. Dichlorvos is expected to be very mobile in soils. It is rapidly degraded by microbial activity and hydrolysis in soil, and does not adsorb to sediments. Degradation in water occurs primarily through hydrolysis. There are relatively few studies on its occurrence in source waters. Exposure from food varies widely, depending on local circumstances and usage. Dichlorvos can be inhaled from its use as a domestic insecticide.

Reason for not establishing a guideline value	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Health-based value*	0.02 mg/l
Acute health-based value**	3 mg/l
Occurrence	Concentrations in surface water in the range 10–50 ng/l, but sometimes as high as 1500 ng/l, have been measured
ADI	0–0.004 mg/kg bw, based on a NOAEL of 0.04 mg/kg bw per day for the inhibition of erythrocyte acetylcholinesterase activity in a 21-day study in male volunteers and application of a safety factor of 10
ARfD	0.1 mg/kg bw, based on a NOAEL of 1 mg/kg bw for erythrocyte acetylcholinesterase inhibition in an acute oral study in male volunteers and application of a safety factor of 10

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Limit of detection	0.01 µg/l (limit of quantification) based on solvent extraction and GC analysis; 0.1 µg/l (reporting limit) based on GC-MS
Treatment performance	Conventional treatment, including coagulation, filtration and chlorination, not effective; removal by membranes depends on membrane type and operational conditions. Removal by nanofiltration membranes has variable effectiveness (removal rates from 4 to 60%). Reverse osmosis would be expected to be effective (removal rates > 85%) based on removal studies and predictions.
Health-based value derivation	
• allocation to water	20% of upper bound of ADI
• weight	60 kg adult
• consumption	2 litres/day
Acute health-based value derivation	
• allocation to water	100% of ARfD
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The default allocation factor of 20% has been used to account for the fact that the available food exposure data, which suggest that exposure via this route is low, do not generally include information from developing countries, where exposure via this route may be higher, and as potential exposure via inhalation from indoor air resulting from use of dichlorvos as a domestic insecticide is unknown As a general principle, the concentration of pesticides in water, including dichlorvos, should be kept as low as possible and concentrations should not be allowed to increase up to the health-based value Further guidance on interpreting the health-based value and deciding when to monitor can be found in section 8.5.3
Assessment date	2016
Principal references	WHO (2012) <i>Pesticide residues in food – 2011 evaluations</i> WHO (2016) <i>Dichlorvos in drinking-water</i>

* When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary.

** For more information on acute health-based values, see [section 8.7.5](#).

As with other organophosphorus insecticides, the inhibition of cholinesterase activity, causing neurotoxicity, is the most sensitive toxicological end-point following acute or repeated exposures to dichlorvos. Dichlorvos is unlikely to be genotoxic in vivo or to pose a carcinogenic risk to humans. Some reproductive toxicity has been observed in rats, but dichlorvos was not found to cause developmental toxicity or to be teratogenic.

Dicofol

Dicofol (CAS No. 115-32-2) is an organochlorine acaricide that has been registered for broad-spectrum contact, non-systemic control of plant-eating mites in cotton, tea and a wide variety of fruit, vegetable and ornamental crops. Products containing

dicofol, which is manufactured from DDT, are being phased out in the USA and are no longer approved for use in the European Union. Dicofol is unlikely to reach water, but may do so if bound to particulate matter subject to runoff. Dicofol is only slightly soluble in water and binds strongly to soil. There are few data on the occurrence of dicofol in water. Exposure from food varies widely, depending on local circumstances and usage. Dicofol has been proposed as a persistent organic pollutant under the Stockholm Convention.

Reason for not establishing a guideline value	Unlikely to be found in drinking-water or drinking-water sources*
Health-based value**	0.01 mg/l
Acute health-based value***	6 mg/l
Occurrence	Not detected in limited groundwater monitoring
ADI	0–0.002 mg/kg bw, based on a NOAEL of 0.22 mg/kg bw per day for histopathological changes in the liver and adrenal gland in a 2-year toxicity and carcinogenicity study in rats and application of a safety factor of 100
ARfD	0.2 mg/kg bw, based on a NOAEL of 15 mg/kg bw for decreased body weight and decreased feed intake in an acute neurotoxicity study in rats and application of a safety factor of 100
Limit of detection	Solvent extraction followed by GC-ECD may be effective (limit of quantification 5 ng/l)
Treatment performance	Should be removed by adsorption onto activated carbon, and any dicofol adsorbed onto particulate matter would likely be removed during coagulation
Health-based value derivation	
• allocation to water	20% of the upper bound of the ADI
• weight	60 kg adult
• consumption	2 litres/day
Acute health-based value derivation	
• allocation to water	100% of the ARfD
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The default allocation factor of 20% has been used to account for the fact that the available food exposure data, which suggest that exposure via this route is low, do not generally include information from developing countries, where exposure via this route may be higher
	As a general principle, the concentration of pesticides in water, including dicofol, should be kept as low as possible and concentrations should not be allowed to increase up to the health-based value
	Further guidance on interpreting the health-based value and deciding when to monitor can be found in section 8.5.3
Assessment date	2016

Principal references	WHO (2012) <i>Pesticide residues in food – 2011 evaluations</i> WHO (2016) <i>Dicofol in drinking-water</i>
* Although dicofol does not fulfil one of the three criteria for evaluation in the Guidelines, a background document has been prepared, and a health-based value has been established, in response to a request from Member States for guidance. ** When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary. *** For more information on acute health-based values, see section 8.7.5.	

The primary effects of dicofol after short- or long-term exposure of experimental animals were body weight reduction associated with decreased feed intake, and increased liver weight accompanied by changes in liver enzyme activities. Dicofol caused liver tumours in male mice at doses associated with significant enzyme induction and liver hypertrophy. However, on the basis of the absence of genotoxicity in an adequate range of in vitro genotoxicity and in vivo chromosomal aberration tests, the absence of carcinogenic effects in rats and the expectation that the adenomas present in mice will exhibit a threshold, dicofol is unlikely to pose a carcinogenic risk to humans at anticipated dietary exposure levels. There is a margin of 20 000 between the upper bound of the ADI and the LOAEL for liver adenomas in the male mouse.

Di(2-ethylhexyl)adipate

Di(2-ethylhexyl)adipate (DEHA) is used mainly as a plasticizer for synthetic resins such as polyvinyl chloride (PVC). Reports of the presence of DEHA in surface water and drinking-water are scarce, but DEHA has occasionally been identified in drinking-water at levels of a few micrograms per litre. As a consequence of its use in PVC films, food is the most important source of human exposure (up to 20 mg/day).

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal reference	WHO (2003) <i>Di(2-ethylhexyl)adipate in drinking-water</i>

DEHA is of low short-term toxicity; however, dietary levels above 6000 mg/kg of feed induce peroxisomal proliferation in the liver of rodents. This effect is often associated with the development of liver tumours. DEHA induced liver carcinomas in female mice at very high doses, but not in male mice or rats. It is not genotoxic. IARC has placed DEHA in Group 3 (not classifiable as to its carcinogenicity to humans).

A health-based value of 80 µg/l can be calculated for DEHA on the basis of a TDI of 280 µg/kg body weight, based on fetotoxicity in rats, and allocating 1% of the TDI to drinking-water. However, because DEHA occurs at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value.

Di(2-ethylhexyl)phthalate

Di(2-ethylhexyl)phthalate (DEHP) is used primarily as a plasticizer. Exposure among individuals may vary considerably because of the broad nature of products into which DEHP is incorporated. In general, food will be the main exposure route.

Guideline value	0.008 mg/l (8 µg/l)
Occurrence	Found in surface water, groundwater and drinking-water in concentrations of a few micrograms per litre; in polluted surface water and groundwater, concentrations of hundreds of micrograms per litre have been reported
TDI	25 µg/kg body weight, based on a NOAEL of 2.5 mg/kg body weight per day for peroxisomal proliferation in the liver in rats, using an uncertainty factor of 100 for interspecies and intraspecies variation
Limit of detection	0.1 µg/l by GC-MS
Treatment performance	No information found on removal from water
Guideline value derivation	
• allocation to water	1% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The reliability of some data on environmental water samples is questionable because of secondary contamination during sampling and working-up procedures. Concentrations that exceed the solubility more than 10-fold have been reported.
Assessment date	1993
Principal reference	WHO (2003) <i>Di(2-ethylhexyl)phthalate in drinking-water</i>

In rats, DEHP is readily absorbed from the gastrointestinal tract. In primates (including humans), absorption after ingestion is lower. Species differences are also observed in the metabolic profile. Most species excrete primarily the conjugated mono-ester in urine. Rats, however, predominantly excrete terminal oxidation products. DEHP is widely distributed in the body, with highest levels in liver and adipose tissue, without showing significant accumulation. The acute oral toxicity is low. The most striking effect in short-term toxicity studies is the proliferation of hepatic peroxisomes, indicated by increased peroxisomal enzyme activity and histopathological changes. The available information suggests that primates, including humans, are far less sensitive to this effect than rodents. In long-term oral carcinogenicity studies, hepatocellular carcinomas were found in rats and mice. IARC has concluded that DEHP is possibly carcinogenic to humans (Group 2B). In 1988, JECFA evaluated DEHP and recommended that human exposure to this compound in food be reduced to the lowest level attainable. JECFA considered that this might be achieved by using alternative plasticizers or alternatives to plastic material containing DEHP. In a variety of in vitro and in vivo studies, DEHP and its metabolites have shown no evidence of genotoxicity, with the exception of induction of aneuploidy and cell transformation.

Dimethoate

Dimethoate (CAS No. 60-51-5) is an organophosphorus insecticide used to control a broad range of insects in agriculture, as well as the housefly. It has a half-life of 18 hours to 8 weeks and is not expected to persist in water, although it is relatively stable at pH 2–7. A total daily intake from food of 0.001 µg/kg body weight has been estimated.

Guideline value	0.006 mg/l (6 µg/l)
Occurrence	Detected at trace levels in a private well in Canada, but not detected in a Canadian survey of surface water or drinking-water supplies
ADI	0–0.002 mg/kg body weight based on an apparent NOAEL of 1.2 mg/kg body weight per day for reproductive performance in a study of reproductive toxicity in rats, applying an uncertainty factor of 500 (100 for interspecies and intraspecies variation, 5 to take into consideration concern regarding whether the NOAEL could be a LOAEL)
Assessment date	2003
Principal references	FAO/WHO (1997) <i>Pesticide residues in food—1996 evaluations</i> WHO (2004) <i>Dimethoate in drinking-water</i>

In studies with human volunteers, dimethoate has been shown to be a cholinesterase inhibitor and a skin irritant. Dimethoate is not carcinogenic to rodents. JMPR concluded that although in vitro studies indicate that dimethoate has mutagenic potential, this potential does not appear to be expressed in vivo. In a multi-generation study of reproductive toxicity in rats, the NOAEL appeared to be 1.2 mg/kg body weight per day, but there was some indication that reproductive performance may have been affected at lower doses. No data were available to assess whether the effects on reproductive performance were secondary to inhibition of cholinesterase. JMPR concluded that it was not appropriate to base the ADI on the results of the studies of volunteers, as the crucial end-point (reproductive performance) has not been assessed in humans. It was suggested that there may be a need to re-evaluate the toxicity of dimethoate after the periodic review of the residue and analytical aspects of dimethoate has been completed if it is determined that omethoate is a major residue.

1,4-Dioxane

1,4-Dioxane is used as a stabilizer in chlorinated solvents and as a solvent for resins, oils and waxes, for agricultural and biochemical intermediates and for adhesives, sealants, cosmetics, pharmaceuticals, rubber chemicals and surface coatings.

Guideline value	0.05 mg/l (50 µg/l)
Occurrence	Has been measured in surface water at concentrations up to 40 µg/l and in groundwater at concentrations up to 80 µg/l
TDI	16 µg/kg body weight, based on a NOAEL of 16 mg/kg body weight per day for hepatocellular tumours observed in a long-term drinking-water study in rats, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for non-genotoxic carcinogenicity)

Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Basis of guideline value derivation based on carcinogenicity	Linear multistage model applied to data for hepatic tumours from drinking-water studies in rats
Limit of detection	0.1–50 µg/l by GC-MS
Treatment performance	Not removed using conventional water treatment processes; effectively removed by biological activated carbon treatment
Additional comments	Similar guideline values were derived using the TDI approach (assuming 1,4-dioxane is not genotoxic in humans at low doses) and linear multistage modelling (because the compound clearly induces multiple tumours in various organs).
Assessment date	2004
Principal reference	WHO (2005) <i>1,4-Dioxane in drinking-water</i>

1,4-Dioxane caused hepatic and nasal cavity tumours in rodents in most long-term oral studies conducted. Tumours in peritoneum, skin and mammary gland were also observed in rats given a high dose. Lung tumours were specifically detected after intraperitoneal injection. Although cohort studies of workers did not reveal any elevation in the incidence of death by cancer, a significant increase in the incidence of liver cancer was found in a comparative mortality study. However, the evidence is inadequate for human carcinogenicity assessment because of small samples or lack of exposure data. A possibly weak genotoxic potential of 1,4-dioxane has been suggested. IARC has classified 1,4-dioxane in Group 2B (possibly carcinogenic to humans).

Diquat

Diquat (CAS No. 85-00-7; CAS No. 2764-72-9 for diquat ion) is a non-selective, quick-acting contact herbicide that is used for weed control on several food crops, for residential weed control on lawns and ornamental plants, and as an aquatic herbicide for the control of free-floating and submerged aquatic weeds in ponds and irrigation ditches. It is highly soluble in water but is strongly adsorbed to soil and is resistant to degradation in the sorbed state. Photochemical degradation in soil and water occurs in the presence of sunlight. Exposure from food is likely to be low.

Reason for not establishing a guideline value	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Health-based value*	0.03 mg/l
Acute health-based value**	20 mg/l
Occurrence	Rarely detected in surface water
ADI	0–0.006 mg/kg bw (expressed as the diquat ion), based on a NOAEL of 0.58 mg/kg bw per day for cataracts in a 2-year toxicity and carcinogenicity study in rats and application of a safety factor of 100

ARfD	0.8 mg/kg bw (expressed as the diquat ion), based on a NOAEL of 75 mg/kg bw for clinical signs and decreased body weight gain in the 1st week and decreased feed consumption in a neurotoxicity study in rats and application of a safety factor of 100
Limit of detection	1 µg/l using HPLC with UV absorbance detection after solid sorbent cartridge extraction; practical quantification limit of 1 µg/l using LC-MS analysis after solid-phase extraction
Treatment performance	Conventional treatment, including coagulation and filtration, not effective; activated carbon may be effective
Health-based value derivation	
• allocation to water	20% of upper bound of unrounded ADI (0.0058 mg/kg bw)
• weight	60 kg adult
• consumption	2 litres/day
Acute health-based value derivation	
• allocation to water	100% of unrounded ARfD (0.75 mg/kg bw)
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The default allocation factor of 20% has been used to account for the fact that the available food exposure data, , which suggest that exposure via this route is low, do not generally include information from developing countries, where exposure via this route may be higher
	As a general principle, the concentration of pesticides in water, including diquat, should be kept as low as possible and concentrations should not be allowed to increase up to the health-based value
	Further guidance on interpreting the health-based value and deciding when to monitor can be found in section 8.5.3
Assessment date	2016
Principal references	WHO (2014) <i>Pesticide residues in food – 2013 evaluations</i> WHO (2016) <i>Diquat in drinking-water</i>

* When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary.

** For more information on acute health-based values, see [section 8.7.5](#).

The eye is the main target organ following short-term repeated exposure in rats and dogs. Effects on kidney, liver and haematological parameters are also observed. Diquat is not carcinogenic in mice or rats. In tests for genotoxicity, diquat gave equivocal or positive responses in the mammalian cell cytogenetic assay, but was negative in the in vivo mouse micronucleus assay and dominant lethal assay. No reproductive effects were observed in a two-generation reproductive toxicity study in rats, and diquat was not teratogenic in rats or rabbits.

Edetic acid

Human exposure to edetic acid, also known as ethylenediaminetetraacetic acid or EDTA, arises directly from its use in food additives, medicines and personal care and

hygiene products. Exposure to EDTA from drinking-water will be mostly very low in comparison with that from other sources. Once EDTA is present in the aquatic environment, its speciation will depend on the water quality and the presence of trace metals with which it will combine. The removal of EDTA from communal wastewater by biodegradation in sewage purification plants is very limited.

Guideline value	EDTA (as the free acid): 0.6 mg/l (600 µg/l)
Occurrence	Present in surface waters generally at concentrations below 70 µg/l, although higher concentrations (900 µg/l) have been measured; detected in drinking-water prepared from surface waters at concentrations of 10–30 µg/l
ADI	0–1.9 mg/kg body weight as the free acid (ADI of 0–2.5 mg/kg body weight proposed by JECFA for calcium disodium edetate as a food additive)
Limit of detection	1 µg/l by potentiometric stripping analysis
Treatment performance	0.01 mg/l using GAC plus ozonation
Guideline value derivation	
• allocation to water	1% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Concern has been expressed over the ability of EDTA to complex and therefore reduce the availability of zinc. However, this is of significance only at elevated doses substantially in excess of those encountered in the environment.
Assessment date	1998
Principal reference	WHO (2003) <i>Edetic acid (EDTA) in drinking-water</i>

Calcium disodium edetate is poorly absorbed from the gut. The long-term toxicity of EDTA is complicated by its ability to chelate essential and toxic metals. Those toxicological studies that are available indicate that the apparent toxicological effects of EDTA have in fact been due to zinc deficiency as a consequence of complexation. EDTA does not appear to be teratogenic or carcinogenic in experimental animals. The vast clinical experience of the use of EDTA in the treatment of metal poisoning has demonstrated its safety in humans.

Endosulfan

Endosulfan (CAS No. 115-29-7) is an insecticide used in countries throughout the world to control pests on fruit, vegetables and tea and on non-food crops such as tobacco and cotton. In addition to its agricultural use, it is used in the control of the tsetse fly, as a wood preservative and for the control of home garden pests. Endosulfan contamination does not appear to be widespread in the aquatic environment, but the chemical has been found in agricultural runoff and rivers in industrialized areas

where it is manufactured or formulated, as well as in surface water and groundwater samples collected from hazardous waste sites in the USA. Surface water samples in the USA generally contain less than 1 µg/l. The main source of exposure of the general population is food, but residues have generally been found to be well below the FAO/WHO maximum residue limits. Another important route of exposure to endosulfan for the general population is the use of tobacco products.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1999) <i>Pesticide residues in food—1998 evaluations</i> WHO (2004) <i>Endosulfan in drinking-water</i>

JMPR concluded that endosulfan is not genotoxic, and no carcinogenic effects were noted in long-term studies using mice and rats. The kidney is the target organ for toxicity. Several recent studies have shown that endosulfan, alone or in combination with other pesticides, may bind to estrogen receptors and perturb the endocrine system. A health-based value of 20 µg/l can be calculated for endosulfan on the basis of an ADI of 0–0.006 mg/kg body weight, based on results from a 2-year dietary study of toxicity in rats and supported by a 78-week study in mice, a 1-year study in dogs and a developmental toxicity study in rats. However, because endosulfan occurs at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value.

Endrin

Endrin (CAS No. 72-20-8) is a broad-spectrum foliar insecticide that acts against a wide range of agricultural pests. It is also used as a rodenticide. There is now very little use of endrin. Small amounts of endrin are present in some foods, but the total intake from food has decreased significantly.

Guideline value	0.0006 mg/l (0.6 µg/l)
Occurrence	Traces of endrin found in the drinking-water supplies of several countries
PTDI	0.2 µg/kg body weight, based on a NOAEL of 0.025 mg/kg body weight per day in a 2-year study in dogs and applying an uncertainty factor of 100 for interspecies and intraspecies variation
Limit of detection	0.002 µg/l by GC with ECD
Treatment performance	0.2 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of PTDI
• weight	60 kg adult
• consumption	2 litres/day

Additional comments	Endrin is listed under the Stockholm Convention on Persistent Organic Pollutants. Hence, monitoring may occur in addition to that required by drinking-water guidelines.
Assessment date	2003
Principal references	FAO/WHO (1995) <i>Pesticide residues in food—1994 evaluations</i> IPCS (1992) <i>Endrin</i> WHO (2004) <i>Endrin in drinking-water</i>

Toxicological data are insufficient to indicate whether endrin is a carcinogenic hazard to humans. The primary site of action of endrin is the central nervous system.

Epichlorohydrin

Epichlorohydrin is used for the manufacture of glycerol, unmodified epoxy resins and water treatment coagulant polymers and some ion exchange resins. No quantitative data are available on its occurrence in food or drinking-water. Epichlorohydrin is slowly hydrolysed in aqueous media.

Provisional guideline value	0.0004 mg/l (0.4 µg/l)
	The guideline value is considered to be provisional because of the uncertainties surrounding the toxicity of epichlorohydrin and the use of a large uncertainty factor in deriving the guideline value.
Occurrence	No quantitative data available
TDI	0.14 µg/kg body weight, on the basis of a LOAEL of 2 mg/kg body weight per day for forestomach hyperplasia observed in a 2-year gavage study in rats, adjusting for daily dosing and using an uncertainty factor of 10 000 to take into consideration interspecies and intraspecies variation (100), the use of a LOAEL instead of a NOAEL (10) and carcinogenicity (10)
Limit of detection	0.01 µg/l by GC with ECD; 0.1 and 0.5 µg/l by GC-MS; 10 µg/l by GC with FID
Treatment performance	Conventional treatment processes do not remove epichlorohydrin. Epichlorohydrin concentrations in drinking-water are controlled by limiting either the epichlorohydrin content of polyamine flocculants or the dose used, or both.
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Although epichlorohydrin is a genotoxic carcinogen, the use of the linearized multistage model for estimating cancer risk was considered inappropriate because tumours are seen only at the site of administration, where epichlorohydrin is highly irritating.
Assessment date	2003
Principal reference	WHO (2004) <i>Epichlorohydrin in drinking-water</i>

Epichlorohydrin is rapidly and extensively absorbed following oral, inhalation or dermal exposure. It binds easily to cellular components. Major toxic effects are local irritation and damage to the central nervous system. It induces squamous cell carcinomas in the nasal cavity by inhalation and forestomach tumours by the oral route. It has been shown to be genotoxic in vitro and in vivo. IARC has placed epichlorohydrin in Group 2A (probably carcinogenic to humans).

Ethylbenzene

The primary sources of ethylbenzene in the environment are the petroleum industry and the use of petroleum products. Because of its physicochemical properties, more than 96% of ethylbenzene in the environment can be expected to be present in air. Values of up to 26 µg/m³ in air have been reported. Ethylbenzene is found in trace amounts in surface water, groundwater, drinking-water and food.

Guideline value	0.3 mg/l (300 µg/l)
Occurrence	Concentrations in drinking-water generally below 1 µg/l; levels up to 300 µg/l have been reported in groundwater contaminated by point emissions
TDI	97.1 µg/kg body weight, based on a NOAEL of 136 mg/kg body weight per day for hepatotoxicity and nephrotoxicity observed in a limited 6-month study in rats, adjusting for daily dosing and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the limited database and short duration of the study)
Limit of detection	0.002–0.005 µg/l by GC with photoionization detector; 0.03–0.06 µg/l by GC-MS
Treatment performance	0.001 mg/l should be achievable using air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value exceeds the lowest reported odour threshold for ethylbenzene in drinking-water (0.002 mg/l).
Assessment date	1993
Principal reference	WHO (2003) <i>Ethylbenzene in drinking-water</i>

Ethylbenzene is readily absorbed by the oral, inhalation or dermal route. In humans, storage in fat has been reported. Ethylbenzene is almost completely converted to soluble metabolites, which are excreted rapidly in urine. The acute oral toxicity is low. No definite conclusions can be drawn from limited teratogenicity data. No data on reproduction, long-term toxicity or carcinogenicity are available. Ethylbenzene has shown no evidence of genotoxicity in in vitro or in vivo systems.

Fenitrothion

Fenitrothion (CAS No. 122-14-5) is mainly used in agriculture for controlling insects on rice, cereals, fruits, vegetables, stored grains and cotton and in forest areas. It is also

used for the control of flies, mosquitoes and cockroaches in public health programmes and indoor use. Fenitrothion is stable in water only in the absence of sunlight or microbial contamination. In soil, biodegradation is the primary route of degradation, although photolysis may also play a role. Fenitrothion residues detected in water were low (maximum 1.30 µg/l) during the spruce budworm spray programme. Following the spraying of forests to control spruce budworm, water samples did not contain detectable amounts of fenitrothion; post-spray samples contained less than 0.01 µg/l. Levels of fenitrothion residues in fruits, vegetables and cereal grains decline rapidly after treatment, with a half-life of 1–2 days. Intake of fenitrothion appears to be primarily (95%) from food.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (2001) <i>Pesticide residues in food—2000 evaluations</i> WHO (2004) <i>Fenitrothion in drinking-water</i>

On the basis of testing in an adequate range of studies in vitro and in vivo, JMPR concluded that fenitrothion is unlikely to be genotoxic. It also concluded that fenitrothion is unlikely to pose a carcinogenic risk to humans. In long-term studies of toxicity, inhibition of cholinesterase activity was the main toxicological finding in all species. A health-based value of 8 µg/l can be calculated for fenitrothion on the basis of an ADI of 0–0.005 mg/kg body weight, based on a NOAEL of 0.5 mg/kg body weight per day for inhibition of brain and erythrocyte cholinesterase activity in a 2-year study of toxicity in rats and supported by a NOAEL of 0.57 mg/kg body weight per day for inhibition of brain and erythrocyte cholinesterase activity in a 3-month study of ocular toxicity in rats and a NOAEL of 0.65 mg/kg body weight per day for reduced food consumption and body weight gain in a study of reproductive toxicity in rats, and allocating 5% of the upper limit of the ADI to drinking-water. However, because fenitrothion occurs at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value.

Fenoprop

The half-lives for degradation of chlorophenoxy herbicides, including fenoprop (CAS No. 93-72-1), also known as 2,4,5-trichlorophenoxy propionic acid or 2,4,5-TP, in the environment are in the order of several days. Chlorophenoxy herbicides are not often found in food.

Guideline value	0.009 mg/l (9 µg/l)
Occurrence	Chlorophenoxy herbicides not frequently found in drinking-water; when detected, concentrations usually no greater than a few micrograms per litre
TDI	3 µg/kg body weight, based on a NOAEL of 0.9 mg/kg body weight for adverse effects on the liver in a study in which dogs were administered fenoprop in the diet for 2 years, with an uncertainty factor of 300 (100 for interspecies and intraspecies variation and 3 for limitations of the database)

Limit of detection	0.2 µg/l by either packed or capillary column GC with ECD
Treatment performance	0.001 mg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenoxy herbicides (excluding 2,4-D and MCPA) in drinking-water</i>

Chlorophenoxy herbicides, as a group, have been classified in Group 2B (possibly carcinogenic to humans) by IARC. However, the available data from studies in exposed populations and experimental animals do not permit assessment of the carcinogenic potential to humans of any specific chlorophenoxy herbicide. Therefore, drinking-water guidelines for these compounds are based on a threshold approach for other toxic effects. Effects observed in long-term studies with dogs given fenoprop in the diet include mild degeneration and necrosis of hepatocytes and fibroblastic proliferation in one study and severe liver pathology in another study. In rats, increased kidney weight was observed in two long-term dietary studies.

Fluoride¹

Fluorine is a common element that is widely distributed in Earth's crust and exists in the form of fluorides in a number of minerals, such as fluorspar, cryolite and fluorapatite. Traces of fluorides are present in many waters, with higher concentrations often associated with groundwaters. In some areas rich in fluoride-containing minerals, well water may contain up to about 10 mg of fluoride per litre, although much higher concentrations can be found. High fluoride concentrations can be found in many parts of the world, particularly in parts of India, China, Central Africa and South America, but high concentrations can be encountered locally in most parts of the world. Virtually all foodstuffs contain at least traces of fluorine. All vegetation contains some fluoride, which is absorbed from soil and water. Tea in particular can contain high fluoride concentrations, and levels in dry tea are on average 100 mg/kg.

Fluoride is widely used in dental preparations to combat dental caries, particularly in areas of high sugar intake. These can be in the form of tablets, mouthwashes, toothpaste, varnishes or gels for local application. In some countries, fluoride may also be added to table salt or drinking-water in order to provide protection against dental caries. The amounts added to drinking-water are such that final concentrations are usually between 0.5 and 1 mg/l. The fluoride in final water is always present as fluoride ions, whether from natural sources or from artificial fluoridation.

¹ As fluoride is one of the chemicals of greatest health concern in some natural waters, its chemical fact sheet has been expanded.

Total daily fluoride exposure can vary markedly from one region to another. This will depend on the concentration of fluoride in drinking-water and the amount drunk, levels in foodstuffs and the use of fluoridated dental preparations. In addition, fluoride exposure in some areas is considerably higher as a consequence of a range of practices, including the consumption of brick tea and the cooking and drying of food with high-fluoride coal.

Guideline value	1.5 mg/l (1500 µg/l)
Occurrence	In groundwater, concentrations vary with the type of rock through which the water flows but do not usually exceed 10 mg/l; highest natural level reported is 2800 mg/l
Basis of guideline value derivation	Epidemiological evidence that concentrations above this value carry an increasing risk of dental fluorosis and that progressively higher concentrations lead to increasing risks of skeletal fluorosis. The value is higher than that recommended for artificial fluoridation of water supplies, which is usually 0.5–1.0 mg/l.
Limit of detection	0.01 mg/l by ion chromatography; 0.1 mg/l by ion-selective electrodes or the sulfo phenyl azo dihydroxy naphthalene disulfonic acid colorimetric method
Treatment performance	1 mg/l should be achievable using activated alumina (not a “conventional” treatment process, but relatively simple to install filters)
Additional comments	A management guidance document on fluoride is available. In setting national standards for fluoride or in evaluating the possible health consequences of exposure to fluoride, it is essential to consider the intake of water by the population of interest and the intake of fluoride from other sources (e.g. from food, air and dental preparations). Where the intakes from other sources are likely to approach, or be greater than, 6 mg/day, it would be appropriate to consider setting standards at concentrations lower than the guideline value. In areas with high natural fluoride levels in drinking-water, the guideline value may be difficult to achieve, in some circumstances, with the treatment technology available.
Assessment date	2003
Principal references	Fawell et al. (2006) <i>Fluoride in drinking-water</i> IPCS (2002) <i>Fluorides</i> USNRC (2006) <i>Fluoride in drinking water</i> WHO (2004) <i>Fluoride in drinking-water</i>

After oral uptake, water-soluble fluorides are rapidly and almost completely absorbed from the gastrointestinal tract, although this may be reduced by complex formation with aluminium, phosphorus, magnesium or calcium. There is no difference in absorption between natural or added fluoride in drinking-water. Fluoride in inhaled particles—from high-fluoride coal, for example—is also absorbed, depending on the particle size and solubility of the fluoride compounds present. Absorbed fluoride is rapidly distributed throughout the body, where it is incorporated into teeth and bones, with virtually no storage in soft tissues. Fluoride in teeth and bone can be

mobilized after external exposure has ceased or been reduced. Fluoride is excreted via urine, faeces and sweat.

Fluoride may be an essential element for humans; however, essentiality has not been demonstrated unequivocally. Meanwhile, there is evidence of fluoride being a beneficial element with regard to the prevention of dental caries.

To produce signs of acute fluoride intoxication, minimum oral doses of about 1 mg of fluoride per kilogram of body weight were required. Many epidemiological studies of possible adverse effects of the long-term ingestion of fluoride via drinking-water have been carried out. These studies clearly establish that high fluoride intakes primarily produce effects on skeletal tissues (bones and teeth). Low concentrations provide protection against dental caries, both in children and in adults. The protective effects of fluoride increase with concentration up to about 2 mg of fluoride per litre of drinking-water; the minimum concentration of fluoride in drinking-water required to produce it is approximately 0.5 mg/l. However, fluoride can also have an adverse effect on tooth enamel and may give rise to mild dental fluorosis (prevalence: 12–33%) at drinking-water concentrations between 0.9 and 1.2 mg/l, depending on drinking-water intake and exposure to fluoride from other sources. Mild dental fluorosis may not be detectable except by specialist examination. The risk of dental fluorosis will depend on the total intake of fluoride from all sources and not just the concentration in drinking-water.

Elevated fluoride intakes can have more serious effects on skeletal tissues. Skeletal fluorosis (with adverse changes in bone structure) may be observed when drinking-water contains 3–6 mg of fluoride per litre, particularly with high water consumption. Crippling skeletal fluorosis usually develops only where drinking-water contains over 10 mg of fluoride per litre. IPCS concluded that there is clear evidence from India and China that skeletal fluorosis and an increased risk of bone fractures occur at a total intake of 14 mg of fluoride per day. This conclusion was supported by a review by the United States National Research Council in 2006. The relationship between exposure and response for adverse effects in bone is frequently difficult to ascertain because of inadequacies in most of the epidemiological studies. IPCS concluded from estimates based on studies from China and India that for a total intake of 14 mg/day, there is a clear excess risk of skeletal adverse effects; and there is suggestive evidence of an increased risk of effects on the skeleton at total fluoride intakes above about 6 mg/day.

Several epidemiological studies are available on the possible association between fluoride in drinking-water and cancer. IPCS evaluated these studies and concluded that, overall, the evidence of carcinogenicity in laboratory animals is inconclusive and that the available evidence does not support the hypothesis that fluoride causes cancer in humans; however, the data on bone cancer are limited. The results of several epidemiological studies on the possible adverse effects of fluoride in drinking-water on pregnancy outcome indicate that there is no relationship between the rates of Down syndrome or congenital malformation and the consumption of fluoridated drinking-water.

There is no evidence to suggest that the guideline value of 1.5 mg/l set in 1984 and reaffirmed in 1993 needs to be revised. Concentrations above this value carry an increasing risk of dental fluorosis, and much higher concentrations lead to skeletal

fluorosis. The value is higher than that recommended for artificial fluoridation of water supplies, which is usually 0.5–1.0 mg/l.

In setting national standards or local guidelines for fluoride or in evaluating the possible health consequences of exposure to fluoride, it is essential to consider the average daily intake of water by the population of interest and the intake of fluoride from other sources (e.g. from food and air). Where the intakes are likely to approach, or be greater than, 6 mg/day, it would be appropriate to consider setting a standard or local guideline at a concentration lower than 1.5 mg/l.

Practical considerations

Fluoride is usually determined by means of an ion-selective electrode, which makes it possible to measure the total amount of free and complex-bound fluoride dissolved in water. The method can detect fluoride concentrations in water well below the guideline value. However, appropriate sample preparation is a critical step in the accurate quantification of fluoride, especially where only the free fluoride ion is measured.

A range of treatment technologies are available for both large and small supplies. Different methods for small supplies are favoured in different countries; these are based on bone charcoal, contact precipitation, activated alumina and clay. However, in some areas with high natural fluoride levels in drinking-water, the guideline value may be difficult to achieve in some circumstances with the treatment technology available. Large supplies tend to rely on activated alumina or advanced treatment processes such as reverse osmosis.

Formaldehyde

Formaldehyde occurs in industrial effluents and is emitted into air from plastic materials and resin glues. Formaldehyde in drinking-water results primarily from the oxidation of natural organic matter during ozonation and chlorination. Concentrations of up to 30 µg/l have been found in ozonated drinking-water. Formaldehyde can also be found in drinking-water as a result of release from polyacetal plastic fittings. Formaldehyde's physicochemical properties suggest that it is unlikely to volatilize from water, so exposure by inhalation during showering is expected to be low.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2004
Principal references	IPCS (2002) <i>Formaldehyde</i> WHO (2005) <i>Formaldehyde in drinking-water</i>

Rats and mice exposed to formaldehyde by inhalation exhibited an increased incidence of carcinomas of the nasal cavity at doses that caused irritation of the nasal epithelium. Ingestion of formaldehyde in drinking-water for 2 years caused stomach irritation in rats. Papillomas of the stomach associated with severe tissue irritation were observed in one study. IARC has classified formaldehyde in Group 1 (carcinogenic to

humans). The weight of evidence indicates that formaldehyde is not carcinogenic by the oral route.

Owing to formaldehyde's high reactivity, effects in the tissue of first contact following ingestion are more likely to be related to the concentration of the formaldehyde consumed than to its total intake. A tolerable concentration of 2.6 mg/l for ingested formaldehyde has been established based on a NOEL of 260 mg/l for histopathological effects in the oral and gastric mucosa of rats administered formaldehyde in their drinking-water for 2 years, using an uncertainty factor of 100 (for interspecies and intraspecies variation). In view of the significant difference between the expected concentrations of formaldehyde in drinking-water and the tolerable concentration, it is not considered necessary to set a formal guideline value for formaldehyde.

Glyphosate and AMPA

Glyphosate (CAS No. 1071-83-6) is a broad-spectrum herbicide used in both agriculture and forestry and for aquatic weed control. Microbial biodegradation of glyphosate occurs in soil, aquatic sediment and water, the major metabolite being aminomethylphosphonic acid (AMPA) (CAS No. 1066-51-9). Glyphosate is chemically stable in water and is not subject to photochemical degradation. The low mobility of glyphosate in soil indicates minimal potential for the contamination of groundwater. Glyphosate can, however, enter surface and subsurface waters after direct use near aquatic environments or by runoff or leaching from terrestrial applications.

Reason for not establishing guideline values	Occur in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1998) <i>Pesticide residues in food—1997 evaluations</i> IPCS (1994) <i>Glyphosate</i> WHO (2005) <i>Glyphosate and AMPA in drinking-water</i>

Glyphosate and AMPA have similar toxicological profiles, and both are considered to exhibit low toxicity. A health-based value of 0.9 mg/l can be derived based on the group ADI for AMPA alone or in combination with glyphosate of 0–0.3 mg/kg body weight, based upon a NOAEL of 32 mg/kg body weight per day, the highest dose tested, identified in a 26-month study of toxicity in rats fed technical-grade glyphosate and using an uncertainty factor of 100 (for interspecies and intraspecies variation).

Because of their low toxicity, the health-based value derived for AMPA alone or in combination with glyphosate is orders of magnitude higher than concentrations of glyphosate or AMPA normally found in drinking-water. Under usual conditions, therefore, the presence of glyphosate and AMPA in drinking-water does not represent a hazard to human health. For this reason, the establishment of a formal guideline value for glyphosate and AMPA is not deemed necessary.

Halogenated acetonitriles (dichloroacetonitrile, dibromoacetonitrile, bromochloroacetonitrile, trichloroacetonitrile)

Halogenated acetonitriles are produced during water chlorination or chloramination from naturally occurring substances, including algae, fulvic acid and proteinaceous material. In general, increasing temperature or decreasing pH is associated with increasing concentrations of halogenated acetonitriles. Ambient bromide levels appear to influence, to some degree, the speciation of halogenated acetonitrile compounds. Dichloroacetonitrile is by far the most predominant halogenated acetonitrile species detected in drinking-water.

Provisional guideline value	<i>Dichloroacetonitrile</i> : 0.02 mg/l (20 µg/l) The guideline value for dichloroacetonitrile is provisional owing to limitations of the toxicological database.
Guideline value	<i>Dibromoacetonitrile</i> : 0.07 mg/l (70 µg/l)
Occurrence	Concentrations of individual halogenated acetonitriles can exceed 0.01 mg/l, although levels of 0.002 mg/l or less are more usual
TDIs	<i>Dichloroacetonitrile</i> : 2.7 µg/kg body weight based on a LOAEL of 8 mg/kg body weight per day for increased relative liver weight in male and female rats in a 90-day study, using an uncertainty factor of 3000 (taking into consideration intraspecies and interspecies variation, the short duration of the study, the use of a minimal LOAEL and database deficiencies) <i>Dibromoacetonitrile</i> : 11 µg/kg body weight, based on a NOAEL of 11.3 mg/kg body weight per day for decreased body weight in male rats in a 90-day drinking-water study and an uncertainty factor of 1000 (accounting for interspecies and intraspecies variation, subchronic to chronic extrapolation and database insufficiencies)
Limit of detection	0.03 µg/l by GC with ECD
Treatment performance	Reduction of organic precursors will reduce the formation of halogenated acetonitriles.
Guideline value derivation	• allocation to water 20% of TDI • weight 60 kg adult • consumption 2 litres/day
Assessment date	2003
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2004) <i>Halogenated acetonitriles in drinking-water</i>
Reason for not establishing guideline values	Available data inadequate to permit derivation of health-based guideline values for <i>bromochloroacetonitrile</i> and <i>trichloroacetonitrile</i>
Assessment date	2003
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2004) <i>Halogenated acetonitriles in drinking-water</i>

IARC has concluded that dichloroacetonitrile, dibromoacetonitrile, bromochloroacetonitrile and trichloroacetonitrile are not classifiable as to their carcinogenicity in humans. Dichloroacetonitrile and bromochloroacetonitrile have been shown to be mutagenic in bacterial assays, whereas results for dibromoacetonitrile and trichloroacetonitrile were negative. All four of these halogenated acetonitriles induced sister chromatid exchange and DNA strand breaks and adducts in mammalian cells in vitro but were negative in the mouse micronucleus test.

The majority of reproductive and developmental toxicity studies of the halogenated acetonitriles were conducted using tricapyrylin as a vehicle for gavage administration of the compound under study. As tricapyrylin was subsequently demonstrated to be a developmental toxicant that potentiated the effects of trichloroacetonitrile and, presumably, other halogenated acetonitriles, results reported for developmental studies using tricapyrylin as the gavage vehicle are likely to overestimate the developmental toxicity of these halogenated acetonitriles.

Dichloroacetonitrile

Dichloroacetonitrile induced decreases in body weight and increases in relative liver weight in short-term studies. Although developmental toxicity has been demonstrated, the studies used tricapyrylin as the vehicle for gavage administration.

Dibromoacetonitrile

Dibromoacetonitrile is currently under analysis for chronic toxicity in mice and rats. None of the available reproductive or developmental studies were adequate to use in the quantitative dose-response assessment. The data gap may be particularly relevant because cyanide, a metabolite of dibromoacetonitrile, induces male reproductive system toxicity and because of uncertainty regarding the significance of the testes effects observed in a 14-day NTP rat study.

Bromochloroacetonitrile

Available data are insufficient to serve as a basis for derivation of a guideline value for bromochloroacetonitrile.

Trichloroacetonitrile

Available data are also insufficient to serve as a basis for derivation of a guideline value for trichloroacetonitrile. The previous provisional guideline value of 1 µg/l was based on a developmental toxicity study in which trichloroacetonitrile was administered by gavage in tricapyrylin vehicle, and a re-evaluation judged this study to be unreliable in light of the finding in a more recent study that tricapyrylin potentiates the developmental and teratogenic effects of halogenated acetonitriles and alters the spectrum of malformations in the fetuses of treated dams.

Hardness

Hardness in water is caused by a variety of dissolved polyvalent metallic ions, predominantly calcium and magnesium cations. It is usually expressed as milligrams of calcium carbonate per litre. Hardness is the traditional measure of the capacity of

water to react with soap, hard water requiring considerably more soap to produce a lather.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993, revised in 2011
Principal reference	WHO (2011) <i>Hardness in drinking-water</i>

Natural and treated waters have a wide range of mineral content, from very low levels in rainwater and naturally soft and softened water to higher levels in naturally hard waters. Bottled and packaged waters can be naturally mineralized or naturally soft or demineralized. Thus, the mineral consumption from drinking-water and cooking water will vary widely, depending upon location, treatment and water source.

The degree of hardness of drinking-water is important for aesthetic acceptability by consumers (see [chapter 10](#)) and for economic and operational considerations. Many hard waters are softened for those reasons using several applicable technologies. The choice of the most appropriate conditioning technology will depend on local circumstances (e.g. water quality issues, piping materials, corrosion) and will be applied either centrally or in individual homes as a consumer preference.

Consumers should be informed of the mineral composition of their water, whether or not it is modified. The contribution of drinking-water minerals to mineral nutrition should be considered where changes in supply are proposed or where less traditional sources, such as recycled water, seawater or brackish water, are processed and exploited for drinking-water. The treatments used remove most minerals, and stabilization of water is always necessary prior to distribution.

Drinking-water can be a contributor to calcium and magnesium intake and could be important for those who are marginal for calcium and magnesium. Where drinking-water supplies are supplemented with or replaced by demineralized water that requires conditioning, consideration should be given to adding calcium and magnesium salts to achieve concentrations similar to those that the population received from the original supply. Modification of calcium and magnesium concentrations in drinking-water for health reasons should comply with the technical requirements to provide water suitable for distribution.

Although there is evidence from epidemiological studies for a protective effect of magnesium or hardness on cardiovascular mortality, the evidence is being debated and does not prove causality. Further studies are being conducted. There are insufficient data to suggest either minimum or maximum concentrations of minerals at this time, as adequate intake will depend on a range of other factors. Therefore, no guideline values are proposed.

Heptachlor and heptachlor epoxide

Heptachlor (CAS No. 76-44-8) is a broad-spectrum insecticide, the use of which has been banned or restricted in many countries. At present, the major use of heptachlor

is for termite control by subsurface injection into soil. Heptachlor is quite persistent in soil, where it is mainly transformed to its epoxide. Heptachlor epoxide (CAS No. 1024-57-3) is very resistant to further degradation. Heptachlor and heptachlor epoxide bind to soil particles and migrate very slowly. Heptachlor and heptachlor epoxide have been found in drinking-water at nanogram per litre levels. Diet is considered to represent the major source of exposure to heptachlor, although intake is decreasing significantly, as its use has substantially declined.

Reason for not establishing a guideline value	Occur in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1992) <i>Pesticide residues in food—1991 evaluations</i> FAO/WHO (1995) <i>Pesticide residues in food—1994 evaluations</i> WHO (2003) <i>Heptachlor and heptachlor epoxide in drinking-water</i>

Prolonged exposure to heptachlor has been associated with damage to the liver and central nervous system toxicity. In 1991, IARC reviewed the data on heptachlor and concluded that the evidence for carcinogenicity was sufficient in animals and inadequate in humans, classifying it in Group 2B (possibly carcinogenic to humans). A health-based value of 0.03 µg/l can be calculated for heptachlor and heptachlor epoxide on the basis of a PTDI of 0.1 µg/kg body weight, based on a NOAEL for heptachlor of 0.025 mg/kg body weight per day from two studies in the dog, taking into consideration inadequacies of the database and allocating 1% of the PTDI to drinking-water. However, because heptachlor and heptachlor epoxide occur at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value. It should also be noted that concentrations below 0.1 µg/l are generally not achievable using conventional treatment technology.

Hexachlorobenzene

The major agricultural application for hexachlorobenzene (CAS No. 118-74-1), or HCB, was as a seed dressing for crops to prevent the growth of fungi, but its use is now uncommon. At present, it appears mainly as a by-product of several chemical processes or an impurity in some pesticides. HCB is distributed throughout the environment because it is mobile and resistant to degradation. It bioaccumulates in organisms because of its physicochemical properties and its slow elimination. HCB is commonly detected at low levels in food, and it is generally present at low concentrations in ambient air. It has been detected only infrequently, and at very low concentrations (below 0.1 µg/l), in drinking-water supplies.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	IPCS (1997) <i>Hexachlorobenzene</i> WHO (2004) <i>Hexachlorobenzene in drinking-water</i>

IARC has evaluated the evidence for the carcinogenicity of HCB in animals and humans and assigned it to Group 2B (possibly carcinogenic to humans). HCB has been shown to induce tumours in three animal species and at a variety of sites. A health-based value of 1 µg/l can be derived for HCB by applying the linearized multistage low-dose extrapolation model to liver tumours observed in female rats in a 2-year dietary study. Using an alternative (tumorigenic dose₀₅, or TD₀₅) approach, a TDI of 0.16 µg/kg body weight can be calculated, which corresponds to a health-based value of approximately 0.05 µg/l, if one assumes a 1% allocation of the TDI to drinking-water. It should be noted that concentrations in food have been falling steadily, and this allocation factor may be considered very conservative.

Because the health-based values derived from both of these approaches are considerably higher than the concentrations at which HCB is detected in drinking-water (i.e. sub-nanograms per litre), when it is detected, it is not considered necessary to establish a formal guideline value for HCB in drinking-water. HCB is listed under the Stockholm Convention on Persistent Organic Pollutants.

Hexachlorobutadiene

Hexachlorobutadiene, or HCB, is used as a solvent in chlorine gas production, a pesticide, an intermediate in the manufacture of rubber compounds and a lubricant. Concentrations of up to 6 µg/l have been reported in the effluents from chemical manufacturing plants. HCB is also found in air and food.

Guideline value	0.0006 mg/l (0.6 µg/l)
Occurrence	Has been detected in surface water at concentrations of a few micrograms per litre and in drinking-water at concentrations below 0.5 µg/l
TDI	0.2 µg/kg body weight, based on a NOAEL of 0.2 mg/kg body weight per day for renal toxicity in a 2-year feeding study in rats, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for limited evidence of carcinogenicity and genotoxicity of some metabolites)
Limit of detection	0.01 µg/l by GC-MS; 0.18 µg/l by GC with ECD
Treatment performance	0.001 mg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The practical quantification limit for HCB is of the order of 2 µg/l, but concentrations in drinking-water can be controlled by specifying the HCB content of products coming into contact with it.
Assessment date	2003
Principal references	IPCS (1994) <i>Hexachlorobutadiene</i> WHO (2003) <i>Hexachlorobutadiene in drinking-water</i>

HCBD is easily absorbed and metabolized via conjugation with glutathione. This conjugate can be further metabolized to a nephrotoxic derivative. Kidney tumours were observed in a long-term oral study in rats. HCBD has not been shown to be carcinogenic by other routes of exposure. IARC has placed HCBD in Group 3 (not classifiable as to its carcinogenicity to humans). Positive and negative results for HCBD have been obtained in bacterial assays for point mutation; however, several metabolites have given positive results.

Hydrogen sulfide

Hydrogen sulfide is a gas with an offensive “rotten eggs” odour that is detectable at very low concentrations, below 0.8 µg/m³ in air. It is formed when sulfides are hydrolysed in water. However, the level of hydrogen sulfide found in drinking-water will usually be low, because sulfides are readily oxidized in well-aerated or chlorinated water.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Hydrogen sulfide in drinking-water</i>

The acute toxicity to humans of hydrogen sulfide following inhalation of the gas is high; eye irritation can be observed at concentrations of 15–30 mg/m³. Although oral toxicity data are lacking, it is unlikely that a person could consume a harmful dose of hydrogen sulfide from drinking-water. Consequently, no guideline value is proposed. However, hydrogen sulfide can be easily detected in drinking-water by taste or odour (see [chapter 10](#)).

Inorganic tin

Tin is used principally in the production of coatings used in the food industry. Food, particularly canned food, therefore represents the major route of human exposure to tin. For the general population, drinking-water is not a significant source of tin, and levels in drinking-water greater than 1–2 µg/l are exceptional. However, there is increasing use of tin in solder, which may be used in domestic plumbing, and tin has been proposed for use as a corrosion inhibitor.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal reference	WHO (2004) <i>Inorganic tin in drinking-water</i>

Tin and inorganic tin compounds are poorly absorbed from the gastrointestinal tract, do not accumulate in tissues and are rapidly excreted, primarily in faeces.

No increased incidence of tumours was observed in long-term carcinogenicity studies conducted in mice and rats fed tin(II) chloride. Tin has not been shown to be teratogenic or fetotoxic in mice, rats or hamsters. In rats, the NOAEL in a long-term feeding study was 20 mg/kg body weight per day.

The main adverse effect on humans of excessive levels of tin in canned beverages (above 150 mg/kg) or other canned foods (above 250 mg/kg) has been acute gastric irritation. There is no evidence of adverse effects in humans associated with chronic exposure to tin.

In 1989, JECFA established a PTWI of 14 mg/kg body weight from a TDI of 2 mg/kg body weight on the basis that the problem with tin is associated with acute gastrointestinal irritancy, the threshold for which is about 200 mg/kg in food. This was reaffirmed by JECFA in 2000. In view of its low toxicity, the presence of tin in drinking-water does not, therefore, represent a hazard to human health. For this reason, the establishment of a guideline value for inorganic tin is not deemed necessary.

Iodine

Iodine occurs naturally in water in the form of iodide. Traces of iodine are produced by oxidation of iodide during water treatment. Iodine is occasionally used for water disinfection in the field or in emergency situations. The diet is the major source of exposure to iodine for the general human population; the contribution to total exposure from drinking-water is assumed to be low (around 5%).

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value. Additionally, occurrence in drinking-water is usually low. Although higher levels of exposure may occur when iodine is used as a drinking-water disinfectant at the point of use, extended periods of exposure to iodine through water disinfection are unlikely.
Occurrence	Average concentrations have ranged from 0.5 to 20 µg/l in rivers and lakes
Limit of detection	10 µg/l by a leuco crystal violet method
Treatment performance	Not applicable as occurrence in drinking-water is usually low
Additional comments	Iodine is not recommended for use as a primary disinfectant of drinking-water but it can be used as a point-of-use disinfectant. Caution should be exercised for longer-term point-of-use disinfection in susceptible individuals (see Part II of the supporting document <i>Alternative drinking-water disinfectants: bromine, iodine and silver</i> ; Annex 1).
Assessment date	2020
Principal reference	WHO (2020) <i>Iodine in drinking-water</i>

Iodine is an essential element for the synthesis of thyroid hormones. Various national and international organizations have established recommended daily intakes, with recommendations by WHO/FAO set in 2004 at 90 µg/day for infants, 120 µg/day for children, 150 µg/day for adults and 200 µg/day for pregnant or lactating women. In many parts of the world, dietary deficiencies in iodine are a health issue, which can adversely affect neurological development.

In 1989, JECFA set a provisional maximum tolerable daily intake (PMTDI) for iodine of 1000 µg/day (17 µg/kg body weight per day) from all sources, based primarily on data on the effects of iodide. Various national and international organizations have also established recommended upper intake levels for iodine, ranging from 500 to 1100 µg/day. Exposure to excess iodine can lead to hypothyroidism (with or without goitre—enlargement of the thyroid), hyperthyroidism, and changes in the incidence and types of thyroid malignancies.

However, available data are inadequate to establish a dose–response relationship between iodine exposure and changes in thyroid levels, and to clearly identify a threshold associated with thyrotoxicosis. Further, a guideline value cannot be derived using the more robust toxicological dataset for iodide because the effects of iodine and iodide on thyroid hormone concentrations in the blood may differ. Because iodine is generally not recommended for long-term disinfection, lifetime exposure to iodine concentrations such as might occur from water disinfection is unlikely. For these reasons, a guideline value for iodine has not been established at this time.

Disinfection of iodide-containing water by adding chlorine or chloramine can result in the production of iodinated DBPs; most often, these are formed during chloramination when complete oxidation is prevented. Iodinated DBPs are occasionally detected in drinking-water from treatment plants located in coastal saltwater areas. As with all DBPs, their concentrations in drinking-water can be reduced at the treatment plant by removing the natural organic matter from the water before the disinfection process occurs. It is critical that any method used to control DBP levels does not compromise the effectiveness of disinfection.

Iron

Iron is one of the most abundant metals in Earth’s crust. It is found in natural fresh waters at levels ranging from 0.5 to 50 mg/l. Iron may also be present in drinking-water as a result of the use of iron coagulants or the corrosion of steel and cast iron pipes during water distribution.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Iron in drinking-water</i>

Iron is an essential element in human nutrition, particularly in the iron(II) oxidation state. Estimates of the minimum daily requirement for iron depend on age, sex, physiological status and iron bioavailability and range from about 10 to 50 mg/day.

As a precaution against storage in the body of excessive iron, in 1983, JECFA established a PMTDI of 0.8 mg/kg body weight, which applies to iron from all sources except for iron oxides used as colouring agents and iron supplements taken during pregnancy and lactation or for specific clinical requirements. An allocation of 10% of this PMTDI to drinking-water gives a value of about 2 mg/l, which does not present a

hazard to health. The taste and appearance of drinking-water will usually be affected below this level (see [chapter 10](#)).

No guideline value for iron in drinking-water is proposed.

Isoproturon

Isoproturon (CAS No. 34123-59-6) is a selective, systemic herbicide used in the control of annual grasses and broad-leaved weeds in cereals. It can be photodegraded, hydrolysed and biodegraded and persists for periods ranging from days to weeks. It is mobile in soil. There is evidence that exposure to this compound through food is low.

Guideline value	0.009 mg/l (9 µg/l)
Occurrence	Has been detected in surface water and groundwater, usually at concentrations below 0.1 µg/l; levels above 0.1 µg/l have occasionally been detected in drinking-water
TDI	3 µg/kg body weight based on a NOAEL of approximately 3 mg/kg body weight in a 90-day study in dogs and a 2-year feeding study in rats, with an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for evidence of non-genotoxic carcinogenicity in rats)
Limit of detection	10–100 ng/l by reversed-phase HPLC followed by UV or electrochemical detection
Treatment performance	0.1 µg/l should be achievable using ozonation
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Isoproturon in drinking-water</i>

Isoproturon is of low acute toxicity and low to moderate toxicity following short-term and long-term exposures. It does not possess significant genotoxic activity, but it causes marked enzyme induction and liver enlargement. Isoproturon caused an increase in hepatocellular tumours in male and female rats, but this was apparent only at doses that also caused liver toxicity. Isoproturon appears to be a tumour promoter rather than a complete carcinogen.

Lead

Lead is used principally in the production of lead-acid batteries, solder and alloys. The organolead compounds tetraethyl and tetramethyl lead have also been used extensively as antiknock and lubricating agents in petrol, although their use for these purposes in all countries has been phased out. Owing to the decreasing use of lead-containing additives in petrol and of lead-containing solder in the food processing industry, concentrations in air and food are declining; in most countries, lead levels in blood are also declining unless there are specific sources, such as dust from leaded paint or occupational/household recycling of lead-containing materials. Lead is rarely present in

tap water as a result of its dissolution from natural sources; rather, its presence is primarily from corrosive water effects on household plumbing systems containing lead in pipes, solder or fittings (including alloy fittings with high lead content), or from the service connections to homes. The amount of lead dissolved from the plumbing system depends on several factors, including pH, temperature, alkalinity, scale in pipe and standing time of the water, with soft, acidic water being the most plumbosolvent. Free chlorine residuals in drinking-water tend to form more insoluble lead-containing deposits, whereas chloramine residuals may form more soluble sediments in lead pipe. Accordingly, significant changes in the water quality of a supply, resulting from, for example, changes in treatment or changes of source, can result in changes in plumbosolvency or solubilization of lead deposits, or both.

Provisional guideline value	0.01 mg/l (10 µg/l)
	The guideline value is designated as provisional on the basis of treatment performance and analytical achievability. As this is no longer a health-based guideline value, concentrations should be maintained as low as reasonably practical. New sources of lead, such as service connections and lead solder, should not be introduced into any system, and low lead alloy fittings should be used in repairs and new installations.
Occurrence	Concentrations in drinking-water are generally below 5 µg/l, although much higher concentrations (above 100 µg/l) have been measured where lead service connections or fittings are present. The primary source of lead is from service connections and plumbing in buildings; therefore, lead should be measured at the tap. Lead concentrations can also vary according to the period in which the water has been in contact with the lead-containing materials.
Basis of guideline derivation	The guideline value was previously based on a JECFA PTWI, which has since been withdrawn, and no new PTWI has been established, on the basis that there does not appear to be a threshold for the key effects of lead. However, substantial efforts have been made to reduce lead exposure from a range of sources, including drinking-water. The guideline value is maintained at 10 µg/l but is designated as provisional on the basis of treatment performance and analytical achievability because it is extremely difficult to achieve a lower concentration than this by central conditioning, such as phosphate dosing.
Limit of detection	1 µg/l by AAS; practical quantification limit in the region of 1–10 µg/l
Treatment performance	Not a raw water contaminant; treatment not applicable
Additional comments	Infants and children are considered to be the most sensitive subgroups of the population Lead is exceptional compared with other chemical hazards, in that most lead in drinking-water arises from lead service connections and plumbing in buildings, and the remedy consists principally of removing service connections, plumbing and fittings containing lead. This requires much time and money, and it is recognized that not all water will meet the guideline value immediately. Meanwhile, all other practical measures to reduce total exposure to lead, including corrosion control, should be implemented. In new installations or repairs, lead-free service connections and solder and low lead alloy fittings should be used to prevent the introduction of contamination.

The sampling protocol adopted – e.g. first draw, random daytime sampling or flushed – will depend on the objective of taking the samples. Where there is a need to verify that lead solder and/or high-lead fittings have not been installed in new or repaired systems, the approach used is to take a worst-case sample that reflects an extended period of stagnation, to maximize the chance of identifying the presence of lead.

Assessment date	2016
Principal references	FAO/WHO (2011) <i>Evaluation of certain food additives and contaminants</i> WHO (2011, revised 2016) <i>Lead in drinking-water</i>

Exposure to lead is associated with a wide range of effects, including various neurodevelopmental effects, mortality (mainly due to cardiovascular diseases), impaired renal function, hypertension, impaired fertility and adverse pregnancy outcomes. Impaired neurodevelopment in children is generally associated with lower blood lead concentrations than the other effects, the weight of evidence is greater for neurodevelopmental effects than for other health effects and the results across studies are more consistent than those for other effects. For adults, the adverse effect associated with lowest blood lead concentrations for which the weight of evidence is greatest and most consistent is a lead-associated increase in systolic blood pressure. JECFA concluded that the effects on neurodevelopment and systolic blood pressure provided the appropriate bases for dose–response analyses.

Based on the dose–response analyses, JECFA estimated that the previously established PTWI of 25 µg/kg body weight is associated with a decrease of at least 3 intelligence quotient (IQ) points in children and an increase in systolic blood pressure of approximately 3 mmHg (0.4 kPa) in adults. These changes are important when viewed as a shift in the distribution of IQ or blood pressure within a population. JECFA therefore concluded that the PTWI could no longer be considered health protective, and it was withdrawn.

Because the dose–response analyses do not provide any indication of a threshold for the key effects of lead, JECFA concluded that it was not possible to establish a new PTWI that would be considered to be health protective. JECFA reaffirmed that because of the neurodevelopmental effects, fetuses, infants and children are the subgroups that are most sensitive to lead.

It needs to be recognized that lead is exceptional compared with other chemical hazards, in that most lead in drinking-water arises from lead service connections and plumbing in buildings, and the remedy consists principally of removing plumbing and fittings containing lead, which requires much time and money. It is therefore emphasized that practical measures to reduce total exposure to lead should be implemented in the interim. These should include corrosion control if the lead source is attributable to distribution systems. Other interim measures may include flushing and point-of-use or point-of-entry treatment. New sources of lead, such as lead service connections and solder, should not be introduced into any system, and low lead alloy fittings should be used in repairs and new installations.

In terms of monitoring, if the monitoring objective is to identify the presence of lead in the internal plumbing of a building, then the sample should be from the

tap. The sampling protocols also depend on the objective of taking the samples. First-draw samples typically will have the highest lead concentrations, but this may not be reflected in normal use if the same system provides water for toilet flushing, etc. Flushed samples, in contrast, give consistent values, but reflect the minimum contact time between the water and the lead-containing material. The random daytime samples, although most truly reflecting the water that the consumer drinks, give the most variable levels; hence, it is necessary to collect more samples to determine the mean level of exposure. Where there is a need to verify that lead service connections, lead solder and/or high-lead fittings have not been installed in new or repaired systems, the approach used is to take a worst-case sample that reflects an extended period of stagnation and to maximize the chance of identifying the presence of lead. Extended stagnation with sequential volume can also be used to identify sources or locations of lead as an investigative activity.

For further guidance on managing lead contamination of drinking-water, see *Lead in drinking-water: health risks, monitoring and corrective actions* ([Annex 1](#)).

Lindane

Lindane (γ -hexachlorocyclohexane; γ -HCH) (CAS No. 58-89-9) is used as an insecticide on fruit and vegetable crops, for seed treatment and in forestry. It is also used as a therapeutic pesticide in humans and animals. Several countries have restricted the use of lindane. Lindane can be degraded in soil and rarely leaches to groundwater. In surface waters, it can be removed by evaporation. Exposure of humans occurs mainly via food, but this is decreasing. There may also be exposure from its use in public health and as a wood preservative.

Guideline value	0.002 mg/l (2 μ g/l)
Occurrence	Has been detected in both surface water and groundwater, usually at concentrations below 0.1 μ g/l, although concentrations as high as 12 μ g/l have been measured in wastewater-contaminated rivers
ADI	0–0.005 mg/kg body weight on the basis of a NOAEL of 0.47 mg/kg body weight per day in a 2-year toxicity/carcinogenicity study in rats in which an increased incidence of periportal hepatocellular hypertrophy, increased liver and spleen weights and increased mortality occurred at higher doses, using an uncertainty factor of 100 (for interspecies and intraspecies variation)
Limit of detection	0.01 μ g/l using GC
Treatment performance	0.1 μ g/l should be achievable using GAC
Guideline value derivation	
• allocation to water	1% of upper limit of ADI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	It should be noted that concentrations in food have been falling steadily, and the 1% allocation factor may be considered very conservative.
Assessment date	2003

Principal references	FAO/WHO (2003) <i>Pesticide residues in food—2002 evaluations</i> WHO (2003) <i>Lindane in drinking-water</i>

Lindane was toxic to the kidney and liver after administration orally, dermally or by inhalation in short-term and long-term studies of toxicity and reproductive toxicity in rats. The renal toxicity of lindane was specific to male rats and was considered not to be relevant to human risk assessment, as it is a consequence of accumulation of α_{2u} -globulin, a protein that is not found in humans. Hepatocellular hypertrophy was observed in a number of studies in mice, rats and rabbits and was reversed only partially after recovery periods of up to 6 weeks. Lindane did not induce a carcinogenic response in rats or dogs, but it caused an increased incidence of adenomas and carcinomas of the liver in agouti and pseudoagouti mice, but not in black or any other strains of mice, in a study of the role of genetic background in the latency and incidence of tumorigenesis. JMPR concluded that there was no evidence of genotoxicity. In the absence of genotoxicity and on the basis of the weight of the evidence from the studies of carcinogenicity, JMPR concluded that lindane is not likely to pose a carcinogenic risk to humans. Further, in an epidemiological study designed to assess the potential association between breast cancer and exposure to chlorinated pesticides, no correlation with lindane was found.

Malathion

Malathion (CAS No. 121-75-5) is commonly used to control mosquitoes and a variety of insects that attack fruits, vegetables, landscaping plants and shrubs. It can also be found in other pesticide products used indoors, on pets to control ticks and insects and to control human head and body lice. Under least favourable conditions (i.e. low pH and little organic content), malathion may persist in water with a half-life of months or even years. However, under most conditions, the half-life appears to be roughly 7–14 days. Malathion has been detected in surface water and drinking-water at concentrations below 2 µg/l.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1998) <i>Pesticide residues in food—1997 evaluations</i> WHO (2003) <i>Malathion in drinking-water</i>

Malathion inhibits cholinesterase activity in mice, rats and human volunteers. It increased the incidence of liver adenomas in mice when administered in the diet. Most of the evidence indicates that malathion is not genotoxic, although some studies indicate that it can produce chromosomal aberrations and sister chromatid exchange in vitro. JMPR has concluded that malathion is not genotoxic.

A health-based value of 0.9 mg/l can be calculated for malathion based on an allocation of 10% of the upper limit of the JMPR ADI—based on a NOAEL of 29 mg/kg body weight per day in a 2-year study of toxicity and carcinogenicity in rats, using an

uncertainty factor of 100 for interspecies and intraspecies variation and supported by a NOAEL of 25 mg/kg body weight per day in a developmental toxicity study in rabbits—to drinking-water. However, intake of malathion from all sources is generally low and well below the upper limit of the ADI. As the chemical occurs in drinking-water at concentrations much lower than the health-based value, the presence of malathion in drinking-water under usual conditions is unlikely to represent a hazard to human health. For this reason, it is considered unnecessary to derive a formal guideline value for malathion in drinking-water.

Manganese¹

Manganese is one of the most abundant metals in Earth's crust, usually occurring with iron. It can exist in 11 oxidation states, often as chloride, oxides and sulfates. The most common oxidation states for manganese in natural water are manganese(II) and manganese(IV). Manganese is used principally in the manufacture of iron and steel alloys, and manganese compounds such as potassium and sodium permanganate are ingredients in various products used for cleaning, bleaching and disinfection. Manganese compounds are additionally used in some locations for potable water treatment and can also be an impurity in coagulants used during water treatment. Manganese occurs naturally in many surface water and groundwater sources; although naturally occurring manganese is usually the most important source for drinking-water, anthropogenic activities can also contribute to high levels of manganese in water. Manganese also occurs naturally in many food sources, and the greatest exposure to manganese is usually from food.

Provisional guideline value	<i>Total manganese:</i> 0.08 mg/l (80 µg/l), to be protective against neurological effects in the most sensitive subpopulation—bottle-fed infants—and consequently the general population
	This guideline value is provisional because of the high level of uncertainty, as reflected in the composite uncertainty factor of 1000
Occurrence	Levels in fresh waters vary widely. They are typically in the range 1–200 µg/l. Higher levels are usually associated with groundwater, lakes and reservoirs under acidic or reducing conditions, or in aerobic waters with industrial pollution. Very high concentrations (up to 10 mg/l) have been reported in acidic groundwater. In treated drinking-water, concentrations are typically less than 50 µg/l.
TDI	0.025 mg/kg bw, derived by applying an uncertainty factor of 1000 to a LOAEL of 25 mg/kg bw per day identified from studies that reported neurological effects in rats exposed to manganese from birth to postnatal day 21. The uncertainty factor takes into account interpecies variation (10), intraspecies variation (10) and database uncertainties (10, including use of a LOAEL).
Limit of detection	0.002 µg/l by ICP-MS; 0.005–50 µg/l by ICP-AES and GFAA spectrometry; and 10–70 µg/l by colorimetric methods. None of these methods distinguish between the different oxidation states of manganese.

¹ As naturally occurring manganese in drinking-water is a chemical of concern in many areas, its chemical fact sheet has been expanded.

12. CHEMICAL FACT SHEETS

Treatment performance	Manganese concentrations in drinking-water can be easily lowered to less than 0.05 mg/l using several treatment methods, including oxidation/filtration, adsorption/oxidation, softening/ion exchange and biological filtration. Selection of the appropriate treatment system for manganese removal depends on the form of manganese (dissolved or particulate) in the source water.
Guideline value derivation	• allocation to water 50% of TDI • weight 5 kg body weight for a bottle-fed infant • consumption 0.75 litres per day for a bottle-fed infant
Additional comments	Risks to infants arising from exceedance of the provisional guideline value may be mitigated by following the WHO recommendation for exclusive breastfeeding, or by using an alternative safe source of drinking-water (e.g. bottled water that is certified by the responsible authorities) to prepare formula. The presence of particulate manganese in drinking-water systems can cause acceptability problems; concentrations above 0.02 mg/l have caused complaints about discoloured water and staining of plumbing fixtures and laundry. Therefore, aesthetic as well as health aspects should be considered in the management of manganese in drinking-water, and when setting regulations and standards for drinking-water quality.
Assessment date	2020
Principal references	WHO (2021) <i>Manganese in drinking-water</i>

Manganese is an essential trace element. It is a necessary component of a number of enzymes and activates several others. The central nervous system is the primary concern for manganese toxicity in mammals, including humans. Neurodevelopmental toxicity studies in manganese-exposed juvenile rats revealed behavioural and sensorimotor effects, and corresponding neurostructural and neurochemical changes. Several epidemiological studies have also reported neurological effects (including reduced cognitive ability) in adult populations and children following ingestion of manganese-contaminated water. The epidemiological studies have limited utility in risk assessment due to uncertain manganese exposure levels, unclear temporality of effects and other potential confounding factors. However, collectively, they provide qualitative support that the neurological effects reported in animal studies are relevant in humans. Existing studies and reports do not provide adequate evidence to assess potential carcinogenicity from oral exposure to manganese in humans.

Absorption of manganese from the gastrointestinal (GI) tract has been suggested to take place through both an active transport mechanism and passive diffusion. GI absorption is influenced by several factors, including dietary factors: absorption is negatively correlated with intake of dietary fibre, oxalic acids and phytic acids. Some studies and reports have suggested that absorption and bioavailability of manganese are greater from drinking-water than from food, although other studies have reported no differences. However, absorption from drinking-water may be influenced by fast-ing conditions and the chemical form of manganese.

Following GI tract absorption, manganese is distributed via the systemic circulation to all tissues. Levels of manganese can increase in several tissues following oral exposure, including some regions of the brain in infants and adults. The main route of elimination of manganese from the body is faecal elimination via hepatobiliary excretion.

Manganese absorption from the GI tract may be higher in infants than in adults. Infants also retain higher levels of manganese than adults during the early neonatal period, possibly because of the incomplete development of the biliary excretion system. Along with the important neurodevelopmental processes occurring in neonates, this may render them particularly susceptible to toxicity from exposure to manganese. Further, there is potential for increased exposure to manganese in bottle-fed infants compared with breastfed infants—from the concentrated or powdered formula itself as well as the tap water used to prepare the formula.

Practical considerations

Manganese levels in drinking-water can be an issue in both high- and low-income countries, and should be considered in establishing national standards and local guidance. Resource-limited suppliers, in particular, may have difficulty in achieving the provisional guideline value; in such cases, incremental improvements towards meeting the provisional guideline value are encouraged. This is a particular problem for groundwater, for which treatment may be minimal and prohibitively expensive. In such instances, benefits from a reliable, microbiologically safe groundwater source should be assessed against the risks posed by an alternative source that may be subject to faecal contamination. Issues of acceptability of the drinking-water (which varies between different populations) should also be considered, since reduced acceptability may lead consumers to turn to more aesthetically acceptable but less microbiologically safe water supplies. It is vital that a sufficient supply of acceptable, microbiologically safe water is always available, even if some guidelines or standards for chemicals such as manganese cannot be immediately met.

Manganese should be evaluated and managed in the context of developing a WSP (see [chapter 4](#)). Surface waters prone to high and variable concentrations of manganese may require more frequent and targeted monitoring than groundwater. Where manganese is present at concentrations close to the provisional guideline value or the water is treated to remove manganese, routine monitoring should be conducted after treatment. If manganese is detected at the point of collection or use, or aesthetic issues related to manganese are reported by consumers, this indicates that treatment for manganese removal is not optimized or that the distribution system is not appropriately managed.

Options for controlling levels in groundwater include drilling a new well or blending water from different wells. For lake and reservoir sources where there is a thermocline and lower water levels become anoxic, management of the sources to prevent release of manganese from sediment is important.

Selection of the appropriate treatment system for manganese removal depends on the form of manganese (dissolved or particulate) in the source water. Dissolved manganese(II) is most often the predominant form present in anoxic and acidic

groundwater or lakes. However, depending on the pH and the dissolved oxygen content of the water, a combination of dissolved and particulate manganese can be present. In general, treatment methods used for manganese rely on a combination of processes (e.g. oxidation, adsorption, filtration) to remove both the dissolved and particulate forms. At the point of use, reverse osmosis is the most effective and reliable treatment technology; however, point-of-use units using ion exchange media are also moderately effective. To reduce water discoloration and staining of laundry and fixtures, ion exchange and greensand filtration with careful operation and maintenance can be used at the point of entry.

Low levels of manganese in source or treated water can accumulate in the distribution system. Periodic release of manganese can then occur, resulting in high levels at the tap. Releases can occur as a result of physical or hydraulic disturbances to the system (e.g. mains breaks, hydrant flushing) or changes in water chemistry (e.g. changes in pH, temperature, chlorine residual, source water type/blending). Physical and hydraulic disturbances most often release particulate manganese and can cause discoloured water and consumer complaints. Chemical releases can go unnoticed if manganese occurs predominantly in the dissolved form. Other contaminants (e.g. arsenic, barium, chromium, lead, uranium) that deposit with manganese oxides in the distribution system may also be released into the water and reach consumers' taps. Control measures to minimize manganese release events include maintaining stable water chemistry and minimizing manganese levels entering the distribution system, the amount of manganese oxide deposits in the distribution system (through best practices for water mains cleaning), and physical or hydraulic disturbances.

MCPA

MCPA is a phenoxyacetic acid herbicide that is found in various formulations: as the free acid (CAS No. 94-74-6), as a dimethylamine salt (CAS No. 2039-46-5), as a sodium salt (CAS No. 3653-48-3) and as a 2-ethylhexyl ester (CAS No. 29450-45-1). It is a post-emergence herbicide that is widely used against broadleaf weeds in agriculture and horticulture and on grassland and lawns. All forms of MCPA will dissociate in water to the acid (anion) form. MCPA is highly soluble in water. Biological degradation is an important process in determining MCPA's environmental fate. Chlorophenols and chlorocresols are potential soil metabolites and may, if present in water, give rise to unacceptable tastes. Surface water may be contaminated via spray drift and runoff, whereas groundwater may be contaminated via leaching from soil. Exposure from food is likely to be low.

Reason for not establishing a guideline value	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Health-based value*	0.7 mg/l
Acute health-based value**	20 mg/l
Occurrence	Concentrations in surface water usually less than 1 µg/l; concentrations in drinking-water usually below 0.1 µg/l

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ADI	0–0.1 mg/kg bw for MCPA ion, based on an overall NOAEL of 12 mg/kg bw per day for changes in clinical chemistry parameters indicative of effects on the kidneys from four subchronic studies in rats and application of a safety factor of 100 ADI established for the sum of MCPA and its salts and esters, expressed as MCPA acid equivalents
ARfD	0.6 mg/kg bw for MCPA ion, based on the overall NOAEL of 60 mg/kg bw for maternal and developmental toxicity in rats and application of a safety factor of 100 ARfD established for the sum of MCPA and its salts and esters, expressed as MCPA acid equivalents
Limit of detection	0.8 µg/L using HPLC with a photodiode array UV detector; 0.09 µg/l using derivatization and GC with ECD; limit of quantification of 0.0005 µg/l for LC-MS/MS
Treatment performance	Conventional treatment not effective; activated carbon adsorption and/or ozonation and advanced oxidation processes (e.g. UV with hydrogen peroxide) are effective; membrane filtration processes (e.g. reverse osmosis) may be effective
Health-based value derivation	
• allocation to water	20% of upper bound of unrounded ADI (0.12 mg/kg bw)
• weight	60 kg adult
• consumption	2 litres/day
Acute health-based value derivation	
• allocation to water	100% of ARfD
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The default allocation factor of 20% has been used to account for the fact that the available food exposure data, which suggest that exposure via this route is low, do not generally include information from developing countries, where exposure via this route may be higher
	As a general principle, the concentration of pesticides in water, including MCPA, should be kept as low as possible and concentrations should not be allowed to increase up to the health-based value
	Further guidance on interpreting the health-based value and deciding when to monitor can be found in section 8.5.3
Assessment date	2016
Principal references	WHO (2013) <i>Pesticide residues in food – 2012 evaluations</i> WHO (2016) <i>MCPA in drinking-water</i>

* When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary.

** For more information on acute health-based values, see [section 8.7.5](#).

The target organs for the MCPA ion are the kidney, liver and blood. MCPA is not carcinogenic in mice or rats, and the MCPA ion exhibits no genotoxic potential. In multigeneration studies in rats, there was no evidence of reproductive toxicity up to the highest dose tested. The MCPA ion was not teratogenic in rats or rabbits.

Mecoprop

The half-lives for degradation of chlorophenoxy herbicides, including mecoprop (CAS No. 93-65-2; 7085-19-0 racemic mixture), also known as 2(2-methyl-chlorophenoxy) propionic acid or MCP, in the environment are in the order of several days. Chlorophenoxy herbicides are not often found in food.

Guideline value	0.01 mg/l (10 µg/l)
Occurrence	Chlorophenoxy herbicides not frequently found in drinking-water; when detected, concentrations usually no greater than a few micrograms per litre
TDI	3.33 µg/kg body weight, based on a NOAEL of 1 mg/kg body weight for effects on kidney weight in 1- and 2-year studies in rats, with an uncertainty factor of 300 (100 for interspecies and intraspecies variation and 3 for limitations in the database)
Limit of detection	0.01 µg/l by GC-MS; 0.01–0.02 µg/l by GC with ECD
Treatment performance	0.1 µg/l should be achievable using GAC or ozonation
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenoxy herbicides (excluding 2,4-D and MCPA) in drinking-water</i>

Chlorophenoxy herbicides, as a group, have been classified in Group 2B (possibly carcinogenic to humans) by IARC. However, the available data from studies in exposed populations and experimental animals do not permit assessment of the carcinogenic potential to humans of any specific chlorophenoxy herbicide. Therefore, drinking-water guidelines for these compounds are based on a threshold approach for other toxic effects. Effects of dietary administration of mecoprop in short-term and long-term studies include decreased relative kidney weight (rats and dogs), increased relative liver weight (rats), effects on blood parameters (rats and dogs) and depressed body weight gain (dogs).

Mercury

Mercury is used in the electrolytic production of chlorine, in electrical appliances, in dental amalgams and as a raw material for various mercury compounds. Methylation of inorganic mercury has been shown to occur in fresh water and in seawater, although almost all mercury in uncontaminated drinking-water is thought to be in the form of Hg²⁺. Thus, it is unlikely that there is any direct risk of the intake of organic mercury compounds, especially of alkylmercurials, as a result of the ingestion of drinking-water. However, there is a possibility that methylmercury will be converted into inorganic mercury. Food is the main source of mercury in non-occupationally

exposed populations; the mean dietary intake of mercury in various countries ranges from 2 to 20 µg/day per person.

Guideline value	0.006 mg/l (6 µg/l) for inorganic mercury
Occurrence	Mercury is present in the inorganic form in surface water and groundwater at concentrations usually below 0.5 µg/l, although local mineral deposits may produce higher levels in groundwater
TDI	2 µg/kg body weight for inorganic mercury based on a NOAEL of 0.23 mg/kg body weight per day for kidney effects in a 26-week study in rats and applying an uncertainty factor of 100 (for interspecies and intraspecies variation) after adjusting for daily dosing
Limit of detection	0.05 µg/l by cold vapour AAS; 0.6 µg/l by ICP; 5 µg/l by flame AAS
Treatment performance	It should be possible to achieve a concentration below 1 µg/l by treatment of raw waters that are not grossly contaminated with mercury using methods that include coagulation/sedimentation/filtration, PAC and ion exchange.
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	A similar TDI may be obtained by applying an uncertainty factor of 1000 (an additional uncertainty factor of 10 for adjustment from a LOAEL to a NOAEL) to the LOAEL for renal effects of 1.9 mg/kg body weight per day in a 2-year NTP study in rats.
	The current guideline value applies to inorganic mercury, which is the form found in drinking-water, whereas the previous guideline value applied to total (inorganic and organic) mercury.
Assessment date	2004
Principal references	IPCS (2003) <i>Elemental mercury and inorganic mercury compounds</i> WHO (2005) <i>Mercury in drinking-water</i>

The toxic effects of inorganic mercury compounds are seen mainly in the kidney in both humans and laboratory animals following short-term and long-term exposure. In rats, effects include increased absolute and relative kidney weights, tubular necrosis, proteinuria and hypoalbuminaemia. In humans, acute oral poisoning results primarily in haemorrhagic gastritis and colitis; the ultimate damage is to the kidney. The overall weight of evidence is that mercury(II) chloride has the potential to increase the incidence of some benign tumours at sites where tissue damage is apparent and that it possesses weak genotoxic activity but does not cause point mutations.

Methoxychlor

Methoxychlor (CAS No. 72-43-5) is an insecticide used on vegetables, fruit, trees, fodder and farm animals. It is poorly soluble in water and highly immobile in most agricultural soils. Under normal conditions of use, methoxychlor does not seem to be of environmental concern. Daily intake from food and air is expected to be below 1

µg per person. Environmental metabolites are formed preferentially under anaerobic rather than aerobic conditions and include mainly the dechlorinated and demethylated products. There is some potential for the accumulation of the parent compound and its metabolites in surface water sediments.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Detected occasionally in drinking-water, at concentrations as high as 300 µg/l in rural areas
TDI	5 µg/kg body weight, based on a systemic NOAEL of 5 mg/kg body weight in a teratology study in rabbits, with an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 reflecting concern for threshold carcinogenicity and the limited database)
Limit of detection	0.001–0.01 µg/l by GC
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2004) <i>Methoxychlor in drinking-water</i>

The genotoxic potential of methoxychlor appears to be negligible. In 1979, IARC assigned methoxychlor to Group 3. Subsequent data suggest a carcinogenic potential of methoxychlor for liver and testes in mice. This may be due to the hormonal activity of proestrogenic mammalian metabolites of methoxychlor and may therefore have a threshold. The study, however, was inadequate, because only one dose was used and because this dose may have been above the maximum tolerated dose. The database for studies on long-term, short-term and reproductive toxicity is inadequate. A teratology study in rabbits reported a systemic NOAEL of 5 mg/kg body weight per day, which is lower than the LOAELs and NOAELs from other studies. This NOAEL was therefore selected for use in the derivation of a TDI.

Methyl parathion

Methyl parathion (CAS No. 298-00-0) is a non-systemic insecticide and acaricide that is produced throughout the world and has been registered for use on many crops, in particular cotton. It partitions mainly to air and soil in the environment. There is virtually no movement through soil, and neither the parent compound nor its breakdown products will reach groundwater. By far the most important route for the environmental degradation of methyl parathion is microbial degradation. Half-lives of methyl parathion in water are in the order of weeks to months. Concentrations of methyl parathion in natural waters of agricultural areas in the USA ranged up to 0.46 µg/l, with highest levels in summer. The general population can come into contact with methyl parathion via air, water or food.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1996) <i>Pesticide residues in food—1995 evaluations</i> . IPCS (1992) <i>Methyl parathion</i> WHO (2004) <i>Methyl parathion in drinking-water</i>

A NOAEL of 0.3 mg/kg body weight per day was derived from the combined results of several studies conducted in humans, based on the depression of erythrocyte and plasma cholinesterase activities. Methyl parathion decreased cholinesterase activities in long-term studies in mice and rats, but did not induce carcinogenic effects. Methyl parathion was mutagenic in bacteria, but there was no evidence of genotoxicity in a limited range of studies in mammalian systems.

A health-based value of 9 µg/l can be calculated for methyl parathion on the basis of an ADI of 0–0.003 mg/kg body weight, based on a NOAEL of 0.25 mg/kg body weight per day in a 2-year study in rats for retinal degeneration, sciatic nerve demyelination, reduced body weight, anaemia and decreased brain acetylcholinesterase activity, using an uncertainty factor of 100 for interspecies and intraspecies variation. As the toxicological end-points seen in experimental animals were other than acetylcholinesterase inhibition, it was considered more appropriate to use these data rather than the NOAEL derived for cholinesterase inhibition in humans.

Intake of methyl parathion from all sources is generally low and well below the upper limit of the ADI. As the health-based value is much higher than concentrations of methyl parathion likely to be found in drinking-water, the presence of methyl parathion in drinking-water under usual conditions is unlikely to represent a hazard to human health. For this reason, the establishment of a formal guideline value for methyl parathion is not deemed necessary.

Methyl tertiary-butyl ether

The major use of methyl *tert*-butyl ether, or MTBE, is as a gasoline additive. Surface water can be contaminated by gasoline spills; however, owing to the high volatility of MTBE, most is lost to evaporation. Spills and leaking storage tanks can cause more serious problems in groundwater, where MTBE is more persistent. MTBE has been detected in groundwater and drinking-water at concentrations in the nanogram to microgram per litre range.

Reason for not establishing a guideline value	Any guideline that would be derived would be significantly higher than concentrations at which MTBE would be detected by odour
Assessment date	2004
Principal references	IPCS (1998) <i>Methyl tertiary-butyl ether</i> WHO (2005) <i>Methyl tertiary-butyl ether (MTBE) in drinking-water</i>

No human cancer studies have been published for either the general population or occupationally exposed cohorts. There have been a number of human studies of

neurological and clinical effects of exposure to MTBE by inhalation, with mixed results. In general, no objective changes could be seen at levels of MTBE normally found, even in such microenvironments as gasoline filling stations.

The weight of evidence suggests that MTBE is not genotoxic. A large number of studies using in vitro and in vivo mammalian and non-mammalian systems have been conducted to assess the mutagenicity of MTBE, almost all of which have produced negative results. These results suggest that the mechanism of action of MTBE is more likely to be non-genotoxic than genotoxic, although no one mechanism appears to explain all of the observed effects.

It has been concluded that MTBE should be considered a rodent carcinogen but that it is not genotoxic, and the carcinogenic response is evident only at high levels of exposure that also induce other adverse effects. The available data are therefore considered inconclusive and prohibit their use for human carcinogenic risk assessment. A health-based guideline value has not been derived for MTBE, owing to the fact that any guideline value that would be derived would be significantly higher than the concentration at which it would be detected by odour (15 µg/l is the lowest level eliciting a response in a study using taste- and odour-sensitive participants).

Metolachlor

Metolachlor (CAS No. 51218-45-2) is a selective pre-emergence herbicide used on a number of crops. It can be lost from the soil through biodegradation, photodegradation and volatilization. It is fairly mobile and under certain conditions can contaminate groundwater, but it is mostly found in surface water.

Guideline value	0.01 mg/l (10 µg/l)
Occurrence	Detected in surface water and groundwater at concentrations that can exceed 10 µg/l
TDI	3.5 µg/kg body weight, based on a NOAEL of 3.5 mg/kg body weight for an apparent decrease in kidney weight at the two highest dose levels in a 1-year dog study, with an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 reflecting some concern regarding carcinogenicity)
Limit of detection	0.75–0.01 µg/l by GC with nitrogen–phosphorus detection
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Metolachlor in drinking-water</i>

In a 1-year study in dogs, administration of metolachlor resulted in decreased kidney weight at the two highest dose levels. In 2-year studies with rodents fed metolachlor in the diet, the only toxicological effects observed in mice were decreased body

weight gain and decreased survival in females at the highest dose level, whereas rats showed decreased body weight gain and food consumption at the highest dose level. There is no evidence from available studies that metolachlor is carcinogenic in mice. In rats, an increase in liver tumours in females as well as a few nasal tumours in males have been observed. Metolachlor is not genotoxic.

Microcystins (cyanobacterial toxins)¹

Microcystins (MCs) are naturally occurring heptapeptides produced by strains of various species of cyanobacteria, primarily in freshwater environments. They occur widely around the globe, particularly with strains of the frequently occurring cyanobacterial genera *Microcystis*, *Planktothrix* and *Dolichospermum* (see also [section 11.5](#)). Among the more than 250 MCs identified to date, only a few occur frequently and in high concentrations. MCs are usually cell-bound, and substantial amounts are released to the surrounding water only during cell rupture (i.e. lysis). One of the most common MCs, and the one most studied, is microcystin-LR (MC-LR).

Drinking-water is the most likely route of exposure to MCs where surface water with cyanobacterial blooms is the drinking-water source. Recreational activities in lakes with cyanobacterial blooms may also be a relevant exposure pathway, potentially to high, usually intermittent concentrations (see WHO *Guidelines on recreational water quality*, 2021). Limited data suggest that MCs may also accumulate in some food items.

Provisional guideline value (lifetime)	<i>Total MCs (sum of all congeners, free plus cell-bound):</i> 0.001 mg/l (1 µg/l) The guideline value is provisional because of the high level of uncertainty—it is based on data for only MC-LR, and the database is limited, as reflected in the composite uncertainty factor of 1000
Provisional short-term guideline value	<i>Total MCs (sum of all congeners, free plus cell-bound):</i> 0.012 mg/l (12 µg/l)
Occurrence	Although concentrations in scums can reach the range of mg/l (even up to 100 mg/L), outside of scum areas they rarely exceed several tens of µg/l. MCs largely occur cell-bound unless cell damage causes release
TDI	0.04 µg/kg bw, based on a NOAEL of 40 µg/kg bw per day for liver pathology observed in a 13-week study in mice and applying an uncertainty factor of 1000 (10 each for inter- and intra-species variability and 10 for database deficiencies,* taking into consideration limitations in the database—in particular, limited data on chronic toxicity, reproductive toxicity and carcinogenicity)
Limit of detection	<1 µg/l by LC-MS/MS, or LC (including HPLC) followed by UV/PDA detection. LC-MS/MS has the highest specificity and sensitivity but requires quantitative reference standards for all relevant MCs in the sample. For UV/PDA detection, standards for a few MCs whose signals are representative of all are sufficient. Prior extraction of cells with freeze–thaw cycles and 75% aqueous methanol is necessary for cell-bound MCs; neglecting extraction from cells will lead to dramatic underestimation of concentrations.

¹ As cyanobacteria and their toxins, in particular MCs, are a concern in many areas and considering the complexities in their management, its chemical fact sheet has been expanded

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<1 µg/l by commercially available immunoassay (ELISA) or enzyme assay (PPA) kits; although these are less precise than LC with the above-mentioned detection methods, they capture all MCs and thus are useful for most monitoring purposes.

Monitoring	The likelihood of blooms can be assessed by understanding water body conditions (in particular, nutrient concentrations, water body depth, water retention time, patterns of mixing and stratification; see section 11.5). Where conditions render blooms likely, visual monitoring of source water (including microscopy for potentially MC-containing genera) for evidence of increasing cyanobacterial biomass (blooms) is important because biomass can increase rapidly. Exceeding alert values for biomass indicators or MC concentrations should trigger management responses to prevent exposure to elevated toxin concentrations (see the alert level framework in section 11.5). Analysis of cyanotoxins is particularly useful for validating and optimizing the efficacy of control measures such as riverbank filtration or treatment.
Prevention and treatment	Actions to decrease the probability of bloom occurrence include catchment and source water management, such as reducing nutrient loading or changing reservoir stratification and mixing. Filtration is effective for removing intact cyanobacterial cells, and in most cases this removes the major fraction of MCs. For dissolved MCs, oxidation with chlorine or ozone at sufficient concentrations and contact times, as well as GAC and some PAC applications, are effective (see chapters 7–10 of <i>Toxic cyanobacteria in water</i> ; Annex 1).
Guideline value derivation	• allocation to water 80% of TDI; for short-term exposure, 100% of TDI • weight 60 kg adult • consumption 2 litres/day
Additional comments	Total MCs as gravimetric or molar equivalents should be evaluated against the guideline values since MCs usually occur as mixtures. The guideline values are based on MC-LR, which is one of the most toxic and common MCs. The provisional short-term drinking-water guideline value is intended to indicate the extent to which the lifetime value can be exceeded for periods of up to about 2 weeks until water treatment can be augmented to bring the concentration of MCs back under control. It is not intended to allow repeated seasonal exceedances of the lifetime value. It is recommended, as a precautionary measure, that bottle-fed infants and small children be provided with an alternative safe drinking-water source (e.g. bottled water that is certified by the responsible authorities) if concentrations are greater than 3 µg/L, even for short periods.
Assessment date	2020
Principal references	WHO (2020) <i>Cyanobacterial toxins: microcystins</i> Chorus & Welker (2021) <i>Toxic cyanobacteria in water</i>

* For the short-term guideline value, a database uncertainty factor was not applied because chronic toxicity, carcinogenicity (tumour promotion) and the limited evidence for reproductive toxicity following subchronic dosing are not relevant to short-term exposures.

MCs are actively transported into cells by specific organic anion transport proteins (OATPs). Because the liver has high expression of OATPs, MCs are considered to be primarily hepatotoxins, although distribution to other organs and tissues expressing OATPs does occur. Once in the cell, MCs bind with high affinity to certain protein phosphatases that are involved in a wide range of regulatory pathways, including those responsible for cytoskeletal structure, cell replication, stress responses and DNA repair. Acute effects of MC poisoning include intrahepatic haemorrhage. Chronic effects include a reduced efficacy of proliferation control mechanisms within tumour cells. Cases of liver damage have been reported from MCs in drinking-water that was inadequately treated. As well, human fatalities clearly attributable to MCs have occurred through renal dialysis using inadequately treated water, including drinking-water that contained high MC levels. In 2006, IARC classified MC-LR as a possible human carcinogen (Group 2B) based on evidence for tumour promotion.

The provisional guideline values for MCs are based on a study in mice using MC-LR (the only congener with sufficiently comprehensive oral toxicity data for guideline value derivation), corroborated by a similar study in rats. The liver was identified as the most sensitive organ in these studies. Although some recent research has suggested that certain reproductive organs may be affected by prolonged exposure to lower concentrations, these reports suffer from a number of methodological and reporting deficiencies and, further, are contradicted by earlier studies. Therefore, these findings of adverse effects require corroboration.

Practical considerations

Where nutrient (phosphorus and nitrogen) concentrations are elevated in lakes, reservoirs or slowly flowing rivers, cyanobacteria occur widely. Where their excessive growth leads to high biomass, sometimes termed “bloom” events, MCs can reach concentrations in raw water that are potentially hazardous to human health. Such blooms tend to recur in the same water bodies. Cells of some cyanobacterial species may accumulate at the surface as scums (particularly of *Microcystis*) or at the thermocline of thermally stratified reservoirs (i.e. *Planktothrix rubescens*). Such accumulations may develop rapidly and may be of very variable duration (hours to weeks). In many circumstances, blooms and accumulations are seasonal, whereas others occur perennially.

Cyanobacteria are most effectively controlled in the context of developing a WSP (see [chapter 4](#)). Control measures to manage potential risks from cyanobacteria, and in particular from their toxins, in drinking-water should include not only adequate treatment, but also measures to control cyanobacterial bloom development. See [section 11.5](#) for more information on cyanobacteria, including further details on monitoring cyanobacterial blooms, the alert level framework, and prevention and management of cyanobacteria in source waters. Effectively minimizing the formation of blooms and locating the raw water intake away from blooms reduce the treatment steps required to remove cyanotoxins.

Drinking-water treatment that removes particles—that is, soil, slow sand or river-bank filtration, conventional water treatment (coagulation, flocculation and filtration or dissolved air flotation)—can remove cell-bound MCs effectively. Soil, slow sand and

riverbank filtration can also remove dissolved cyanotoxins. For all these processes, it is important that they are optimized to target the removal of cells and dissolved toxins. Chlorination and ozonation at sufficiently high doses and contact times are effective for degrading dissolved MCs; however, elevated organic carbon in bloom situations will substantially increase the disinfectant demand. Chlorine dioxide and chloramine are ineffective for degrading MCs. Both for pre-oxidation and conventional treatment, cell rupture and toxin release should be avoided. GAC and PAC can be effective for removing dissolved MCs, with efficacy dependent on several factors, including the type of activated carbon, contact times (PAC), flow rates (GAC) and water quality. As the challenges that blooms present for treatment are complex, periodic validation of efficacy during bloom situations and under the specific local conditions is particularly important. Avoiding bloom occurrence and intake is therefore the preferred option.

Molinate

Molinate (CAS No. 2212-67-1) is a herbicide used to control broad-leaved and grassy weeds in rice. The available data suggest that groundwater pollution by molinate is restricted to some rice growing regions. Data on the occurrence of molinate in the environment are limited. Molinate is of low persistence in water and soil, with a half-life of about 5 days.

Guideline value	0.006 mg/l (6 µg/l)
Occurrence	Concentrations in water rarely exceed 1 µg/l
TDI	2 µg/kg body weight, based on a NOAEL for reproductive toxicity in the rat of 0.2 mg/kg body weight, with an uncertainty factor of 100 (for inter-species and intraspecies variation)
Limit of detection	0.01 µg/l by GC-MS
Treatment performance	0.001 mg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Molinate in drinking-water</i>

On the basis of the limited information available, molinate does not seem to be carcinogenic or mutagenic in experimental animals. Evidence suggests that impairment of the reproductive performance of the male rat represents the most sensitive indicator of molinate exposure. However, epidemiological data based on the examination of workers involved in molinate production do not indicate any effect on human fertility.

Molybdenum

Molybdenum is found naturally in soil and is used in the manufacture of special steels and in the production of tungsten and pigments, and molybdenum compounds are used as lubricant additives and in agriculture to prevent molybdenum deficiency in

crops. Concentrations in drinking-water are usually less than 0.01 mg/l, although concentrations as high as 200 µg/l have been reported in areas near mining sites.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	1993, revised in 2011
Principal references	WHO (2011) <i>Molybdenum in drinking-water</i>

Molybdenum is considered to be an essential element, with an estimated daily requirement of 0.1–0.3 mg for adults.

As molybdenum occurs at very low concentrations in drinking-water, it is not considered necessary to set a formal guideline value. For guidance purposes, a health-based value can be derived.

In a 2-year study of humans exposed via drinking-water, the NOAEL was found to be 0.2 mg/l, but there are some concerns about the quality of this study. As molybdenum is an essential element, a factor of 3 is considered to be adequate to reflect intraspecies variation. This gives a health-based value of 0.07 mg/l (rounded figure), which is in the same range as that derived on the basis of the results of toxicological studies in experimental animals and is consistent with the essential daily requirement for molybdenum.

Monochloroacetic acid

Chlorinated acetic acids are formed from organic material during water chlorination.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Present in surface water–derived drinking-water at concentrations up to 82 µg/l (mean 2.1 µg/l)
TDI	3.5 µg/kg body weight, based on a LOAEL of 3.5 mg/kg body weight per day from a study in which increased absolute and relative spleen weights were observed in male rats exposed to monochloroacetic acid in drinking-water for 2 years, using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for use of a minimal LOAEL instead of a NOAEL and database deficiencies, including the lack of a multigeneration reproductive toxicity study)
Limit of detection	2 µg/l by GC with ECD; 5 µg/l by GC-MS
Treatment performance	No information available
Guideline value derivation	
• allocation to water	20% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	2003
Principal reference	WHO (2004) <i>Monochloroacetic acid in drinking-water</i>

No evidence of carcinogenicity of monochloroacetate was found in 2-year gavage bioassays with rats and mice. Monochloroacetate has given mixed results in a limited

number of mutagenicity assays and has been negative for clastogenicity in genotoxicity studies. IARC has not classified the carcinogenicity of monochloroacetic acid.

Monochlorobenzene

Releases of monochlorobenzene (MCB) to the environment are thought to be mainly due to volatilization losses associated with its use as a solvent in pesticide formulations, as a degreasing agent and from other industrial applications. MCB has been detected in surface water, groundwater and drinking-water; mean concentrations were less than 1 µg/l in some potable water sources (maximum 5 µg/l) in Canada. The major source of human exposure is probably air.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern, and health-based value would far exceed lowest reported taste and odour threshold
Assessment date	2003
Principal reference	WHO (2004) <i>Monochlorobenzene in drinking-water</i>

MCB is of low acute toxicity. Oral exposure to high doses of MCB results in effects mainly on the liver, kidneys and haematopoietic system. There is limited evidence of carcinogenicity in male rats, with high doses increasing the occurrence of neoplastic nodules in the liver. The majority of evidence suggests that MCB is not mutagenic; although it binds to DNA *in vivo*, the level of binding is low.

A health-based value of 300 µg/l can be calculated for MCB on the basis of a TDI of 85.7 µg/kg body weight, based on neoplastic nodules identified in a 2-year rat study with dosing by gavage, and taking into consideration the limited evidence of carcinogenicity. However, because MCB occurs at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value. It should also be noted that the health-based value far exceeds the lowest reported taste and odour threshold for MCB in water.

MX

MX, which is the common name for 3-chloro-4-dichloromethyl-5-hydroxy-2-(5H)-furanone, is formed by the reaction of chlorine with complex organic matter in drinking-water. It has been identified in chlorinated humic acid solutions and drinking-water in Finland, the United Kingdom and the USA and was found to be present in 37 water sources at levels of 2–67 ng/l. Five drinking-water samples from different Japanese cities contained MX at concentrations ranging from less than 3 to 9 ng/l.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> WHO (2003) <i>MX in drinking-water</i>

MX is a potent mutagen in bacteria and in cells in vitro and has undergone a lifetime study in rats in which some tumorigenic responses were observed. These data indicate that MX induces thyroid and bile duct tumours. IARC has classified MX in Group 2B (possibly carcinogenic to humans) on the basis of rat tumorigenicity and its strong mutagenicity.

A health-based value of 1.8 µg/l can be calculated for MX on the basis of the increase in cholangiomas and cholangiocarcinomas in female rats using the linearized multistage model (without a body surface area correction). However, this is significantly above the concentrations that would be found in drinking-water, and, in view of the analytical difficulties in measuring this compound at such low concentrations, it is considered unnecessary to propose a formal guideline value for MX in drinking-water.

Nickel

Nickel is a naturally occurring element, which is used mainly in the production of stainless steel and nickel alloys. Food is the dominant source of nickel exposure in the non-smoking, non-occupationally exposed population; water is generally a minor contributor to the total daily oral intake. However, the nickel contribution from water may be significant where there is heavy pollution, where nickel that occurs naturally in groundwater is mobilized, or where nickel leaches from nickel- or chromium-plated taps or stainless steel devices or materials that are in contact with water. The primary source of nickel in drinking-water is leaching from metals that are in contact with drinking-water.

Guideline value	0.07 mg/l (70 µg/l)
Occurrence	Concentration in drinking-water are typically less than 25 µg/l, although concentrations may be elevated (up to 5 mg/l) where nickel is released from metal alloys that are in contact with drinking-water, such as fittings, including taps, or as a result of anthropogenic contamination or mobilization from natural deposits
Basis of guideline value derivation	The guideline value is based on achievability by available source control measures and treatment technologies, measurability by analytical methods and toxicology. Reproductive toxicity (increased incidence of rat litters with post-implantation loss) was considered the most sensitive end-point and is therefore the basis for the TDI derivation of 13 µg/kg body weight. An uncertainty factor of 100 was applied to account for interspecies differences (10) and intraspecies variation (10) to the BDML10 of 1.3 mg/kg bw per day, from a two-generation study in rats. Although the 2021 risk assessment supports a health-based value of 80 µg/l based on this TDI, the guideline value was retained at 70 µg/l considering the above factors. Further, 80 µg/l is only slightly higher than the previous guideline value of 70 µg/l (established in 2014); factoring in the imprecision inherent in risk assessment procedures, this difference is not judged significant enough to warrant a revised, minimally relaxed guideline value.
Limit of detection	0.5–5 µg/l for ICP-MS, ICP-AES and graphite furnace AAS; 0.1 mg/l for flame AAS

Prevention and treatment	For surface water, conventional water treatment (coagulation, sedimentation, filtration) may be effective under certain circumstances, depending on a number of factors, including the coagulant dosage and pH. For groundwater, ion-exchange resins have been shown to be effective. The most important means of control is by product specifications through an appropriate certification scheme for materials in contact with drinking-water. Consumers, particularly nickel-sensitive people, should flush chromium- or nickel-plated taps before using the water, particularly after periods of stagnation.
Health-based value derivation	
• allocation to water	20% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value is also protective of any possible acute effects, including systemic contact dermatitis As nickel is usually found in drinking-water below the guideline value, monitoring and inclusion in drinking-water regulations and standards would usually only be necessary if there were indications that a specific pollution or problem might exist
Assessment date	2021
Principal reference	WHO (2021) <i>Nickel in drinking-water</i> EFSA (2020) <i>Update of the risk assessment of nickel in food</i>

IARC concluded that inhaled nickel compounds are carcinogenic to humans (Group 1) and that metallic nickel is possibly carcinogenic (Group 2B). However, there is a lack of evidence of a carcinogenic risk from oral exposure to nickel. In a well-conducted two-generation reproductive study in rats administered nickel by gavage, an increase in the frequency of litters with post-implantation loss was reported; thus, reproductive and developmental toxicity was identified as the relevant and sensitive end-point for derivation of the health-based value.

Exposure to nickel through the skin or by inhalation may lead to nickel sensitization. Whereas oral ingestion of nickel is not known to lead to systemic contact dermatitis (SCD) in the general population, it can elicit SCD in nickel-sensitized individuals. However, the guideline value of 70 µg/L is considered adequately protective of SCD that may result from drinking-water exposure based on a margin of exposure (MOE) assessment. Acute consumption of water containing nickel at the chronic health-based value of 80 µg/l would result in an MOE of approximately 16. The MOE of 16 was calculated from an acute LOAEL for SCD of 4.3 µg/kg bw and an acute exposure of 0.27 µg/kg bw (based on a scenario of a person weighing 60 kg, drinking a glass of tap water (about 200 ml) containing nickel at the health-based value of 80 µg/l). The guideline value is further considered adequately protective considering that SCD elicitation was associated with a bolus exposure, in contrast to the intermittent nature of a drinking-water exposure scenario.

Nitrate and nitrite¹

Nitrate (NO_3^-) is found naturally in the environment and is an important plant nutrient. It is present at varying concentrations in all plants and is a part of the nitrogen cycle. Nitrite (NO_2^-) is not usually present in significant concentrations except in a reducing environment, because nitrate is the more stable oxidation state. It can be formed by the microbial reduction of nitrate and in vivo by reduction from ingested nitrate. Nitrite can also be formed chemically in distribution pipes by *Nitrosomonas* bacteria during stagnation of nitrate-containing and oxygen-poor drinking-water in galvanized steel pipes, or if chloramination is used to provide a residual disinfectant. An excess of free ammonia entering the distribution system can lead to nitrification and the potential increase of nitrate and nitrite in drinking-water. Nitrate can reach both surface water and groundwater as a consequence of agricultural activity (including excess application of inorganic nitrogenous fertilizers and manures), from wastewater disposal and from oxidation of nitrogenous waste products in human and other animal excreta, including septic tanks. Nitrate can also occasionally reach groundwater as a consequence of natural vegetation. Surface water nitrate concentrations can change rapidly owing to surface runoff of fertilizer, uptake by phytoplankton and denitrification by bacteria, but groundwater concentrations generally show relatively slow changes. Nitrate and nitrite can also be produced as a result of nitrification in source water or distribution systems.

In general, the most important source of human exposure to nitrate and nitrite is through vegetables (nitrate and nitrite) and through meat in the diet (nitrite is used as a preservative in many cured meats). In some circumstances, however, drinking-water can make a significant contribution to nitrate and, occasionally, nitrite intake. In the case of bottle-fed infants, drinking-water can be the major external source of exposure to nitrate and nitrite.

Guideline values ²	<i>Nitrate</i>: 50 mg/l as nitrate ion, to be protective against methaemoglobinemia and thyroid effects in the most sensitive subpopulation, bottle-fed infants, and, consequently, other population subgroups
	<i>Nitrite</i>: 3 mg/l as nitrite ion, to be protective against methaemoglobinaemia induced by nitrite from both endogenous and exogenous sources in bottle-fed infants, the most sensitive subpopulation, and, consequently, the general population
	<i>Combined nitrate plus nitrite</i>: The sum of the ratios of the concentrations of each of nitrate and nitrite to its guideline value should not exceed 1
Occurrence	Nitrate levels vary significantly, but levels in well water are often higher than those in surface water and, unless heavily influenced by surface water, are less likely to fluctuate. Concentrations often approach or exceed 50 mg/l where there are significant sources of contamination. Nitrite levels are normally lower, less than a few milligrams per litre.

¹ As nitrate and nitrite are chemicals of significant concern in some natural waters, the chemical fact sheet on nitrate and nitrite has been expanded.

² Conversion factors: 1 mg/l as nitrate = 0.226 mg/l as nitrate-nitrogen; 1 mg/l as nitrite = 0.304 mg/l as nitrite-nitrogen.

12. CHEMICAL FACT SHEETS

Basis of guideline value derivation	<i>Nitrate (bottle-fed infants):</i> In epidemiological studies, no adverse health effects (methaemoglobinaemia or thyroid effects) were reported in infants in areas where drinking-water consistently contained nitrate at concentrations below 50 mg/l <i>Nitrite (bottle-fed infants):</i> Based on: 1) no incidence of methaemoglobinaemia at nitrate concentrations below 50 mg/l (as nitrate ion) in drinking-water for bottle-fed infants less than 6 months of age (assuming body weight of 2 kg); 2) converting 50 mg/l as nitrate to corresponding molar concentration for nitrite; 3) multiplying by a factor of 0.1 to account for the estimated conversion rate of nitrate to nitrite in infants where nitrite is formed endogenously from nitrate at a rate of 5–10%; and 4) multiplying by a source allocation factor for drinking-water of 100% or 1, as a bottle-fed infant's primary exposure to nitrite is through consumption of formula reconstituted with drinking-water that contains nitrate or nitrite. As the guideline value is based on the most sensitive subgroup of the population (bottle-fed infants less than 6 months of age), application of an uncertainty factor is not deemed necessary. <i>Combined nitrate plus nitrite:</i> To account for the possibility of the simultaneous occurrence of nitrate and nitrite in drinking-water
Limit of detection	MDLs of 0.009 mg/l as nitrate ion and 0.013 mg/l as nitrite ion by IC; MDL of 0.04–4.4 mg/l as nitrate ion by automated cadmium reduction with colorimetry (recommended for the analysis of nitrate at concentrations below 0.4 mg/l)
Treatment performance	<i>Nitrate:</i> Effective central treatment technologies involve the physical/chemical and biological removal of nitrate and include ion exchange, reverse osmosis, biological denitrification and electrodialysis, which are capable of removing over 80% of nitrate from water to achieve effluent nitrate concentrations as low as 13 mg/l; conventional treatment processes (coagulation, sedimentation, filtration and chlorination) are not effective <i>Nitrite:</i> Treatment usually focuses on nitrate, because nitrite is readily converted to nitrate by many disinfectants
Additional comments	The guideline values for both nitrate and nitrite are based on short-term effects; however, they are also considered protective for any possible long-term effects . Methaemoglobinaemia is complicated by the presence of microbial contamination and subsequent gastrointestinal infection, which can increase the risk for bottle-fed infants significantly. Authorities should therefore be all the more vigilant that water to be used for bottle-fed infants is microbiologically safe when nitrate is present at concentrations near or above the guideline value. It is particularly important to ensure that these infants are not currently exhibiting symptoms of gastrointestinal infection (diarrhoea). Also, as excessive boiling of water to ensure microbiological safety can concentrate levels of nitrate in the water, care should be taken to ensure that water is heated only until it reaches a rolling boil. In extreme situations, alternative sources of water (e.g. bottled water) can be used. Nitrite is relatively unstable and can be rapidly oxidized to nitrate. Nitrite can occur in the distribution system at higher concentrations when chloramination is used, but the occurrence is almost invariably intermittent. Methaemoglobinaemia is therefore the most important consideration, and the guideline value derived for protection against methaemoglobinaemia would be the most appropriate under these circumstances, allowing for any nitrate that may also be present.

All water systems that practise chloramination should closely and regularly monitor their systems to verify disinfectant levels, microbiological quality and nitrite levels. If nitrification is detected (e.g. reduced disinfectant residuals and increased nitrite levels), steps can be taken to modify the treatment train or water chemistry in order to minimize nitrite formation. Effective disinfection must never be compromised. Excessively high levels may occur in small supplies; where this is suspected from the risk assessment, testing may be appropriate.

Assessment date	2016
Principal references	Health Canada (2013) <i>Guidelines for Canadian drinking water quality: guideline technical document – nitrate and nitrite</i> WHO (2016) <i>Nitrate and nitrite in drinking-water</i>

Absorption of nitrate ingested from vegetables, meat or water is rapid and in excess of 90%; final excretion is in the urine. In humans, about 25% of ingested nitrate is recirculated in saliva, of which about 20% is converted to nitrite by the action of bacteria in the mouth. There is also endogenous formation of nitrate from nitric oxide and protein breakdown as part of normal metabolism. In normal healthy adults, this endogenous synthesis leads to the excretion of about 62 mg of nitrate ion per day in the urine. Endogenous formation of nitrate or nitrite can be significantly increased in the presence of infections, particularly gastrointestinal infections. When nitrate intake is low, endogenous formation may be the major source of nitrate in the body. Nitrate metabolism is different in humans and rats, as rats may not actively secrete nitrate in their saliva.

Nitrate probably has a role in protecting the gastrointestinal tract against a variety of gastrointestinal pathogens, as nitrous oxide and acidified nitrite have antibacterial properties. It may have other beneficial physiological roles. Hence, there may be a benefit from exogenous nitrate uptake, and there remains a need to balance the potential risks with the potential benefits.

Significant bacterial reduction of nitrate to nitrite does not normally take place in the stomach, except in individuals with low gastric acidity or with gastrointestinal infections. These may include individuals using antacids, particularly those that block acid secretion. In humans, methaemoglobinaemia is a consequence of the reaction of nitrite with haemoglobin in the red blood cells to form methaemoglobin, which binds oxygen tightly and does not release it, thus blocking oxygen transport. Although most absorbed nitrite is oxidized to nitrate in the blood, residual nitrite can react with haemoglobin. High levels of methaemoglobin (>10%) formation in infants can give rise to cyanosis, referred to as blue-baby syndrome. Although clinically significant methaemoglobinaemia can occur as a result of extremely high nitrate intake in adults and children, the most familiar situation is its occurrence in bottle-fed infants. This was considered to be primarily a consequence of high levels of nitrate in water, although there have been cases of methaemoglobinaemia in weaned infants, associated with high nitrate intake from vegetables. Bottle-fed infants are considered to be at greater risk because the intake of water in relation to body weight is high and, in infants, the development of repair enzymes is limited. In clinical epidemiological studies of

methaemoglobinaemia and subclinical increases in methaemoglobin levels associated with drinking-water nitrate, 97% of cases occurred at concentrations in excess of 44.3 mg/l, with clinical symptoms associated with the higher concentrations. The affected individuals were almost exclusively under 3 months of age.

Although drinking-water nitrate may be an important risk factor for methaemoglobinaemia in bottle-fed infants, there is compelling evidence that the risk of methaemoglobinaemia is primarily increased in the presence of simultaneous gastrointestinal infections, which increase endogenous nitrite formation, may increase reduction of nitrate to nitrite and may also increase the intake of water in combating dehydration. Cases have been described in which gastrointestinal infection seems to have been the primary cause of methaemoglobinaemia. Most cases of methaemoglobinaemia reported in the literature are associated with contaminated private wells (predominantly when the drinking-water is anaerobic) that also have a high probability of microbial contamination, which should not occur if it is properly disinfected.

Although numerous epidemiological studies have investigated the relationship between exposure to nitrate or nitrite in drinking-water and cancer occurrence, the weight of evidence does not support an association between cancer and exposure to nitrate or nitrite per se. Nitrite can react with nitrosatable compounds, primarily secondary amines, in the body to form *N*-nitroso compounds. A number of these are considered to be carcinogenic to humans, whereas others, such as *N*-nitrosoproline, are not. Several studies have been carried out on the formation of *N*-nitroso compounds in relation to nitrate intake in humans, but there is large variation in the intake of nitrosatable compounds and in gastric physiology. Higher mean levels of *N*-nitroso compounds, along with high nitrate levels, have been found in the gastric juice of individuals who are achlorhydric (i.e. have very low levels of hydrochloric acid in the stomach). However, other studies have been largely inconclusive, and there appears to be no clear relationship with drinking-water nitrate compared with overall nitrate intake in relation to formation of *N*-nitroso compounds. Moderate consumption of a number of dietary antioxidant components, such as ascorbic acid and green tea, appears to reduce endogenous *N*-nitrosamine formation.

A significant number of epidemiological studies have been carried out on the association of nitrate intake with primarily gastric cancers. Although the epidemiological data are considered to be inadequate to allow definitive conclusions to be drawn regarding all cancers, there is no convincing evidence of a causal association with any cancer site. The weight of evidence indicates that there is unlikely to be a causal association between gastric cancer and nitrate in drinking-water. This is consistent with the conclusion by IARC that ingested nitrate or nitrite under conditions that result in endogenous nitrosation is probably carcinogenic to humans (Group 2A), but not nitrate alone.

There have been suggestions that nitrate in drinking-water could be associated with congenital malformations, but the overall weight of evidence does not support this.

Nitrate appears to competitively inhibit iodine uptake, with the potential for an adverse effect on the thyroid. Current evidence also suggests that exposure to nitrate in drinking-water may alter human thyroid gland function by competitively inhibiting

thyroidal iodide uptake, leading to altered thyroid hormone concentrations and functions. Although studies found that exposure to nitrate concentrations above 50 mg/l are weakly associated with altered thyroid function, the evidence is limited, conflicting and based on studies with important methodological limitations. Mode of action data suggest that pregnant women and infants are the most sensitive populations, owing primarily to the importance of adequate thyroid hormones for normal neurodevelopment in the fetus and infant, but also to increased thyroid hormone turnover and low intrathyroidal stores in fetal and early life.

There have been suggestions of an association between nitrate in drinking-water and the incidence of childhood diabetes mellitus. However, subsequent studies have not found a significant relationship, and no mechanism has been identified.

In some studies on rats treated with high doses of nitrite, a dose-related hypertrophy of the zona glomerulosa of the adrenal was seen; one strain of rats appeared to be more sensitive than others. However, this minimal hyperplasia was considered to be due to physiological adaptation to small fluctuations in blood pressure in response to high nitrite doses.

Nitrate is not carcinogenic in laboratory animals. Nitrite has been frequently studied, and there have been suggestions of carcinogenic activity, but only at very high doses. The most recent long-term studies have shown only equivocal evidence of carcinogenicity in the forestomach of female mice, but not in rats or male mice. In view of the lack of evidence for genotoxicity, this led to the conclusion that sodium nitrite was not carcinogenic in mice and rats. In addition, as humans do not possess a forestomach and the doses were high, the significance of these data for humans is very doubtful.

The guideline value for nitrate of 50 mg/l, as nitrate ion, is based on an absence of health effects (methaemoglobinaemia and thyroid effects) in epidemiological studies and is protective for bottle-fed infants and, consequently, other parts of the population. Methaemoglobinaemia is complicated by the presence of microbial contamination and subsequent gastrointestinal infection, which can increase the risk for this group significantly. Authorities should therefore be all the more vigilant that water to be used for bottle-fed infants is microbiologically safe when nitrate is present at concentrations near the guideline value. It is particularly important to ensure that these infants are not currently exhibiting symptoms of significant gastrointestinal infection (diarrhoea). Also, as excessive boiling of water to ensure microbiological safety can concentrate levels of nitrate in the water, care should be taken to ensure that water is heated only until it reaches a rolling boil. In extreme situations, alternative sources of water (e.g. bottled water) can be used.

The guideline for nitrite of 3 mg/l, as nitrite ion, is based on: 1) no incidence of methaemoglobinaemia at nitrate concentrations below 50 mg/l in drinking-water for bottle-fed infants less than 6 months of age (assuming body weight of 2 kg), 2) converting 50 mg/l nitrate to the corresponding molar concentration for nitrite, 3) multiplying by a factor of 0.1 to account for the estimated conversion rate of nitrate to nitrite in infants where nitrite is formed endogenously from nitrate at a rate of 5–10% and 4) multiplying by a source allocation factor for drinking-water of 100% or 1, as a bottle-fed infant's primary exposure to nitrite is through consumption of for-

mula reconstituted with nitrate- or nitrite-containing drinking-water. As the health-based value is based on the most sensitive subgroup of the population (bottle-fed infants less than 6 months of age), application of an uncertainty factor is not deemed necessary.

Because of the possibility of the simultaneous occurrence of nitrate and nitrite in drinking-water, the sum of the ratios of the concentration (C) of each to its guideline value (GV) should not exceed 1:

$$\frac{C_{\text{nitrate}}}{GV_{\text{nitrate}}} + \frac{C_{\text{nitrite}}}{GV_{\text{nitrite}}} \leq 1$$

The guideline values are based on short-term effects; however, they are also considered protective for long-term effects.

Practical considerations

The most appropriate means of controlling nitrate concentrations, particularly in groundwater, is the prevention of contamination. This may take the form of appropriate management of agricultural practices (e.g. management of fertilizer and manure application and storage of animal manures) and sanitation practices (e.g. the careful siting of pit latrines and septic tanks, sewer leakage control).

Methaemoglobinemia has most frequently been associated with private wells. It is particularly important to ensure that septic tanks and pit latrines are not sited near a well or where a well is to be dug and to ensure that animal manure is kept at a sufficient distance to ensure that runoff cannot enter the well or the ground near the well. It is particularly important that the household use of manures and fertilizers on small plots near wells should be managed with care to avoid potential contamination. The well should be sufficiently protected to prevent runoff from entering the well. Where there are elevated concentrations of nitrate or where inspection of the well indicated that there are sources of nitrate close by that could be causing contamination, particularly where there are also indications that microbiological quality might also be poor, a number of actions can be taken. As noted above, water should be heated only until the water reaches a rolling boil or disinfected by an appropriate means before consumption. Where alternative supplies are available for bottle-fed infants, these can be used, taking care to ensure that they are microbiologically safe. Steps should then be taken to protect the well and ensure that sources of both nitrate and microbial contamination are removed from the vicinity of the well.

In areas where household wells are common, health authorities may wish to take a number of steps to ensure that nitrate contamination is not or does not become a problem. Such steps could include targeting mothers, particularly expectant mothers, with appropriate information about water safety, assisting with visual inspection of wells to determine whether a problem may exist, providing testing facilities where a problem is suspected, providing guidance on disinfecting water or, where nitrate levels are particularly high, providing bottled water from safe sources or providing advice as to where such water can be obtained.

With regard to piped supplies, where nitrate is present, the first potential approach to treatment of drinking-water supplies, if source substitution is not feasible,

is to dilute the contaminated water with a low-nitrate source. Where blending is not feasible, a number of treatment techniques are available for drinking-water. The first is disinfection, which may serve to oxidize nitrite to the less toxic nitrate as well as minimize the pathogenic and non-pathogenic reducing bacterial population in the water. Nitrate removal methods include ion exchange, biological denitrification, reverse osmosis and electrodialysis. However, there are disadvantages associated with all of these approaches, including cost, operational complexities and the need for disposal of resin, brine or reject water. Conventional municipal water treatment processes (coagulation, sedimentation, filtration and chlorination) are not effective for nitrate removal, as nitrate is a stable and highly soluble ion with low potential for co-precipitation and adsorption.

In systems with a water source containing naturally occurring ammonia or that add ammonia for chloramination, free ammonia entering the distribution system can be one of the causative factors of nitrification and the potential increase of nitrate and nitrite in the distribution system. Care should be taken with the use of chloramination for providing a residual disinfectant in the distribution system. It is important to manage this to minimize nitrite formation, either in the main distribution system or in the distribution systems of buildings.

Nitritotriacetic acid

Nitritotriacetic acid, or NTA, is used primarily in laundry detergents as a replacement for phosphates and in the treatment of boiler water to prevent accumulation of mineral scale.

Guideline value	0.2 mg/l (200 µg/l)
Occurrence	Concentrations in drinking-water usually do not exceed a few micrograms per litre, although concentrations as high as 35 µg/l have been measured
TDI	10 µg/kg body weight, based on nephritis and nephrosis in a 2-year study in rats and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for carcinogenic potential at high doses)
Limit of detection	0.2 µg/l using GC with a nitrogen-specific detector
Treatment performance	No information found on removal from water
Guideline value derivation	
• allocation to water	50% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Nitritotriacetic acid in drinking-water</i>

NTA is not metabolized in experimental animals and is rapidly eliminated, although some may be briefly retained in bone. It is of low acute toxicity to experimental animals, but it has been shown to produce kidney tumours in rodents following long-term exposure to doses higher than those required to produce nephrotoxicity. IARC

has placed NTA in Group 2B (possibly carcinogenic to humans). It is not genotoxic, and the reported induction of tumours is believed to be due to cytotoxicity resulting from the chelation of divalent cations such as zinc and calcium in the urinary tract, leading to the development of hyperplasia and subsequently neoplasia.

Nitrobenzene

Nitrobenzene is used primarily in the production of aniline, but it is also used as a solvent, as an ingredient of metal polishes and soaps and in the synthesis of other organic compounds, including acetaminophen. Nitrobenzene can be released to water during these production processes.

Concentrations of nitrobenzene in environmental samples, such as surface water, groundwater and air, are generally low, except in areas with industrial pollution. Based on limited data, it appears that the potential for contamination is greater for groundwater than for surface water.

The general population can be exposed to variable concentrations of nitrobenzene in air and possibly drinking-water. Only populations in the vicinity of manufacturing activities and petroleum refining plants are likely to have any significant exposure to nitrobenzene; however, people living in and around abandoned hazardous waste sites may also have potential for higher exposure, due to possible groundwater and soil contamination and uptake of nitrobenzene by plants.

Reason for not establishing a guideline value	Rarely found in drinking-water at concentrations of health concern
Assessment date	2009
Principal reference	WHO (2009) <i>Nitrobenzene in drinking-water</i>

Nitrobenzene is toxic to humans by the inhalation, dermal and oral routes of exposure. The main systemic effect associated with human exposure to nitrobenzene is methaemoglobinaemia. Although some recent studies have reported positive results in mutagenicity tests, it cannot be excluded that nitrobenzene is a non-genotoxic chemical. No long-term oral administration studies are available. Based on inhalation studies, IARC concluded that there was inadequate evidence in humans but sufficient evidence in experimental animals for the carcinogenicity of nitrobenzene and classified nitrobenzene in Group 2B (possibly carcinogenic to humans).

Because nitrobenzene occurrence in drinking-water at concentrations above trace levels is infrequent, it is not considered necessary to derive a formal guideline value. However, health-based values can be calculated to provide guidance in the event of spills and where there are higher concentrations in industrial areas. Two health-based values are derived based on the limited available information: one for short-term exposure (30 µg/l) and the other for long-term exposure (8–63 µg/l, depending on end-point and approach used). It should be emphasized that the derivation of the long-term health-based values includes large uncertainties because of the dose metric conversion from inhalation studies and the possibility of increased metabolism to aniline in the gastrointestinal tract.

It should be emphasized that nitrobenzene is a potent methaemoglobinaemic agent in humans, which is of particular concern for bottle-fed infants. Currently, data are not adequate to determine a separate health-based value for this end-point.

It should also be noted that the reported odour threshold for nitrobenzene in water is 30–110 µg/l.

N-Nitrosodimethylamine

N-Nitrosodimethylamine, or NDMA, can occur in drinking-water through the degradation of dimethylhydrazine (a component of rocket fuel) as well as from several other industrial processes. It is also a contaminant of certain pesticides. NDMA has recently been identified as a disinfection by-product of chloramination (by the reaction of monochloramine with dimethylamine, a ubiquitous component of waters affected by wastewater discharges) and, to some extent, chlorination. NDMA can also be formed as a by-product of anion exchange treatment of water.

Guideline value	0.0001 mg/l (0.1 µg/l)
Occurrence	Where chloramination is used, distribution system samples can have much higher levels of NDMA than the finished water at the treatment plant; levels as high as 0.16 µg/l have been measured in the distribution system, but concentrations in water at the treatment plant are generally less than 0.01 µg/l
Basis of guideline value derivation	Hepatic biliary cystadenomas in female rats, the most sensitive carcinogenic end-point, observed in a drinking-water study, using a multistage model
Limit of detection	0.028 ng/l by capillary column GC and chemical ionization tandem MS; 0.4 ng/l by capillary column GC and high-resolution MS; 0.7–1.6 ng/l by GC-MS and ammonia positive chemical ionization detection
Treatment performance	The most common process for NDMA removal is UV irradiation. A concentration below 0.005 µg/l should be achievable by UV irradiation provided that the water is not grossly contaminated. NDMA is not removable by air stripping, activated carbon adsorption, reverse osmosis or biodegradation.
Additional comments	Potential methods for reducing the formation of NDMA during disinfection include avoiding the use of chloramination, use of breakpoint chlorination and removal of ammonia prior to chlorination.
Assessment date	2006
Principal references	IPCS (2002) <i>N-Nitrosodimethylamine</i> WHO (2008) <i>N-Nitrosodimethylamine in drinking-water</i>

There is conclusive evidence that NDMA is a potent carcinogen in experimental animals by several routes of exposure, including through ingestion of drinking-water. NDMA has been classified by IARC as probably carcinogenic to humans. The mechanism by which NDMA produces cancer is well understood to involve biotransformation by liver microsomal enzymes, generating the methyldiazonium ion. This reactive metabolite forms DNA adducts, with most evidence pointing to O⁶-methylguanine as

the likely proximal carcinogenic agent. As a consequence of the clear evidence of carcinogenicity, there have been few studies of other possible toxicity end-points.

There is also ample evidence that NDMA is genotoxic both in vivo and in vitro. Activation by liver microsomal S9 fractions is necessary for a positive in vitro result. The recent observation that human S9 fractions are much more active in promoting genotoxicity in the Ames test than rat S9 fractions suggests that humans may be especially sensitive to the carcinogenicity of NDMA.

Although there have been several case-control studies and one cohort study of NDMA in humans, none of them can be used to derive a quantitative risk of cancer. The results are supportive of the assumption that NDMA consumption is positively associated with either gastric or colorectal cancer. However, none of the studies focused on drinking-water as the route of exposure; instead, they used estimations of total dietary intake of NDMA.

Organotins

The group of chemicals known as the organotins is composed of a large number of compounds with differing properties and applications. Uses for the mono- and di-substituted compounds include as heat stabilizers in plastics, including PVC and chlorinated PVC (CPVC) water pipes. The tri-substituted compounds have been widely used as biocides and as antifouling agents in paint.

Reason for not establishing a guideline value	TBT, TPT, DBT and DOT occur in drinking-water or drinking-water sources at concentrations well below those of health concern
	Establishing a guideline value for MMT, DMT and DMTC is unnecessary because their use as stabilizers in PVC and CPVC is normally controlled by product specifications
	For other organotins, available data inadequate to permit derivation of health-based guideline values
Health-based value*	Sum of TBT, TPT, DBT and DOT: 1.5 µg/l (equivalent to approximately 0.6 µg/l tin)
Occurrence	Concentrations in drinking-water from PVC pipes are usually below a few hundred nanograms. However, many organotins have been detected as contaminants in environmental waters, including fresh waters that are possible drinking-water sources; the concentrations were mostly between the detection limits and hundreds of nanograms per litre.
TDI	Sum of TBT, TPT, DBT and DOT: 0.25 µg/kg bw (0.1 µg/kg bw as tin) derived by applying an uncertainty factor of 100 to account for interspecies (10) and intraspecies (10) variation to the NOAEL for tributyltin oxide (the reference organotin), based on chronic immunotoxicity studies in rats
Limit of detection	24–51 pg as tin for six organotin compounds (chlorides of DMT, DBT, TMT, TBT, DPT and TPT) by ICP-MS; 0.2–0.4 ng/l for DBT, TBT, DPT and TPT; and 2 ng/l for MPT, by derivatization–liquid extraction followed by GC and MS

Prevention and treatment	The effectiveness of water treatment appears to be significantly different for different compounds. For example, conventional water treatment (coagulation, sedimentation, filtration) has been shown to be effective in removing TBT in shipyard waters under optimal conditions, whereas it was ineffective in removing DPT and TPT. In contrast, advanced treatment processes were effective in removing these compounds but ineffective in removing other organotin compounds. However, available data and quantification information are still limited. Where the organotins originate from plastic service water pipes and fittings, particularly monoalkyltins and dialkyltins in PVC and CPVC pipes and fittings, the most important means of control is by product specifications through an appropriate certification scheme.
Health-based value derivation	• allocation to water 20% of TDI • weight 60 kg adult • consumption 2 litres/day
Assessment date	2020
Principal reference	WHO (2020) <i>Organotins in drinking-water</i>

DBT, dibutyltin; DMT, dimethyltin; DMTC, dimethyltin dichloride; DOT, di-n-octyltin; DPT, diphenyltin; MMT, monomethyltin; MPT, monophenyltin; TBT, tributyltin; TMT, trimethyltin; TPT, triphenyltin

* When a formal guideline value is not established, a “health-based value” may be determined in order to provide guidance to Member States when there is reason for local concern. Establishing a formal guideline value for such substances may encourage Member States to incorporate a value into their national standards when this may be unnecessary.

Reliable lifetime TDI values for monomethyltin, dimethyltin and dimethyltin dichloride could not be derived because of a lack of long-term studies with systematic experimental data. However, there is no need to establish a guideline value for these organotins because their use as stabilizers in PVC and CPVC is normally controlled by product specifications. The data available are insufficient to permit the proposal of guideline or health-based values for other organotins, including trimethyltin, tetrabutyltin, mono-n-octyltin, tetraoctyltin, monophenyltin, diphenyltin and tetraphenyltin.

Parathion

Parathion (CAS No. 56-38-2) is a non-systemic insecticide that is used in many countries throughout the world. It is used as a fumigant and acaricide and as a pre-harvest soil and foliage treatment on a wide variety of crops, both outdoors and in greenhouses. Parathion released to the environment will adsorb strongly to the top layer of soil and is not likely to leach significantly. Parathion disappears from surface waters in about a week. The general population is not usually exposed to parathion from air or water. Parathion residues in food are the main source of exposure.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (1996) <i>Pesticide residues in food—1995 evaluations</i> WHO (2004) <i>Parathion in drinking-water</i>

Parathion inhibits cholinesterase activity in all species tested. There has been no evidence of carcinogenicity in 2-year rat studies. JMPR concluded that parathion is not genotoxic.

A health-based value of 10 µg/l can be calculated for parathion on the basis of an ADI of 0–0.004 mg/kg body weight based on a NOAEL of 0.4 mg/kg body weight per day in a 2-year study in rats for retinal atrophy and inhibition of brain acetylcholinesterase at the next higher dose, and using an uncertainty factor of 100 for interspecies and intraspecies variation. Lower NOAELs in experimental animals, based only on inhibition of erythrocyte or brain acetylcholinesterase, were not considered relevant because of the availability of a NOAEL for erythrocyte acetylcholinesterase inhibition in humans, which was 0.1 mg/kg body weight per day.

Intake of parathion from all sources is generally low and well below the upper limit of the ADI. As the health-based value is much higher than concentrations of parathion likely to be found in drinking-water, the presence of parathion in drinking-water under usual conditions is unlikely to represent a hazard to human health. For this reason, the establishment of a formal guideline value for parathion is not deemed necessary.

Pendimethalin

Pendimethalin (CAS No. 40487-42-1) is a pre-emergence herbicide that is fairly immobile and persistent in soil. It is used in large amounts in Japan (5000 tonnes per year). It is lost through photodegradation, biodegradation and volatilization. The leaching potential of pendimethalin appears to be very low, but little is known about its more polar degradation products.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Rarely found in drinking-water in the limited studies available
TDI	5 µg/kg body weight, based on evidence of slight liver toxicity even at the lowest dose tested (5 mg/kg body weight) in a long-term rat feeding study, with an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for a combination of the use of a LOAEL instead of a NOAEL and limitations of the database)
Limit of detection	0.01 µg/l by GC-MS
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Pendimethalin in drinking-water</i>

In a short-term dietary study in rats, a variety of indications of hepatotoxicity as well as increased kidney weights in males were observed at the highest dose level. In a long-term dietary study, some toxic effects (hyperglycaemia in the mouse and hepatotoxicity in the rat) were present even at the lowest dose level. On the basis of

available data, pendimethalin does not appear to have significant mutagenic activity. Long-term studies in mice and rats have not provided evidence of carcinogenicity; however, these studies have some important methodological limitations.

Pentachlorophenol

Pentachlorophenol (CAS No. 87-86-5), or PCP, and other chlorophenols are used primarily for protecting wood from fungal growth. Food is usually the major source of exposure to PCP unless there is a specific local contamination of drinking-water by PCP or exposure from log homes treated with PCP.

Provisional guideline value	0.009 mg/l (9 µg/l) The guideline value is considered provisional because of the variations in metabolism between experimental animals and humans.
Occurrence	Concentrations in water samples are usually below 10 µg/l, although much higher concentrations in groundwater may be measured under certain conditions
Basis of guideline value derivation	Multistage modelling of tumour incidence in an NTP bioassay without incorporation of a body surface area correction, recognizing that there are interspecies differences in metabolism between experimental animals and humans, with an important metabolite formed in rats being only a minor metabolite in humans
Limit of detection	0.005–0.01 µg/l by GC with ECD
Treatment performance	0.4 µg/l should be achievable using GAC
Additional comments	The concentration of PCP associated with a 10 ⁻⁵ upper-bound excess lifetime cancer risk is similar to the guideline value established in the second edition, so that guideline value is retained.
Assessment date	1998
Principal reference	WHO (2003) <i>Pentachlorophenol in drinking-water</i>

IARC classified PCP in Group 2B (possibly carcinogenic to humans) on the basis of inadequate evidence of carcinogenicity in humans but sufficient evidence in experimental animals. There is suggestive, although inconclusive, evidence of the carcinogenicity of PCP from epidemiological studies of populations exposed to mixtures that include PCP. Conclusive evidence of carcinogenicity has been obtained in one animal species (mice). Although there are notable variations in metabolism between experimental animals and humans, it was considered prudent to treat PCP as a potential carcinogen.

Perchlorate

Perchlorate is a naturally occurring anion that is frequently detected in the environment. It is used primarily as an oxidizer for solid rocket fuels, automotive airbags, fireworks and road flares. Perchlorate is found in water due to contamination from perchlorate manufacturing or use, natural deposits of perchlorate, use of fertilizers containing natural deposits of perchlorate, and natural formation of perchlorate in the atmosphere and its deposition during rain or snow events. It also forms in hypochlo-

rite solutions to varying degrees, depending on the hypochlorite concentration, age and storage conditions.

Guideline value	0.07 mg/l (70 µg/l)
Occurrence	Generally found in drinking-water at concentrations below 10 µg/l, although concentrations above 40 µg/l have been measured
PMTDI	0.01 mg/kg bw, based on a BMDL ₅₀ of 0.11 mg/kg bw per day for 50% inhibition of iodide uptake, derived from a human clinical study on healthy adult volunteers administered perchlorate in drinking-water, and using an uncertainty factor of 10 to account for inter-individual differences
Limit of detection	20–50 ng/l (method reporting limits) by LC-MS; 4 µg/l (method reporting limit) by IC with suppressed conductivity detection
Treatment performance	The perchlorate anion is highly stable in water and is difficult to remove using conventional water treatment technologies. Treatment technologies that have been shown to effectively remove perchlorate from water include nanofiltration and reverse osmosis membranes, anaerobic biodegradation and ion exchange.
Guideline value derivation	
• allocation to water	20% of unrounded PMTDI (0.011 mg/kg bw)
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	2016
Principal references	EFSA (2014) <i>Scientific opinion on the risks to public health related to the presence of perchlorate in food, in particular fruits and vegetables</i> FAO/WHO (2011) <i>Safety evaluation of certain contaminants in food</i> WHO (2016) <i>Perchlorate in drinking-water</i>

The primary effect of perchlorate is its ability to competitively inhibit uptake of iodide by the thyroid gland. Inhibition of iodide uptake by perchlorate reduces the amount of iodide available for the synthesis of thyroid hormones. Sustained reduction in iodide uptake by the thyroid may result in hypothyroidism, which has adverse implications for structural and functional brain development in the fetus, infant and child, and for metabolism and the functioning of the cardiovascular, gastrointestinal, skeletal, neuromuscular and reproductive systems in adults. As the rat is not a good model for humans for substances known to affect the thyroid and having a mode of action involving inhibition of the uptake of iodide, the guideline value was derived from human studies.

Petroleum products

Petroleum products are used in large quantities, primarily as fuels. They are complex mixtures of chemicals derived from crude oil by distillation and fractionation. They consist primarily of a wide range of aliphatic and aromatic hydrocarbons, many of which are of extremely low solubility in water. Petroleum products are widely stored and handled and are often spilt. The primary concern for drinking-water is the potential for spills into source water, penetration of distribution systems and contamination of drinking-water treatment works.

Reason for not establishing a guideline value	Taste and odour will in most cases be detectable at concentrations below those of health concern, particularly with short-term exposure
Assessment date	2004
Principal reference	WHO (2008) <i>Petroleum products in drinking-water</i>

Exposure to the constituents of petroleum products through drinking-water is frequently short term, as the result of an accidental spill or short-term incident. Such incidents may lead to high concentrations of total petroleum hydrocarbons. However, a number of the most soluble aromatic hydrocarbons will be detectable by taste or odour at concentrations below those concentrations of concern for health, particularly for short-term exposure. Substances such as the alkyl benzenes and the alkyl naphthalenes have taste and odour thresholds of a few micrograms per litre. In view of the above, it is not considered appropriate to set a formal health-based guideline value for petroleum products in drinking-water.

In the event of a spill, it may be necessary to carry out a context-specific assessment of the risk to health. The fact that petroleum products are complex mixtures of many individual hydrocarbons is a complicating factor in determining the potential risks to consumers. The traditional approach of evaluating individual chemicals in assessing the risks from drinking-water is therefore largely inappropriate. In order to overcome this difficulty, it is more practical to consider a series of hydrocarbon fractions and to determine appropriate tolerable concentrations for those fractions. The most widely accepted approach is that developed by the Total Petroleum Hydrocarbons Criteria Working Group in the USA, which divided total petroleum hydrocarbons into a series of aliphatic and aromatic fractions based on the number of carbon atoms and the boiling point, to give equivalent carbon numbers.

This pragmatic approach provides a suitable basis for assessing the potential health risks associated with larger-scale contamination of drinking-water by petroleum products. The allocation of 10% of each of the reference doses, equivalent to TDIs, for the various fractions to drinking-water provides a conservative assessment of the risks. Although the approach is based on the analysis of hydrocarbon fractions, most are of low solubility, and the most soluble fractions, consisting largely of lower molecular weight aromatic hydrocarbons, will be present in the greatest concentration.

pH

No health-based guideline value is proposed for pH. Although pH usually has no direct impact on consumers, it is one of the most important operational water quality parameters (see [chapter 10](#)).

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	An important operational water quality parameter
Assessment date	1993
Principal reference	WHO (2007) <i>pH in drinking-water</i>

2-Phenylphenol and its sodium salt

2-Phenylphenol (CAS No. 90-43-7) is used as a disinfectant, bactericide and virucide. In agriculture, it is used in disinfecting fruits, vegetables and eggs. It is also used as a general surface disinfectant in hospitals, nursing homes, veterinary hospitals, poultry farms, dairy farms, commercial laundries, barbershops and food processing plants. 2-Phenylphenol is readily degraded in surface waters, with a half-life of about 1 week in river water.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal references	FAO/WHO (2000) <i>Pesticide residues in food—1999 evaluations</i> WHO (2003) <i>2-Phenylphenol and its sodium salt in drinking-water</i>

2-Phenylphenol has been determined to be of low toxicity. Both 2-phenylphenol and its sodium salt are carcinogenic in male rats, and 2-phenylphenol is carcinogenic in male mice. However, urinary bladder tumours observed in male rats and liver tumours observed in male mice exposed to 2-phenylphenol appear to be threshold phenomena that are species and sex specific. JMPR concluded that 2-phenylphenol is unlikely to represent a carcinogenic risk to humans. Although a working group convened by IARC classified 2-phenylphenol, sodium salt, in Group 2B (possibly carcinogenic to humans) and 2-phenylphenol in Group 3 (not classifiable as to its carcinogenicity to humans), JMPR noted that the IARC classification is based on hazard identification, not risk assessment, and is furthermore limited to published literature, excluding unpublished studies on toxicity and carcinogenicity. JMPR also concluded that there are unresolved questions about the genotoxic potential of 2-phenylphenol.

A health-based value of 1 mg/l can be calculated for 2-phenylphenol on the basis of an ADI of 0–0.4 mg/kg body weight, based on a NOAEL of 39 mg/kg body weight per day in a 2-year toxicity study on the basis of decreased body weight gain and hyperplasia of the urinary bladder and carcinogenicity of the urinary bladder in male rats, using an uncertainty factor of 100 for interspecies and intraspecies variation. Because of its low toxicity, however, the health-based value derived for 2-phenylphenol is much higher than concentrations of 2-phenylphenol likely to be found in drinking-water. Under usual conditions, therefore, the presence of 2-phenylphenol in drinking-water is unlikely to represent a hazard to human health. For this reason, the establishment of a formal guideline value for 2-phenylphenol is not deemed necessary.

Polynuclear aromatic hydrocarbons

Polynuclear aromatic hydrocarbons, or PAHs, form a class of diverse organic compounds each containing two or more fused aromatic rings of carbon and hydrogen atoms. Most PAHs enter the environment via the atmosphere from a variety of combustion processes and pyrolysis sources. Owing to their low solubility and

high affinity for particulate matter, they are not usually found in water in notable concentrations. The main source of PAH contamination in drinking-water is usually the coal tar coating of drinking-water distribution pipes, used to protect the pipes from corrosion. Fluoranthene is the most commonly detected PAH in drinking-water and is associated primarily with coal tar linings of cast iron or ductile iron distribution pipes. PAHs have been detected in a variety of foods as a result of the deposition of airborne PAHs and in fish from contaminated waters. PAHs are also formed during some methods of food preparation, such as char-broiling, grilling, roasting, frying or baking. For the general population, the major routes of exposure to PAHs are from food and ambient and indoor air. The use of open fires for heating and cooking, which is common especially in developing countries, may increase PAH exposure. Where there are elevated levels of contamination by coal tar coatings of water pipes, PAH intake from drinking-water could equal or even exceed that from food.

Guideline value	<i>Benzo[a]pyrene</i> : 0.0007 mg/l (0.7 µg/l)
Occurrence	PAH levels in uncontaminated groundwater usually in range 0–5 ng/l; concentrations in contaminated groundwater may exceed 10 µg/l; typical concentration range for sum of selected PAHs in drinking-water is from about 1 ng/l to 11 µg/l
Basis of guideline value derivation	Based on an oral carcinogenicity study in mice and calculated using a two-stage birth–death mutation model, which incorporates variable dosing patterns and time of killing; quantification of dose–response for tumours, on the basis of new studies in which the carcinogenicity of benzo[a]pyrene was examined following oral administration in mice, but for which the number of dose groups was smaller, confirms this value
Limit of detection	0.01 µg/l by GC-MS and reversed-phase HPLC with a fluorescence detector
Treatment performance	0.05 µg/l should be achievable using coagulation
Additional comments	The presence of significant concentrations of benzo[a]pyrene in drinking-water in the absence of very high concentrations of fluoranthene indicates the presence of coal tar particles, which may arise from seriously deteriorating coal tar pipe linings. It is recommended that the use of coal tar–based and similar materials for pipe linings and coatings on storage tanks be discontinued.
Assessment date	1998
Principal reference	WHO (2003) <i>Polynuclear aromatic hydrocarbons in drinking-water</i>
Reason for not establishing a guideline value	<i>Fluoranthene</i> : Occurs in drinking-water at concentrations well below those of health concern
Assessment date	1998
Principal reference	WHO (2003) <i>Polynuclear aromatic hydrocarbons in drinking-water</i>

Evidence that mixtures of PAHs are carcinogenic to humans comes primarily from occupational studies of workers following inhalation and dermal exposure. No data are available for humans for the oral route of exposure. There are few data on the oral toxicity of PAHs other than benzo[*a*]pyrene, particularly in drinking-water. Relative potencies of carcinogenic PAHs have been determined by comparison of data from dermal and other studies. The order of potencies is consistent, and this scheme therefore provides a useful indicator of PAH potency relative to that of benzo[*a*]pyrene.

A health-based value of 4 µg/l can be calculated for fluoranthene on the basis of a NOAEL of 125 mg/kg body weight per day for increased serum glutamate-pyruvate transaminase levels, kidney and liver pathology, and clinical and haematological changes in a 13-week oral gavage study in mice, using an uncertainty factor of 10 000 (100 for interspecies and intraspecies variation, 10 for the use of a subchronic study and inadequate database and 10 because of clear evidence of co-carcinogenicity with benzo[*a*]pyrene in mouse skin painting studies). However, this health-based value is significantly above the concentrations normally found in drinking-water. Under usual conditions, therefore, the presence of fluoranthene in drinking-water does not represent a hazard to human health. For this reason, the establishment of a formal guideline value for fluoranthene is not deemed necessary.

Potassium

Potassium is an essential element in humans and is seldom, if ever, found in drinking-water at levels that could be a concern for healthy humans. The recommended daily requirement is greater than 3000 mg. Potassium occurs widely in the environment, including all natural waters. It can also occur in drinking-water as a consequence of the use of potassium permanganate as an oxidant in water treatment. In some countries, potassium chloride is being used in ion exchange for household water softening in place of, or mixed with, sodium chloride, so potassium ions would exchange with calcium and magnesium ions. Possible replacement or partial replacement of sodium salts with potassium salts for conditioning desalinated water has been suggested. The latter seems to be an unlikely development at this stage, in view of the cost difference.

Reason for not establishing a guideline value	Occurs in drinking-water at concentrations well below those of health concern
Assessment date	2009
Principal reference	WHO (2009) <i>Potassium in drinking-water</i>

Currently, there is no evidence that potassium levels in municipally treated drinking-water, even water treated with potassium permanganate, are likely to pose any risk for the health of consumers. It is not considered necessary to establish a health-based guideline value for potassium in drinking-water.

Although potassium may cause some health effects in susceptible individuals, potassium intake from drinking-water is well below the level at which adverse health effects may occur. Health concerns would be related to the consumption of drinking-water treated by potassium-based water treatment (principally potassium

chloride for regeneration of ion exchange water softeners), affecting only individuals in high-risk groups (i.e. individuals with kidney dysfunction or other diseases, such as heart disease, coronary artery disease, hypertension, diabetes, adrenal insufficiency, pre-existing hyperkalaemia; people taking medications that interfere with normal potassium-dependent functions in the body; and older individuals or infants). It is recommended that susceptible individuals seek medical advice to determine whether they should avoid the consumption of water (for drinking or cooking) treated by water softeners using potassium chloride.

When high-risk individuals have been advised by a physician to avoid elevated potassium intake from water, the recommended strategy is to limit the addition of potassium to water that will be ingested or to avoid ingesting such water. This can be done by having a proportion of the water bypass the softener altogether; this approach is recommended by several countries. Although technologies are available to remove potassium, they are generally more expensive and redundant when combined with the softening treatment.

Propanil

Propanil (CAS No. 709-98-8) is a contact post-emergence herbicide used to control broad-leaved and grassy weeds, mainly in rice. It is a mobile compound with affinity for the water compartment. Propanil is not, however, persistent, being easily transformed under natural conditions to several metabolites. Two of these metabolites, 3,4-dichloroaniline and 3,3',4,4'-tetrachloroazobenzene, are more toxic and more persistent than the parent compound. Although used in a number of countries, propanil has only occasionally been detected in groundwater.

Reason for not establishing a guideline value	Readily transformed into metabolites that are more toxic; a guideline value for the parent compound is considered inappropriate, and there are inadequate data to enable the derivation of guideline values for the metabolites
Assessment date	2003
Principal reference	WHO (2003) <i>Propanil in drinking-water</i>

Although a health-based value for propanil can be derived, this has not been done, because propanil is readily transformed into metabolites that are more toxic. Therefore, a guideline value for the parent compound is considered inappropriate, and there are inadequate data on the metabolites to allow the derivation of guideline values for them. Authorities should consider the possible presence in water of more toxic environmental metabolites.

Saxitoxins (cyanobacterial toxins)¹

Saxitoxins (STXs) are naturally occurring alkaloids produced by some marine dinoflagellates and by strains of various species of freshwater cyanobacteria. They have

¹ As cyanobacteria and their toxins are a concern in many areas and considering the complexities in their management, this chemical fact sheet has been expanded.

been found in cyanobacterial blooms around the globe and from a number of cyanobacterial genera, including *Dolichospermum*, *Raphidiopsis* and *Aphanizomenon* (see also [section 11.5](#)).

Whereas STX-contaminated marine shellfish are the most likely cause of the severe illness known as paralytic shellfish poisoning, drinking-water is the most likely exposure route where surface water with cyanobacterial blooms is the drinking-water source. However, in locations where marine shellfish are contaminated with STXs, they are likely to be the greater source of exposure. Limited data suggest that STXs may also accumulate in some freshwater food items. Recreational activities may also be a relevant exposure pathway, potentially to high concentrations (see WHO *Guidelines on recreational water quality*, 2021).

Acute guideline value	<i>Total STXs (sum of all congeners, free plus cell-bound):</i> 0.003 mg/l (3 µg/l)
Occurrence	Although concentrations usually range well below 1 µg/l, several µg/l are occasionally reported, and up to almost 200 µg/l has been found in scums. STXs largely occur cell-bound unless cell damage causes release
ARfD	FAO (2004) and EFSA (2009) reviewed a large dataset of human STX poisoning cases. Exposure was to a range of STX analogues. An ARfD for saxitoxins of 0.5 µg/kg bw was derived based on a LOAEL for mild symptoms of 1.5 µg/kg bw of STX equivalents (EFSA, 2009). A database uncertainty factor of 3 was applied for use of a LOAEL.
Limit of detection	<3 µg/l by LC-MS or HPLC methods with either pre- or post-column derivatisation and fluorescence detection; LC-MS methods require quantitative reference standards, which are available for only some congeners. A commercially available receptor-binding assay circumvents this problem and provides more reliable results than HPLC. Prior extraction of cells with freeze–thaw cycles and acetic acid, hydrochloric acid or acidified aqueous methanol is necessary for cell-bound STX; neglecting extraction from cells will lead to dramatic underestimation of concentrations. STX analysis requires particular care in the laboratory because congeners may interconvert; furthermore, the chemical variability of congeners presents challenges. 0.02 µg/l for STX alone by commercially available immunoassay kits (ELISA), but cross-reactivity with other congeners is highly variable.
Monitoring	The likelihood of blooms can be assessed by understanding water body conditions (in particular, nutrient concentrations, water body depth, water retention time, patterns of mixing and stratification; see section 11.5). Where conditions render blooms likely, visual monitoring of source water (including microscopy for potentially STX-containing genera) is important because biomass can increase rapidly. Exceeding alert values for biomass indicators or STX concentrations should trigger management responses to prevent exposure to elevated toxin concentrations (see the alert level framework in section 11.5). Analysis of cyanotoxins is particularly useful for validating and optimizing the efficacy of control measures such as riverbank filtration or treatment.

Prevention and treatment	Actions to decrease the probability of bloom occurrence include catchment and source water management, such as reducing nutrient loading or changing reservoir stratification and mixing. Filtration is effective for removing intact cyanobacterial cells, but some dissolved STX may be present. For this dissolved fraction, oxidation with chlorine at sufficient concentrations and contact times, as well as GAC and some PAC applications, are effective (see chapters 7–10 of <i>Toxic cyanobacteria in water</i> ; Annex 1). Ozone may also be effective, but research is limited.
Guideline value derivation	• allocation to water 100% of ARfD • weight 5 kg infant • consumption 0.75 litres/day
Additional comments	The total amount of STX and its structural analogues as gravimetric or molar equivalents should be evaluated against the guideline value. The drinking-water guideline value was derived to protect bottle-fed infants because the guideline value is for acute exposure, and a guideline value based on adult exposure could allow exposure of infants to a concentration of STXs close to the LOAEL. For a 60 kg adult consuming 2 litres of drinking-water per day, a 5-fold higher concentration than the acute guideline value would be tolerable. In locations where exposure to STXs may occur through both marine shellfish and drinking-water, health authorities may need to consider the combined exposure in their risk assessments.
Assessment date	2020
Principal references	FAO (2004) <i>Marine biotoxins</i> EFSA (2009) <i>Marine biotoxins in shellfish: saxitoxin group</i> WHO (2020) <i>Cyanobacterial toxins: saxitoxins</i> Chorus & Welker (2021) <i>Toxic cyanobacteria in water</i>

STXs act by blocking sodium channels in nerve cell axons, which inhibits propagation of an action potential along the axon. When this occurs in sensory neurons, symptoms such as tingling and numbness are induced; in motor neurons, muscle weakness or paralysis ensues. The effects of human intoxication by STXs have been well described from numerous cases of paralytic shellfish poisoning after consumption of marine shellfish. They range from numbness and tingling in the tongue and mouth through muscular weakness in the limbs to, in severe cases, respiratory failure and death. If a person recovers from the acute toxicity, they appear to make a full recovery and no long-term adverse effects are known. STXs were the cause of massive animal deaths along 1000 km of the Murray–Darling river system in Australia during a heavy bloom of *Dolichospermum* in 1991. Unlike the other cyanotoxins considered in these Guidelines (MCs, CYNs and ATXs), a guideline value for STXs could be derived from human data that are available from incidents of shellfish poisoning.

Practical considerations

Where nutrient (phosphorus and nitrogen) concentrations are elevated in lakes, reservoirs or slowly flowing rivers, cyanobacteria occur widely. Where their excessive

growth leads to high biomass, sometimes termed “bloom” events, STXs can reach concentrations in raw water that are potentially hazardous to human health. Such blooms tend to recur in the same water bodies. Cells of some cyanobacterial species (e.g. *Raphidiopsis*, *Aphanizomenon*, *Dolichospermum*) may accumulate at the surface as scums. Such accumulations may develop rapidly and may be of very variable duration (hours to weeks). In many circumstances, blooms and accumulations are seasonal, whereas others occur perennially.

Cyanobacteria are most effectively controlled in the context of developing a WSP (see [chapter 4](#)). Control measures to manage potential risks from cyanobacteria, and in particular from their toxins, in drinking-water should include not only adequate treatment, but also measures to control cyanobacterial bloom development. See [section 11.5](#) for more information on cyanobacteria, including further details on monitoring cyanobacterial blooms, the alert level framework, and prevention and management of cyanobacteria in source waters. Effectively minimizing the formation of blooms and locating the raw water intake away from blooms reduce the treatment steps required to remove cyanotoxins.

Drinking-water treatment that removes particles—that is, soil, slow sand or riverbank filtration, conventional water treatment (coagulation, flocculation and filtration) or dissolved air flotation—can remove cell-bound STXs effectively. Soil, slow sand and riverbank filtration can also remove dissolved cyanotoxins. For all these processes, it is important that they are optimized to target the removal of cells and dissolved toxins. Chlorination at sufficiently high doses and contact times is effective for degrading dissolved STXs; however, elevated organic carbon in bloom situations will substantially increase the disinfectant demand. Ozonation may also be effective, but research is limited. Chlorine dioxide and chloramine are ineffective for degrading STXs. Both for pre-oxidation and conventional treatment, cell rupture and toxin release should be avoided. GAC and PAC can be effective for removing dissolved STXs, with efficacy dependent on several factors, including the type of activated carbon, contact times (PAC), flow rates (GAC) and water quality. As the challenges that blooms present for treatment are complex, periodic validation of efficacy during bloom situations and under the specific local conditions is particularly important. Avoiding bloom occurrence and intake is therefore the preferred option.

Selenium

Selenium is present in Earth’s crust, often in association with sulfur-containing minerals. Selenium is an essential trace element, and foodstuffs such as cereals, meat and fish are the principal source of selenium for the general population. Levels in food also vary greatly according to geographical area of production. However, even in high-selenium areas, the relative contribution of selenium from drinking-water is likely to be small in comparison with that from locally produced food.

Provisional guideline value	0.04 mg/l (40 µg/l)

The guideline value is designated as provisional because of the uncertainties inherent in the scientific database.

GUIDELINES FOR DRINKING-WATER QUALITY: FOURTH EDITION
INCORPORATING THE FIRST, SECOND AND THIRD ADDENDA

Occurrence	Most drinking-water contains concentrations of selenium that are much lower than 10 µg/l, except in certain seleniferous areas
Basis of guideline value derivation	An allocation of 20% of the upper tolerable intake of 400 µg/day to drinking-water provides a sensible balance that will assist regulators and suppliers in making decisions about whether further action is needed
Limit of detection	0.5 µg/l by hydride generation AAS
Treatment performance	Selenium is not removed by conventional treatment processes; significant removals of selenium from water using activated alumina adsorption, ion exchange, reverse osmosis and nanofiltration have been reported.
Guideline value derivation	
• allocation to water	20% of upper tolerable intake
• consumption	2 litres/day
Additional comments	It is important that a proper balance be achieved between recommended intakes and undesirable intakes in determining an appropriate guideline value for selenium in drinking-water. While for most parts of the world, the concentration of selenium in drinking-water will not exceed 10 µg/l, there are circumstances in which selenium may be elevated significantly above normal concentrations, and guidance may be required. Where selenium intake from the diet is known, this should be used in determining a concentration that ensures that intake is safe and sufficient. Where selenium intake from the diet is not known, guidance may be required. For most Member States, a drinking-water guideline for selenium is unnecessary. Where there are regions of high intake from a number of sources, of which drinking-water may be one, then Member States should take into consideration exposure from all sources in determining actions to reduce exposure. For drinking-water, this may include using alternative sources, blending low-selenium sources with high-selenium sources as well as considering selenium removal.
Assessment date	2010
Principal references	FAO/WHO (2004) <i>Vitamin and mineral requirements in human nutrition</i> WHO (2011) <i>Selenium in drinking-water</i>

Selenium is an essential element for humans, and there are indications that selenium status may be marginal in many parts of the world, including western Europe. The potential for adverse effects from selenium deficiency appears to be dependent on a number of factors, including overall health and nutritional status. Very low selenium status in humans has been associated with a juvenile, multifocal myocarditis called Keshan disease and a chondrodystrophy called Kaschin-Beck disease. Several studies have also found blood selenium levels to be inversely associated with the prevalence of several types of cancer.

High intakes of selenium are also associated with a number of specific diseases and the potential for adverse effects, but, again, this seems to be strongly influenced by other factors. Symptoms in people with high urinary selenium levels included gastrointestinal disturbances, discoloration of the skin, decayed teeth, hair or nail loss, nail abnormalities and changes in peripheral nerves. Slight biochemical changes have also

been observed. One case of selenium toxicity directly attributable to a water source (well water containing selenium at a concentration of 9 mg/l) has been reported. The average dietary intake that is associated with selenosis has been found to be in excess of 900 µg/day.

As selenium is an essential element, various national and international organizations have established recommended daily intakes of selenium. A joint FAO/WHO consultation recommended intakes of 6–21 µg of selenium per day for infants and children, according to age, 26 and 30 µg of selenium per day for adolescent females and males, respectively, and 26 and 35 µg of selenium per day for adult females and males, respectively.

Because of concern about the adverse effects resulting from exposure to excessive levels of selenium, various national and international organizations have established upper limits of exposure for selenium. FAO/WHO established an upper tolerable limit for selenium of 400 µg/day.

Silver

Silver occurs naturally, mainly in the form of its very insoluble and immobile sulfide, oxides and some salts. Silver ions are primarily found in the +1 oxidation state, and the ionic compounds silver nitrate (soluble) and silver chloride (relatively insoluble) are the most important forms of silver for drinking-water. Silver can also occur as nanoparticles, which can be present in water bodies as a result of release into the environment from wastewater or industrial discharges. When silver is used in point-of-use water treatment devices, drinking-water is expected to be the major source of exposure to silver.

Reason for not establishing a guideline value	Available data inadequate to permit derivation of health-based guideline value, and silver usually occurs in drinking-water at concentrations well below those of health concern
Provisional reference value*	0.1 mg/l
Occurrence	In surface water and groundwater, silver concentrations are usually below 5 µg/l; however, detections >100 µg/l, although rare, have been reported
TDI	A formal TDI could not be derived because of database limitations. However, 0.6 mg/kg bw of colloidal silver per day was considered a LOAEL based on a case report of argyria in a woman who ingested this dose of silver for 16 months. An uncertainty factor of 100 (10 for intraspecies variability and 10 for database limitations) was applied. The database limitations include the short duration of the study, uncertainty associated with a dose level derived from human recall and use of a LOAEL instead of a NOAEL.
Limit of detection	2 ng/ by neutron activation analysis; 5 ng/l by ICP-MS; 2 µg/l by graphite furnace AAS; 10 µg/l by a spectrographic and colorimetric method with dithizone for a 20 ml sample. In addition, asymmetric flow field-flow fractionation in combination with single particle ICP-MS can differentiate between silver nanoparticles and dissolved silver, although this method has not yet been standardized. Since ionic silver can be released from silver nanoparticles, it may be difficult to determine whether silver dispersed in water has originated from the ionic or the particulate fraction.

Treatment performance	Conventional treatment (coagulation, sedimentation, filtration) and lime softening are effective for removing ionic silver; conventional treatment is also expected to be effective for removing coated silver nanoparticles
Guideline value derivation	
• allocation to water	80%
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The provisional reference value is applicable particularly where silver is used in point-of-use water treatment devices. However, silver is not recommended for disinfection of drinking-water including in point-of-use water treatment devices (see Part III of the supporting document <i>Alternative drinking-water disinfectants: bromine, iodine and silver</i> ; Annex 1). It is also noted that silver with copper may be necessary to control <i>Legionella</i> in the distribution systems of buildings, and the risks of legionellosis outweigh the risks from low levels of silver ions (usually in the low µg/l range) that may be detected in drinking-water as a result of such use. See <i>Silver in drinking-water</i> , Annex 2 , for further information, including on <i>Legionella</i> control measures.
Assessment date	2021
Principal reference	WHO (2021) <i>Silver in drinking-water</i>

* Although a formal guideline value cannot be established, the provisional reference value was derived based on limited available studies, recognizing that a “bounding value” may be useful, to provide guidance to Member States in the event of need. Reference values are too uncertain to be used for developing regulations or standards.

The only obvious sign of silver overload is argyria, a condition in which skin and hair are heavily discoloured by silver in the tissues. In argyria, silver is deposited in various organs (e.g. skin, kidney, liver) after oral ingestion of silver in its ionic form or as silver nanoparticles, oxidizing to insoluble silver sulfide, which is responsible for the discolouring effects. It is difficult to determine the lowest dose that may lead to development of argyria, and the toxicological database on silver is not adequate to support derivation of a formal guideline value. Furthermore, a formal guideline value is considered unnecessary since silver usually occurs naturally in drinking-water at concentrations well below those of health concern. Nevertheless, a bounding value (provisional reference value) may provide a useful benchmark where elevated concentrations of silver in drinking-water may be expected. The provisional reference value of 0.1 mg/l is supported by the prior assessments that concluded that 10 g of ingested silver can be considered a human NOAEL (WHO, 1984a, b, 1993; US EPA 1992). Over a 70-year period, assuming a drinking-water intake of 2 litres/day, 0.1 mg/l is a concentration in drinking-water that would give a total dose of half this NOAEL.

Simazine

Simazine (CAS No. 122-34-9) is a pre-emergence herbicide used on a number of crops as well as in non-crop areas. It is fairly resistant to physical and chemical dissipation processes in the soil. It is persistent and mobile in the environment.

Guideline value	0.002 mg/l (2 µg/l)
Occurrence	Frequently detected in groundwater and surface water at concentrations of up to a few micrograms per litre
TDI	0.52 µg/kg body weight, based on a NOAEL of 0.52 mg/kg body weight from a long-term study in the rat (based on weight changes, effects on haematological parameters and an increase in mammary tumours) and an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for possible non-genotoxic carcinogenicity)
Limit of detection	0.01 µg/l by GC-MS; 0.1–0.2 µg/l by GC with flame thermionic detection
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Simazine in drinking-water</i>

Simazine does not appear to be genotoxic in mammalian systems. Recent studies have shown an increase in mammary tumours in the female rat but no effects in the mouse. IARC has classified simazine in Group 3 (not classifiable as to its carcinogenicity to humans).

Sodium

Sodium salts (e.g. sodium chloride) are found in virtually all food (the main source of daily exposure) and drinking-water. Although concentrations of sodium in potable water are typically less than 20 mg/l, they can greatly exceed this in some countries. The levels of sodium salts in air are normally low in relation to those in food or water. It should be noted that some water softeners can add significantly to the sodium content of drinking-water.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Sodium in drinking-water</i>

No firm conclusions can be drawn concerning the possible association between sodium in drinking-water and the occurrence of hypertension. Therefore, no health-based guideline value is proposed. However, concentrations in excess of 200 mg/l may give rise to unacceptable taste (see [chapter 10](#)).

Sodium dichloroisocyanurate

Sodium dichloroisocyanurate is the sodium salt of a chlorinated hydroxytriazine and is used as a source of free available chlorine, in the form of hypochlorous acid, for the disinfection of water. It is widely used as a stable source of chlorine for the disinfection of swimming pools and in the food industry. It is also used as a means of disinfecting drinking-water, primarily in emergencies, when it provides an easy-to-use source of free chlorine, and, more recently, as the form of chlorine for household point-of-use water treatment.

Guideline values	<i>Sodium dichloroisocyanurate</i> : 50 mg/l (50 000 µg/l) <i>Cyanuric acid</i> : 40 mg/l (40 000 µg/l)
Occurrence	Where sodium dichloroisocyanurate is used for the disinfection of drinking-water, exposure will be to both the chlorinated species and residual cyanuric acid. The concentrations will relate directly to the quantities added to achieve adequate disinfection.
TDI	2.2 mg/kg body weight for anhydrous sodium dichloroisocyanurate and 1.54 mg/kg body weight for cyanuric acid, based on a NOEL of 154 mg/kg body weight per day (equivalent to 220 mg/kg body weight per day as anhydrous sodium dichloroisocyanurate) for urinary tract and cardiac lesions from a 2-year study of rats exposed to sodium cyanurate and using an uncertainty factor of 100 for interspecies and intraspecies variation
Limit of detection	0.001 mg/l by GC with flame thermionic specific detection; 0.05 mg/l by reversed-phase LC with UV detection; 0.09 mg/l by GC with MS selective ion monitoring
Treatment performance	At very high chlorine doses (up to 10 mg/l), the sodium cyanurate concentration would be below 11 mg/l. In emergency situations, "topping up" might be done in an attempt to maintain a free chlorine residual, but this practice should be discouraged. In this case, it would be possible for the sodium cyanurate concentration to build up to undesirable levels. In such cases, it would be very desirable to monitor the concentration of sodium cyanurate.
Guideline value derivation	• allocation to water 80% of TDI • weight 60 kg adult • consumption 2 litres/day
Additional comments	The controlling factors are the level of free chlorine and the residue of cyanuric acid, particularly if there is topping up of chlorine in a static system under emergency conditions. The concentration of free chlorine should normally be such that it should not give rise to unacceptable tastes and should not normally exceed the guideline value of 5 mg/l for free chlorine. Sodium dichloroisocyanurate used for disinfecting drinking-water should be of adequate purity so that there is no increase in any inorganic or organic contaminants in the drinking-water. The amounts of sodium dichloroisocyanurate used should be the lowest consistent with adequate disinfection, and the concentrations of cyanuric acid should be managed to be kept as low as is reasonably possible.

Assessment date	2007
Principal references	FAO/WHO (2004) <i>Evaluation of certain food additives and contaminants</i> WHO (2008) <i>Sodium dichloroisocyanurate in drinking-water</i>

Studies of the toxicity of sodium cyanurate are appropriate for assessing the safety of sodium dichloroisocyanurate, because any residues of intact sodium dichloroisocyanurate in drinking-water would be rapidly converted to cyanuric acid on contact with saliva. Both sodium dichloroisocyanurate and sodium cyanurate have low acute oral toxicity. Sodium cyanurate does not induce any genotoxic, carcinogenic or teratogenic effects. The NOEL from which the guideline value was derived was based on multiple lesions of the urinary tract (calculi and hyperplasia, bleeding and inflammation of the bladder epithelium, dilated and inflamed ureters and renal tubular nephrosis) and cardiac lesions (acute myocarditis, necrosis and vascular mineralization) in male rats exposed at the next higher dose.

Styrene

Styrene, which is used primarily for the production of plastics and resins, is found in trace amounts in surface water, drinking-water and food. In industrial areas, exposure via air can result in intake of a few hundred micrograms per day. Smoking may increase daily exposure by up to 10-fold.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Has been detected in drinking-water and surface water at concentrations below 1 µg/l
TDI	7.7 µg/kg body weight, based on a NOAEL of 7.7 mg/kg body weight per day for decreased body weight observed in a 2-year drinking-water study in rats, and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the carcinogenicity and genotoxicity of the reactive intermediate styrene-7,8-oxide)
Limit of detection	0.3 µg/l by GC with photoionization detection and confirmation by MS
Treatment performance	0.02 mg/l may be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	May affect the acceptability of drinking-water at the guideline value
Assessment date	1993
Principal reference	WHO (2003) <i>Styrene in drinking-water</i>

Following oral or inhalation exposure, styrene is rapidly absorbed and widely distributed in the body, with a preference for lipid depots. It is metabolized to the active intermediate styrene-7,8-oxide, which is conjugated with glutathione or further metabolized. Metabolites are rapidly and almost completely excreted in urine. Styrene

has a low acute toxicity. In short-term toxicity studies in rats, impairment of glutathione transferase activity and reduced glutathione concentrations were observed. In in vitro tests, styrene has been shown to be mutagenic in the presence of metabolic activation only. In in vitro as well as in vivo studies, chromosomal aberrations have been observed, mostly at high doses of styrene. The reactive intermediate styrene-7,8-oxide is a direct-acting mutagen. In long-term studies, orally administered styrene increased the incidence of lung tumours in mice at high dose levels but had no carcinogenic effect in rats. Styrene-7,8-oxide was carcinogenic in rats after oral administration. IARC has classified styrene in Group 2B (possibly carcinogenic to humans). The available data suggest that the carcinogenicity of styrene is due to overloading of the detoxification mechanism for styrene-7,8-oxide (e.g. glutathione depletion).

Sulfate

Sulfates occur naturally in numerous minerals and are used commercially, principally in the chemical industry. They are discharged into water in industrial wastes and through atmospheric deposition; however, the highest levels usually occur in ground-water and are from natural sources. In general, the average daily intake of sulfate from drinking-water, air and food is approximately 500 mg, food being the major source. However, in areas with drinking-water supplies containing high levels of sulfate, drinking-water may constitute the principal source of intake.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	2003
Principal reference	WHO (2004) <i>Sulfate in drinking-water</i>

The existing data do not identify a level of sulfate in drinking-water that is likely to cause adverse human health effects. The data from a liquid diet study with piglets and from tap water studies with human volunteers indicate a laxative effect at concentrations of 1000–1200 mg/l, but no increase in diarrhoea, dehydration or weight loss.

No health-based guideline is proposed for sulfate. However, because of the gastrointestinal effects resulting from ingestion of drinking-water containing high sulfate levels, it is recommended that health authorities be notified of sources of drinking-water that contain sulfate concentrations in excess of 500 mg/l. The presence of sulfate in drinking-water may also cause noticeable taste (see [chapter 10](#)) and may contribute to the corrosion of distribution systems.

2,4,5-T

The half-lives for degradation of chlorophenoxy herbicides, including 2,4,5-T (CAS No. 93-76-5), also known as 2,4,5-trichlorophenoxyacetic acid, in the environment are in the order of several days. Chlorophenoxy herbicides are not often found in food.

Guideline value	0.009 mg/l (9 µg/l)
Occurrence	Chlorophenoxy herbicides not frequently found in drinking-water; when detected, concentrations usually no greater than a few micrograms per litre
TDI	3 µg/kg body weight, based on a NOAEL of 3 mg/kg body weight for reduced body weight gain, increased liver and kidney weights and renal toxicity in a 2-year study in rats, with an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 to take into consideration the suggested association between 2,4,5-T and soft tissue sarcoma and non-Hodgkin lymphoma in epidemiological studies)
Limit of detection	0.02 µg/l by GC with ECD
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1993
Principal reference	WHO (2003) <i>Chlorophenoxy herbicides (excluding 2,4-D and MCPA) in drinking-water</i>

Chlorophenoxy herbicides, as a group, have been classified in Group 2B (possibly carcinogenic to humans) by IARC. However, the available data from studies in exposed populations and experimental animals do not permit assessment of the carcinogenic potential to humans of any specific chlorophenoxy herbicide. Therefore, drinking-water guidelines for these compounds are based on a threshold approach for other toxic effects. The NOAEL for reproductive effects (reduced neonatal survival, decreased fertility, reduced relative liver weights and thymus weights in litters) of dioxin-free (< 0.03 µg/kg) 2,4,5-T in a three-generation reproduction study in rats is the same as the NOAEL for reduced body weight gain, increased liver and kidney weights and renal toxicity in a toxicity study in which rats were fed 2,4,5-T (practically free from dioxin contamination) in the diet for 2 years.

Terbuthylazine

Terbuthylazine (CAS No. 5915-41-3), or TBA, a herbicide that belongs to the chlorotriazine family, is used in both pre-emergence and post-emergence treatment of a variety of agricultural crops and in forestry. Degradation of TBA in natural water depends on the presence of sediments and biological activity.

Guideline value	0.007 mg/l (7 µg/l)
Occurrence	Concentrations in water seldom exceed 0.2 µg/l, although higher concentrations have been observed.
TDI	2.2 µg/kg body weight, based on a NOAEL of 0.22 mg/kg body weight for decreased body weight gain at the next higher dose in a 2-year toxicity/carcinogenicity study in rats, with an uncertainty factor of 100 (for interspecies and intraspecies variation)

Limit of detection	0.1 µg/l by HPLC with UV detection
Treatment performance	0.1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Assessment date	1998
Principal reference	WHO (2003) <i>Terbutylazine in drinking-water</i>

There is no evidence that TBA is carcinogenic or mutagenic. In long-term dietary studies in rats, effects on red blood cell parameters in females, an increased incidence of non-neoplastic lesions in the liver, lung, thyroid and testis and a slight decrease in body weight gain were observed.

Tetrachloroethene

Tetrachloroethene (PCE) has been used primarily as a solvent in dry-cleaning industries, and to a lesser extent as a degreasing solvent. Since the 1980s, as a result of regulations in North America, Europe and elsewhere, its use has substantially decreased. PCE is widespread in the environment and is found in trace amounts in water, aquatic organisms, air, foodstuffs and human tissue. The most relevant routes of exposure are considered to be inhalation of contaminated air and ingestion of contaminated drinking-water, particularly from groundwater sources. Poor handling and improper disposal of PCE in landfills have been the main causes of water contamination. Higher levels of PCE are expected in groundwater than in surface water because of the lack of volatilization that occurs from groundwater. However, drinking-water is not a major source of exposure, unless in the vicinity of a contaminated site.

Guideline value	0.1 mg/l (100 µg/l)
Occurrence	Concentrations in drinking-water are generally below 10 µg/l, although much higher concentrations have been detected in well water (23 mg/l) and in contaminated groundwater (1.5 mg/l)
TDI	16 µg/kg bw, based on a BMDL10 of 4.7 mg/kg bw per day for neurological effects (decreased colour vision) observed in an occupational study and applying an uncertainty factor of 300 (10 each for inter- and intra-species variability and 3 for extrapolation from an occupational study with intermittent exposure)
Limit of detection	0.002–0.008 µg/l by GC with ECD after liquid–liquid extraction; 0.02–0.05 µg/l by purge-and-trap capillary GC with PD and ECD in series; 0.036 µg/l by volatile organic compound analysis with GC-MS; and 0.05–0.14 µg/l by purge-and-trap capillary GC-MS

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Prevention and treatment	Source control, by improving handling and disposal practices, should be the priority action since PCE can persist in waters where volatilization cannot occur. Treatment of surface water sources is not needed because PCE volatilizes to the atmosphere. GAC, packed tower aeration and diffused aeration are effective central treatment technologies, although diffused aeration achieves lower removal efficiencies than packed tower aeration systems. Advanced oxidation processes may also be effective, but effectiveness depends on the physical and chemical properties of the water.
Guideline value derivation	
• allocation to water	20% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value is considered protective against both cancer and noncancer effects. In developing national standards, authorities may take into consideration the additional exposures through the dermal and inhalation routes from showering and bathing, especially in countries with low rates of ventilation in houses. Authorities may also consider overall exposure in developing national standards, noting the continued decline in human exposure to PCE by all probable exposure routes. Requirements for monitoring PCE in drinking-water regulations and standards should be limited to groundwater sources, where a possibility of PCE contamination is indicated. Monitoring is not needed for surface water sources because PCE volatilizes to the atmosphere.
Assessment date	2020
Principal reference	WHO (2020) <i>Tetrachloroethene in drinking-water</i>

IARC has classified PCE in Group 2A (probably carcinogenic to humans). PCE has been reported to produce liver tumours in male and female mice following inhalation, and there is some evidence of mononuclear cell leukaemia in male and female rats and kidney tumours in male rats. However, absorption, metabolic pathways, excretion and pattern of effects are similar for inhalation and oral exposure. The weight of evidence for PCE suggests that a nonlinear, nonmutagenic mode of action is predominant for liver tumours, and this effect is considered the most relevant end-point for cancer risk assessment of PCE exposure. Additionally, noncancer effects resulting from inhalation exposure, including evidence of neurotoxicity, were observed in human occupational studies and in laboratory animal studies. The nervous system is the most sensitive organ for noncancer effects. To protect against both cancer and noncancer effects, comparative points of departure were identified using benchmark dose modelling and physiologically based pharmacokinetic modelling (to account for the first-pass effects and inhalation-to-ingestion extrapolation). The most sensitive human-relevant effects were determined to be reductions in colour vision, and changes in cognitive function and reaction time in exposed workers; neurological effects were therefore the basis for the TDI derivation.

Toluene

Most toluene (in the form of benzene–toluene–ethylbenzene–xylene mixtures) is used in the blending of petrol. It is also used as a solvent and as a raw material in chemical production. The main exposure is via air. Exposure is increased by smoking and in traffic.

Guideline value	0.7 mg/l (700 µg/l)
Occurrence	Concentrations of a few micrograms per litre have been found in surface water, groundwater and drinking-water; point emissions can lead to higher concentrations in groundwater (up to 1 mg/l); it may also penetrate plastic pipes from contaminated soil
TDI	223 µg/kg body weight, based on a LOAEL of 312 mg/kg body weight per day for marginal hepatotoxic effects observed in a 13-week gavage study in mice, adjusting for daily dosing and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the short duration of the study and use of a LOAEL instead of a NOAEL)
Limit of detection	0.13 µg/l by GC with FID; 6 µg/l by GC-MS
Treatment performance	0.001 mg/l should be achievable using air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value exceeds the lowest reported odour threshold for toluene in water.
Assessment date	2003
Principal reference	WHO (2003) <i>Toluene in drinking-water</i>

Toluene is absorbed completely from the gastrointestinal tract and rapidly distributed in the body, with a preference for adipose tissue. Toluene is rapidly metabolized and, following conjugation, excreted predominantly in urine. With occupational exposure to toluene by inhalation, impairment of the central nervous system and irritation of mucous membranes are observed. The acute oral toxicity is low. Toluene exerts embryotoxic and fetotoxic effects, but there is no clear evidence of teratogenic activity in laboratory animals and humans. In long-term inhalation studies in rats and mice, there is no evidence for carcinogenicity of toluene. Genotoxicity tests in vitro were negative, whereas in vivo assays showed conflicting results with respect to chromosomal aberrations. IARC has concluded that there is inadequate evidence for the carcinogenicity of toluene in both experimental animals and humans and classified it as Group 3 (not classifiable as to its carcinogenicity to humans).

Total dissolved solids

Total dissolved solids (TDS) comprise inorganic salts (principally calcium, magnesium, potassium, sodium, bicarbonates, chlorides and sulfates) and small amounts of organic matter that are dissolved in water. TDS in drinking-water originates from

natural sources, sewage, urban runoff and industrial wastewater. Salts used for road de-icing in some countries may also contribute to the TDS content of drinking-water. Concentrations of TDS in water vary considerably in different geological regions owing to differences in the solubilities of minerals.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Total dissolved solids in drinking-water</i>

Reliable data on possible health effects associated with the ingestion of TDS in drinking-water are not available, and no health-based guideline value is proposed. However, the presence of high levels of TDS in drinking-water may be objectionable to consumers (see [chapter 10](#)).

Trichloroacetic acid

Chlorinated acetic acids are formed from organic material during water chlorination.

Guideline value	0.2 mg/l (200 µg/l)
Occurrence	Detected in groundwater and surface water distribution systems in the USA at mean concentrations of 5.3 µg/l (up to a maximum of 80 µg/l) and 16 µg/l (up to a maximum of 174 µg/l), respectively; maximum concentration (200 µg/l) measured in chlorinated water in Australia
TDI	32.5 µg/kg body weight, based on a NOAEL of 32.5 mg/kg body weight per day from a study in which decreased body weight, increased liver serum enzyme activity and liver histopathology were seen in rats exposed to trichloroacetate in drinking-water for 2 years, incorporating an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for database deficiencies, including the absence of a multigeneration reproductive study, the lack of a developmental study in a second species and the absence of full histopathological data in a second species)
Limit of detection	1 µg/l by GC-MS or GC-ECD
Treatment performance	Concentrations may be reduced by installing or optimizing coagulation to remove precursors or by controlling the pH during chlorination.
Guideline value derivation	
• allocation to water	20% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	A similar TDI for trichloroacetate was established by IPCS based on a NOAEL for hepatic toxicity in a long-term study in mice.
Assessment date	2003
Principal reference	WHO (2003) <i>Trichloroacetic acid in drinking-water</i>

Trichloroacetic acid has been shown to induce tumours in the liver of mice. It has given mixed results in in vitro assays for mutations and chromosomal aberrations and has been reported to cause chromosomal aberrations in in vivo studies. IARC has classified trichloroacetic acid in Group 3, not classifiable as to its carcinogenicity to humans. The weight of evidence indicates that trichloroacetic acid is not a genotoxic carcinogen.

Trichlorobenzenes (total)

Releases of trichlorobenzenes (TCBs) into the environment occur through their manufacture and use as industrial chemicals, chemical intermediates and solvents. TCBs are found in drinking-water, but rarely at levels above 1 µg/l. General population exposure will primarily result from air and food.

Reason for not establishing a guideline value	Occur in drinking-water at concentrations well below those of health concern, and health-based value would exceed lowest reported odour threshold
Assessment date	2003
Principal reference	WHO (2003) <i>Trichlorobenzenes in drinking-water</i>

The TCBs are of moderate acute toxicity. After short-term oral exposure, all three isomers show similar toxic effects, predominantly on the liver. Long-term toxicity and carcinogenicity studies via the oral route have not been carried out, but the data available suggest that all three isomers are non-genotoxic.

A health-based value of 20 µg/l can be calculated for total TCBs on the basis of a TDI of 7.7 µg/kg body weight, based on liver toxicity identified in a 13-week rat study, taking into consideration the short duration of the study. However, because TCBs occur at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value. It should be noted that the health-based value exceeds the lowest reported odour threshold in water.

1,1,1-Trichloroethane

1,1,1-Trichloroethane is widely used as a cleaning solvent for electrical equipment, as a solvent for adhesives, coatings and textile dyes and as a coolant and lubricant. It is found mainly in the atmosphere, although it is mobile in soils and readily migrates to groundwaters. 1,1,1-Trichloroethane has been found in only a small proportion of surface waters and groundwaters, usually at concentrations of less than 20 µg/l; higher concentrations (up to 150 µg/l) have been observed in a few instances. There appears to be increasing exposure to 1,1,1-trichloroethane from other sources.

Reason for not establishing a guideline value	Occur in drinking-water at concentrations well below those of health concern
Assessment date	2003
Principal reference	WHO (2003) <i>1,1,1-Trichloroethane in drinking-water</i>

1,1,1-Trichloroethane is rapidly absorbed from the lungs and gastrointestinal tract, but only small amounts—about 6% in humans and 3% in experimental animals—are metabolized. Exposure to high concentrations can lead to hepatic steatosis (fatty liver) in both humans and laboratory animals. In a well-conducted oral study in mice and rats, effects included reduced liver weight and changes in the kidney consistent with hyaline droplet neuropathy. IARC has placed 1,1,1-trichloroethane in Group 3. 1,1,1-Trichloroethane does not appear to be mutagenic.

A health-based value of 2 mg/l can be calculated for 1,1,1-trichloroethane on the basis of a TDI of 0.6 mg/kg body weight, based on changes in the kidney that were consistent with hyaline droplet nephropathy observed in a 13-week oral study in male rats, and taking into account the short duration of the study. However, because 1,1,1-trichloroethane occurs at concentrations well below those of health concern, it is not considered necessary to derive a formal guideline value.

Trichloroethene

Trichloroethene (TCE) is used primarily in metal degreasing. However, its use has been substantially declining since the 1990s as a result of increased environmental regulations on TCE emissions. It is emitted mainly to the atmosphere, but it may also be introduced into groundwater and, to a lesser extent, surface water in industrial effluents. Poor handling and improper disposal of TCE in landfills have been the main causes of water contamination. Higher levels of TCE are expected in groundwater than in surface water because of the lack of volatilization that occurs from groundwater. Therefore, the most relevant routes of exposure are considered inhalation of contaminated air and ingestion of contaminated drinking-water, particularly from groundwater sources.

Guideline value	0.008 mg/l (8 µg/l)
Occurrence	Typically present at low or undetectable concentrations in surface water (≤ 1 µg/l) due to high volatility and continued decline in TCE production. Concentrations may be higher (usually below 100 µg/l) in groundwater systems where volatilization and biodegradation are limited.
TDI	0.5 µg/kg bw, based on the TDI values derived from three key studies showing decreased thymus weight in female mice, increased incidence of developmental immunotoxicity in mice and increased incidence of fetal cardiac malformations in rats. The narrow range of TDIs among the three key studies was further supported by two other studies in rats evidencing renal effects. Where applicable, uncertainty factors were applied to the points of departure from each of the three studies to account for interspecies differences, intraspecies variation and use of a LOAEL instead of a NOAEL.
Limit of detection	0.01 µg/l by GC with ECD after liquid–liquid extraction; 0.01–3.0 µg/l by purge-and-trap capillary GC with PD or with PD and ECD in series; and 0.5 µg/l by purge-and-trap capillary GC-MS

Prevention and treatment	Source control is by improved handling and disposal practices. Treatment of surface water sources is not needed because TCE volatilizes to the atmosphere. For groundwater sources, aeration (packed tower aeration and air stripping) and GAC are effective central treatment technologies. Ozone and advanced oxidation processes may also be effective.
Guideline value derivation	• allocation to water 50% of TDI • weight 60 kg adult • consumption 2 litres/day
Additional comments	The guideline value is considered protective against both cancer and noncancer effects. In developing national standards, authorities may take into consideration the additional exposures through the dermal and inhalation routes from showering and bathing, especially in countries with low rates of ventilation in houses. Requirements for monitoring TCE in drinking-water regulations and standards should be limited to groundwater sources where a possibility of TCE contamination is indicated. Monitoring is not needed for surface water sources because TCE volatilizes to the atmosphere.
Assessment date	2020
Principal reference	WHO (2020) <i>Trichloroethene in drinking-water</i>

Available human and animal data after repeated exposure to TCE identify the kidney, liver, immune system, male reproductive system and developing fetus as potential targets of TCE toxicity and/or carcinogenicity. IARC has classified TCE as carcinogenic to humans (Group 1), concluding that sufficient epidemiological data are available for an association between exposure to TCE and human kidney cancer. Associations reported for liver cancer and non-Hodgkin lymphoma are characterized as limited, and evidence for other tumours is classified as inadequate. Multiple points of departure from different studies, rather than a single key study, were included in the derivation of the TDI. The overall TDI is based on three key studies with TDIs ranging from 0.3 to 0.6 µg/kg bw per day, and supported by other two studies in rats. Critical effects include increased incidence of heart malformations in rats (TDI of 0.6 µg/kg bw per day), decreased thymus weights in mice (TDI of 0.6 µg/kg bw per day) and developmental immunotoxicity (TDI of 0.37 µg/kg bw per day). The further supporting data in the database include studies reporting toxic nephropathy in rats (TDI of 0.3 µg/kg bw per day) and increased kidney weight in rats (TDI of 0.8 µg/kg bw per day by using route-to-route extrapolation from the inhalation study). Whenever possible, benchmark dose modelling and physiologically based pharmacokinetic modelling (to account for first-pass effects and inhalation-to-ingestion extrapolation) were applied.

Trifluralin

Trifluralin (CAS No. 1582-09-8) is a pre-emergence herbicide used in a number of crops. It has low water solubility and a high affinity for soil. However, biodegradation

and photodegradation processes may give rise to polar metabolites that may contaminate drinking-water sources. Although this compound is used in many countries, relatively few data are available concerning contamination of drinking-water.

Guideline value	0.02 mg/l (20 µg/l)
Occurrence	Not detected in the small number of drinking-water samples analysed; has been detected in surface water at concentrations above 0.5 µg/l and rarely in groundwater
TDI	7.5 µg/kg body weight, based on a NOAEL of 0.75 mg/kg body weight for mild hepatic effects in a 1year feeding study in dogs, with an uncertainty factor of 100 (for interspecies and intraspecies variation)
Limit of detection	0.05 µg/l by GC with nitrogen–phosphorus detection
Treatment performance	1 µg/l should be achievable using GAC
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	Authorities should note that some impure technical grades of trifluralin could contain potent carcinogenic compounds and therefore should not be used.
Assessment date	1993
Principal reference	WHO (2003) <i>Trifluralin in drinking-water</i>

Trifluralin of high purity does not possess mutagenic properties. Technical trifluralin of low purity may contain nitroso contaminants and has been found to be mutagenic. No evidence of carcinogenicity was demonstrated in a number of long-term toxicity/carcinogenicity studies with pure (99%) test material. IARC has assigned technical-grade trifluralin to Group 3 (not classifiable as to its carcinogenicity to humans).

Trihalomethanes (bromoform, bromodichloromethane, chloroform, dibromochloromethane)

THMs are formed in drinking-water primarily as a result of chlorination of organic matter present naturally in raw water supplies. The rate and degree of THM formation increase as a function of the chlorine and humic acid concentration, temperature, pH and bromide ion concentration. Chloroform is the most common THM and the principal disinfection by-product in chlorinated drinking-water. In the presence of bromides, brominated THMs are formed preferentially, and chloroform concentrations decrease proportionally. It is assumed that most THMs present in water are ultimately transferred to air as a result of their volatility. For chloroform, for example, individuals may be exposed during showering to elevated concentrations from chlorinated tap water. For the volatile THMs, approximately equal contributions to total exposure come from four areas: ingestion of drinking-water, inhalation of indoor air largely due to volatilization from drinking-water, inhalation and dermal exposure during shower-

ing or bathing and ingestion of food, with all but food exposure arising primarily from drinking-water. Indoor air exposure to the volatile THMs is particularly important in countries with low rates of ventilation in houses and high rates of showering and bathing.

Guideline values	<i>Chloroform</i> : 0.3 mg/l (300 µg/l)
	<i>Bromoform</i> : 0.1 mg/l (100 µg/l)
	<i>Dibromochloromethane (DBCM)</i> : 0.1 mg/l (100 µg/l)
	<i>Bromodichloromethane (BDCM)</i> : 0.06 mg/l (60 µg/l)
Occurrence	THMs are not expected to be found in raw water (unless near a pollution source), but are usually present in finished or chlorinated water; concentrations are generally below 100 µg/l; in most circumstances, chloroform is the dominant compound
TDIs	<i>Chloroform</i> : 15 µg/kg body weight, derived from the lower 95% confidence limit for 5% incidence of hepatic cysts, generated by physiologically based pharmacokinetic modelling, in dogs that ingested chloroform in toothpaste for 7.5 years, using an uncertainty factor of 25 (10 for intraspecies differences in toxicokinetics and toxicodynamics and 2.5 for differences in interspecies toxicodynamics)
	<i>Bromoform</i> : 17.9 µg/kg body weight, based on the absence of histopathological lesions in the liver in a well-conducted and well-documented 90-day study in rats, using an uncertainty factor of 1000 (100 for intraspecies and interspecies variation and 10 for possible carcinogenicity and short duration of exposure)
	<i>DBCM</i> : 21.4 µg/kg body weight, based on the absence of histopathological effects in the liver in a well-conducted and well-documented 90-day study in rats, using an uncertainty factor of 1000 (100 for intraspecies and interspecies variation and 10 for the short duration of the study); an additional uncertainty factor for potential carcinogenicity was not applied because of the questions regarding mouse liver tumours from corn oil vehicles and inconclusive evidence of genotoxicity
Basis of guideline value derivation	<i>BDCM</i> : Application of the linearized multistage model for the observed increases in incidence of kidney tumours in male mice observed in an NTP bioassay
Limit of detection	0.1–0.2 µg/l (method detection limits) by purge-and-trap and liquid–liquid extraction and direct aqueous injection in combination with a chromatographic system; 0.1 µg/l by GC-ECD; 2.2 µg/l by GC-MS
Treatment performance	Concentrations can be reduced by changes to disinfection practice (e.g. reducing organic THM precursors) or using air stripping.
Guideline value derivation	
• allocation to water	20% of TDI for bromoform and DBCM 75% of TDI for chloroform
• weight	60 kg adult
• consumption	2 litres/day

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Additional comments on THMs	For authorities wishing to establish a total THM standard to account for additive toxicity, the following fractionation approach could be taken: $\frac{C_{\text{bromoform}}}{GV_{\text{bromoform}}} + \frac{C_{\text{DBCM}}}{GV_{\text{DBCM}}} + \frac{C_{\text{BDCM}}}{GV_{\text{BDCM}}} + \frac{C_{\text{chloroform}}}{GV_{\text{chloroform}}} \leq 1$ where C = concentration and GV = guideline value. Authorities wishing to use a guideline value for total THMs should not simply add up the guideline values for the individual compounds in order to arrive at a standard. It is emphasized that adequate disinfection should never be compromised in attempting to meet guidelines for THMs. Nevertheless, in view of the potential link between adverse reproductive outcomes and THMs, particularly brominated THMs, it is recommended that THM levels in drinking-water be kept as low as practicable.
Additional comments on chloroform	In countries with low rates of ventilation in houses and high rates of showering and bathing, the guideline value could be lowered to account for the additional exposures from inhalation of indoor air largely due to volatilization from drinking-water and inhalation and dermal exposure during showering or bathing. The guideline value is based on the same study as in the third edition; the increase in value is primarily a result of an increase in the allocation of exposure in drinking-water from 50% to 75% to account for the fact that chloroform is used less now than it was in 1993 when the original guideline was developed.
Additional comments on BDCM	Although a health-based value of 21 µg/l is derived, the previous guideline value of 60 µg/l has been retained for two reasons: 1) both calculations were based on the same study, the only differences being the model and model assumptions used to derive the guideline value; there is therefore no scientific basis on which to justify a change in the guideline value; and 2) BDCM concentrations below 50 µg/l may be difficult to achieve using currently available technology without compromising the effectiveness of disinfection. As with chloroform, countries with low rates of ventilation and high rates of showering and bathing may wish to lower the guideline value to account for dermal and inhalation exposures, although, as noted above, concentrations below 50 µg/l may be difficult to achieve using currently available technology without compromising the effectiveness of disinfection. As BDCM was negative for carcinogenicity in a recent NTP bioassay in which it was dosed in drinking-water, exceedances of the guideline value are not likely to result in an increased risk of cancer.
Assessment date	2004
Principal references	IPCS (2000) <i>Disinfectants and disinfectant by-products</i> IPCS (2004) <i>Chloroform</i> USNTP (1987) <i>Toxicology and carcinogenesis studies of bromodichloromethane in F344/N rats and B6C3F1 mice (gavage studies)</i> WHO (2005) <i>Trihalomethanes in drinking-water</i>

Chloroform

The weight of evidence for genotoxicity of chloroform is considered negative. IARC has classified chloroform as possibly carcinogenic to humans (Group 2B) based on limited evidence of carcinogenicity in humans but sufficient evidence of carcinogenicity in experimental animals. The weight of evidence for liver tumours in mice is consistent with a threshold mechanism of induction. Although it is plausible that kidney tumours in rats may similarly be associated with a threshold mechanism, there are some limitations of the database in this regard. The most universally observed toxic effect of chloroform is damage to the centrilobular region of the liver. The severity of these effects per unit dose administered depends on the species, vehicle and method by which the chloroform is administered.

Bromoform

In an NTP bioassay, bromoform induced a small increase in relatively rare tumours of the large intestine in rats of both sexes but did not induce tumours in mice. Data from a variety of assays on the genotoxicity of bromoform are equivocal. IARC has classified bromoform in Group 3 (not classifiable as to its carcinogenicity to humans).

Dibromochloromethane

In an NTP bioassay, DBCM induced hepatic tumours in female mice and possibly in male mice but not in rats. The genotoxicity of DBCM has been studied in a number of assays, but the available data are considered inconclusive. IARC has classified DBCM in Group 3 (not classifiable as to its carcinogenicity to humans).

Bromodichloromethane

IARC has classified BDCM in Group 2B (possibly carcinogenic to humans). BDCM gave both positive and negative results in a variety of in vitro and in vivo genotoxicity assays. In an NTP bioassay, BDCM induced renal adenomas and adenocarcinomas in both sexes of rats and male mice, rare tumours of the large intestine (adenomatous polyps and adenocarcinomas) in both sexes of rats and hepatocellular adenomas and adenocarcinomas in female mice. However, BDCM was negative for carcinogenicity in a recent NTP bioassay in which it was dosed in drinking-water. Exposure to BDCM has also been linked to a possible increase in reproductive effects (increased risk for spontaneous abortion or stillbirth).

Uranium

Uranium is widespread in nature, occurring in granites and various other mineral deposits. It is used mainly as fuel in nuclear power stations. Uranium is present in the environment as a result of leaching from natural deposits, release in mill tailings, emissions from the nuclear industry, the combustion of coal and other fuels and the use of phosphate fertilizers that contain uranium. Intake of uranium through air is low, and it appears that intake through food is between 1 and 4 µg/day. Intake through drinking-water is normally extremely low; however, in circumstances in which uranium is present in a drinking-water source, the majority of intake can be through drinking-water.

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Provisional guideline value	0.03 mg/l (30 µg/l)
	The guideline value is designated as provisional because of scientific uncertainties surrounding uranium toxicity.
Occurrence	Levels in drinking-water are generally less than 1 µg/l, although concentrations as high as 700 µg/l have been measured in private supplies.
TDI	60 µg, derived from the lower 95% confidence limit on the 95th percentile uranium exposure distribution in a study from Finland, using an uncertainty factor of 10 for intraspecies variation
Limit of detection	0.01 µg/l by ICP-MS; 0.1 µg/l by solid fluorimetry with either laser excitation or UV light; 0.2 µg/l by ICP using adsorption with chelating resin
Treatment performance	1 µg/l should be achievable using conventional treatment (e.g. coagulation or ion exchange)
Guideline value derivation	
• consumption	2 litres/day
Additional comments	Where supplies exceed 30 µg/l, it is important that precipitate action be avoided. Consideration should first be given to exposure from all sources and the availability of alternative safe sources.
	Only chemical, not radiological, aspects of uranium toxicity have been addressed here.
Assessment date	2003, revised in 2011
Principal reference	WHO (2012) <i>Uranium in drinking-water</i>

There are insufficient data regarding the carcinogenicity of uranium in humans and experimental animals. Nephritis is the primary chemically induced effect of uranium in humans. Little information is available on the chronic health effects of exposure to environmental uranium in humans. A number of epidemiological studies of populations exposed to uranium in drinking-water have shown a correlation with alkaline phosphatase and β -microglobulin in urine along with modest alterations in proximal tubular function. However, the actual measurements were still within the normal physiological range, and these findings are not consistent between studies.

No clear no-effect concentration has emerged from the human studies to date. This is not surprising, as most of the study populations are quite small, and there is substantial normal variation in the measured parameters in the human population. However, the overall indications are that there is no clear evidence of effects below an exposure concentration of 30 µg/l. In fact, the evidence for effects on the kidney, which appears to be the most sensitive organ, is equivocal until much higher exposure concentrations.

The provisional guideline value of 30 µg/l, which is derived from new epidemiological studies on populations exposed to high uranium concentrations, replaces the previous value derived from experimental animal studies and designated as provisional on the basis of uncertainties regarding the toxicology and epidemiology of uranium as well as difficulties concerning its technical achievability in smaller supplies. It is noted that studies on human populations, when available and of good quality, are

the preferred source of health-related information to be used in deriving guideline values.

Vinyl chloride

Vinyl chloride is used primarily for the production of PVC. Owing to its high volatility, vinyl chloride has rarely been detected in surface waters, except in contaminated areas. Unplasticized PVC is increasingly being used in some countries for water mains supplies. Migration of vinyl chloride monomer from unplasticized PVC is a possible source of vinyl chloride in drinking-water. It appears that inhalation is the most important route of vinyl chloride intake, although drinking-water may contribute a substantial portion of daily intake where PVC piping with a high residual content of vinyl chloride monomer is used in the distribution network. Vinyl chloride has been reported in groundwater as a degradation product of the chlorinated solvents trichloroethene and tetrachloroethene.

Guideline value	0.0003 mg/l (0.3 µg/l)
Occurrence	Rarely detected in surface waters, the concentrations measured generally not exceeding 10 µg/l; much higher concentrations found in groundwater and well water in contaminated areas; concentrations up to 10 µg/l detected in drinking-water
Basis of guideline value derivation	Application of a linear extrapolation by drawing a straight line between the dose, determined using a pharmacokinetic model, resulting in tumours in 10% of animals in rat bioassays involving oral exposure and the origin (zero dose), determining the value associated with the upper-bound risk of 10^{-5} and assuming a doubling of the risk for exposure from birth
Limit of detection	0.01 µg/l by GC-ECD or GC-FID with MS for confirmation
Treatment performance	0.001 mg/l should be achievable using air stripping
Additional comments	The results of the linear extrapolation are nearly identical to those derived using the linearized multistage model. As vinyl chloride is a known human carcinogen, exposure to this compound should be avoided as far as practicable, and levels should be kept as low as technically feasible. Vinyl chloride is primarily of concern as a potential contaminant from some grades of PVC pipe and is best controlled by specification of material quality.
Assessment date	2003
Principal references	IPCS (1999) <i>Vinyl chloride</i> WHO (2004) <i>Vinyl chloride in drinking-water</i>

There is sufficient evidence of the carcinogenicity of vinyl chloride in humans from industrial populations exposed to high concentrations via the inhalation route, and IARC has classified vinyl chloride in Group 1 (carcinogenic to humans). Studies of workers employed in the vinyl chloride industry showed a marked exposure–response

for all liver cancers, angiosarcomas and hepatocellular carcinoma, but no strong relationship between cumulative vinyl chloride exposure and other cancers. Experimental animal data show vinyl chloride to be a multisite carcinogen. When administered orally or by inhalation to mice, rats and hamsters, it produced tumours in the mammary gland, lungs, Zymbal gland and skin, as well as angiosarcomas of the liver and other sites. Evidence indicates that vinyl chloride metabolites are genotoxic, interacting directly with DNA. DNA adducts formed by the reaction of DNA with a vinyl chloride metabolite have also been identified. Occupational exposure has resulted in chromosomal aberrations, micronuclei and sister chromatid exchanges; response levels were correlated with exposure levels.

Xylenes

Xylenes are used in blending petrol, as a solvent and as a chemical intermediate. They are released to the environment largely via air. Exposure to xylenes is mainly from air, and exposure is increased by smoking.

Guideline value	0.5 mg/l (500 µg/l)
Occurrence	Concentrations of up to 8 µg/l have been reported in surface water, groundwater and drinking-water; levels of a few milligrams per litre were found in groundwater polluted by point emissions; xylenes can also penetrate plastic pipe from contaminated soil
TDI	179 µg/kg body weight, based on a NOAEL of 250 mg/kg body weight per day for decreased body weight in a 103-week gavage study in rats, adjusting for daily dosing and using an uncertainty factor of 1000 (100 for interspecies and intraspecies variation and 10 for the limited toxicological end-points)
Limit of detection	0.1 µg/l by GC-MS; 1 µg/l by GC-FID
Treatment performance	0.005 mg/l should be achievable using GAC or air stripping
Guideline value derivation	
• allocation to water	10% of TDI
• weight	60 kg adult
• consumption	2 litres/day
Additional comments	The guideline value exceeds the lowest reported odour threshold for xylenes in drinking-water.
Assessment date	1993
Principal reference	WHO (2003) <i>Xylenes in drinking-water</i>

Xylenes are rapidly absorbed by inhalation. Data on oral exposure are lacking. Xylenes are rapidly distributed in the body, predominantly in adipose tissue. They are almost completely metabolized and excreted in urine. The acute oral toxicity of xylenes is low. No convincing evidence for teratogenicity has been found. Long-term carcinogenicity studies have shown no evidence for carcinogenicity. In vitro as well as in vivo mutagenicity tests have proved negative.

Zinc

Zinc is an essential trace element found in virtually all food and potable water in the form of salts or organic complexes. The diet is normally the principal source of zinc. Although levels of zinc in surface water and groundwater normally do not exceed 0.01 and 0.05 mg/l, respectively, concentrations in tap water can be much higher as a result of dissolution of zinc from pipes.

Reason for not establishing a guideline value	Not of health concern at levels found in drinking-water
Additional comments	May affect acceptability of drinking-water
Assessment date	1993
Principal reference	WHO (2003) <i>Zinc in drinking-water</i>

In 1982, JECFA proposed a PMTDI for zinc of 1 mg/kg body weight. The daily requirement for adult men is 15–20 mg/day. It was considered that, taking into account recent studies on humans, the derivation of a formal guideline value is not required at this time. However, drinking-water containing zinc at levels above 3 mg/l may not be acceptable to consumers (see [chapter 10](#)).

ANNEX 1

Supporting documentation to the Guidelines¹

The *Guidelines for drinking-water quality* are accompanied by separate texts that provide background information substantiating the derivation of the Guidelines and providing guidance on good practice towards effective implementation (<https://www.who.int/teams/environment-climate-change-and-health/water-sanitation-and-health/water-safety-and-quality/publications>).

Published supporting documents

A field guide to improving small drinking-water supplies: water safety planning for rural communities

Published in 2022 by the World Health Organization Regional Office for Europe
Contains short explanations of the water safety planning process and includes a range of practical templates to support water safety plan (WSP) development and implementation in rural water supplies.

<https://iris.who.int/handle/10665/363510>

A guide to equitable water safety planning: ensuring no one is left behind

Published in 2019 by the World Health Organization
Provides step-by-step guidance and good practice examples on how the WSP approach can bring tangible improvements in water quality and availability for all users.

<https://iris.who.int/handle/10665/311148>

A practical guide to auditing water safety plans

Published in 2015 by the World Health Organization
Provides guidance and tools on WSP auditing – a key surveillance activity – to support the continuous improvement and sustainability of WSPs, including practical examples from diverse settings.

<https://iris.who.int/handle/10665/204280>

¹ The web links given in this annex were current as of 13 March 2026.

A toolkit for monitoring and evaluating household water treatment and safe storage programmes
Published in 2012 by the World Health Organization and United Nations Children's Fund
Provides a framework to support planning for, conducting and utilizing monitoring and evaluation data from, household water treatment and safe storage programmes.
<https://iris.who.int/handle/10665/76568>

Alternative drinking-water disinfectants: bromine, iodine and silver
Published in 2018 by the World Health Organization
A review of the evidence on the efficacy and toxicity of bromine, iodine and silver as drinking-water disinfectants.
<https://iris.who.int/handle/10665/260545>

Arsenic primer: guidance on the investigation and mitigation of arsenic contamination
Published in 2018 by United Nations Children's Fund and the World Health Organization
Provides practical advice for UN agencies, government counterparts and development workers responding to the challenge of arsenic contamination of drinking-water.
<https://www.who.int/publications/m/item/arsenic-primer>

Assessing microbial safety of drinking water: improving approaches and methods
Edited by A. Dufour et al.
Published in 2003 by IWA Publishing on behalf of the World Health Organization and the Organisation for Economic Co-operation and Development
A state-of-the-art review of approaches and methods used in assessing the microbial safety of drinking-water.
<https://iris.who.int/handle/10665/42790>

Boil water
Published in 2015 by the World Health Organization
Provides the scientific basis for the efficacy of boiling water
<https://iris.who.int/handle/10665/155821>

Calcium and magnesium in drinking-water: public health significance
Edited by J. Cotruvo and J. Bartram
Published in 2009 by the World Health Organization
A review of the contribution of drinking-water to total daily intake of calcium and magnesium, and an assessment of possible health benefits, including reducing cardiovascular disease mortality and osteoporosis.
<https://iris.who.int/handle/10665/43836>

Chemical mixtures in source water and drinking-water
Published in 2017 by the World Health Organization
Provides an overview of available tools and practical recommendations to support the assessment and management of risks to human health associated with chemical mixtures in drinking-water and its sources.
<https://iris.who.int/handle/10665/255543>

Chemical safety of drinking-water: assessing priorities for risk management
T. Thompson et al.
Published in 2007 by the World Health Organization
A tool to assist in undertaking a systematic assessment of water supply systems to prioritize, control or eliminate chemicals in drinking-water.
<https://iris.who.int/handle/10665/43285>

Climate-resilient water safety plans: managing health risks associated with climate variability and change

Published in 2017 by the World Health Organization

Provides guidance on how climate considerations can be integrated into water safety planning to provide greater resilience to the current and predicted impacts of climate change and variability on water supplies.

<https://iris.who.int/handle/10665/258722>

Compendium of drinking-water systems and technologies from source to consumer

Published in 2025 by the World Health Organization

Provides an overview of drinking-water systems and technologies to support informed selection of context appropriate technologies.

<https://iris.who.int/handle/10665/383116>

Developing drinking-water quality regulations and standards: general guidance with a special focus on countries with limited resources

Published in 2018 by the World Health Organization

Provides practical guidance to support the development or revision of customized national or subnational drinking-water quality regulations and standards.

<https://iris.who.int/handle/10665/272969>

Domestic water quantity, service level and health, 2nd edition

G. Howard et al.

Published in 2020 by the World Health Organization

Reviews the evidence about the relationships between water quantity, water accessibility and health, providing updated guidance, including recommended targets on domestic water supply to ensure beneficial health outcomes.

<https://iris.who.int/handle/10665/338044>

Evaluating household water treatment options: health-based targets and microbiological performance specifications

J. Brown and M. Sobsey

Published in 2011 by the World Health Organization

Establishes health-based targets and testing protocols for point-of-use water treatment approaches, including to inform development of country certification programmes.

<https://iris.who.int/handle/10665/44693>

Evaluation of the H₂S method for detection of faecal contamination of drinking water

M. Sobsey and F. Pfaender

Published in 2002 by the World Health Organization

Provides the scientific basis, validity, available data and other information concerning the use of “H₂S tests” as measures or indicators of faecal contamination in drinking-water.

<https://iris.who.int/handle/10665/205188>

Fluoride in drinking-water

J.K. Fawell et al.

Published in 2006 by IWA Publishing on behalf of the World Health Organization

Provides information on the occurrence of fluoride in drinking-water, its health effects, ways of reducing excess levels and methods for analysis of fluoride in water.

<https://iris.who.int/handle/10665/43514>

Guide to hygiene and sanitation in aviation, 3rd edition. Module 1: water; Module 2: cleaning and disinfection of facilities

Published in 2009 by the World Health Organization

Addresses water and cleaning and disinfection of facilities with the ultimate goal of assisting all types of airport and aircraft operators and other responsible bodies in achieving high standards of hygiene and sanitation, to protect travellers.

<https://iris.who.int/handle/10665/44164>

Guide to ship sanitation, 3rd edition

Published in 2011 by the World Health Organization

Presents the public health significance of ships in terms of disease and highlights the importance of applying appropriate control measures.

<https://iris.who.int/handle/10665/43193>

Guidelines for drinking-water quality: small water supplies

Published in 2024 by the World Health Organization

A companion publication to the *Guidelines for drinking-water quality* – offers guidance on applying the framework for safe drinking-water in the context of small water supplies ranging from private household wells to community piped supplies.

<https://iris.who.int/handle/10665/375822>

Hazard characterization for pathogens in food and water: guidelines

Published in 2003 by the Food and Agriculture Organization of the United Nations and the World Health Organization

A practical framework and structured approach for the characterization of microbial hazards in food and water, to assist governmental and research scientists.

<https://iris.who.int/handle/10665/42693>

Health aspects of plumbing

Published in 2006 by the World Health Organization and the World Plumbing Council

A description of the processes involved in the design, installation and maintenance of effective plumbing systems and consideration of the microbial, chemical, physical and financial concerns associated with plumbing.

<https://iris.who.int/handle/10665/43423>

Heterotrophic plate counts and drinking-water safety: the significance of HPCs for water quality and human health

Edited by J. Bartram et al.

Published in 2003 by IWA Publishing on behalf of the World Health Organization

Assessment of the role of the heterotrophic plate count measurement in drinking-water safety management.

<https://iris.who.int/handle/10665/42612>

Lead in drinking-water: Health risks, monitoring and corrective actions: technical brief

Published in 2022 by the World Health Organization

Guidance on the assessment and management of lead contamination in drinking-water supplies, including actions to take when elevated lead levels are detected.

<https://iris.who.int/handle/10665/361821>

Legionella and the prevention of legionellosis

Edited by J. Bartram et al.

Published in 2007 by the World Health Organization

An overview of the sources, ecology and laboratory detection of *Legionella* bacteria, risk assessment and risk management of susceptible environments, the necessary measures to prevent or adequately control the risks and the policies and practices for outbreak management.

<https://iris.who.int/handle/10665/43233>

Management of cyanobacteria in drinking-water supplies: information for regulators and water suppliers, 2nd edition

Published in 2024 by the World Health Organization

Guidance on the assessment and management of cyanobacteria in drinking-water supplies including measures to prevent the formation of cyanobacterial blooms and options to manage such blooms and their toxins when they occur.

<https://iris.who.int/handle/10665/379618>

Management of radioactivity in drinking-water

Published in 2018 by the World Health Organization

Guidance to support radionuclide management in drinking-water in emergency and non-emergency situations.

<https://iris.who.int/handle/10665/272995>

Managing water in the home: accelerated health gains from improved water supply

M. Sobsey

Published in 2002 by the World Health Organization

A review of the various methods and systems for household water collection, treatment and storage.

<https://iris.who.int/handle/10665/67319>

Microplastics in drinking-water

Published in 2019 by the World Health Organization

Critically examines the evidence related to the occurrence of microplastics in the water cycle (including both tap and bottled drinking-water and its sources), the potential health impacts from microplastic exposure and the removal of microplastics during wastewater and drinking-water treatment.

<https://iris.who.int/handle/10665/326499>

Pathogenic mycobacteria in water: a guide to public health consequences, monitoring and management

Edited by J. Bartram et al.

Published in 2004 by IWA Publishing on behalf of the World Health Organization

A description of the distribution, routes of transmission and infection, and guidance on the control of pathogenic environmental mycobacteria in water and other parts of the environment.

<https://iris.who.int/handle/10665/42854>

Pharmaceuticals in drinking-water

Published in 2012 by the World Health Organization

Provides evidence-based, practical guidance and recommendations for managing pharmaceuticals in drinking-water.

<https://iris.who.int/handle/10665/44630>

Potable reuse: guidance for producing safe drinking-water

Published in 2017 by the World Health Organization

Highlights specific aspects of potable reuse including the quality of source wastewaters and public acceptance and provides guidance on appropriate risk assessment and risk management procedures in order to ensure the safety of drinking-water from municipal wastewater.

<https://iris.who.int/handle/10665/258715>

Protecting groundwater for health: managing the quality of drinking-water sources

Edited by O. Schmoll et al.

Published in 2006 by the World Health Organization

An analysis of the hazards to groundwater quality, and the risk they may present to a specific supply. This is a tool for developing strategies to protect groundwater for health by managing the quality of drinking-water sources.

<https://iris.who.int/handle/10665/43186>

Protecting surface water for health: identifying, assessing and managing drinking-water quality risks in surface water catchments

Published in 2016 by the World Health Organization

Provides a structured approach to understanding surface waters and their catchments to support the identification, assessment and prioritization of the risks, and the development of management strategies for their control, as a basis for providing safe drinking-water

<https://iris.who.int/handle/10665/246196>

Quantifying public health risk in the WHO guidelines for drinking-water quality: a burden of disease approach

A.H. Havelaar and J.M. Melse

Published in 2003 by the National Institute for Public Health and the Environment of the Kingdom of the Netherlands

A discussion paper on the concepts and methodology of disability-adjusted life years as a common public health metric and its usefulness for drinking-water quality.

<https://www.who.int/publications/m/item/quantifying-public-health-risks-in-the-who-guidelines-for-drinking-water-quality>

Quantitative microbial risk assessment: application for water safety management

Published in 2016 by the World Health Organization

Synthesizes the current knowledge on quantitative microbial risk assessment to facilitate its application in the practice of water supply, water reuse and water recreation to support the management of risks associated with faecal pathogens in the water-related context.

<https://iris.who.int/handle/10665/246195>

Review of latest available evidence on potential transmission of avian influenza (H5N1) through water and sewage and ways to reduce the risks to human health

Published in 2006 by the World Health Organization

A summary of the latest available studies and findings on avian influenza (H5N1) pertaining to water resources, water supplies, sanitation (human excreta, sewerage systems and health-care waste) and hygiene.

<https://iris.who.int/handle/10665/204275>

Risk assessment of Cryptosporidium in drinking water

G. Medema et al.

Published in 2009 by the World Health Organization

A text supporting the *Guidelines for drinking-water quality* by providing further data on *Cryptosporidium* to assist country authorities in setting health-based targets and water suppliers in determining required performance of water treatment processes as part of a system-specific WSP.

<https://iris.who.int/handle/10665/70117>

Safe drinking-water from desalination

Published in 2011 by the World Health Organization

Highlights the principal health risks related to different desalination processes and provides guidance on appropriate risk assessment and risk management procedures in order to ensure the safety of desalinated drinking-water.

<https://iris.who.int/handle/10665/70621>

Safe piped water: managing microbial water quality in piped distribution systems

Edited by R. Ainsworth

Published in 2004 by IWA Publishing on behalf of the World Health Organization

A report on microbial contaminants and growth of microorganisms in distribution networks and the practices that contribute to ensuring drinking-water safety in piped distribution systems.

<https://iris.who.int/handle/10665/42785>

Sanitary inspection packages – A supporting tool for the Guidelines for drinking-water quality: small water supplies

Published in 2024 by the World Health Organization

Simple field checklists and associated guidance to support proactive risk management by water suppliers and drinking-water quality surveillance across a range of water delivery scenarios.

<https://iris.who.int/handle/10665/375824>

Scaling up household water treatment among low-income populations

T. Clasen

Published in 2009 by the World Health Organization

Examines the evidence to date regarding the scalability of household water treatment systems. Its primary aims are to review the development and evolution of leading household water treatment technologies in their efforts to achieve scale, identify the main constraints that they have encountered and recommend ways forward.

<https://iris.who.int/handle/10665/70049>

Think big, start small and scale up: a roadmap to support country-level implementation of water safety plans

Published in 2010 by the World Health Organization

Guidance on introducing and scaling up the implementation of WSPs within a country, presented as a number of steps.

<https://www.who.int/publications/m/item/think-big-start-small-scale-up>

Toxic cyanobacteria in water, 2nd edition

Edited by I. Chorus and M. Welker

Published in 2021 by CRC Press on behalf of the World Health Organization

Provides background information and practical guidance to support the assessment and management of the risks from cyanobacteria and their toxins in drinking-water and recreational water bodies.

<https://www.who.int/publications/m/item/toxic-cyanobacteria-in-water---second-edition>

Upgrading water treatment plants

E.G. Wagner and R.G. Pinheiro

Published in 2001 by Spon Press on behalf of the World Health Organization

A practical guide to improving the performance of water treatment plants.

<https://iris.who.int/handle/10665/42409>

Water quality and health – Review of turbidity: information for regulators and operators of water supplies

Published in 2017 by the World Health Organization

Provides information on the uses and significance of turbidity, and is intended for regulators and operators of drinking-water supplies.

<https://iris.who.int/handle/10665/254631>

Water quality—Guidelines, standards and health: assessment of risk and risk management for water-related infectious disease

Edited by L. Fewtrell and J. Bartram

Published in 2001 by IWA Publishing on behalf of the World Health Organization

Guidance on issues relating to microbial water quality and health, including environmental and public health scientists, water scientists, policy-makers and those responsible for developing standards and regulations.

<https://iris.who.int/handle/10665/42442>

Water safety in buildings

Edited by D. Cunliffe et al.

Published in 2011 by the World Health Organization

Guidance for managing water supplies in buildings (e.g. hospitals, schools, care facilities, hotels) where people may drink water; use water for food preparation; wash, shower, swim or use water for other recreational activities; or be exposed to aerosols produced by water-using devices, such as cooling towers.

<https://iris.who.int/handle/10665/76145>

Water safety in distribution systems

Published in 2014 by the World Health Organization

A reference tool to help water suppliers and regulators who are familiar with the WSP approach to enhance risk assessment and management and investment planning for their water distribution systems.

<https://iris.who.int/handle/10665/204422>

Water safety plan manual: step-by-step risk management for drinking-water suppliers, 2nd edition

Published in 2023 by the World Health Organization

Guidance on developing and implementing a WSP, primarily intended for piped water supplies that are professionally managed by a water supplier or equivalent management entity.

<https://iris.who.int/handle/10665/366148>

ANNEX 1. SUPPORTING DOCUMENTATION TO THE GUIDELINES

Water safety planning for small community water supplies: step-by-step risk management guidance for drinking-water supplies in small communities

Published in 2012 by the World Health Organization

Guidance for the planning, design and implementation of WSPs by and for rural and remote communities, including communities with piped schemes, those served by point sources and community-wide water supply services using various technical options.

<https://iris.who.int/handle/10665/75145>

Water safety plans: managing drinking-water quality from catchment to consumer

A. Davison et al.

Published in 2005 by the World Health Organization

Guidance on improved strategies for the preventive management, control and monitoring of drinking-water quality.

<https://iris.who.int/handle/10665/42890>

Water treatment and pathogen control: process efficiency in achieving safe drinking-water

M.W. LeChevallier and K.K. Au

Published in 2004 by IWA Publishing on behalf of the World Health Organization

A critical analysis of the removal and inactivation of pathogenic microbes in water to aid the water quality specialist and design engineer in making decisions regarding microbial water quality.

<https://iris.who.int/handle/10665/42796>

Waterborne zoonoses: identification, causes and control

Edited by J.A. Cotruvo et al.

Published in 2004 by IWA Publishing on behalf of the World Health Organization

An invaluable tool for all professionals concerned with assessing and managing water-borne zoonoses, which are diseases caused by microorganisms of animal origin that also infect humans.

<https://iris.who.int/handle/10665/42977>

ANNEX 2

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¹ This list includes all references cited in the text, except for the supporting documents to the Guidelines, which are listed separately in [Annex 1](#), and the selected bibliographic references in [chapter 11](#), which are cited following each microbial fact sheet in that chapter.

² The web links given in this annex were current as of 13 March 2026.

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Chapter 12¹

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ANNEX 3

Chemical summary tables

Table A3.1 Chemicals excluded from guideline value derivation

Chemical	Reason for exclusion
Amitraz	Degrades rapidly in the environment and is not expected to occur at measurable concentrations in drinking-water supplies
Chlorobenzilate	Unlikely to occur in drinking-water
Chlorothalonil	Unlikely to occur in drinking-water
Cypermethrin	Unlikely to occur in drinking-water
Deltamethrin	Unlikely to occur in drinking-water
Diazinon	Unlikely to occur in drinking-water
Dinoseb	Unlikely to occur in drinking-water
Ethylene thiourea	Unlikely to occur in drinking-water
Fenamiphos	Unlikely to occur in drinking-water
Formothion	Unlikely to occur in drinking-water
Hexachlorocyclohexanes (mixed isomers)	Unlikely to occur in drinking-water
MCPB ^a	Unlikely to occur in drinking-water
Methamidophos	Unlikely to occur in drinking-water
Methomyl	Unlikely to occur in drinking-water
Mirex	Unlikely to occur in drinking-water
Monocrotophos	Has been withdrawn from use in many countries and is unlikely to occur in drinking-water
Oxamyl	Unlikely to occur in drinking-water
Phorate	Unlikely to occur in drinking-water
Propoxur	Unlikely to occur in drinking-water
Pyridate	Not persistent and only rarely found in drinking-water
Quintozene	Unlikely to occur in drinking-water
Toxaphene	Unlikely to occur in drinking-water
Triazophos	Unlikely to occur in drinking-water
Trichlorfon	Unlikely to occur in drinking-water

^a 4-(4-chloro-o-tolyloxy)butyric acid.

ANNEX 3. CHEMICAL SUMMARY TABLES

Table A3.2 Chemicals for which guideline values have not been established^a

Chemical	Reason for not establishing a guideline value
Aluminium	The health-based value exceeds practicable levels based on optimization of the coagulation process in drinking-water plants using aluminium-based coagulants: 0.1 mg/l or less in large supplies and 0.2 mg/l or less in small supplies
Ammonia	Occurs in drinking-water at concentrations well below those of health concern
Anatoxins (cyanobacterial toxins)	Available data inadequate to permit derivation of a health-based guideline value
Asbestos	No consistent evidence that ingested asbestos is hazardous to health and available data inadequate to permit derivation of health-based guideline value
Bentazone	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Beryllium	Rarely found in drinking-water at concentrations of health concern
Bromide	Occurs in drinking-water at concentrations well below those of health concern
Bromochloroacetate	Available data inadequate to permit derivation of health-based guideline value
Bromochloroacetonitrile	Available data inadequate to permit derivation of health-based guideline value
<i>Bacillus thuringiensis israelensis</i> (Bti)	Not considered appropriate to set guideline values for pesticides used for vector control in drinking-water
Carbaryl	Occurs in drinking-water at concentrations well below those of health concern
Chloral hydrate	Occurs in drinking-water at concentrations well below those of health concern
Chloride	Not of health concern at levels found in drinking-water ^b
Chlorine dioxide	Reduced primarily to chlorite, chlorate and chloride in drinking-water, and to chlorite and chloride upon ingestion; the provisional guideline values for chlorite and chlorate are protective for potential toxicity from chlorine dioxide
Chloroacetones	Available data inadequate to permit derivation of health-based guideline values for any of the chloroacetones
2-Chlorophenol	Available data inadequate to permit derivation of health-based guideline value
Chloropicrin	Available data inadequate to permit derivation of health-based guideline value
Cyanide	Occurs in drinking-water at concentrations well below those of health concern, except in emergency situations following a spill to a water source
Cyanogen chloride	Occurs in drinking-water at concentrations well below those of health concern
Dibromoacetate	Available data inadequate to permit derivation of health-based guideline value

Table A3.2 (continued)

Chemical	Reason for not establishing a guideline value
Dichloramine	Available data inadequate to permit derivation of health-based guideline value
1,3-Dichlorobenzene	Available data inadequate to permit derivation of health-based guideline value
1,1-Dichloroethane	Available data inadequate to permit derivation of health-based guideline value
1,1-Dichloroethene	Occurs in drinking-water at concentrations well below those of health concern
2,4-Dichlorophenol	Available data inadequate to permit derivation of health-based guideline value
1,3-Dichloropropane	Available data inadequate to permit derivation of health-based guideline value
Dichlorvos	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Dicofol	Unlikely to occur in drinking-water or drinking-water sources ^c
Di(2-ethylhexyl)adipate	Occurs in drinking-water at concentrations well below those of health concern
Diflubenzuron	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Diquat	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Endosulfan	Occurs in drinking-water at concentrations well below those of health concern
Fenitrothion	Occurs in drinking-water at concentrations well below those of health concern
Fluoranthene ^d	Occurs in drinking-water at concentrations well below those of health concern
Formaldehyde	Occurs in drinking-water at concentrations well below those of health concern
Glyphosate and AMPA ^e	Occur in drinking-water at concentrations well below those of health concern
Hardness	Not of health concern at levels found in drinking-water ^b
Heptachlor and heptachlor epoxide	Occur in drinking-water at concentrations well below those of health concern
Hexachlorobenzene	Occurs in drinking-water at concentrations well below those of health concern
Hydrogen sulfide	Not of health concern at levels found in drinking-water ^b
Inorganic tin	Occurs in drinking-water at concentrations well below those of health concern
Iodine	Available data inadequate to permit derivation of health-based guideline value. Additionally, occurrence in drinking-water is usually low. Although higher levels of exposure may occur when iodine is used as a drinking-water disinfectant at the point of use, extended periods of exposure to iodine through water disinfection are unlikely

Table A3.2 (continued)

Chemical	Reason for not establishing a guideline value
Iron	Not of health concern at levels causing acceptability problems in drinking-water ^a
Malathion	Occurs in drinking-water at concentrations well below those of health concern
MCPA ^f	Occurs in drinking-water or drinking-water sources at concentrations well below those of health concern
Methoprene	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Methyl parathion	Occurs in drinking-water at concentrations well below those of health concern
Methyl <i>tertiary</i> -butyl ether (MTBE)	Any guideline that would be derived would be significantly higher than concentrations at which MTBE would be detected by odour
Molybdenum	Occurs in drinking-water at concentrations well below those of health concern
Monobromoacetate	Available data inadequate to permit derivation of health-based guideline value
Monochlorobenzene	Occurs in drinking-water at concentrations well below those of health concern, and health-based value would far exceed lowest reported taste and odour threshold
MX ^g	Occurs in drinking-water at concentrations well below those of health concern
Nitrobenzene	Rarely found in drinking-water at concentrations of health concern
Novaluron	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Organotins	
TBT, TPT, DBT, DOT ^h	Occurs in drinking-water at concentrations well below those of health concern
MMT, DMT, DMTC ⁱ	Unnecessary since their use as stabilizers in polyvinyl chloride and chlorinated polyvinyl chloride are normally controlled by product specification
Other organotins	Available data inadequate to permit derivation of health-based guideline value
Parathion	Occurs in drinking-water at concentrations well below those of health concern
Permethrin	Not recommended for direct addition to drinking-water as part of WHO's policy to exclude the use of any pyrethroids for larviciding of mosquito vectors of human disease
Petroleum products	Taste and odour will in most cases be detectable at concentrations below those of health concern, particularly with short-term exposure
pH ⁱ	Not of health concern at levels found in drinking-water ^j
2-Phenylphenol and its sodium salt	Occurs in drinking-water at concentrations well below those of health concern

Table A3.2 (continued)

Chemical	Reason for not establishing a guideline value
Pirimiphos-methyl	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Potassium	Occurs in drinking-water at concentrations well below those of health concern
Propanil	Readily transformed into metabolites that are more toxic; a guideline value for the parent compound is considered inappropriate, and there are inadequate data to enable the derivation of guideline values for the metabolites
Pyriproxyfen	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Silver	Available data inadequate to permit derivation of health-based guideline value and usually occurs in drinking-water at concentrations well below those of health concern
Sodium	Not of health concern at levels found in drinking-water ^b
Spinosad	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Sulfate	Not of health concern at levels found in drinking-water ^b
Temephos	Not considered appropriate to set guideline values for pesticides potentially used for vector control in drinking-water
Total dissolved solids	Not of health concern at levels found in drinking-water ^b
Trichloramine	Available data inadequate to permit derivation of health-based guideline value
Trichloroacetonitrile	Available data inadequate to permit derivation of health-based guideline value
Trichlorobenzenes (total)	Occurs in drinking-water at concentrations well below those of health concern, and health-based value would exceed lowest reported odour threshold
1,1,1-Trichloroethane	Occurs in drinking-water at concentrations well below those of health concern
Zinc	Not of health concern at levels found in drinking-water ^b

^a Health-based values have been developed for a number of these chemicals, whereas, for anatoxins and silver, provisional reference values have been established. These values may be useful to guide actions by Member States when there is a reason for local concern. For further information on guideline values, health-based values and provisional reference values, see [section 8.2](#). Where health-based values or provisional reference values have been established, this information can be found in the chemical fact sheets in [chapter 12](#).

^b May affect acceptability of drinking-water (see [chapter 10](#)).

^c Although dicofol does not fulfil one of the three criteria for evaluation in the Guidelines, a background document has been prepared and a health-based value has been established, in response to a request from Member States for guidance.

^d See fact sheet on polynuclear aromatic hydrocarbons.

^e Aminomethylphosphonic acid.

^f (2-Methyl-4-chlorophenoxy)acetic acid.

^g 3-Chloro-4-dichloromethyl-5-hydroxy-2(5H)-furanone.

^h Tributyltin, triphenyltin, dibutyltin, and di-*n*-octyltin respectively.

ⁱ Monomethyltin dimethyltin and dimethyltin dichloride, respectively.

^j An important operational water quality parameter.

Table A3.3 Guideline values for chemicals that are of health significance in drinking-water

Chemical	Guideline value		Remarks
	mg/l	µg/l	
Acrylamide	0.0005 ^a	0.5 ^a	
Alachlor	0.02 ^a	20 ^a	
Aldicarb and its sulfoxide and sulfone metabolites	0.01	10	
Aldrin and dieldrin	0.000 03	0.03	For combined aldrin plus dieldrin
Antimony	0.02	20	
Arsenic	0.01 (A, T)	10 (A, T)	
Atrazine and its chloro-s-triazine metabolites	0.1	100	
Barium	1.3	1300	
Benzene	0.01 ^a	10 ^a	
Benzo[a]pyrene	0.0007 ^a	0.7 ^a	
Boron	2.4	2 400	
Bromate	0.01 ^a (A, T)	10 ^a (A, T)	
Bromodichloromethane	0.06 ^a	60 ^a	
Bromoform	0.1	100	
Cadmium	0.003	3	
Carbofuran	0.007	7	
Carbon tetrachloride	0.004	4	
Chlorate	0.7 (D)	700 (D)	
Chlordane	0.0002	0.2	
Chlorine	5 (C)	5 000 (C)	For free chlorine. For effective disinfection, there should be a residual concentration of free chlorine of ≥0.5 mg/l after at least 30 min contact time at pH <8.0. A chlorine residual should be maintained throughout the distribution system. At the point of delivery, the minimum residual concentration of free chlorine should be 0.2 mg/l.
Chlorite	0.7 (D)	700 (D)	
Chloroform	0.3	300	
Chlorotoluron	0.03	30	
Chlorpyrifos	0.03	30	
Chromium	0.05	50	For total chromium
Copper	2	2 000	Staining of laundry and sanitary ware may occur below guideline value
Cyanazine	0.0006	0.6	

Table A3.3 (continued)

Chemical	Guideline value		Remarks
	mg/l	µg/l	
Cylindrospermopsins (cyanobacterial toxin)	0.0007 (P) 0.003 (P)	0.7 (P) 3 (P)	For short-term exposure ^b Values are for total cylindrospermopsins (sum of all congeners, free plus-cell bound)
2,4-D ^c	0.03	30	Applies to free acid
2,4-DB ^d	0.09	90	
DDT ^e and metabolites	0.001	1	
Dibromoacetonitrile	0.07	70	
Dibromochloromethane	0.1	100	
1,2-Dibromo-3- chloropropane	0.001 ^a	1 ^a	
1,2-Dibromoethane	0.0004 ^a (P)	0.4 ^a (P)	
Dichloroacetate	0.05 ^a (D)	50 ^a (D)	
Dichloroacetonitrile	0.02 (P)	20 (P)	
1,2-Dichlorobenzene	1 (C)	1 000 (C)	
1,4-Dichlorobenzene	0.3 (C)	300 (C)	
1,2-Dichloroethane	0.03 ^a	30 ^a	
1,2-Dichloroethene	0.05	50	
Dichloromethane	0.02	20	
1,2-Dichloropropane	0.04 (P)	40 (P)	
1,3-Dichloropropene	0.02 ^a	20 ^a	
Dichloroprop	0.1	100	
Di(2-ethylhexyl)phthalate	0.008	8	
Dimethoate	0.006	6	
1,4-Dioxane	0.05 ^a	50 ^a	Derived using tolerable daily intake approach as well as linearized multistage modelling
Edetic acid	0.6	600	Applies to the free acid
Endrin	0.0006	0.6	
Epichlorohydrin	0.0004 (P)	0.4 (P)	
Ethylbenzene	0.3 (C)	300 (C)	
Fenoprop	0.009	9	
Fluoride	1.5	1 500	Volume of water consumed and intake from other sources should be considered when setting national standards
Hexachlorobutadiene	0.0006	0.6	
Hydroxyatrazine	0.2	200	Atrazine metabolite
Isoproturon	0.009	9	

ANNEX 3. CHEMICAL SUMMARY TABLES

Table A3.3 (continued)

Chemical	Guideline value		Remarks
	mg/l	µg/l	
Lead	0.01 (A, T)	10 (A, T)	
Lindane	0.002	2	
Manganese	0.08 (P)	80 (P)	For total manganese. Aesthetic as well as health aspects should be considered when setting national standards
Mecoprop	0.01	10	
Mercury	0.006	6	For inorganic mercury
Methoxychlor	0.02	20	
Metolachlor	0.01	10	
Microcystins (cyanobacterial toxin)	0.001 (P) 0.012 (P)	1 (P) 12 (P)	For short-term exposure ^b Values are for total microcystins (sum of all congeners, free plus-cell bound)
Molinate	0.006	6	
Monochloramine	3	3 000	
Monochloroacetate	0.02	20	
Nickel	0.07	70	Based on long-term effects, but protective for short-term effects
Nitrate (as NO ₃ ⁻)	50	50 000	Based on short-term effects, but protective for long-term effects
Nitrilotriacetic acid	0.2	200	
Nitrite (as NO ₂ ⁻)	3	3 000	Based on short-term effects, but protective for long-term effects
N-Nitrosodimethylamine	0.0001	0.1	
Pendimethalin	0.02	20	
Pentachlorophenol	0.009 ^a (P)	9 ^a (P)	
Perchlorate	0.07	70	
Saxitoxins (cyanobacterial toxin)	0.003	3	For acute exposure For total saxitoxins (sum of all congeners, free plus-cell bound)
Selenium	0.04 (P)	40 (P)	
Simazine	0.002	2	
Sodium dichloroisocyanurate	50 40	50 000 40 000	As sodium dichloroisocyanurate As cyanuric acid
Styrene	0.02 (C)	20 (C)	
2,4,5-T ^f	0.009	9	
Terbutylazine	0.007	7	
Tetrachloroethene	0.1	100	
Toluene	0.7 (C)	700 (C)	
Trichloroacetate	0.2	200	

Table A3.3 (continued)

Chemical	Guideline value		Remarks
	mg/l	µg/l	
Trichloroethene	0.008	8	
2,4,6-Trichlorophenol	0.2 ^a (C)	200 ^a (C)	
Trifluralin	0.02	20	
Trihalomethanes			The sum of the ratio of the concentration of each to its respective guideline value should not exceed 1
Uranium	0.03 (P)	30 (P)	Only chemical, not radiological, aspects of uranium addressed
Vinyl chloride	0.0003 ^a	0.3 ^a	
Xylenes	0.5 (C)	500 (C)	

A, provisional guideline value because calculated guideline value is below the achievable quantification level; C, concentrations of the substance at or below the health-based guideline value may affect the appearance, taste or odour of the water, leading to consumer complaints; D, provisional guideline value because effective disinfection may result in the guideline value being exceeded; P, provisional guideline value because of uncertainties in the health database; T, provisional guideline value because calculated guideline value is below the level that can be achieved through practical treatment methods, source protection, etc.

^a For substances that are considered to be carcinogenic, the guideline value is the concentration in drinking-water associated with an upper-bound excess lifetime cancer risk of 10^{-5} (one additional case of cancer per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). Concentrations associated with upper-bound estimated excess lifetime cancer risks of 10^{-4} and 10^{-6} can be calculated by multiplying and dividing, respectively, the guideline value by 10.

^b See the respective chemical fact sheet in [chapter 12](#) for considerations for bottle-fed infants.

^c 2,4-Dichlorophenoxyacetic acid.

^d 2,4-Dichlorophenoxybutyric acid.

^e Dichlorodiphenyltrichlorethane.

^f 2,4,5-Trichlorophenoxyacetic acid.

Analytical methods and achievability

A4.1 Analytical methods

In *volumetric titration*, chemicals are analysed by titration with a standardized titrant. The titration end-point is identified by the development of colour resulting from the reaction with an indicator, by the change of electrical potential or by the change of pH value.

Colorimetric methods are based on measuring the intensity of colour of a coloured target chemical or reaction product. The optical absorbance is measured using light of a suitable wavelength. The concentration is determined by means of a calibration curve obtained using known concentrations of the determinant. The ultra-violet (UV) method is similar to this method except that UV light is used. For ionic materials, the ion concentration can be measured using an *ion selective electrode*. The measured potential is proportional to the logarithm of the ion concentration. Some organic compounds absorb UV light (wavelength 190–380 nm) in proportion to their concentration. *UV absorption* is useful for qualitative estimation of organic substances, because a strong correlation may exist between UV absorption and organic carbon content.

Atomic absorption spectrometry (AAS) is used for the determination of metals. It is based on the phenomenon that the atom in the ground state absorbs the light of wavelengths that are characteristic to each element when light is passed through the atoms in the vapour state. Because this absorption of light depends on the concentration of atoms in the vapour, the concentration of the target element in the water sample is determined from the measured absorbance. The Beer-Lambert law describes the relationship between concentration and absorbance.

In *flame atomic absorption spectrometry (FAAS)*, a sample is aspirated into a flame and atomized. A light beam from a hollow cathode lamp of the same element as the target metal is radiated through the flame, and the amount of absorbed light is measured by the detector. This method is much more sensitive than other methods and free from spectral or radiation interference by co-existing elements. Pretreatment is either unnecessary or straightforward. However, it is not suitable for simultaneous analysis of many elements, because the light source is different for each target element.

Electrothermal atomic absorption spectrometry (EAAS) is based on the same principle as FAAS, but an electrically heated atomizer or graphite furnace replaces the standard burner head for determination of metals. In comparison with FAAS, EAAS gives higher sensitivities and lower detection limits, and a smaller sample volume is required. EAAS suffers from more interference through light scattering by co-existing elements and requires a longer analysis time than FAAS.

The principle of *inductively coupled plasma atomic emission spectrometry (ICP-AES)* for determination of metals is as follows. An ICP source consists of a flowing stream of argon gas ionized by an applied radio frequency. A sample aerosol is generated in a nebulizer and spray chamber and then carried into the plasma through an injector tube. A sample is heated and excited in the high-temperature plasma. The high temperature of the plasma causes the atoms to become excited. On returning to the ground state, the excited atoms produce ionic emission spectra. A monochromator is used to separate specific wavelengths corresponding to different elements, and a detector measures the intensity of radiation of each wavelength. A significant reduction in chemical interference is achieved. In the case of water with low pollution, simultaneous or sequential analysis is possible without special pretreatment to achieve low detection limits for many elements. This, coupled with the extended dynamic range from three digits to five digits, means that multielement determination of metals can be achieved. ICP-AES has similar sensitivity to FAAS or EAAS.

In *inductively coupled plasma mass spectrometry (ICP-MS)*, elements are atomized and excited as in ICP-AES, then passed to a mass spectrometer. Once inside the mass spectrometer, the ions are accelerated by high voltage and passed through a series of ion optics, an electrostatic analyser and, finally, a magnet. By varying the strength of the magnet, ions are separated according to mass/charge ratio and passed through a slit into the detector, which records only a very small atomic mass range at a given time. By varying the magnet and electrostatic analyser settings, the entire mass range can be scanned within a relatively short period of time. In the case of water with low pollution, simultaneous or sequential analysis is possible without special pretreatment to achieve low detection limits for many elements. This, coupled with the extended dynamic range from three digits to five digits, means that multielement determination of metals can be achieved.

Chromatography is a separation method based on the affinity difference between two phases, the stationary and mobile phases. A sample is injected into a column, either packed or coated with the stationary phase, and separated by the mobile phase based on the difference in interaction (distribution or adsorption) between compounds and the stationary phase. Compounds with a low affinity for the stationary phase move more quickly through the column and elute earlier. The compounds that elute from the end of the column are determined by a suitable detector.

In *ion chromatography*, an ion exchanger is used as the stationary phase, and the eluant for determination of anions is typically a dilute solution of sodium hydrogen carbonate and sodium carbonate. Colorimetric, electrometric or titrimetric detectors can be used for determining individual anions. In suppressed ion chromatography, anions are converted to their highly conductive acid forms; in the carbonate–bicarbonate

eluant, anions are converted to weakly conductive carbonic acid. The separated acid forms are measured by conductivity and identified on the basis of retention time as compared with their standards.

High-performance liquid chromatography (HPLC) is an analytical technique using a liquid mobile phase and a column containing a liquid stationary phase. Detection of the separated compounds is achieved through the use of absorbance detectors for organic compounds and through conductivity or electrochemical detectors for metallic and inorganic compounds.

Gas chromatography (GC) permits the identification and quantification of trace organic compounds. In GC, gas is used as the mobile phase, and the stationary phase is a liquid that is coated either on an inert granular solid or on the walls of a capillary column. When the sample is injected into the column, the organic compounds are vaporized and moved through the column by the carrier gas at different rates depending on differences in partition coefficients between the mobile and stationary phases. The gas exiting the column is passed to a suitable detector. A variety of detectors can be used, including flame ionization (FID), electron capture (ECD) and nitrogen-phosphorus. As separation ability is good in this method, mixtures of substances with similar structure are systematically separated, identified and determined quantitatively in a single operation.

The *gas chromatography/mass spectrometry (GC-MS)* method is based on the same principle as the GC method, using a mass spectrometer as the detector. As the gas emerges from the end of the GC column opening, it flows through a capillary column interface into the MS. The sample then enters the ionization chamber, where a collimated beam of electrons impacts the sample molecules, causing ionization and fragmentation. The next component is a mass analyser, which uses a magnetic field to separate the positively charged particles according to their mass. Several types of separating techniques exist; the most common are quadrupoles and ion traps. After the ions are separated according to their masses, they enter a detector.

The *purge-and-trap packed column GC-MS* method or purge-and-trap packed column GC method is applicable to the determination of various purgeable organic compounds that are transferred from the aqueous to the vapour phase by bubbling purge gas through a water sample at ambient temperature. The vapour is trapped with a cooled trap. The trap is heated and backflushed with the same purge gas to desorb the compounds onto a GC column. The principles of GC or GC-MS are as referred to above.

The principle of *enzyme-linked immunosorbent assay (ELISA)* is as follows. The protein (antibody) against the chemical of interest (antigen) is coated onto the solid material. The target chemical in the water sample binds to the antibody, and a second antibody with an enzyme attached is also added that will attach to the chemical of interest. After washing to remove any of the free reagents, a chromogen is added that will give a colour reaction due to cleavage by the enzyme that is proportional to the quantity of the chemical of interest. The ELISA method can be used to determine microcystin and synthetic surfactants.

A4.2 Analytical achievability for chemicals for which guideline values have been established

Analytical achievability for chemicals for which guideline values have been established is given in Tables A4.1–A4.6.

Table A4.1 Analytical achievability for inorganic chemicals for which guideline values have been established, by source category^a

	Field methods		Laboratory methods				
	Col	Absor	IC	FAAS	EAAS	ICP	ICP-MS
Naturally occurring chemicals							
Arsenic	+++	#		++(H)	+	++(H)	+++
Barium				++	+++	+++	+++
Boron		++				+++	+++
Chromium		#			++	++	+++
Fluoride	#	+	+++				
Manganese	#			++	+++	++ ^b	+++
Selenium		#		++(H)	++	++(H)	+++
Uranium							+++
Chemicals from industrial sources and human dwellings							
Cadmium		#			++	++	+++
Mercury				+++			
Chemicals from agricultural activities							
Nitrate/nitrite	+++	+++	+++				
Chemicals used in water treatment or materials in contact with drinking-water							
Antimony				+++ (H)		++ (H)	+++
Copper	#	+++		+++	+++	+++	+++
Lead		#			+	+	+++
Nickel		+		+	++	++	+++

^a For definitions and notes to Table A4.1, see below [Table A4.6](#).

^b For ICP-AES

Table A4.2 Analytical achievability for organic chemicals from industrial sources and human dwellings for which guideline values have been established^a

	Col	GC	(PT-) GC-PD	(PT-) GC-ECD	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC- MS	HPLC	HPLC -FD	HPLC- UVPAD	EAAS	IC-FD	IC-SCD	LC-MS
Benzene			+++						+++							
Carbon tetrachloride				+++					+++							
1,2-Dichlorobenzene			+++	+++				+++	+++							
1,4-Dichlorobenzene			+++	+++				+++	+++							
1,2-Dichloroethane				+++					+++							
1,2-Dichloroethene			+++	+++					+++							
Dichloromethane				+++					+++							
Di(2-ethylhexyl)phthalate								++								
1,4-Dioxane								+++								
Edetic acid								+++								
Ethylbenzene			+++						+++							
Hexachlorobutadiene			++	++					++							
Nitrilotriacetic acid		+++						+++								
Pentachlorophenol				+++				+			+					
Perchlorate															++	+++
Styrene			+++						+++							
Tetrachloroethene			+++	+++				+++	+++							
Toluene			+++						+++							
Trichloroethene			+++	+++				+++	+++							
Xylenes			+++						+++							

^a For definitions and notes to Table A4.2, see below [Table A4.6](#).

Table A4.3 Analytical achievability for organic chemicals from agricultural activities for which guideline values have been established^{a,b}

	CoI	GC	(PT-) GC-PD	(PT-) GC-ECD	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC-MS	HPLC	HPLC-FD	HPLC-UVPAD	EAAS	IC-FD
Alachlor				+++				+++						
Aldicarb											+++			
Aldrin and dieldrin				++				++						
Atrazine and its chloro- s-triazine metabolites				+++				+++					+++	
Carbofuran		++												
Chlordane				+++				+++						
Chlorotoluron								+++					+++	
Cyanazine				+++				+++					+	
2,4-D				+++				+++					++	
2,4-DB				+++				++					++	
1,2-Dibromo-3-chloro- propane				+++				+++	+++					
1,2-Dibromoethane				++				++	+++					
1,2-Dichloropropane				+++					+++					
1,3-Dichloropropene				+++					+++					
Dichlorprop				+++				+++						
Dimethoate								+++						
Endrin				+++				+++						
Fenoprop				+++									+	
Hydroxyatrazine							+++						+++	
Isoproturon								+++					+++	
Lindane				+++				+++						

Table A4.3 (continued)

	CoI	GC	(PT-) GC-PD	(PT-) GC-ECD	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC-MS	HPLC	HPLC-FD	HPLC-UVPAD	EAAS	IC-FD
Mecoprop				+++				+++						
Methoxychlor								+++						
Metolachlor				+++				+++						
Molinate		+++						+++						
Pendimethalin								+++						
Simazine				+++				+++						
2,4,5-T				+++								+		
Terbuthylazine								+++				++		
Trifluralin		+++		+++				+++						

^a For definitions and notes to Table A4.3, see below [Table A4.6](#).

^b LC-MS is also applicable for many of these agricultural chemicals.

Table A4.4 Analytical achievability for chemicals used in water treatment or from materials in contact with water for which guideline values have been established^a

	CoI	GC	(PT-) GC-PD	(PT-) GC-ECD	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC-MS	HPLC	HPLC-FD	HPLC-UVPAD	EAAS	IC
Disinfectants														
Monochloramine	+++													
Chlorine	+++													
Sodium dichloroisocyanurate							+++	+++				+++		
Disinfection by-products														
Bromate														++
Bromodichloromethane				+++				+++	+++					
Bromoform				+++				+++	+++					
Chlorate														+++

Table A4.4 (continued)

	Col	GC	(PT-) GC-PD	(PT-) GC-ECD	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC-MS	HPLC	HPLC-FD	HPLC-UVPAD	EAAS	IC
Chlorite														+++
Chloroform				+++				+++	+++					
Dibromoacetonitrile				+++				+++						
Dibromochloromethane				+++				+++	+++					
Dichloroacetic acid				+++				+++						
Dichloroacetonitrile				+++				+++						
Monochloroacetic acid				+++				++						
<i>N</i> -Nitrosodimethylamine								+++						
Trichloroacetic acid				+++				+++						
2,4,6-Trichlorophenol				+++				+++						
Trihalomethanes ^b				+++				+++	+++					
Organic contaminants from treatment chemicals														
Acrylamide							+				+			
Epichlorohydrin				+++	+++				+					
Organic contaminants from pipes and fittings														
Benzo[<i>a</i>]pyrene								++			++			
Vinyl chloride			++	++					+					

^a For definitions and notes to Table A4.4, see below [Table A4.6](#).^b See also individual trihalomethanes.**Table A4.5 Analytical achievability for pesticides used in water for public health purposes for which guideline values have been established^a**

	Col	GC	GC-PD	GC-EC	GC-FID	GC-FPD	GC-TID	GC-MS	PT-GC-MS	HPLC	HPLC-FD	HPLC-UVPAD	EAAS	IC/FD
Chlorpyrifos				+++		++	++	+++						
DDT (and metabolites)				++				++						

^a For definitions and notes to Table A4.5, see below [Table A4.6](#).

Table A4.6 Analytical achievability for cyanobacterial toxins for which guideline values have been established^{a,b,c}

Cyanotoxin group	PPA	RBA	ELISA	HPLC-UVPAD	LC-MS/PAD	HPLC-FD with pre- or post-column derivatization	LC-MS/MS ^d
Anatoxin-a variants		+	++			+++	+++
Cylindrospermopsins			++	++	++		+++
Microcystins	+		++	++	++		+++
Saxitoxins		++	+			+++	+++

^a For definitions and notes to Table A4.6, see below this table.

^b Note that most cyanotoxins are contained within the cell material and prior extraction is crucial. See the [chapter 12](#) fact sheets for further details.

^c Note that for anatoxin-a variants, although the data were inadequate to establish a guideline value, a provisional reference value was derived.

^d Quantitative reference standards are needed for congeners. See the [chapter 12](#) fact sheets for further details.

Definitions to Tables A4.1–A4.6

Absor	Absorptiometry	HPLC-UVPAD	High-performance liquid chromatography–ultraviolet photodiode array detector
Col	Colorimetry	IC	Ion chromatography
EAAS	Electrothermal atomic absorption spectrometry	IC-FAAS	Ion chromatography–flame atomic absorption spectrometry
ELISA	Enzyme-linked immunosorbent assay	IC-FD	Ion chromatography–fluorescence detector
FAAS	Flame atomic absorption spectrometry	IC-SCD	Ion chromatography–suppressed conductivity detection
GC	Gas chromatography	ICP	Inductively coupled plasma
GC-ECD	Gas chromatography–electron capture detector	ICP-AES	Inductively coupled plasma atomic emission spectrometry
GC-FID	Gas chromatography–flame ionization detector	ICP-MS	Inductively coupled plasma mass spectrometry
GC-FPD	Gas chromatography–flame photodiode detector	LC-MS	Liquid chromatography–mass spectrometry
GC-MS	Gas chromatography–mass spectrometry	LC-MS/MS	Liquid chromatography–tandem mass spectrometry
GC-PD	Gas chromatography–photoionization detector	LC-MS/PAD	Liquid chromatography–mass spectrometry–photodiode array detector
GC-TID	Gas chromatography–thermal ionization detector	PPA	Protein phosphatase assay
HPLC	High-performance liquid chromatography	PT-GC-MS	Purge-and-trap gas chromatography–mass spectrometry
HPLC-FD	High-performance liquid chromatography–fluorescence detector	RBA	Receptor-binding assay

Notes to Tables A4.1–A4.6

- + The detection limit is between the guideline value and 1/10th of its value.
- ++ The detection limit is between 1/10th and 1/50th of the guideline value.
- +++ The detection limit is under 1/100th of the guideline value.
- # The analytical method is available for detection of the guideline value concentration, but it is difficult to detect the concentration of 1/10 of the guideline value.
- (H) This method is applicable to the determination by conversion to their hydrides by hydride generator.

Treatment methods and performance

A5.1 Treatment methods

A5.1.1 Chlorination

Chlorination can be achieved by using liquefied chlorine gas, sodium hypochlorite solution or calcium hypochlorite granules and on-site chlorine generators. Liquefied chlorine gas is supplied in pressurized containers. The gas is withdrawn from the cylinder and dosed into water by a chlorinator, which both controls and measures the gas flow rate. Sodium hypochlorite solution is dosed using a positive-displacement electric dosing pump or gravity feed system. Calcium hypochlorite has to be dissolved in water, then mixed with the main supply. Chlorine, whether in the form of chlorine gas from a cylinder, sodium hypochlorite or calcium hypochlorite, dissolves in water to form hypochlorous acid (HOCl) and hypochlorite ion (OCl^-).

Different techniques of chlorination can be used, including breakpoint chlorination, marginal chlorination and superchlorination/dechlorination. Breakpoint chlorination is a method in which the chlorine dose is sufficient to rapidly oxidize all the ammonia nitrogen in the water and to leave a suitable free residual chlorine available to protect the water against reinfection from the point of chlorination to the point of use. Superchlorination/dechlorination is the addition of a large dose of chlorine to effect rapid disinfection and chemical reaction, followed by reduction of excess free chlorine residual. Removing excess chlorine is important to prevent taste problems. It is used mainly when the bacterial load is variable or the detention time in a tank is not enough. Marginal chlorination is used where water supplies are of high quality and is the simple dosing of chlorine to produce a desired level of free residual chlorine. The chlorine demand in these supplies is very low, and a breakpoint might not even occur.

Chlorination is employed primarily for microbial disinfection. However, chlorine also acts as an oxidant and can remove or assist in the removal or chemical conversion of some chemicals—for example, decomposition of easily oxidized pesticides, such as aldicarb; oxidation of dissolved species (e.g. manganese(II)) to form insoluble products that can be removed by subsequent filtration; and oxidation of dissolved species to more easily removable forms (e.g. arsenite to arsenate).

A disadvantage of chlorine is its ability to react with natural organic matter to produce trihalomethanes and other halogenated disinfection by-products. However, by-product formation may be controlled by optimization of the treatment system.

A5.1.2 Ozonation

Ozone is a powerful oxidant and has many uses in water treatment, including oxidation of organic chemicals. Ozone can be used as a primary disinfectant. Ozone gas (O_3) is formed by passing dry air or oxygen through a high-voltage electric field. The resultant ozone-enriched air is dosed directly into the water by means of porous diffusers at the base of baffled contactor tanks. The contactor tanks, typically about 5 m deep, provide 10–20 minutes of contact time. Dissolution of at least 80% of the applied ozone should be possible, with the remainder contained in the off-gas, which is passed through an ozone destructor and vented to the atmosphere.

The performance of ozonation relies on achieving the desired concentration after a given contact period. For oxidation of organic chemicals, such as some oxidizable pesticides, a residual of about 0.5 mg/l after a contact time of up to 20 minutes is typically used. The doses required to achieve this vary with the type of water but are typically in the range 2–5 mg/l. Higher doses are needed for untreated waters, because of the ozone demand of the natural background organics.

Ozone reacts with natural organics to increase their biodegradability, measured as assimilable organic carbon. To avoid undesirable bacterial growth in distribution, ozonation is normally used with subsequent treatment, such as biological filtration or granular activated carbon (GAC), to remove biodegradable organics, followed by a chlorine residual, as ozone does not provide a disinfectant residual. Ozone is effective for the degradation of a wide range of pesticides and other organic chemicals.

A5.1.3 Other disinfection processes

Other disinfection methods include chloramination, the use of chlorine dioxide and UV radiation, as well as alternative disinfection techniques that may be used in smaller-scale applications, such as for household water.

Chloramines (monochloramine, dichloramine and trichloramine, or nitrogen trichloride) are produced by the reaction of aqueous chlorine with ammonia. Monochloramine is the only useful chloramine disinfectant, and conditions employed for chloramination are designed to produce only monochloramine. Monochloramine is a less effective disinfectant than free chlorine, but it is persistent, and it is therefore an attractive secondary disinfectant for the maintenance of a stable distribution system residual.

Chlorine dioxide has been used in recent years because of concerns about disinfection by-product production associated with chlorine disinfection. Typically, chlorine dioxide is generated immediately prior to application by the addition of chlorine gas or an aqueous chlorine solution to aqueous sodium chlorite. Chlorine dioxide decomposes in water to form chlorite and chlorate.

UV radiation, emitted by a low-pressure or medium-pressure mercury arc lamp, is biocidal between wavelengths of 180 and 320 nm. It can be used to inactivate protozoa, bacteria, bacteriophage, yeast, viruses, fungi and algae. Turbidity can inhibit UV

disinfection. UV radiation can act as a catalyst in oxidation reactions when used in conjunction with ozone or hydrogen peroxide.

Numerous possible disinfection techniques are being developed and are typically used in smaller-scale applications, such as household point-of-use and point-of-entry water treatment systems. Some of these, including bromine and iodine, show promise for expanded use. Bromine and iodine are halogens, like chlorine, and they are well-known biocides. Iodine is commonly used for short-term applications, such as by travellers in areas where water quality is questionable. Some forms of silver may have applications as bacteriostats or possibly as slow-acting disinfectants for some micro-organisms; however, there are not good peer-reviewed published data to quantify the latter. It will be necessary to develop a more thorough analysis of the biocidal efficacy, potential disinfection by-products and risks from long-term exposures and application conditions for these lesser-used treatment chemicals to provide appropriate guidance as to their potential for wider applications.

A5.1.4 Filtration

Particulate matter can be removed from raw waters by rapid gravity, horizontal, pressure or slow sand filters. Slow sand filtration is essentially a biological process, whereas the others are physical treatment processes.

Rapid gravity, horizontal and pressure filters can be used for filtration of raw water, without pretreatment. Rapid gravity and pressure filters are commonly used to filter water that has been pretreated by coagulation and sedimentation. An alternative process is direct filtration, in which coagulation is added to the water, which then passes directly onto the filter where the precipitated floc (with contaminants) is removed; the application of direct filtration is limited by the available storage within the filter to accommodate solids.

Rapid gravity filters

Rapid gravity sand filters usually consist of open rectangular tanks (usually < 100 m²) containing silica sand (size range 0.5–1.0 mm) to a depth of between 0.6 and 2.0 m. The water flows downwards, and solids become concentrated in the upper layers of the bed. The flow rate is generally in the range 4–20 m³/m²·h. Treated water is collected via nozzles in the floor of the filter. The accumulated solids are removed periodically by backwashing with treated water, sometimes preceded by scouring of the sand with air. A dilute sludge that requires disposal is produced.

In addition to single-medium sand filters, dual-media or multimedia filters are used. Such filters incorporate different materials, such that the structure is from coarse to fine as the water passes through the filter. Materials of suitable density are used in order to maintain the segregation of the different layers following backwashing. A common example of a dual-media filter is the anthracite–sand filter, which typically consists of a 0.2 m deep layer of 1.5 mm anthracite over a 0.6 m deep layer of silica sand. Anthracite, sand and garnet can be used in multimedia filters. The advantage of dual-media and multimedia filters is that there is more efficient use of the whole bed depth for particle retention—the rate of headloss development can be half that of

single-medium filters, which can allow higher flow rates without increasing headloss development.

Rapid gravity filters are most commonly used to remove floc from coagulated waters (see [section A5.1.6](#)). They may also be used to reduce turbidity (including adsorbed chemicals) and oxidized iron and manganese from raw waters.

Roughing filters

Roughing filters can be applied as pre-filters prior to other processes such as slow sand filters. Roughing filters with coarse gravel or crushed stones as the filter medium can successfully treat water of high turbidity (> 50 nephelometric turbidity units). The main advantage of roughing filtration is that as the water passes through the filter, particles are removed by both filtration and gravity settling. Horizontal filters can be up to 10 m long and are operated at filtration rates of $0.3\text{--}1.0\text{ m}^3/\text{m}^2\cdot\text{h}$.

Pressure filters

Pressure filters are sometimes used where it is necessary to maintain head in order to eliminate the need for pumping into supply. The filter bed is enclosed in a cylindrical shell. Small pressure filters, capable of treating up to about $15\text{ m}^3/\text{h}$, can be manufactured in glass-reinforced plastics. Larger pressure filters, up to 4 m in diameter, are manufactured in specially coated steel. Operation and performance are generally as described for the rapid gravity filter, and similar facilities are required for backwashing and disposal of the dilute sludge.

Slow sand filters

Slow sand filters usually consist of tanks containing sand (effective size range $0.15\text{--}0.3\text{ mm}$) to a depth of between 0.5 and 1.5 m. The raw water flows downwards, and turbidity and microorganisms are removed primarily in the top few centimetres of the sand. A biological layer, known as the “schmutzdecke”, develops on the surface of the filter and can be effective in removing microorganisms. Treated water is collected in underdrains or pipework at the bottom of the filter. The top few centimetres of sand containing the accumulated solids are removed and replaced periodically. Slow sand filters are operated at a water flow rate of between 0.1 and $0.3\text{ m}^3/\text{m}^2\cdot\text{h}$.

Slow sand filters are more suitable for low-turbidity water or water that has been pre-filtered. They are used to remove algae and microorganisms, including protozoa, and, if preceded by microstraining or coarse filtration, to reduce turbidity (including adsorbed chemicals). Slow sand filtration is effective for the removal of some organics, including certain pesticides and also ammonia.

Bank filtration

Bank filtration is a process that produces an influx of surface water through the groundwater, via the bed and banks of the surface water body. This is commonly achieved through abstraction from boreholes adjacent to the surface water source. It is a relatively simple and low-cost means for removing particulates and microorganisms from surface water by placing pumping wells in alluvial sediments of the river or stream banks. The sediments act as both a filter and biofilter, trapping and reducing the concentrations of microorganisms and many organic pollutants. Bank filtration wells can

be either horizontal or vertical, depending upon the hydrogeological circumstances and required production rate. Horizontal wells are often used where alluvial deposits are shallow or where high pumping rates are required.

Bank filtration can remove particles, bacteria, viruses, parasites, heavy metals and easily biodegradable compounds. Bank filtration attenuates concentration peaks, providing uniform quality of raw water feed to downstream treatment. The performance of bank filtration can be highly dependent upon several factors, including soil and geological conditions as well as the quality of the source water. Bank filters can become clogged, resulting in pressure drops. Site-specific testing is needed to determine whether the appropriate geology is present as well as the effectiveness and operational parameters.

A5.1.5 Aeration

Aeration processes are designed to achieve removal of gases and volatile compounds by air stripping. Transfer can usually be achieved using a simple cascade or diffusion of air into water, without the need for elaborate equipment. Stripping of gases or volatile compounds, however, may require a specialized plant that provides a high degree of mass transfer from the liquid phase to the gas phase.

Cascade or step aerators are designed so that water flows in a thin film to achieve efficient mass transfer. Cascade aeration may introduce a significant headloss; design requirements are between 1 and 3 m to provide a loading of 10–30 m³/m²·h. Alternatively, compressed air can be diffused through a system of submerged perforated pipes. These types of aerator are used for oxidation and precipitation of iron and manganese.

Air stripping can be used for removal of volatile organics (e.g. solvents), some taste- and odour-causing compounds and radon. Aeration processes to achieve air stripping need to be much more elaborate to provide the necessary contact between the air and water. The most common technique is cascade aeration, usually in packed towers in which water is allowed to flow in thin films over plastic media with air blown counter-current. The required tower height and diameter are functions of the volatility and concentration of the compounds to be removed and the flow rate. Increasing the dissolved oxygen content of a water can increase its corrosivity towards some metallic materials used in distribution pipes and plumbing, and this should be taken into account when considering aeration as a treatment process.

A5.1.6 Chemical coagulation

Chemical coagulation-based treatment is the most common approach for treatment of surface waters and is almost always based on the following unit processes.

Chemical coagulants, usually salts of aluminium or iron, are dosed to the raw water under controlled conditions to form a solid flocculent metal hydroxide. Typical coagulant doses are 2–5 mg/l as aluminium or 4–10 mg/l as iron. The precipitated floc removes suspended and dissolved contaminants by mechanisms of charge neutralization, adsorption and entrapment. The efficiency of the coagulation process depends on raw water quality, the coagulant or coagulant aids used and operational factors, including mixing conditions, coagulation dose and pH. The floc is removed from the

treated water by subsequent solid–liquid separation processes such as sedimentation or flotation and/or rapid or pressure gravity filtration.

Effective operation of the coagulation process depends on selection of the optimum coagulant dose and also the pH value. The required dose and pH can be determined by using small-scale batch coagulation tests, often termed “jar tests”. Increasing doses of coagulant are applied to raw water samples that are stirred and allowed to settle. The optimum dose is selected as that which achieves adequate removal of colour and turbidity; the optimum pH can be selected in a similar manner. These tests have to be conducted at a sufficient frequency to keep pace with changes in raw water quality and hence coagulant demand.

Powdered activated carbon (PAC) may be dosed during coagulation to adsorb organic chemicals, such as some hydrophobic pesticides. The PAC will be removed as an integral fraction of the floc and disposed of with the waterworks sludge.

The floc may be removed by sedimentation to reduce the solids loading to the subsequent rapid gravity filters. Sedimentation is most commonly achieved in horizontal flow or floc blanket clarifiers. Alternatively, floc may be removed by dissolved air flotation, in which solids are contacted with fine bubbles of air that attach to the floc, causing them to float to the surface of the tank, where they are removed periodically as a layer of sludge. The treated water from either process is passed to rapid gravity filters (see [section A5.1.4](#)), where remaining solids are removed. Filtered water may be passed to a further stage of treatment, such as additional oxidation and filtration (for removal of manganese), ozonation and/or GAC adsorption (for removal of pesticides and other trace organics), prior to final disinfection before the treated water enters the supply.

Coagulation is suitable for removal of particulates and bound microorganisms, certain heavy metals and low-solubility organic chemicals, such as certain organo-chlorine pesticides. For other organic chemicals, coagulation is generally ineffective, except where the chemical is bound to humic material or adsorbed onto particulates.

A5.1.7 Activated carbon adsorption

Activated carbon is produced by the controlled thermalization of carbonaceous material, normally wood, coal, coconut shells or peat. This activation produces a porous material with a large surface area (500–1500 m²/g) and a high affinity for organic compounds. It is normally used in either powdered (PAC) or granular (GAC) form. When the adsorption capacity of the carbon is exhausted, it can be reactivated by burning off the organics in a controlled manner. However, PAC (and some GAC) is normally used only once before disposal. Different types of activated carbon have different affinities for types of contaminants.

The choice between PAC and GAC will depend upon the relative cost-effectiveness, frequency and dose required. PAC would generally be preferred in the case of seasonal or intermittent contamination or where low dosage rates are required.

PAC is dosed as a slurry into the water and removed by subsequent treatment processes, together with the waterworks sludge. Its use is therefore restricted to surface water treatment works with existing filters. GAC in fixed-bed adsorbers is used much more efficiently than PAC dosed into the water, and the effective carbon use per water

volume treated would be much lower than the dose of PAC required to achieve the same removal.

GAC is used for taste and odour control. It is normally used in fixed beds, either in purpose-built adsorbers for chemicals or in existing filter shells by replacement of sand with GAC of a similar particle size. Although at most treatment works it would be cheaper to convert existing filters rather than build separate adsorbers, use of existing filters usually allows only short contact times, and they are not capable of facile reactivation. It is therefore common practice to install additional GAC adsorbers (in some cases preceded by ozonation) between the rapid gravity filters and final disinfection. Most groundwater sources do not have existing filters, and separate adsorbers would need to be installed.

The service life of a GAC bed is dependent on the capacity of the carbon used and the contact time between the water and the carbon, the empty bed contact time, controlled by the flow rate of the water. Empty bed contact times are usually in the range 5–30 minutes. GACs vary considerably in their capacity for specific organic compounds, which can have a significant effect upon their service life. A guide to capacity can be obtained from published isotherm data. Carbon capacity is strongly dependent on the water source and is greatly reduced by the presence of background organic compounds. The properties of a chemical that influence its adsorption onto activated carbon include the water solubility and octanol–water partition coefficient. As a general rule, chemicals with low solubilities and high log octanol–water partition coefficients are well adsorbed.

Activated carbon is used for the removal of pesticides and other organic chemicals, taste and odour compounds, cyanobacterial toxins and total organic carbon.

A5.1.8 Ion exchange

Ion exchange is a process in which ions of like charge are exchanged between the water phase and the solid resin phase. Water softening is achieved by cation exchange. Water is passed through a bed of cationic resin, and the calcium ions and magnesium ions in the water are replaced by sodium ions. When the ion exchange resin is exhausted (i.e. the sodium ions are depleted), it is regenerated using a solution of sodium chloride. The process of “dealkalization” can also soften water. Water is passed through a bed of weakly acidic resin, and the calcium and magnesium ions are replaced by hydrogen ions. The hydrogen ions react with the carbonate and bicarbonate ions to produce carbon dioxide. The hardness of the water is thus reduced without any increase in sodium levels. Anion exchange can be used to remove contaminants such as nitrate, fluoride, arsenate and uranium (as the uranyl anion), which are exchanged for chloride. Several appropriate resins are available for this purpose.

An ion exchange plant normally consists of two or more resin beds contained in pressure shells with appropriate pumps, pipework and ancillary equipment for regeneration. The pressure shells are typically up to 4 m in diameter, containing 0.6–1.5 m depth of resin.

Cation exchange can be used for removal of certain heavy metals. Potential applications of anionic resins, in addition to nitrate removal, are for removal of arsenic and selenium species.

A5.1.9 Membrane processes

The membrane processes of most significance in water treatment are reverse osmosis, ultrafiltration, microfiltration and nanofiltration. These processes have traditionally been applied to the production of water for industrial or pharmaceutical applications, but are now being applied to the treatment of drinking-water.

High-pressure processes

If two solutions are separated by a semipermeable membrane (i.e. a membrane that allows the passage of the solvent but not of the solute), the solvent will naturally pass from the lower-concentration solution to the higher-concentration solution. This process is known as osmosis. It is possible, however, to force the flow of solvent in the opposite direction, from the higher to the lower concentration, by increasing the pressure on the higher-concentration solution. The required pressure differential is known as the osmotic pressure, and the process is known as reverse osmosis.

Reverse osmosis results in the production of a treated water stream and a relatively concentrated waste stream. Typical operating pressures are in the range 15–50 bar, depending on the application. Reverse osmosis rejects monovalent ions and organics of molecular weight greater than about 50 daltons (membrane pore sizes are less than 0.002 μm). The most common application of reverse osmosis is desalination of brackish water and seawater.

Nanofiltration uses a membrane with properties between those of reverse osmosis and ultrafiltration membranes; pore sizes are typically 0.001–0.01 μm . Nanofiltration membranes allow monovalent ions such as sodium or potassium to pass but reject a high proportion of divalent ions such as calcium and magnesium and some higher molecular weight organics. Operating pressures are typically about 5 bar. Nanofiltration may be effective for the removal of colour-forming organic compounds.

Lower-pressure processes

Ultrafiltration is similar in principle to reverse osmosis, but the membranes have much larger pore sizes (typically 0.002–0.03 μm) and operate at lower pressures. Ultrafiltration membranes reject organic molecules of molecular weight above about 800 daltons and usually operate at pressures less than 5 bar.

Microfiltration is a direct extension of conventional filtration into the sub-micrometre range. Microfiltration membranes have pore sizes typically in the range 0.01–12 μm and do not separate molecules but reject colloidal and suspended material at operating pressures of 1–2 bar. Microfiltration is capable of sieving out particles greater than 0.05 μm . It has been used for water treatment in combination with coagulation or PAC to remove particulates and some dissolved organic carbon prior to reverse osmosis membranes and to improve permeate flux.

A5.1.10 Other treatment processes

Processes aimed at generating hydroxyl radicals are known collectively as advanced oxidation processes and can be effective for the destruction of chemicals that are difficult to treat using other methods, such as ozone alone. Hydrogen peroxide with UV is also a source of hydroxyl radicals. Chemicals can react either directly with molecular

ozone or with the hydroxyl radical ($\text{HO}\cdot$), which is a product of the decomposition of ozone in water and is an exceedingly powerful indiscriminate oxidant that reacts readily with a wide range of organic chemicals. The formation of hydroxyl radicals can be encouraged by using ozone at high pH. One advanced oxidation process using ozone or UV plus hydrogen peroxide involves dosing hydrogen peroxide simultaneously with ozone at a rate of approximately 0.4 mg of hydrogen peroxide per litre per milligram of ozone dosed per litre (the theoretical optimum ratio for hydroxyl radical production) and bicarbonate.

Other treatment processes that can be used in certain applications include:

- precipitation softening (addition of lime, lime plus sodium carbonate or sodium hydroxide to precipitate hardness at high pH);
- ion exchange softening;
- biological denitrification for removal of nitrate from surface waters;
- biological nitrification for removal of ammonia from surface waters;
- activated alumina (or other adsorbents) for specialized applications, such as removal of fluoride and arsenic.

A5.2 Treatment performance for chemicals for which guideline values have been established

Treatment performance for chemicals for which guideline values have been established is given in [Tables A5.1–A5.5](#).

A5.3 Corrosion of metals used in water treatment and distribution

A5.3.1 Brass

The main corrosion problem with brasses is dezincification, which is the selective dissolution of zinc from duplex brass, leaving behind copper as a porous mass of low mechanical strength. Meringue dezincification, in which a voluminous corrosion product of basic zinc carbonate forms on the brass surface, largely depends on the ratio of chloride to alkalinity. Meringue dezincification can be controlled by maintaining a low zinc to copper ratio (1:3 or lower) and by keeping pH below 8.3.

General dissolution of brass can also occur, releasing metals, including lead, into the water. Impingement attack can occur under conditions of high water velocity with waters that form poorly protective corrosion product layers and that contain large amounts of dissolved or entrained air.

A5.3.2 Concrete and cement

Concrete is a composite material consisting of a cement binder in which an inert aggregate is embedded. Cement is primarily a mixture of calcium silicates and aluminates together with some free lime. Cement mortar, in which the aggregate is fine sand, is used as a protective lining in iron and steel water pipes. In asbestos–cement pipe, the aggregate is asbestos fibres, which are not of concern in drinking-water (see also asbestos fact sheet in [chapter 12](#)). Cement is subject to deterioration on prolonged exposure to aggressive water, due either to the dissolution of lime and other soluble compounds

Table A5.1 Treatment performance for naturally occurring chemicals for which guideline values have been established^{a,b}

	Coagulation	Ion exchange	Precipitation softening	Activated alumina	Activated carbon	Ozonation	Adsorption/oxidation	Membranes
Arsenic ^c	++ <0.005	+++ <0.005	++ <0.005	+++ <0.005				+++ ^d <0.005
Fluoride	++			+++ <1				+++ <1
Manganese		++ ^e <0.05	+++ <0.02			++ ^f <0.05	+++ ^g <0.02	+++ ^h <0.01
Selenium	++	+++ <0.01		+++ <0.01				+++ <0.01
Uranium	++	+++ <0.001	++	+++ <0.001				

^a Symbols are as follows:

++ Approximately 50% or more removal

+++ Approximately 80% or more removal

^b The table includes chemicals for which some treatment data are available. A blank entry in the table indicates either that the process is completely ineffective or that there are no data on the effectiveness of the process. For the most effective processes, the table estimates the concentration of the chemical (in mg/l) that could be achievable in an ideal water.

^c Iron oxide-based and iron hydroxide-based media have been shown to be very effective for both arsenate and arsenite forms.

^d Reverse osmosis membranes are more effective for removal of arsenate than arsenite. However, arsenite is readily oxidized to arsenate by disinfectants (e.g. chlorine).

^e Dissolved manganese(II) can be removed through cation exchange in zeolite softening processes.

^f Oxidation of manganese using ozone must be followed by a filtration step.

^g Includes manganese greensand and other filter media coated with manganese oxides (such as conventional filtration media with free chlorine applied across the filter bed).

^h Low pressure membrane filtration preceded by oxidation using potassium permanganate.

Table A5.2 Treatment performance for chemicals from industrial sources and human dwellings for which guideline values have been established^{a,b}

	Air stripping	Coagulation	Ion exchange	Precipitation softening	Activated carbon	Ozonation	Advanced oxidation	Membranes	Biological treatment ^c	UV irradia- tion
Cadmium		+++ <0.002	+++ <0.002	+++ <0.002				+++ <0.002		
Mercury		+++ <0.0001		+++ <0.0001	+++ <0.0001			+++ <0.0001		
Benzene	+++ <0.01				+++ <0.01	+++ <0.01	Yes ^d			
Carbon tetrachloride	+++ <0.001				+++ <0.001			+		
1,2-Dichlorobenzene	+++ <0.01				+++ <0.01	+++ <0.01		Yes ^d		
1,4-Dichlorobenzene	+++ <0.01				+++ <0.01	+++ <0.01		Yes ^d		
1,2-Dichloroethane	+++				+++ <0.01		+			
1,2-Dichloroethene	+++ <0.01				+++ <0.01	+++ <0.01				
1,4-Dioxane					+		+++ 0.05			
Edetic acid					+++ <0.01					
Ethylbenzene	++ <0.001	+			+++ <0.001	+++ <0.001	++	+	++	
Hexachlorobutadiene					+++ <0.001			+		

Table A5.2 (continued)

	Air stripping	Coagulation	Ion exchange	Precipitation softening	Activated carbon	Ozonation	Advanced oxidation	Membranes	Biological treatment ^c	UV irradia- tion
Nitrilotriacetic acid					++				++	
N-Nitrosodimethylamine					+		++			+
Pentachlorophenol					+++ <0.0004			++		
Perchlorate			yes ^d					yes ^d	yes ^d	
Styrene	+++ <0.02				+++ <0.002	++		+	+	
Tetrachloroethene	+++ <0.001				+++ <0.001			+		
Toluene	+++ <0.001				+++ <0.001	+++	+++ ^e <0.001		++ <0.001	
Trichloroethene	+++ <0.02				+++ <0.02	+++	+++ ^e <0.02			
Xylenes	+++ <0.005				+++ <0.005		+++ ^e <0.005		++	

^a Symbols are as follows:

- + Limited removal
- ++ Approximately 50% or more removal
- +++ Approximately 80% or more removal

^b The table includes only those chemicals for which some treatment data are available. A blank entry in the table indicates either that the process is completely ineffective or that there are no data on the effectiveness of the process. For the most effective processes, where data are available, the table indicates the concentration of the chemical (in mg/l) that should be achievable.

^c Biological treatment includes slow sand filtration and bank filtration.

^d Yes means known or likely to be effective, but performance was not quantified.

^e Might be effective, but other techniques would be more likely to be applied due to cost.

Table A5.3 Treatment performance for chemicals from agricultural activities for which guideline values have been established^{a,b}

	Chlorination	Air stripping	Coagulation	Ion exchange	Activated carbon	Ozonation	Advanced oxidation	Membranes	Biological treatment ^c
Nitrate				+++ <5				+++ <5	+++ <5
Nitrite	+++ <0.1							+	+++
Alachlor					+++ <0.001	++	+++ <0.001	+++ <0.001	
Aldicarb					+++ <0.001	+++ <0.001		+++ <0.001	
Aldrin/dieldrin			+		+++ <0.000 02	++ <0.000 02		+++ <0.00002	
Atrazine and its chloro-s-triazine metabolites			+		+++ <0.0001	Yes ^d	+++ <0.0001	+++ <0.0001	+++ ^e <0.0001
Carbofuran	+				+++ <0.001	Yes ^d		+++ <0.001	
Chlordane					+++ <0.0001	++ <0.0001		Yes ^d	
Chlorotoluron					+++ <0.0001	+++ <0.0001			
Cyanazine					+++ <0.0001	+		+++ <0.0001	
2,4-D					+++ <0.001	+++ <0.001		Yes ^d	
1,2-Dibromo-3-chloropropane		++ <0.001			+++ <0.0001				

Table A5.3 (continued)

	Chlorination	Air stripping	Coagulation	Ion exchange	Activated carbon	Ozonation	Advanced oxidation	Membranes	Biological treatment ^c
1,2-Dibromoethane		+++ <0.0001			+++ <0.0001				
1,2-Dichloropropane		Yes			+++ <0.001	+			
Dimethoate	+++ <0.001				++	++			
Endrin			+		+++ <0.0002			Yes ^d	
Hydroxyatrazine							+++ <0.001	Yes ^d	
Isoproturon	++				+++ <0.0001	+++ <0.0001	+++ <0.0001	+++ <0.0001	+
Lindane					+++ <0.0001	++		Yes ^d	++
Mecoprop					+++ <0.0001	+++ <0.0001			+++ <0.0001
Methoxychlor			++		+++ <0.0001	+++ <0.0001		Yes ^d	
Metalochlor					+++ <0.0001	++		Yes ^d	++
Simazine					+++ <0.0001	++	+++ <0.0001	+++ <0.0001	
2,4,5-T					+++ <0.001			Yes ^d	

Table A5.3 (continued)

	Chlorination	Air stripping	Coagulation	Ion exchange	Activated carbon	Ozonation	Advanced oxidation	Membranes	Biological treatment ^c
Terbutylazine			+		+++ <0.0001	++			
Trifluralin					+++ <0.0001			+++ ^f <0.0001	

^a Symbols are as follows:

- + Limited removal
- ++ Approximately 50% or more removal
- +++ Approximately 80% or more removal

^b The table includes only those chemicals for which some treatment data are available. A blank entry in the table indicates either that the process is completely ineffective or that there are no data on the effectiveness of the process. For the most effective processes, the table indicates the concentration of the chemical (in mg/l) that should be achievable.

^c Biological treatment includes slow sand filtration, bank filtration and biological denitrification (for nitrate removal).

^d Yes means known or likely to be effective, but performance was not quantified.

^e For bank filtration; slow sand filtration is not effective.

^f Might be effective, but other techniques would be more likely to be applied due to cost.

Table A5.4 Treatment performance for pesticides used in water for public health for which guideline values have been established^{a,b}

	Chlorination	Coagulation	Activated carbon	Ozonation	Advanced oxidation	Membranes
DDT and metabolites		+	+++ <0.0001	+	+++ ^c <0.0001	+++ ^c <0.0001

^a Symbols are as follows:

+ Limited removal

+++ Approximately 80% or more removal

^b For the most effective processes, the table indicates the concentration of the chemical (in mg/l) that should be achievable.

^c Might be effective, but other techniques would be more likely to be applied due to cost.

Table A5.5 Treatment performance for cyanobacterial cells and cyanotoxins for which guideline values or a reference value have been established^{a,b}

	Chlorination	Coagulation	Activated carbon	Ozonation	Advanced oxidation	Membranes	Sediment passage ^{c,d}
Cyanobacterial cells		+++				+++	+++
Cyanotoxins	+++ ^d		+++	+++ ^e	+++ ^f	+++ ^g	+++

^a +++ 80% or more removal.

^b The table includes only those chemicals for which some treatment data are available. A blank entry in the table indicates limited removal or that there are no data on the effectiveness of the process.

^c Sediment passage includes slow sand filtration and bank filtration. Percolation through soils, such as during artificial groundwater recharge, can also be effective in removing cyanobacteria and cyanotoxins.

^d May release cyanotoxins from cells, which may not be sufficiently degraded. Not effective for anatoxins.

^e May release cyanotoxins from cells, which may not be sufficiently degraded. Effectiveness for STXs is uncertain.

^f May release cyanotoxins from cells, which may not be sufficiently degraded. Experience with the various advanced oxidation methods in real-scale drinking-water treatment is limited.

^g May be effective, depending on pore size and toxin. Nanofiltration and reverse osmosis are effective.

or to chemical attack by aggressive ions such as chloride or sulfate, and this may result in structural failure. Newly installed cement materials will leach lime, with consequent increases in pH, alkalinity and hardness. Cement contains a variety of metals that can be leached into the water. Aggressiveness to cement is related to the “aggressivity index”, which has been used specifically to assess the potential for the dissolution of concrete. A pH of 8.5 or higher may be necessary to control cement corrosion.

A5.3.3 Copper

The corrosion of copper pipework and hot water cylinders can cause blue water, blue or green staining of bathroom fittings and, occasionally, taste problems. Copper tubing may be subject to general corrosion, impingement attack and pitting corrosion.

General corrosion is most often associated with soft, acidic waters; waters with pH below 6.5 and hardness of less than 60 mg of calcium carbonate per litre are very aggressive to copper. Copper, like lead, can enter water by dissolution of the corrosion product, basic copper carbonate. The solubility is mainly a function of pH and total inorganic carbon. Solubility decreases with increase in pH, but increases with increase in concentrations of carbonate species. Raising the pH to between 8 and 8.5 is the usual procedure to overcome these difficulties.

Impingement attack is the result of excessive flow velocities and is aggravated in soft water at high temperature and low pH.

The pitting of copper is commonly associated with hard groundwaters having a carbon dioxide concentration above 5 mg/l and high dissolved oxygen. Phosphates have been used to suppress copper corrosion in those cases. Surface waters with organic colour may also be associated with pitting corrosion. Copper pipes can fail by pitting corrosion, which involves highly localized attacks leading to perforations with negligible loss of metal. Two main types of attack are recognized. Type I pitting affects cold water systems (below 40 °C) and is associated, particularly, with hard borehole waters and the presence of a carbon film in the bore of the pipe, derived from the manufacturing process. Tubes that have had the carbon removed by cleaning are immune from Type I pitting. Type II pitting occurs in hot water systems (above 60 °C) and is associated with soft waters. A high proportion of general and pitting corrosion problems are associated with new pipe in which a protective oxide layer has not yet formed. Calcium carbonate precipitation indices such as Langelier and Ryznar are not good predictors of corrosion for copper systems.

A5.3.4 Iron

Iron (either cast or ductile) is frequently used in water distribution systems, and its corrosion is of concern. While structural failure as a result of iron corrosion is rare, water quality problems (e.g. “red water”) can arise as a result of excessive corrosion of iron pipes. The corrosion of iron is a complex process that involves the oxidation of the metal, normally by dissolved oxygen, ultimately to form a precipitate of iron(III). This leads to the formation of tubercles on the pipe surface. The major water quality factors that determine whether the precipitate forms a protective scale are pH and alkalinity. The concentrations of calcium, chloride and sulfate also influence iron corrosion. Successful control of iron corrosion has been achieved by adjusting the pH to

the range 6.8–7.3, hardness and alkalinity to at least 40 mg/l (as calcium carbonate), oversaturation with calcium carbonate of 4–10 mg/l and a ratio of alkalinity to chloride plus sulfate of at least 5 (when both are expressed as calcium carbonate).

Silicates and polyphosphates are often described as “corrosion inhibitors”, but there is no guarantee that they will inhibit corrosion in water distribution systems. However, they can complex dissolved iron (in the iron(II) state) and prevent its precipitation as visibly obvious red “rust”. These compounds may act by masking the effects of corrosion rather than by preventing it. Orthophosphate is a possible corrosion inhibitor and, like polyphosphates, is used to prevent “red water”.

A5.3.5 Lead

Lead corrosion (plumbosolvency) is of particular concern. Lead piping is still common in old houses in some countries, lead solders have been used widely for jointing copper tubing and brass fittings can contain substantial amounts of lead. Galvanized iron pipe plumbing can accumulate incoming lead and release it at a later time as particulates. The solubility of lead is governed by the formation of lead carbonates as pipe deposits. Wherever practicable, lead pipework should be replaced. Lead can also leach from lead-based solders and brass and bronze fittings.

The solubility of corrosion-related lead salts increases markedly as the pH increases above or decreases below 8.3 because of the substantial decrease in the equilibrium carbonate concentration. Thus, plumbosolvency tends to be at a maximum in waters with a low pH and low alkalinity, and a useful interim control procedure, pending pipe replacement, is to increase the pH to 8.0–8.5 after chlorination prior to distribution. Orthophosphate and other phosphates are effective in suppressing dissolution of lead.

Lead concentrations increase with increasing standing time of water in lead pipe. Flushing the pipework before drawing water for consumption may be used as an interim measure to reduce exposure to lead. Showering, bathing and flushing the toilet can be used to flush out the system.

Lead can corrode more rapidly when it is coupled to other metals such as copper. The rate of such galvanic corrosion is faster than that of simple oxidative corrosion, and lead concentrations are not limited by the solubility of the corrosion products. The rate of galvanic corrosion is affected principally by chloride concentration. Galvanic corrosion is less easily controlled but can be reduced by dosing zinc in conjunction with orthophosphate and by adjustment of pH.

Treatment to reduce plumbosolvency usually involves pH adjustment. When the water is very soft (calcium carbonate concentration less than 50 mg/l), the optimum pH is about 8.0–8.5. Alternatively, dosing with orthophosphoric acid or sodium orthophosphate might be more effective, particularly when plumbosolvency occurs in non-acidic waters. Calcium carbonate precipitation indices such as Langelier and Ryznar are not considered to be necessarily good predictors of corrosion for lead.

For further guidance on managing lead contamination of drinking-water, see *Lead in drinking-water: health risks, monitoring and corrective actions* ([Annex 1](#)).

A5.3.6 Nickel

Nickel in water may arise due to leaching from nickel- or chromium-plated taps as well as from stainless steel pipes and fittings. Nickel leaching is particularly influenced by stagnation time. There is some evidence that corrosion control measures (e.g. pH control) may reduce leaching of nickel from stainless steel materials. Product certification is the most important means of control. Consumers, particularly nickel-sensitive people, should flush nickel- or chromium-plated taps before using the water, particularly after periods of stagnation.

A5.3.7 Zinc

Galvanized pipes will release zinc (from the galvanizing layer) and can also leach cadmium and lead. Corrosion can be a particular problem where galvanized steel or iron piping is connected to dissimilar materials, such as brass, in taps and fittings.

The solubility of zinc in water is a function of pH and total inorganic carbon concentrations; the solubility of basic zinc carbonate decreases with increase in pH and concentrations of carbonate species. For low-alkalinity waters, an increase of pH to 8.5 should be sufficient to control the dissolution of zinc.

With galvanized iron, the zinc layer initially protects the steel by corroding preferentially. In the long term, a protective deposit of basic zinc carbonate forms; however, galvanized pipe is also prone to uncontrolled deposition and clogging. Recent findings have shown that lead can accumulate on galvanized pipe particulates and become resuspended by physical disruption, such as water hammer. Protective deposits do not form in soft waters where the alkalinity is less than 50 mg/l as calcium carbonate or waters containing high carbon dioxide concentrations (> 25 mg/l), and galvanized steel is unsuitable for these waters. Electrolytic corrosion can occur where galvanized steel or iron pipes or fittings are connected with copper tube or brass fittings.

ANNEX 6

Supporting information on radionuclides

A6.1 Guidance levels for radionuclides in drinking-water

Table A6.1 Guidance levels for radionuclides in drinking-water

Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a
³ H	10 000	⁷¹ Ge	10 000	¹⁰⁵ Rh	1 000	¹²⁹ Cs	1 000
⁷ Be	10 000	⁷³ As	1 000	¹⁰³ Pd	1 000	¹³¹ Cs	1 000
¹⁴ C	100	⁷⁴ As	100	¹⁰⁵ Ag	100	¹³² Cs	100
²² Na	100	⁷⁶ As	100	^{110m} Ag	100	¹³⁴ Cs	10
³² P	100	⁷⁷ As	1 000	¹¹¹ Ag	100	¹³⁵ Cs	100
³³ P	1 000	⁷⁵ Se	100	¹⁰⁹ Cd	100	¹³⁶ Cs	100
³⁵ S	100	⁸² Br	100	¹¹⁵ Cd	100	¹³⁷ Cs	10
³⁶ Cl	100	⁸⁶ Rb	100	^{115m} Cd	100	¹³¹ Ba	1 000
⁴⁵ Ca	100	⁸⁵ Sr	100	¹¹¹ In	1 000	¹⁴⁰ Ba	100
⁴⁷ Ca	100	⁸⁹ Sr	100	^{114m} In	100	¹⁴⁰ La	100
⁴⁶ Sc	100	⁹⁰ Sr	10	¹¹³ Sn	100	¹³⁹ Ce	1 000
⁴⁷ Sc	100	⁹⁰ Y	100	¹²⁵ Sn	100	¹⁴¹ Ce	100
⁴⁸ Sc	100	⁹¹ Y	100	¹²² Sb	100	¹⁴³ Ce	100
⁴⁸ V	100	⁹³ Zr	100	¹²⁴ Sb	100	¹⁴⁴ Ce	10
⁵¹ Cr	10 000	⁹⁵ Zr	100	¹²⁵ Sb	100	¹⁴³ Pr	100
⁵² Mn	100	^{93m} Nb	1 000	^{123m} Te	100	¹⁴⁷ Nd	100
⁵³ Mn	10 000	⁹⁴ Nb	100	¹²⁷ Te	1 000	¹⁴⁷ Pm	1 000
⁵⁴ Mn	100	⁹⁵ Nb	100	^{127m} Te	100	¹⁴⁹ Pm	100
⁵⁵ Fe	1 000	⁹³ Mo	100	¹²⁹ Te	1 000	¹⁵¹ Sm	1 000
⁵⁹ Fe	100	⁹⁹ Mo	100	^{129m} Te	100	¹⁵³ Sm	100
⁵⁶ Co	100	⁹⁶ Tc	100	¹³¹ Te	1 000	¹⁵² Eu	100
⁵⁷ Co	1 000	⁹⁷ Tc	1 000	^{131m} Te	100	¹⁵⁴ Eu	100
⁵⁸ Co	100	^{97m} Tc	100	¹³² Te	100	¹⁵⁵ Eu	1 000
⁶⁰ Co	100	⁹⁹ Tc	100	¹²⁵ I	10	¹⁵³ Gd	1 000
⁵⁹ Ni	1 000	⁹⁷ Ru	1 000	¹²⁶ I	10	¹⁶⁰ Tb	100
⁶³ Ni	1 000	¹⁰³ Ru	100	¹²⁹ I	1	¹⁶⁹ Er	1 000
⁶⁵ Zn	100	¹⁰⁶ Ru	10	¹³¹ I	10	¹⁷¹ Tm	1 000

Table A6.1 (continued)

Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a	Radio-nuclide	Guidance level (Bq/l) ^a
¹⁷⁵ Yb	1 000	²¹⁰ Pb ^b	0.1	²³¹ U	1 000	²⁴³ Am	1
¹⁸² Ta	100	²⁰⁶ Bi	100	²³² U	1	²⁴² Cm	10
¹⁸¹ W	1 000	²⁰⁷ Bi	100	²³³ U	1	²⁴³ Cm	1
¹⁸⁵ W	1 000	²¹⁰ Bi ^b	100	²³⁴ U ^b	1	²⁴⁴ Cm	1
¹⁸⁶ Re	100	²¹⁰ Po ^b	0.1	²³⁵ U ^b	1	²⁴⁵ Cm	1
¹⁸⁵ Os	100	²²³ Ra ^b	1	²³⁶ U ^b	1	²⁴⁶ Cm	1
¹⁹¹ Os	100	²²⁴ Ra ^b	1	²³⁷ U	100	²⁴⁷ Cm	1
¹⁹³ Os	100	²²⁵ Ra	1	²³⁸ U ^{b,c}	10	²⁴⁸ Cm	0.1
¹⁹⁰ Ir	100	²²⁶ Ra ^b	1	²³⁷ Np	1	²⁴⁹ Bk	100
¹⁹² Ir	100	²²⁸ Ra ^b	0.1	²³⁹ Np	100	²⁴⁶ Cf	100
¹⁹¹ Pt	1 000	²²⁷ Th ^b	10	²³⁶ Pu	1	²⁴⁸ Cf	10
^{193m} Pt	1 000	²²⁸ Th ^b	1	²³⁷ Pu	1 000	²⁴⁹ Cf	1
¹⁹⁸ Au	100	²²⁹ Th	0.1	²³⁸ Pu	1	²⁵⁰ Cf	1
¹⁹⁹ Au	1 000	²³⁰ Th ^b	1	²³⁹ Pu	1	²⁵¹ Cf	1
¹⁹⁷ Hg	1 000	²³¹ Th ^b	1 000	²⁴⁰ Pu	1	²⁵² Cf	1
²⁰³ Hg	100	²³² Th ^b	1	²⁴¹ Pu	10	²⁵³ Cf	100
²⁰⁰ Tl	1 000	²³⁴ Th ^b	100	²⁴² Pu	1	²⁵⁴ Cf	1
²⁰¹ Tl	1 000	²³⁰ Pa	100	²⁴⁴ Pu	1	²⁵³ Es	10
²⁰² Tl	1 000	²³¹ Pa ^b	0.1	²⁴¹ Am	1	²⁵⁴ Es	10
²⁰⁴ Tl	100	²³³ Pa	100	²⁴² Am	1 000	^{254m} Es	100
²⁰³ Pb	1 000	²³⁰ U	1	^{242m} Am	1		

^a Guidance levels were rounded to the nearest order of magnitude by averaging the log scale values (to 10ⁿ if the calculated value was below 3 × 10ⁿ and to 10ⁿ⁺¹ if the value was 3 × 10ⁿ or above). For example, if the calculated value was 2 Bq/L (i.e. 2 × 10⁰), the guidance level was rounded to 10⁰ (i.e. = 1) whereas, if the calculated value was 3 Bq/L (i.e. 3 × 10⁰ or above), the guidance level was rounded to 10¹ (i.e. = 10).

^b Natural radionuclides.

^c The provisional guideline value for uranium in drinking-water is 30 µg/l based on its chemical toxicity for the kidney (see [section 8.5](#)).

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ANNEX 7

Contributors to the development of the *Guidelines for drinking-water quality:* *fourth edition incorporating* *the first, second and third addenda*

This annex lists the names of those who have contributed to the development of the fourth edition of the *Guidelines for drinking-water quality* and to the first, second and third addenda to the fourth edition, through participation at relevant meetings, authorship or peer review of text in the Guidelines themselves or its supporting documents, or through provision of intellectual advice. The list of contributors begins with the first meeting at which the fourth edition was discussed, held in Berlin, Germany, in 2007. Contributors from US government institutions preceded 20 January 2025. All those who contributed to the third edition of the Guidelines as well as the first and second addenda to the third edition, which constitute a major portion of this fourth edition, are listed in Annex 2 of the third edition incorporating the first and second addenda, available on the WHO web site at: <https://www.who.int/publications/i/item/9789241547611>. Sincere apologies are extended to any contributors whose names have inadvertently been omitted from these lists.

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GUIDELINES FOR DRINKING-WATER QUALITY: FOURTH EDITION
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This fourth edition incorporating the first, second and third addenda, of the World Health Organization's *Guidelines for Drinking-water Quality* builds on over 60 years of guidance by WHO on drinking-water quality, which has formed an authoritative basis for the setting of national regulations and standards for water safety in support of public health.

It is the product of significant revisions to clarify and elaborate on ways of implementing its recommendations of contextual hazard identification and risk management, through the establishment of health-based targets, catchment-to-consumer water safety plans and independent surveillance. It reflects the renewed focus on primary prevention.

Significant additional guidance on good practice is presented, incorporating changes introduced in the third edition. Emerging water management issues are comprehensively addressed for a range of circumstances, from household water treatment and safe storage and the bulk supply of water over long distances to the implications of climate change.

Additional risk assessments are presented for a number of new chemical and microbial hazards. Existing reviews on chemicals and waterborne pathogens have been revised to account for new scientific information. The chapter on radiological aspects of drinking-water quality has been comprehensively updated.

Even more than the previous edition, this new edition incorporating the first, second and third addenda, emphasizes achievable practices and the formulation of sound regulations, applicable to low-income, middle-income and industrialized countries alike, from large, to small water supplies.