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Water Treatment and Pathogen Control

Process Efficiency in Achieving Safe Drinking Water

Mark W LeChevallier and Kwok-Keung Au

World Health Organization

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Foreword

Microbial contamination of drinking-water contributes to disease outbreaks and background rates of disease in developed and developing countries worldwide. Control of waterborne disease is an important element of public health policy and an objective of water suppliers.

The World Health Organization (WHO) has developed *Guidelines for Drinking-water Quality*. These guidelines, which are now in their third edition (WHO, 2004), provide an internationally harmonized basis to help countries to develop standards, regulations and norms that are appropriate to national and local circumstances. The latest edition of the WHO *Guidelines for Drinking-water Quality* is structured around an overall "water safety framework", used to develop supply-specific "water safety plans". The framework, which focuses on health protection and preventive management from catchment to consumer, has five key components:

- health-based targets, based on an evaluation of health concerns;
- system assessment to determine whether the drinking-water supply (from source through treatment to the point of consumption) as a whole can deliver water of a quality that meets the health-based targets;

- operational monitoring of the control measures in the drinking-water supply that are of particular importance in securing drinking-water safety;
- management plans that document the system assessment and monitoring plans, and describe actions to be taken in normal operation and incident conditions (including upgrade and improvement, and documentation and communication);
- a system of independent surveillance to verify that the above are operating properly.

Understanding the effectiveness of water treatment is necessary for:

- design of cost-effective interventions
- review of the adequacy of existing structures
- operation of facilities to maximum benefit.

WHO has also developed a series of expert reviews covering various aspects of microbial water quality and health (listed below). This publication forms part of this series of reviews.

- *Managing Water in the Home: Accelerated Health Gains from Improved Water Supply* (M Sobsey, 2002)
- *Pathogenic Mycobacteria in Water: A Guide to Public Health Consequences, Monitoring and Management* (S Pedley et al, eds, 2004)
- *Quantifying Public Health Risk in the WHO Guidelines for Drinking-water Quality: A Burden of Disease Approach* (AH Havelaar and JM Melse, 2003)
- *Safe, Piped Water: Managing Microbial Water Quality in Piped Distribution Systems* (R Ainsworth, 2004)
- *Toxic Cyanobacteria in Water: A Guide to their Public Health Consequences, Monitoring and Management* (I Chorus and J Bartram, eds, 1999)
- *Upgrading Water Treatment Plants* (EG Wagner and RG Pinheiro, 2001)
- *Water Safety Plans* (A Davison et al., 2004)
- *Assessing Microbial Safety of Drinking Water: Improving Apporoaches and Methods* (A Dufour et al., 2003).

Further texts are in preparation or in revision:

- *Arsenic in Drinking-water* (in preparation)
- *Fluoride in Drinking-water* (in preparation)
- *Guide to Hygiene and Sanitation in Aviation* (in revision)
- *Guide to Ship Sanitation* (in revision)
- *Health Aspects of Plumbing* (in preparation)

- *Legionella and the Prevention of Legionellosis* (in preparation)
- *Protecting Groundwaters for Health — Managing the Quality of Drinking-water Sources* (in preparation)
- *Protecting Surface Waters for Health — Managing the Quality of Drinking-water Sources* (in preparation)
- *Rapid Assessment of Drinking-water Quality: A Handbook for Implementation* (in preparation)
- *Safe Drinking-water for Travellers and Emergencies* (in preparation)

Water safety management demands a quantitiative understanding of how processes and actions affect water quality, which in turn requires an understanding of risk assessment. This volume is intended to provide guidance on using risk assessment when selecting appropriate treatment processes, to ensure the production of high quality drinking-water. It is hoped that it will be useful to water utilities, water quality specialists and design engineers.

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This text is one of the supporting documents to the rolling revision of the *WHO Guidelines on Drinking-water Quality*. Its preparation was overseen by the working group on microbial aspects of the guidelines, and thanks are also due to its members:

- Ms T Boonyakarnkul, Department of Health, Thailand (*Surveillance and control*)
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- Prof M Sobsey, University of North Carolina, USA (*Risk assessment*).

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Acronyms and abbreviations used in the text

AOC	assimilable organic carbon
asu	areal standard unit
AWWA	American Water Works Association
AWWARF	AWWA Research Foundation
BDL	below detection limit
BDOC	biodegradable dissolved organic carbon
CC-PCR	cell culture-polymerase chain reaction
cfu	colony forming unit
DAF	dissolved air flotation
DE	diatomaceous earth
DNA	deoxyribonucleic acid
FAC	free available chlorine
FMEA	failure mode and effects analysis
HACCP	hazard analysis critical control point
HPC	heterotrophic plate count
IDDF	integrated disinfection design framework
IFA	immunofluorescence assay
MF	microfiltration

NA	not applicable
NF	nanofiltration
NR	not reported
NTU	nephelometric turbidity unit
PACl	polyaluminium chloride
pfu	plaque forming unit
PVC	polyvinylchloride
RO	reverse osmosis
RNA	ribonucleic acid
SFBW	spent filter backwash
THM	trihalomethane
UF	ultrafiltration
USEPA	United States Environmental Protection Agency
UV	ultraviolet
WHO	World Health Organization
WTP	water treatment plant

Executive summary

This document is part of a series of expert reviews on different aspects of microbial water quality and health, developed by the World Health Organization (WHO) to inform development of guidelines for drinking-water quality, and to help countries and suppliers to develop and implement effective water safety plans.

Contamination of drinking-water by microbial pathogens can cause disease outbreaks and contribute to background rates of disease. There are many treatment options for eliminating pathogens from drinking-water. Finding the right solution for a particular supply involves choosing from a range of processes. This document is a critical review of some of the literature on removal and inactivation of pathogenic microbes in water. The aim is to provide water quality specialists and design engineers with guidance on selecting appropriate treatment processes, to ensure the production of high quality drinking-water. Specifically, the document provides information on choosing appropriate treatment in relation to raw water quality, estimating pathogen concentrations in drinking-water, assessing the ability of treatment processes to achieve health-based water safety targets and identifying control measures in process operation.

Processes for removal of microbes from water include pretreatment; coagulation, flocculation and sedimentation; and filtration. Pretreatment can broadly be defined as any process to modify microbial water quality before, or at the entry to, the treatment plant. Pretreatment processes include application of roughing filters, microstrainers, off-stream storage and bank infiltration, each with a particular function and water quality benefit. Applications of these pretreatment processes include removal of algal cells, high levels of turbidity, viruses and protozoan cysts.

For conventional treatment processes, chemical coagulation is critical for effective removal of microbial pathogens. Together, coagulation, flocculation and sedimentation can result in 1–2 log removals of bacteria, viruses and protozoa. For waters with high levels of algae, care must be taken to remove these organisms without disrupting the cells, which may release liver or nerve toxins. High-rate clarification using solids contact clarification, ballasted-floc, or contact clarification systems can be as, or more, effective than conventional basins for removal of microbes. Dissolved air flotation can be particularly effective for removal of algal cells and *Cryptosporidium* oocysts. Lime softening can provide good microbial treatment through a combination of inactivation by high pH and removal by sedimentation.

Granular media filtration is widely used in drinking-water treatment. It removes microbes through a combination of physical–hydrodynamic properties and surface and solution chemistry. Under optimal conditions, the combination of coagulation, flocculation, sedimentation and granular media filtration can result in 4-log or better removal of protozoan pathogens. However, without proper chemical pretreatment, this type of rapid rate filtration works as a simple strainer and is not an effective barrier to microbial pathogens. Slow sand filtration works through a combination of biological and physical–chemical interactions. The biological layer of the filter, termed *schmutzdecke*, is important for effective removal of microbial pathogens. Precoat filtration was initially developed as a portable unit to remove *Entamoeba histolytica*, a protozoan parasite. In this process, water is forced under pressure or by vacuum through a uniformly thin layer of filtering material, typically diatomaceous earth. As with granular media filtration, proper chemical conditioning of the water improves the treatment efficiency of precoat filtration. In contrast, membrane filtration removes microbial pathogens primarily by size exclusion (without the need for coagulation), and is effective in removing microbes larger than the membrane pore size.

Oxidants may be added to water for a variety of purposes, such as control of taste and odour compounds, removal of iron and manganese, control of zebra mussel and removal of particles. For microbial pathogens, application of strong oxidizing compounds such as chlorine, chlorine dioxide or ozone will act as

disinfectants, inactivating microbial cells through a variety of chemical pathways. Principal factors that influence inactivation efficiency of these agents are the disinfectant concentration, contact time, temperature and pH. In applying disinfectants, it is important to take into account data on CT (disinfectant concentration multiplied by the contact time) for the specific disinfectant. Ultraviolet light (UV) inactivates microorganisms through reactions with microbial nucleic acids and is particularly effective for control of *Cryptosporidium*.

For control of microbes within the distribution system, disinfectants must interact with bacteria growing in pipeline biofilms or contaminating the system. The mechanism of disinfection within the distribution system differs from that of primary treatment. Factors important in secondary disinfection include disinfectant stability and transport into biofilms, disinfectant type and residual, pipe material, corrosion and other engineering and operational parameters.

Performance models can help in understanding and predicting the effectiveness of granular media filtration processes for removal of particles and microbes. Similarly, equations can be useful in predicting microbial inactivation by disinfectants. It is also useful to consider variability in processes and in measurements to determine the overall effectiveness of treatment to control microbial risk. At present, performance models cannot precisely define microbial treatment effectiveness. This leads the operator back to the monitoring and control of critical points within the treatment process. The combined effect of these control measures ensures that the microbial water quality of the treated water meets or surpasses risk goals for the potable water supply.

A water safety plan combines elements of a “hazard analysis and critical control point” (HACCP) approach, quality managment and the “multiple barriers” principle, to provide a preventive management approach specifically developed for drinking-water supply. It can provide a framework for evaluating microbial control measures by helping to focus attention on process steps such as coagulation, filtration and disinfection, which are important for ensuring the microbial safety of water. Many current practices already employ some elements of a water safety plan, and this type of approach is likely to become more clearly defined in water treatment practices in the future.

1
Introduction

1.1 PURPOSE AND SCOPE

This publication is a critical review of removal and inactivation of microbial pathogens by drinking-water treatment processes. Chapters 2 and 3 focus on removal and inactivation processes respectively, in terms of their operational principles, mechanisms and efficiency. Chapter 4 presents performance models for granular filtration and disinfection, two of the most important barriers for microbes, and illustrates how these models can be used to determine the effects of process variables on treatment efficiency. Chapter 5 looks at measures of process variation, including uncertainty in treatment effects and problems associated with the use of surrogates. Finally, Chapter 6 illustrates how an approach based on a water safety framework can be used to minimize microbial hazards in water.

 Water Treatment and Pathogen Control: Process Efficiency in Achieving Safe Drinking Water. Edited by Mark W LeChevallier and Kwok-Keung Au. ISBN: 1 84339 069 8. Published by IWA Publishing, London, UK.

The review focuses on bacteria, viruses, protozoan parasites and microbial toxins, and their removal from source water by various treatment processes. The aim is help water utilities to:

- choose appropriate treatment in relation to raw water quality
- estimate pathogen concentrations in drinking-water
- assess the ability of treatment processes to achieve health-based water safety targets
- identify control measures in process operation.

This review does not attempt to cite all the relevant literature; rather, it highlights information that illustrates the performance of each treatment process. Where possible, it provides quantitative information on the removal or inactivation of pathogenic microorganisms and toxins. Also, it considers (and, where possible, quantifies) interactions between the effects of different treatment processes.

The information is given at different levels of detail:

- The first level estimates the order of magnitude of the expected effect under typical process conditions and proper operating conditions. This level of detail allows simple decision trees for the choice of a treatment chain to be constructed.
- The second level identifies the process parameters (both design and monitoring) that are most relevant to the treatment effect, and quantifies the effect of these parameters. Where possible, mathematical models are used to describe these relations. This level of detail allows control measures and their operational limits to be identified. There is an emphasis on physical and chemical parameters; microbiological indicators are discussed in a separate review (Dufour et al., 2003).
- The third level identifies and quantifies any variability and uncertainty in the treatment effect that is not explained by the process parameters. This level of detail allows exposure to pathogens to be assessed within the framework of a formal risk assessment procedure.

1.2 MULTIPLE BARRIERS

For centuries, the process of providing safe drinking-water has relied on the application of the ‘multiple barrier concept’. Hippocrates (460–354 B.C.), writes in Air, Water and Places — the first treatise on public hygiene, that ‘qualities of the waters differ from one another in taste and weight’. One should ‘consider the waters which the inhabitants use, whether they be marshy and soft, or hard and running from elevated and rocky situations, and then if saltish and unfit for cooking …. for water contributes much to health’ (Baker, 1948).

The concept of multiple barriers for water treatment is the cornerstone of safe drinking-water production. The barriers are selected so that the removal capabilities of different steps in the treatment process are duplicated. This approach provides sufficient backup to allow continuous operation in the face of normal fluctuations in performance, which will typically include periods of ineffectiveness. Having multiple barriers means that a failure of one barrier can be compensated for by effective operation of the remaining barriers, minimizing the likelihood that contaminants will pass through the treatment system and harm consumers. Traditionally, the barriers have included:

- protection of source water (water used for drinking-water should originate from the highest quality source possible);
- coagulation, flocculation and sedimentation;
- filtration;
- disinfection;
- protection of the distribution system.

If these conventional barriers are thought to be inadequate, it may be advisable to consider adding multiple stages of filtration or disinfection.

The benefit of multiple treatment barriers is illustrated by a recent epidemiological study of a karstic groundwater system where one well was filtered and chlorinated while a second was only chlorinated (Beaudeau et al., 1999). Increases in sales of antidiarrheal drugs correlated strongly with lapses in chlorination of the well that had disinfection as the only treatment. In contrast, no effect could be traced to lapses in chlorination of the filtered well. The combination of filtration and chlorination appeared to provide sufficient duplication in removal of contaminants that temporary lapses in disinfection did not generate a measurable adverse outcome (Beaudeau et al., 1999).

1.3 PROCESS CONTROL MEASURES

There are many different microbes that may be of concern in source waters or within the distribution system. Developing a monitoring scheme for each would be an impossible task; therefore, another approach is needed. The food and beverage industry has used the “hazard analysis critical control point” (HACCP) approach to determine the key points within the manufacturing chain where contamination can be measured and prevented. A similar concept can be used by water utilities, to prioritize the key contamination points within the treatment and distribution system (Bryan, 1993; Sobsey et al., 1993). This approach allows utilities to focus their resources on monitoring these points and correcting any deviations from acceptable limits. The latest edition of the World Health Organization (WHO) *Guidelines for Drinking-Water Quality* (WHO,

2004) incorporates such an approach, providing guidance on the development of a water safety plan. The plan is developed using a water safety framework, which combines HACCP principles with water quality management and the multiple barrier concept.

Most microbiological monitoring programs for drinking-water have not been designed using such a framework. However, many of the relevant concepts are found in the overall process control of water treatment plants and distribution systems. For example, maintaining a disinfectant residual within the distribution system can be considered a control procedure.

The water safety framework is not only applicable to microbial monitoring of drinking-water treatment; it can also be applied to aspects such as turbidity, disinfectant residuals, pressure and particle counts. A strength of the framework is that it allows water utilities to allocate limited laboratory resources to monitoring points within the water supply process where the results will provide the greatest information and benefit.

2

Removal processes

This chapter considers various processes for removal of microbes from water. In particular, it discusses:

- *pretreatment* — broadly defined as any process to modify microbial water quality before, or at the entry to, a treatment plant;
- *coagulation, flocculation and sedimentation* — by which small particles interact to form larger particles and settle out by gravity;
- *ion exchange* — used for removal of calcium, magnesium and some radionuclides;
- *granular filtration* — in which water passes through a bed of granular materials after coagulation pretreatment;
- *slow sand filtration* — in which water is passed slowly through a sand filter by gravity, without the use of coagulation pretreatment.

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2.1 PRETREATMENT

This section describes some of the processes that can be used in pretreatment of water (roughing filters, microstrainers, off-stream storage and bank infiltration), each of which has a particular function and water quality benefit. Applications of pretreatment include removal of algal cells, high levels of turbidity, viruses and protozoan cysts The various options for pretreatment may be compatible with a variety of treatment processes, ranging in complexity from simple disinfection to membrane filtration.

2.1.1 Roughing filters

A roughing filter is a coarse media (typically rock or gravel) filter used to reduce turbidity levels before processes such as slow sand filtration, diatomaceous earth (DE) or membrane filtration. The American Water Works Association Research Foundation (AWWARF) has reviewed design variables for roughing filters (Collins et al., 1994). Such filters typically have a filter box divided into multiple sections containing gravel beds of decreasing particle size, inlet and outlet structures, and flow-control devices. Examples of common configurations are shown in Figure 2.1.

Roughing filters have achieved peak turbidity removals ranging from 60 to 90%; generally, the more turbid the water initially, the greater the reduction that can be achieved (Galvis, Fernandez & Visscher, 1993; Collins et al., 1994; Ahsan, Alaerts & Buiteman, 1996). These filters can achieve similar reductions of coliform bacteria. Pilot studies of various roughing filter configurations (horizontal-flow, up-flow and down-flow) reduced faecal coliform bacteria by 93–99.5% (Galvis, Fernandez & Visscher, 1993). These filters were also combined with a dynamic roughing filter (which contains a thin layer of fine gravel on top of a shallow bed of coarse gravel, with a system of underdrains) to pretreat high turbidity events, and achieved faecal coliform removal of 86.3%. When followed by slow sand filtration, the removal reached 99.8%, with an overall combined treatment efficiency of 4.9–5.5 log units. In a five-month pilot study of a medium gravel (5.5 mm) horizontal roughing filter in Texas City, United States of America (USA), the filter removed on average 47% of total bacteria (as measured by epifluorescence microscopy), 37% of the source water algal cells and 53% of the total chlorophyll (Collins et al., 1994). The researchers found that the roughing filters removed clay particles more effectively when the filter was ripened with algal cells. Addition of alum coagulant before treatment with a horizontal roughing filter improved the filter's performance for turbidity, colour, organic carbon, head loss and filter run length (Ahsan, Alaerts & Buiteman, 1996).

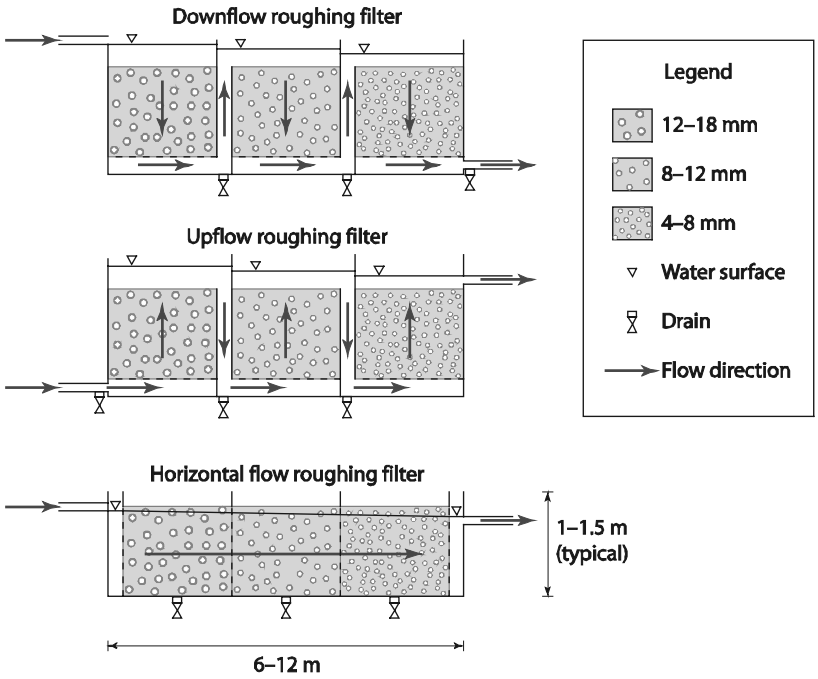


Figure 2.1 Typical roughing filter configurations (Collins et al., 1994)

2.1.2 Microstrainers

Microstrainers are fabric meshes woven of stainless steel or polyester wires, with apertures ranging from 15 to 45 µm (usually 30–35 µm). Such meshes are useful for removing algal cells and large protozoa (e.g. *Balantidium coli*), but have no significant impact on bacteria or viruses. Microstrainers generally remove about 40–70% of algae and, at the same time, about 5–20% of turbidity (Mouchet & Bonnelye, 1998). The performance of microstrainers for specific applications varies, depending on the type of algae present, as summarized in Table 2.1. Although microstrainers can reduce the amount of coagulant needed, they do not remove smaller species or reproductive forms of algae.

Table 2.1 Performance of microstrainers for various algae

Organism	Type	Percentage removal
Diatoms		
Cyclotella	Unicellular	10–70
Stephanodiscus	Unicellular	10–60
Melosira	Filamentous	80–90
Synedra	Unicellular	40–90
Asterionella	Colonial	75–100
Fragilaria	Filamentous	85–100
Chlorophyceae		
Chlorella	Unicellular	10–50
Scenedesmus	Cenobia (4–8 cells)	15–60
Pediastrum	Cenobia (4–64 cells)	80–95
Blue-green algae		
Oscillatoria	Filamentous	40–50
Anabaena	Filamentous	50–70

Adapted from Mouchet & Bonnelye (1998)

2.1.3 Off-stream storage

In this discussion, off-stream storage refers to a storage reservoir that directly or indirectly feeds a potable water intake. The effects of off-stream storage are difficult to generalize because important physical, biological and chemical processes are influenced by hydrological and limnological characteristics of the reservoir. For example, 'round' reservoirs and lowland impoundments influenced by strong winds can be represented as homogeneous biotypes because they are mixed effectively. On the other hand, long reservoirs whose depth increases with length are best represented as a series of interconnected individual basins (Bernhardt, 1995). The characteristics of reservoirs created by construction of a dam will differ from those of a natural or artificial lake.

Oskam (1995) summarized the self-purification processes that improve water quality in off-stream reservoirs (Table 2.2). The major factors that influence these processes are the degree of compartmentalization, the hydraulic residence time, the shape and flow through the reservoir, and the quality of the source water. Certain processes can also degrade water quality; for example, poorer quality of the impounded water can result from failure to:

- manage algal growth;
- control influx of nitrogen, phosphorus or other contaminants;
- limit faecal contamination from run-off of surrounding areas or roosting birds.

Table 2.2 Self-purification processes that improve off-stream reservoir water quality

Type of process	Effects
Physical	Equalization of peak concentrations (e.g. chemicals, microbes) Exchange of oxygen and carbon dioxide with the atmosphere Evaporation of volatile substances (e.g. solvents) Settling of suspended solids and adsorbed substances (e.g. turbidity, heavy metals)
Biological	Biodegradation of organic substances Die-off of faecal bacteria and viruses Nitrification of ammonium to nitrate Denitrification of nitrate to nitrogen Phosphorus elimination by phytoplankton uptake (in pre-reservoirs)
Chemical	Oxidation of divalent iron and manganese Hydrolysis of polyphosphates and organic esters (e.g. phthalates) Photolysis of humic substances and polynuclear aromatic hydrocarbons

Adapted from Oskam (1995)

In a study by Bernhardt (1995), coliform bacteria in dammed reservoirs were reduced by 80–99% when residence times were greater than 40 days, and allochthonous bacteria were reduced by 90–99% when retention times exceeded about 100 days. Kors & Bosch (1995) reported reductions of enteroviruses (1.5 logs), Kjeldahl nitrogen (50%), total phosphorus (60%) and ammonium (70%) for a pumped, off-stream reservoir after about 100 days retention time. Stewart et al. (1997) examined storm events that washed high levels of *Giardia* cysts (up to 17 000 cysts/100 l) and *Cryptosporidium* oocysts (up to 42 000 oocysts/100 l) into receiving reservoirs. Only one of 29 reservoir effluent samples was positive, suggesting that the cysts and oocysts had become trapped in sediments that settled to the bottom of the reservoir, because unattached organisms settle slowly (Medema et al., 1998). Hawkins et al. (2000) reported complete elimination of *Cryptosporidium* spikes (i.e. high concentrations) within three weeks in the 2 million megalitre Lake Burragorang reservoir that provides source water for Sydney (Australia). The authors calculated a settling rate of 5–10 metres/day and postulated that sedimentation was accelerated by oocysts clumping with other suspended particles. In a study of three reservoirs in Biesbosch (Netherlands), storage with long residence times (average 24 weeks) resulted in reductions of 2.3 logs for *Giardia*, 1.4–1.9 logs for *Cryptosporidium*, 2.2 logs for *Escherichia coli* and 1.7 logs for faecal streptococci (Ketelaars et al., 1995; van Breemen & Waals, 1998).

The die-off kinetics for microbes can be modelled as a first-order reaction dependent on the residence time and short-circuiting (i.e. the decrease in hydraulic

residence time in a vessel) (Oskam, 1995). For relatively rapid removal rates (*k*-values > 0.05/day), the degree of compartmentalization has a positive effect on water quality. Therefore, a series of three or four smaller reservoirs would be better than one large impoundment. With estimated *k*-values of 0.07/day for removal of *Giardia* and *Cryptosporidium*, and 0.13/day for enteric viruses, compartmentalization in three or four reservoirs would increase the removal effect to 15–230 times that achieved by a single basin (Oskam, 1995).

For reservoirs with short retention times (and therefore limited self-purification), the raw water pumping schedule can be used to improve water quality, by avoiding periods of source water contamination. For example, in a study of the Delaware River (USA), peak levels of microbial contaminants were associated with high levels of turbidity following rainfall events (LeChevallier et al., 1998). By operating the source water pumps to avoid these peak events, levels of *Giardia* and *Cryptosporidium* 12–16 times higher than normal were avoided.

2.1.4 Bank infiltration

Bank infiltration refers to the process of surface water seeping from the bank or bed of a river or lake to the production wells of a water treatment plant. During the water's passage through the ground, its quality changes due to microbial, chemical and physical processes, and due to mixing with groundwater. The process can also be described as 'induced infiltration,' because the well-field pumping lowers the water table, causing surface water to flow into the aquifer under a hydraulic gradient. Bank infiltration can be accomplished through natural seepage into receiving ponds, shallow vertical or horizontal wells placed in alluvial sand and gravel deposits adjacent to surface waters, and infiltration galleries.

Bank infiltration has been widely used in European countries and is of increased interest in many other countries. Variations on the underground passage concept include soil aquifer treatment, injection of surface water for underground passage and aquifer recharge.

The advantages of bank infiltration are summarized in Table 2.3. The efficiency of the process depends on a number of factors: the quality of the surface water (turbidity, dissolved organic matter, oxygen, ammonia and nutrients), the composition and porosity of the soil, the residence time of the water in the soil and the temperature. This efficiency can vary over time, depending on the difference in level between the source water (e.g. river stage) and groundwater. This difference can influence the degree of groundwater mixing and the residence time of the infiltrated surface water.

Table 2.3 Advantages of bank infiltration

A natural pretreatment step requiring little chemical addition
Reduced turbidity and particles
Removal of biodegradable compounds
Reduction of natural organic matter and less formation of disinfection by-products
Reduction of bacteria, viruses and protozoa
Equalization of concentration peaks (e.g. moderation of spills, temperature, etc.)
Dilution with groundwater

Adapted from Kuhn (1999)

Concern about groundwater under the direct influence of surface water has caused some confusion about how to regard bank infiltration. Clearly, this process is under the direct influence of surface water; however, in the USA, the Surface Water Treatment Rule (USEPA, 1989a) does not consider the infiltration process as contributing to water treatment.

In a study of the Grand River in Ontario (Canada), removal of algae and diatoms ranged from 4.8 to 7.2 logs when the quality of the collection well was compared to the raw water (Clancy & Stendahl, 1997). No *Giardia* or *Cryptosporidium* were detected in the collector wells, although these protozoa were frequently detected in the river water. Figure 2.2 shows the relationship between the concentration of algae and the theoretical flow-path distance for wells along the Great Miami River at Cincinnati (USA), with approximately 1 log reduction for every 8.5 m (28 ft) of separation from the source water (Gollnitz, Cossins & DeMarco, 1997). Schijven and Rietveld (1997) measured the removal of male-specific coliphage, enteroviruses and reoviruses at three infiltration sites, and compared the measured values to those predicted by a virus transport model. They found a 3.1-log reduction of bacteriophage within 2 m (6.6 ft) and a 4.0-log reduction within 4 m (13.2 ft) of very fine dune sand. Phage levels were reduced by 6.2 logs through riverbank infiltration over 30 m (98 ft) of sandy soil. In all cases, enteroviruses and reoviruses were eliminated to below detection limits (> 2.6 to > 4.8 log removals). The virus transport model corresponded reasonably well with the measured results, producing calculated removals ranging from 2.5 to 15 logs.

In studies being conducted by the American Water Works Service Company and the Johns Hopkins University, monitoring of three river bank infiltration systems along the Wabash, Ohio and Missouri rivers (USA) have shown complete removal of *Clostridium* and bacteriophage indicators (Table 2.4) and substantial reductions in biodegradable dissolved organic carbon (BDOC) and assimilable organic carbon (AOC), which can stimulate bacterial growth in distribution system pipelines (Ainsworth, 2004). These data indicate that bank infiltration can be highly effective for removal of microbial contaminants.

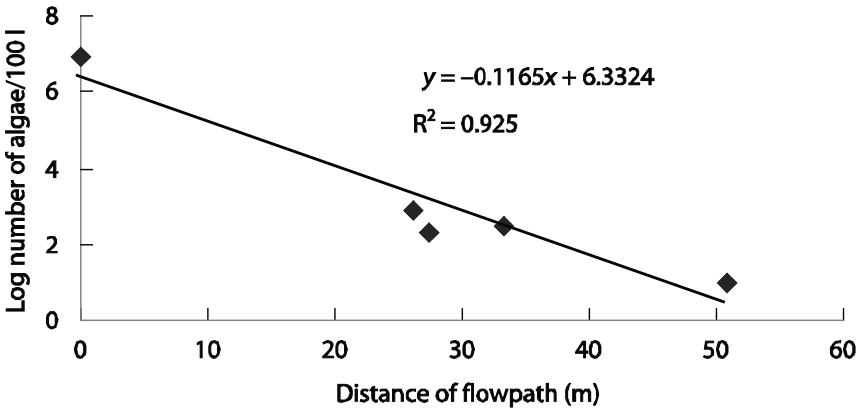


Figure 2.2 Relationship between algae concentration and theoretical flowpath. Adapted from Gollnitz, Cossins & DeMarco (1997)

2.2 COAGULATION, FLOCCULATION AND SEDIMENTATION

Coagulation, flocculation and sedimentation are used in conjunction with subsequent filtration. These processes are summarized below.

- *Coagulation* promotes the interaction of small particles to form larger particles. In practice, the term refers to coagulant addition (i.e. addition of a substance that will form the hydrolysis products that cause coagulation), particle destabilization and interparticle collisions.
- *Flocculation* is the physical process of producing interparticle contacts that lead to the formation of large particles.
- *Sedimentation* is a solid–liquid separation process, in which particles settle under the force of gravity.

Excellent reviews of these processes are available (Gregory, Zabel & Edzwald, 1999; Letterman, Amirtharajah & O'Melia, 1999). With respect to coagulation and flocculation, most bacteria and protozoa can be considered as particles, and most viruses as colloidal organic particles.

Table 2.4 Effects of bank infiltration

Sample	Distance from river (m)	BDOC (mg/l)	Total AOC (μg/l)	Clost-ridium cfu/100 ml	Bacteriophage	
					Somatic pfu/100 ml	Male-specific pfu/100 ml
Site — Terre Haute						
Wabash River	–	1.35	193	253	129	12
Collector	21–27	0.14	23	0.06	< 0.13	< 0.13
Well #3*	122	0.07	15	< 0.13	< 0.13	< 0.13
Site — Jeffersonville						
Ohio River	–	0.35	58	116	46	10
Well #9	61	0.04	32	< 0.13	< 0.13	0.2
Well #2	177	0.03	19	< 0.13	< 0.13	< 0.13
Site — Parkville						
Missouri River	–	0.33	233	132	42	5.5
Well #4	37	0.28	290	< 0.13	< 0.13	< 0.13
Well #5	37	0.25	201	< 0.13	< 0.13	< 0.13

AOC = assimilable organic carbon; BDOC = biodegradable dissolved organic carbon; cfu = colony forming units.
* Water from this well is not dominated by infiltration.

2.2.1 Conventional clarification

Efficiency of conventional clarification

Conventional clarification typically refers to chemical addition, rapid mixing, flocculation and sedimentation (usually in a rectangular basin). Removal of particles depends mainly on the terminal settling velocity of the particles and the rate of basin surface loading or overflow. The efficiency of the sedimentation process may be improved by using inclined plates or tubes. For conventional treatment processes, chemical coagulation is critical for effective removal of microbial pathogens. In the absence of a chemical coagulant, removal of microbes is low because sedimentation velocities are low (Medema et al., 1998). A chemical coagulant destabilizes microbial particles (e.g. by neutralizing or reducing their surface electrical charge, enmeshing them in a floc particle or creating bridges between them) and allows particles to come into contact with one another. Flocculation of microbial particles creates aggregates with sufficient settling velocities to be removed in the sedimentation basin.

When properly performed, coagulation, flocculation and sedimentation can result in 1–2 log removals of bacteria, viruses and protozoa. However, performance of full-scale, conventional clarification processes may be highly

variable, depending on the degree of optimization. For example, in a report summarizing the performance of treatment plants from various countries, average microbial removals for coagulation and sedimentation ranged from 27 to 74% for viruses, 32 to 87% for bacteria (total coliforms or faecal streptococci) and 0 to 94% for algae (Gimbel & Clasen, 1998). It is difficult to interpret full-scale data for *Cryptosporidium* and *Giardia* because these protozoa are found at very low levels, and methods for their detection have limitations (LeChevallier et al., 1991).

Factors that can result in poor clarification efficiency include variable plant flow rates, improper dose of coagulant, poor process control with little monitoring, shear of formed floc, inappropriate mixing of chemicals, poor mixing and flocculation, and inadequate sludge removal (USEPA, 1991). In addition to metallic coagulants (e.g. alum or ferric), it may be necessary to use polymeric coagulation, filter aids or both to produce low turbidity levels (< 0.1 nephelometric turbidity unit, NTU) especially for high-rate filtration (> 2.71 $l/m^2 \cdot s$). Preoxidation with chlorine or ozone can improve particle removal by sedimentation and filtration (Yates et al., 1997; Becker, O'Melia & Croker, 1998). In some cases, treatment plants are being designed with intermediate ozonation, specifically to aid in particle removal by sedimentation and filtration (Langlais, Reckhow & Brink, 1991).

Using jar tests, Bell et al. (2000) reported removal of bacteria (*E. coli* vegetative cells and *Clostridium perfringens* spores) and protozoa (*Giardia* cysts and *Cryptosporidium* oocysts) as typically of 1–2 logs (Figure 2.3). Overall, iron-based coagulants were slightly more efficient than alum (aluminum hydroxide) or polyaluminium chloride (PACl); however, site-specific water-quality conditions had a greater effect on removal efficiencies than did the type of coagulant. Coagulation conditions (i.e. dose, pH, temperature, alkalinity, turbidity and the level and type of natural organic matter) affected the efficiency of removal, with slightly better overall microbial reductions under pH conditions optimal for removal of total organic carbon (i.e. pH 5–6.5).

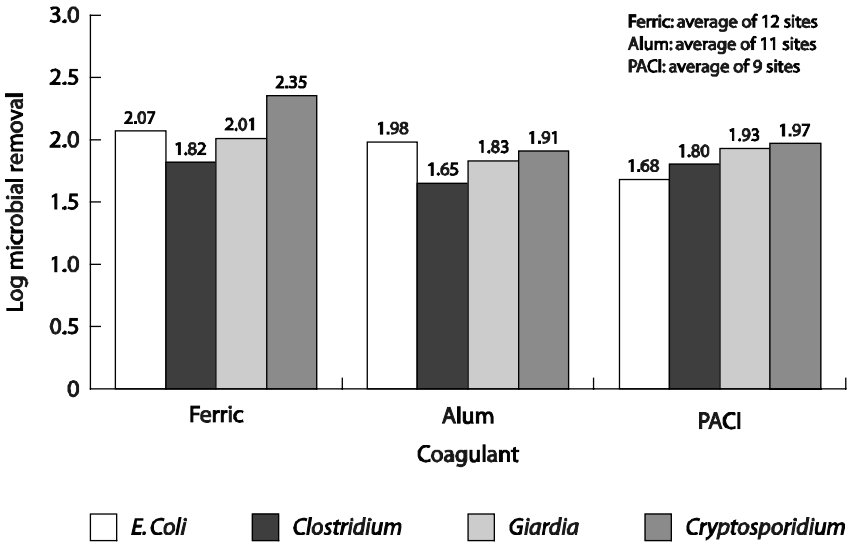


Figure 2.3 Removal of bacteria and protozoa under optimal coagulation conditions. Adapted from Bell et al. (2000)

Viruses

Figure 2.4 shows that different viruses may respond quite differently to coagulation conditions. For example, the bacteriophage MS2 and human enteric poliovirus are removed at a fairly high efficiency (2.6–3.4 logs), whereas the phage PRD-1 and enteric echovirus are removed at a much lower rate (1.1–1.9 logs). The differences in virus removal are most pronounced for alum. Similar differences in virus adsorption have been observed in granulated gels (Mouillot & Netter, 1977). It is evident that the effect of coagulation differs for various viruses, and that it may be unwise to extrapolate the data on viruses to other, untested viruses.

Protozoa

Haas et al. (2000) reviewed data from four bench-scale or pilot-plant studies for coagulation, flocculation and sedimentation of *Cryptosporidium* oocysts. The authors selected data from studies where the coagulant type, coagulant dose, pH, temperature and mixing conditions were described. Using 24 data points, they found that oocyst removal depended on coagulant concentration, polymer concentration and process pH. The model had an excellent fit to the data (R^2 of

0.94); however, the fit decreased when data from other studies were added to the model. The authors concluded that additional data are needed, especially from studies that fully describe coagulation and flocculation conditions.

An optimal coagulation dose is the most important factor for ensuring effective removal of cysts and oocysts by sedimentation and filtration (Logsdon et al., 1985; Al-Ani et al., 1986; Logsdon, 1990; Bellamy et al., 1993). Impaired flocculation was one of the factors in the 1987 outbreak of cryptosporidiosis in Carrollton, Georgia (USA) (Bellamy et al., 1993). In a study of eight water filtration plants, Hendricks et al. (1988) concluded:

> … without proper chemical pretreatment *Giardia* cysts will pass the filtration process. Lack of chemical coagulation or improper coagulation was the single most important factor in the design or operation of those rapid rate filtration plants where *Giardia* cysts were found in finished water … with proper chemical coagulation, the finished water should be free of *Giardia* cysts, have few microscopic particles and have turbidity levels less than 0.1 NTU [nephelometric turbidity units].

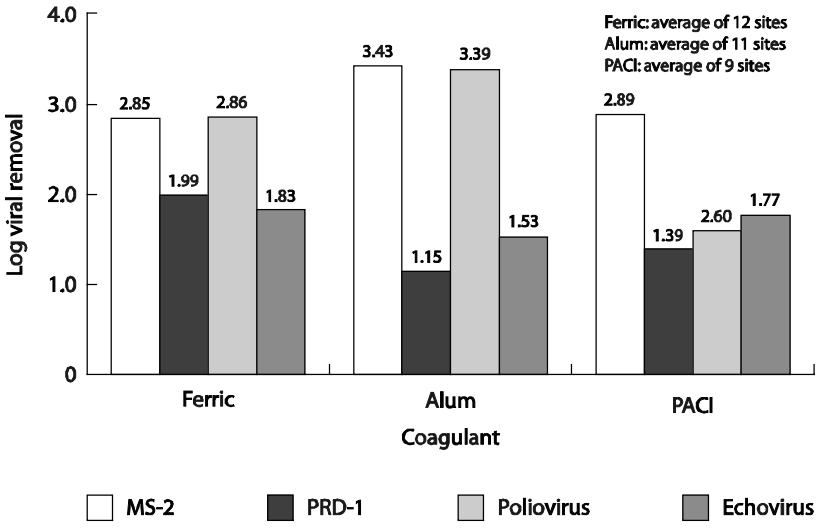


Figure 2.4 Removal of viruses under optimized coagulation conditions. Adapted from Bell et al. (2000).

Algae

Coagulation and sedimentation can be effective for removal of algae, although care must be taken to remove these organisms without disrupting the cells, as this may release liver or nerve toxins. Generally, coagulation appears not to cause the release of algal toxins, provided that oxidants are not added (Yoo et al., 1995b). Coagulation and sedimentation are not very effective at removing algal toxins; studies have shown removal levels ranging from 0 to 49%. However, addition of powdered activated carbon to the clarification process can increase removal levels to 90% or more, depending on the carbon dose, type of carbon, toxin level and organic matrix (Yoo et al., 1995b). A natural coagulant derived from shrimp shells (termed chitosan) was shown to be effective, removing more than 90% of the algae *Chlorella* and *Scenedesmus quadricuda* at neutral to alkaline pH conditions, using chitosan doses of more than 10 mg/l (Chen, Liu & Ju, 1996).

2.2.2 High-rate clarification

High-rate clarification was first used in the 1930s, and it grew in popularity during the 1970s and 1980s. It involves using smaller basins and higher surface loading rates than conventional clarifiers, and is therefore referred to as high-rate clarification. Processes include floc-blanket sedimentation (also known as 'solids-contact clarification'), ballasted-floc sedimentation, and adsorption or contact clarification.

In floc-blanket sedimentation, a fluidized blanket increases the particle concentration, thus increasing the rate of flocculation and sedimentation. Ballasted-floc systems combine coagulation with sand, clay, magnetite or carbon to increase the particle sedimentation rate. Adsorption or contact clarification involves passing coagulated water through a bed where particles attach to previously adsorbed material.

High-rate clarifiers can be as effective as or even more effective than conventional basins for removal of microbes. The choice of an appropriate blanket polymer is important for optimal operation (Gregory, Zabel & Edzwald, 1999). Bell, Bienlien & LeChevallier (1998) reported turbidity removals of 98% for a solids-contact, sludge blanket clarifier (raw water turbidity 20–50 NTU, settled water 0.6–0.75 NTU), 89% for internal slurry recirculation (raw water turbidity 4–10 NTU, settled water 0.5–0.9 NTU) and 61% for circular floc-blanket purification unit clarification (raw water turbidity 1.2–16 NTU, settled water average 0.97 NTU). Baudin & Laîné (1998) evaluated three full-scale treatment plants and found complete removal (> 2–2.8 logs) of *Giardia* and *Cryptosporidium* by pulsator clarifiers. The units produced a 1.0–2.7 log removal of turbidity. Other investigators (Hall, Pressdee & Carrington, 1994)

have reported similar efficiencies for floc-blanket clarifiers. A combination of preozonation and use of a solids-contact sludge blanket reportedly improved clarification of *Giardia* and *Cryptosporidium*-sized particles by about 1.5–2.5 logs (Wilczak et al., 1991). Pilot plant studies of a sand ballasted-floc system showed effective removal of turbidity and particle counts (Jeschke, 1998). In addition, microscopic particulate analysis of raw and settled water showed an average 3.9-log removal of algae, and 4.5-log removal of diatoms (Jeschke, 1998). Floc formed on magnetic particles can be rapidly removed by using magnets within the sedimentation process (Gregory, Maloney & Stockley, 1988; Bolto, 1990; Anderson et al., 1993). The magnetic particles can be collected and regenerated for reuse.

2.2.3 Dissolved air flotation

In dissolved air flotation (DAF), bubbles are produced by reducing pressure in a water stream saturated with air. The rising bubbles attach to floc particles, causing the agglomerate to float to the surface, where the material is skimmed off (Gregory, Zabel & Edzwald, 1999). DAF can be particularly effective for removal of algal cells and *Cryptosporidium* oocysts. It is most applicable to waters with heavy algal blooms or those with low turbidity, low alkalinity and high color, which are difficult to treat by sedimentation because the floc produced has a low settling velocity.

The effectiveness of DAF for treating algal-laden, humic, coloured water is illustrated by the comments of Kiuru (1998), who indicated that the only type of treatment plants built in Finland since the mid-1960s have been DAF plants. A 1.8-log removal of the algae *Aphanizomenon* and *Microcystis* was achieved by pilot-scale DAF. Similar results (1.4–2.0 log removals) have been obtained in full-scale studies (Mouchet and Bonnelye, 1998). DAF is also effective in the removal of cell-associated algal toxins (Mouchet and Bonnelye, 1998).

Plummer, Edzwald & Kelley (1995) reported that, depending on the coagulant dose, DAF achieved 2–2.6 log removal of *Cryptosporidium* oocysts, whereas conventional sedimentation resulted in 0.6–0.8 log removal. The performance of DAF for oocyst removal depended on the pH, coagulant dose, flocculation time and recycle ratio of the saturated water stream. Other researchers have confirmed the effectiveness of DAF for oocyst removal, particularly when polyelectrolyte coagulant aids are added to help stabilize the floc (Hall, Pressdee & Carrington, 1994).

2.2.4 Lime softening

Precipitative lime softening is a process in which the pH of the water is increased (usually through the addition of lime or soda ash) to precipitate high concentrations of calcium and magnesium. Typically, calcium can be reduced at pH 9.5–10.5, although magnesium requires pH 10.5–11.5. This distinction is important because the pH of lime softening can inactivate many microbes at the higher end (e.g. pH 10–11), but may have less impact at more moderate levels (e.g. pH 9.5). In precipitative lime softening, the calcium carbonate and magnesium hydroxide precipitates are removed in a clarifier before the water is filtered. The microbial impact of lime softening can, therefore, be a combination of inactivation by elevated pH and removal by settling.

Logsdon et al. (1994) evaluated the effects of lime softening on the removal and disinfection efficiency of *Giardia*, viruses and coliform bacteria. Coliform bacteria in river water (spiked with raw sewage) were inactivated by 0.1 log at pH 9.5, 1.0 log at pH 10.5 and 0.8–3.0 logs at pH 11.5 for 6 hours at 2–8°C. Bacteriophage MS2 was sensitive to lime softening conditions, demonstrating more than 4-log inactivation in the pH range of 11–11.5 within 2 hours. Hepatitis A virus was reduced by 99.8% when exposed to pH 10.5 for 6 hours. Poliovirus was the most resistant virus tested, requiring exposure to a pH level of 11 for 6 hours to achieve a 2.5-log inactivation. Reductions were less than 1 log when exposed for 6 hours to a pH of less than 11. The viability of *Giardia muris* cysts (measured by excystation) was not significantly affected by exposure to pH 11.5 for 6 hours. *Cryptosporidium* viability (measured using dye exclusion) was not affected by exposure to pH 9 for 5 hours (Robert, Campbell & Smith, 1992).

Jar tests of precipitative lime softening at pH 11.5 resulted in 4-log removal of viruses and bacteria, and 2-log removal of *Giardia* and *Cryptosporidium*, due to combined effects of removal by sedimentation and inactivation through high pH (Bell et al., 2000). Limited full-scale data suggest that 2-log removal can be achieved through sedimentation by precipitative lime softening (Logsdon et al. 1994).

2.2.5 In-line coagulation

In-line coagulation can be used with high-quality source waters (e.g. those where turbidity and other contaminant levels are low). The coagulants are added directly to the raw water pipeline before direct filtration. Typically, the coagulants are added before an in-line static mixer, and it is not necessary to use a basin for sedimentation. In-line coagulation permits the particle destabilization

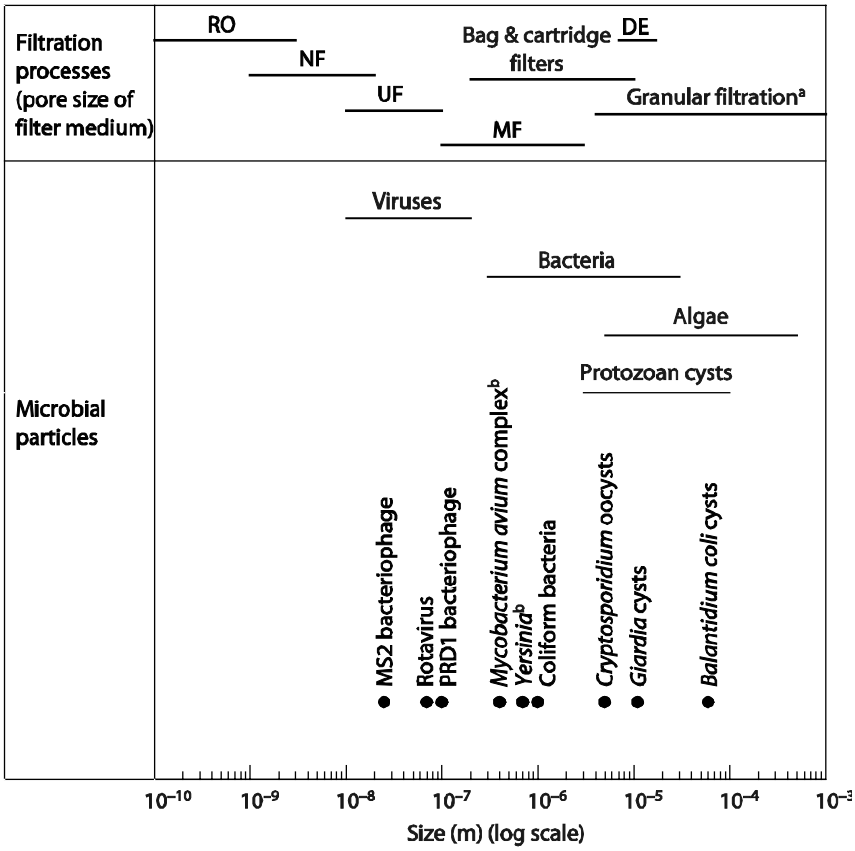
necessary for effective particle removal by filtration, but does not remove microbes by sedimentation.

2.3 ION EXCHANGE

Ion exchange is a treatment process in which a solid phase presaturant ion is exchanged for an unwanted ion in the untreated water. The process is used for water softening (removal of calcium and magnesium), removal of some radionuclides (e.g. radium and barium) and removal of various other contaminants (e.g. nitrate, arsenate, chromate, selenate and dissolved organic carbon). The effectiveness of the process depends on the background water quality, and the levels of other competing ions and total dissolved solids. Although some ion exchange systems can be effective for adsorbing viruses and bacteria (Semmens, 1977), such systems are not generally considered a microbial treatment barrier, because the organisms can be released from the resin by competing ions. Also, ion exchange resins may become colonized by bacteria, which can then contaminate treated effluents (Flemming, 1987; Parsons, 2000). Backflushing and other rinsing procedures, even regeneration, will not remove all of the attached microbes. Impregnation of the resin with silver suppresses bacterial growth initially, but eventually a silver-tolerant population develops. Disinfection of ion exchange resins using 0.01% peracetic acid (1 hour contact time) has been suggested (Flemming, 1987).

2.4 FILTRATION

Various filtration processes are used in drinking-water treatment. Filtration can act as a consistent and effective barrier for microbial pathogens. Figure 2.5 shows the most commonly used filtration processes in potable water treatment, the pore size of the filter media and the sizes of different microbial particles. These size spectra are useful for understanding removal mechanisms and efficiencies, and for developing strategies to remove microbes by different filtration processes.



DE = diatomaceous earth; MF = microfiltration; NF = nanofiltration; RO = reverse osmosis; UF = ultrafiltration.

Figure 2.5 Pore size of filter medium and size of microbial particles

2.5 GRANULAR HIGH-RATE FILTRATION

Granular media filtration is the most widely used filtration process in drinking-water treatment. A comprehensive review of granular media filtration processes is provided by Cleasby and Logsdon (1999). Under optimal conditions, a combination of coagulation, flocculation, sedimentation and granular media filtration can result in 4 logs or better removal of protozoan pathogens with chlorine-resistant cysts. This section discusses granular filtration other than slow

sand filtration (described in Section 2.6). Slow sand filtration is discussed separately because the low filtration rate (< 0.4 m/h) used in this process means that the design and operating criteria, and the mechanisms for removal of microbes are considerably different from those of 'rapid-rate' granular filtration.

2.5.1 Design of granular filtration

In granular filtration, water passes through a filter consisting of a packed bed of granular materials. Microbes or microbe-associated particles are removed as they deposit on the filter medium. The removal occurs within the granular medium (depth filtration) rather than on the top layer only (cake filtration). After a period of operation, the head loss increases (i.e. the pressure increases) or the effluent quality deteriorates to an unacceptable level. The filter then has to be cleaned by backwashing, after which it performs poorly during a 'ripening period' before achieving a stable level of performance. Passage of microbial pathogens during the ripening period can be very high. Various strategies are used to minimize this effect, including:

- *filter to waste* — wasting the initial filtered water;
- *slow start* — limiting the initial filtration rate until the filtrate quality is acceptable;
- *delayed start* — leaving the filter inactive for a time following backwash, before bringing it into operation;
- *filter aid* — adding a filter aid to the wash water supply.

Granular filters can be constructed as monomedium (e.g. silica sand), dual media (e.g. anthracite coal and sand) and trimedia (e.g. coal, sand and garnet). Granular activated carbon is used when both filtration of particles and adsorption of organic compounds are desired. Depending on raw water quality, granular filtration can be operated in three different modes:

- *conventional*, which includes addition of coagulants (rapid mixing), flocculation (slow mixing), sedimentation and filtration;
- *direct filtration*, in which the sedimentation step is omitted;
- *in-line filtration,* in which both flocculation and sedimentation steps are omitted.

Conventional treatment is appropriate for most source waters, whereas direct and in-line filtration are used for raw waters with a consistently good quality (low turbidity and colour).

2.5.2 Mechanism of action of granular filtration

Removal of microbial pathogens by granular filtration does not rely on physical processes alone. Comparing the pore size of granular filters with the size of most types of microbe (as in Figure 2.5), it is evident that effective removal of microbes by granular filtration cannot rely on physical straining alone, at least at the initial stage of a filter run. The removal of particles by granular filtration is considered to involve two steps: transport of particles from suspension to filter medium, followed by attachment of particles to the medium (Yao, Habibian & O'Melia, 1971).

The transport step depends on the physical and hydrodynamic properties of the system. Transport mechanisms include diffusion, interception and sedimentation. Factors such as size and density of microbes, size and depth of filter medium, and filtration rate affect transport efficiency. In the case of motile microorganisms, an additional mechanism is the active movement of the cell (Hozalski & Bouwer, 1998). Attachment is determined by the surface and solution chemistry of the system (Tobiason & O'Melia, 1988). Unfavorable interactions between particles and the filter medium must be avoided so that particles can attach to the medium. Chemical coagulation is used before filtration to destabilize particles; this step is the single most important factor in determining filtration efficiency. Without proper chemical pretreatment, rapid rate filtration works as a simple strainer and is not an effective barrier for microbial pathogens.

2.5.3 Importance of chemical coagulation pretreatment

The importance of chemical coagulation pretreatment for removal of microbes by granular filtration has been emphasized by numerous studies. Al-Ani et al. (1986) conducted a pilot-scale filtration study for low turbidity waters (< 1 NTU). Without chemical pretreatment, the removal by filtration averaged 69% (0.51 logs) for *Giardia* and 28% (0.14 logs) for turbidity. Adding alum and polymer filter aids increased the removal efficiency to more than 95% (1.30 logs) for *Giardia*, 99% (2 logs) for total coliform bacteria and 70% (0.52 logs) for turbidity. Other process variables such as filtration mode (direct and in-line filtrations), filter media (monomedium with sand, and dual-media with sand and anthracite) and temperature (5°C and 18°C) did not significantly affect the filtered water quality. Varying the filtration rate from 5 to about 20 m/h had little effect on removal of *Giardia*, total coliform bacteria and heterotrophic bacteria, but increased turbidity in the filtered water.

Robeck, Clarke & Dostal (1962) reported that, when alum was used as a coagulant, the removal of poliovirus type 1 by a pilot-scale dual-media filter was more than 98% (1.70 logs). Ongerth (1990) conducted pilot studies for conventional and in-line filtration. Without any chemical addition, removal of *Giardia* cysts averaged 75% (0.60 logs) for conventional treatment and 64% (0.44 logs) for in-line filtration. With optimal chemical pretreatment, the removal increased to 98% (1.70 logs) for conventional treatment and 93.6% (1.19 logs) for in-line filtration.

Nieminski & Ongerth (1995) evaluated the removal of *Cryptosporidium* oocysts and *Giardia* cysts over two years at pilot and full-scale filtration plants, operated under direct filtration and conventional treatment modes. Dual-media filters with anthracite and sand were used. *Giardia* and *Cryptosporidium* were effectively removed when coagulation conditions produced filtered water of low turbidity (0.1—0.2 NTU). Under optimal coagulation conditions, the average removal of *Giardia* was 3.3 logs or more, and the average removal of *Cryptosporidium* was 2.3 logs or more in both pilot and full-scale plants, regardless of the treatment modes (direct filtration or conventional treatment). The authors also investigated correlations between the removal of cysts and surrogate parameters. A high correlation was found between removal of cysts and particles of 4–7 μm and 7–11 μm ($R^2 \geq 0.79$). There was a lower correlation between removal of *Giardia* and *Cryptosporidium* and removal of turbidity ($R^2 \leq 0.64$). Particle counting was a better indicator of cyst and oocyst removal than turbidity. Log removal of seeded cysts did not correlate with log removal of heterotrophic bacteria ($R^2 \leq 0.08$), suggesting that heterotrophic plate counts (HPCs) are not a good surrogate to use in evaluating cyst removal. A recent WHO publication (Bartram et al., 2003) discusses the significance of HPCs for water quality and human health.

2.5.4 Effect of filter media design

Swertfeger et al. (1999) evaluated the effect of filter media design on cyst and oocyst removal. Designs included monomedium (sand with a depth of 750 mm), fine dual-media (anthracite and sand with depths of 900 mm and 300 mm, respectively) and deep dual-media (anthracite and sand with depths of 1500 mm and 300 mm, respectively). The feedwater to the pilot systems was taken from the effluent of a sedimentation unit in a full-scale water treatment plant and was in optimal coagulation condition. The authors found no statistical difference in the filtration performance for the different media. Removal of *Giardia* was 4.4 logs or better, with greater removal efficiency

in the summer than in the winter. Removal of *Cryptosporidium* was similar in summer and winter, and averaged 2.7 logs or more.

Payment et al. (2000) reported water-quality monitoring results for a full-scale conventional water treatment plant using dual-media filters, with coagulation provided by alum and activated silica. Prechlorination was applied at 1 mg/l. The results confirmed that a properly operated conventional treatment plant provided a substantial barrier to microbial pathogens. *Giardia* cysts were detected in only 1 of 32 filtered water samples, with a mean removal of 3.6 logs after filtration (including removal by coagulation and sedimentation). Removal of *Cryptosporidium* oocysts was lower than for *Giardia*. Oocysts were detected in 7 of 32 filtered water samples, with a mean removal of 2 logs. *Clostridium perfringens* was detected in 9 of 33 filtered water samples, with average removal of 4.4 logs. No human enteric virus was detected in 32 filtered water samples, with average removal of 3.1 logs (assuming that the concentration of humic enteric virus in filtered water was equal to the detection limit). Somatic coliphage were detected in 24 of 32 filtered water samples, with average removal of 3.5 logs.

2.5.5 Importance of filter backwash

When solids accumulate within a filter bed, they create a resistance to flow. This resistance is measured as loss of head (pressure increase) for the filter bed. The filter is backwashed, usually with finished water, to remove the accumulated particles. The need for backwashing may be determined using various criteria — a terminal head loss, a fixed time interval, or a breakthrough of solids (measured as turbidity or particle counts). Options for disposal of the spent filter backwash water may include discharge to a sewer or a receiving stream. Because backwash water may contain disinfectants and other chemicals that may be harmful to the biological life of a stream, direct discharge to streams may be restricted. Similarly, discharge to sewers may be restricted, based on the constituents and total quantity of the backwash water.

For many water treatment plants, particularly in arid or water-scarce areas with limited raw water resources, it is often necessary to reuse backwash water. When the water is recycled, accumulation of microbial and algal contaminants is a concern. For example, algal toxins may be released from stored treatment sludges when the overlying water is recycled (Drikas et al., 2001). Because of the resistance of oocysts to conventional disinfectants, *Cryptosporidium* has been a major concern for the handling and operation of recycled process streams. Table 2.5 summarizes data on the

occurrence and concentration of *Giardia* cysts and *Cryptosporidium* oocysts in filter backwash water. The level of treatment required for spent filter backwash water before recycle will vary from site to site depending on the treatment process and water-quality objectives. Equalization of the recycle flow and sedimentation of the backwash solids, aided by the addition of a polymer coagulant, is sufficient to reduce cyst concentrations to raw water levels in most cases (Cornwell & Lee, 1993; Arora, Di Giovanni & LeChevallier, 1999; McTigue et al., 2000).

2.6 SLOW SAND FILTRATION

The use of slow sand filtration to protect drinking-water consumers from microbial risk was well established more than 100 years ago. Two of the earliest successful cases were reductions in cholera in Altona (Germany) and typhoid fever in Lawrence, Massachusetts (USA) in the 1890s (Bellamy et al., 1985). Numerous disease outbreaks due to chlorine-resistant protozoan pathogens in the past two decades have increased interest in slow sand filtration because of its ability to remove parasites.

2.6.1 Design and action of slow sand filters

Slow sand filtration involves passing water through a sand filter by gravity at a very low filtration rate, without the use of coagulation pretreatment. The filter typically consists of a layer of sand supported on a layer of graded gravel. Typical design criteria for slow sand filtration are given in Table 2.6. Detailed design guidelines can be found in Hendricks (1991). As water passes through the filter, microbes and other substances are removed. The removal mechanisms are not well understood, although they are believed to be a combination of biological, physical and chemical mechanisms (Weber-Shirk & Dick, 1997ab). Specific mechanisms may include biological action (e.g. ciliate protozoa acting as bacterial predators), attachment of microbes to sand media (e.g. by electrochemical forces and through bridging by microbial extracellular polymers) and physical straining.

Table 2.5 *Giardia* and *Cryptosporidium* occurrence in filter backwash water

Reference	Location (No. of WTPs sampled)	Sample type (No. of samples)	*Cryptosporidium* oocysts/100l		*Giardia* cysts/100l	
Rose et al. (1986)	USA (2)	SFBW (2)	Sample 1: 686,900 Sample 2: 2,430,600		NR	
Colbourne (1989)	Thames Water, UK (1)	Raw (unknown) SFBW (1) Supernatant [a] (1)	0.2–1400 > 1,000,000 >100,000		NR	
Rose et al. (1991)	USA (17 states)	SFBW (subset of 257 samples)	217[b]		NR	
LeChevallier et al. (1991)	USA (66 in 14 states)	Raw Initial SFBW	7–108 57–61 times raw water level		4–32 12–16 times raw water level	
Cornwell & Lee (1993, 1994)	USA (2)	Plant 1: Raw (1) Mixed influent [c] (1) SFBW (1) Supernatant [a] (1)	Round 1 6 40 902 141	Round 2 140 45 850 750	Round 1 3 7 1350 86	Round 2 BDL NR BDL BDL

Table 2.5 (continued) *Giardia* and *Cryptosporidium* occurrence in filter backwash water

Reference	Location (No. of WTPs sampled)	Sample type (No. of samples)	*Cryptosporidium* oocysts/100l		*Giardia* cysts/100l	
Cornwell & Lee (1993, 1994)	USA (2)	Plant 2:	Round 1	Round 2	Round 1	Round 2
		Raw (1)	13	20	290	60
		Mixed influent [c](1)	30	476	160	79
		SFBW (1)	16,613	NR	16,513	NR
		Supernatant [a](1)	80	420	70	BDL
Karanis, Schoenen & Seitz (1996)	Germany (1)	*Centrifugation method*				
		Raw (8 positive out of 12)	0.8 to 109			
		SFBW (8 positive out of 11)	1–69		NR	
		Cartridge filter				
		SFBW (33 positive out of 39)	0.8 to 252			
Karanis, Schoenen & Seitz (1998)	Germany (1)	SFBW [d](1)	150		NR	
States et al. (1995)	Pittsburgh, USA (1)	Raw (11 positive out of 15)	43		42	
		Filtered (2 positive out of 15)	0.4		BDL	
		SFBW (8 positive out of 15)	321			
		SFBW (2 positive out of 15)			59	

Table 2.5 (continued) *Giardia* and *Cryptosporidium* occurrence in filter backwash water

Reference	Location (No. of WTPs sampled)	Sample type (No. of samples)	*Cryptosporidium* oocysts/100l	*Giardia* cysts/100l
Arora, Di Giovanni & LeChevallier (1999)	USA (25)	*IFA method*		
		Raw (17 positive out of 146)	108	
		Raw (44 positive out of 146)		89
		SFBW (7 positive out of 148)	175	
		SFBW (12 positive out of 148)		203
		CC-PCR method [e]	Qualitative method	NA
		Raw (6 positive out of 122)	Qualitative method	NA
		SFBW (9 positive out of 121)		

BDL = below detection level; CC-PCR = cell culture-polymerase chain reaction; IFA = immunofluorescence assay; NA = not applicable; NR = not reported; SFBW = spent filter backwash.

Notes:
[a] Supernatant from settling basin treating spent filter backwash water
[b] Geometric mean concentration
[c] Sample after addition of recycle stream
[d] Sample taken 10 minutes after start of backwash cycle
[e] Cell culture-polymerase chain reaction method identifies live, infectious *Cryptosporidium*

Table 2.6 Typical design criteria for slow sand filtration

Design criterion	Normal range
Filtration rate	0.04–0.4 m/h
Sand media	
Depth	0.5–1.5 m
Effective size	0.15–0.40 mm
Uniformity coefficient	1.5–3.6
Gravel media	
Depth	0.2–1 m
Graded	Fine to coarse (top to bottom)

Source: Letterman, 1991; Cleasby & Logsdon (1999)

Removal of particles by slow sand filtration occurs predominantly, if not entirely, in a thin layer on the top of the sand bed. This biologically active layer, composed of living and dead microorganisms and macroorganisms, is termed *schmutzdecke*. As operation progresses, deposited materials and biological growth on the sand medium increase the head loss across the filter. When the head loss reaches the operational limit (normally 1–2 m), the filter is removed from service. It is then usually cleaned by scraping about 2 cm of accumulated material and sand from the top layer of the sand bed, before being returned to service. A typical filter run is from one to six months, depending on the raw water quality and filtration rate. After the sand bed is reduced to a lowest acceptable depth by repeated scrapings, it is necessary to replace the sand down to the gravel support level.

2.6.2 Protection provided by slow sand filtration

Slow sand filtration can provide some degree of protection against microbial pathogens. As coagulation pretreatment is not required, slow sand filtration has little maintenance or chemical cost. If the raw water has a high concentration of suspended particles or algae, physical pretreatment processes (e.g. roughing filter or microstrainers) can be used to prevent clogging of the filter and maintain a reasonable filter run period.

Removal of microbes

In a review by Ellis (1985), virus removal ranging from about 1 to 5 logs was reported for bench and full-scale slow sand filters. Various studies have reported the effective removal of bacteria and protozoa by slow sand filtration in pilot and full-scale systems.

In a pilot-scale study, a new filter removed 0.82 logs of total coliform bacteria and more than 1.7 logs of *Giardia* (Bellamy et al., 1985). Once a microbiological population was established within the sand bed (after two weeks), the removal of total coliforms increased to 4 logs and no *Giardia* was detected in the filtered water. The calculated cyst reduction was more than 2.6 logs, depending on influent cyst concentration. Similar results were found in another pilot study, where the removal of total coliform bacteria, heterotrophic bacteria and turbidity increased with the biological activity of the *schmutzdecke* (Bellamy, Hendricks & Logsdon, 1985).

In a full-scale study of a slow sand filter in Empire, Colorado (USA), *Giardia* cysts were detected in almost half of the influent samples, but not in the effluent (Seelaus, Hendricks & Janonis, 1986). In a full-scale study for three slow sand filtration plants in Idaho (USA), no samples positive for *Giardia* were found in the filtered water from two of the three treatment plants (Tanner & Ongerth, 1990). For the one positive sample found in one plant, 1-log removal of *Giardia* was achieved. In the same study, removal of total coliforms and faecal coliforms varied from 84.35 to 99.5% (0.81–2.30 logs) and from 48.1 to 70.0% (0.29–0.52 logs), respectively. Removal of heterotrophic bacteria (as measured by HPC) varied from 65.8 to 91.0% (0.47–1.05 logs). These differences in removal efficiency were influenced by raw water quality, filtration rate, media size and depth. Removal of *Cryptosporidium* by slow sand filtration is often more difficult than removal of *Giardia*. In a full-scale study in British Columbia, Fogel et al. (1993) reported that the average removal of *Giardia* was 93% (1.16 logs) but was only 48% (0.28 logs) for *Cryptosporidium*.

Removal of turbidity

Although the removal of microbes by slow sand filtration can be substantial, reduction of turbidity may be site specific. In one pilot study, turbidity removal was 97.8% (1.66 logs) or more after a filter-ripening period of about two days (Cleasby, Hilmoe & Dimitracopoulos, 1984); similar to the removal of total coliform bacteria (≥ 99.4%) and chlorophyll-a (≥ 95%). Another pilot study found a 27–39% (0.14–0.22 log) removal of turbidity, whereas the reduction of *Giardia* was up to 4 logs (Bellamy et al., 1985). The authors concluded that the low removal of turbidity was due to the fine clay particles present in the raw water, which penetrated the filter. In a full-scale study, turbidity removal was between 0 and 63% (0.43 logs), due to the fine particles present in the raw water and to the large fraction (4% by weight) of fines in the new sand media used in the study (Tanner & Ongerth, 1990). The fact that slow sand filtration can achieve effective removal of microbial pathogens but not necessarily decreased turbidity indicates that turbidity may not be a suitable surrogate for evaluation of the removal of pathogens by slow sand filtration.

2.7 PRECOAT FILTRATION

Precoat filtration was developed by the US Army during World War II as a portable unit for the removal of *Entamoeba histolytica* (a protozoan parasite prevalent in the Pacific war zone) from drinking-water. The process involves forcing water under pressure or by vacuum through a uniformly thin layer of filtering material precoated onto a permeable, rigid, supporting structure (referred to as a septum). Precoat materials include DE and perlite, with DE more commonly used in drinking-water treatment. As water passes through the filter media and septum, the precoat materials (filter cake) capture microbes and other particles, mainly by physical straining. Often, a "bodyfeed" solution containing the filter media slurry is added continuously to the system, to maintain the permeability of the filter cake. As the cake becomes thicker due to the captured particles, head loss increases until further filtration is impractical. The filter cake is removed from the support septum and disposed of. The filter is then cleaned and precoated with a new layer of coating materials, and a new filter cycle starts. A detailed design and operating manual for precoat filtration has been published by the American Water Works Association (AWWA, 1995).

Because the major removal mechanism is physical straining, efficiency of precoat filtration depends to a large extent on the grade (size) of the coating materials and on the size of the microbes. Other factors influencing the removal efficiency are chemical pretreatment of the filter media, filtration rate and bodyfeed rate. Chemical pretreatment of the raw water is usually not necessary; however, the raw water must be of high quality (low turbidity) to maintain a reasonable filter run time.

2.7.1 Removal of microbes

Diatomite grades used for drinking-water treatment have a mean pore diameter of 7–17 µm (Figure 2.5). Precoat filtration can remove protozoan parasites such as *Giardia* very effectively. A pilot study showed complete removal of *Giardia* for both coarse and fine grades of DE over a wide range of operating conditions (Lang et al., 1986). Removal of *Cryptosporidium* can be significant, but because this organism is smaller than *Giardia*, it is more difficult to remove. Removal of *Cryptosporidium* oocysts by a bench-scale DE filter ranged from 3.60 to 6.68 logs, depending on the media grade and the filtration rate (Ongerth & Hutton, 1997). In a pilot-plant study, filtration with DE gave a consistently complete removal of *Giardia* cysts and a 3-log removal of *Cryptosporidium* oocysts (Schuler & Ghosh, 1990).

2.7.2 Importance of chemical pretreatment

Precoat filters remove smaller microbial particles (e.g. bacteria and viruses) less effectively than they do parasites, unless the coating materials are chemically pretreated; for example, with aluminium or iron coagulants, or with cationic polymers. In the pilot study by Schuler & Ghosh (1990) mentioned above, removal of coliforms with untreated DE was about 0.36 logs, increasing to 0.82 logs with a coating of alum at 1 mg/g DE, and to 2 logs at 3 mg/g DE. This increase was probably due to the trapping of bacteria by the alum. A similar beneficial effect was observed using cationic polymers; at 3.5 mg/g DE, removal of coliforms increased to 3.3 logs. The authors concluded that this increase in removal could be due to an increased site density on the polymer-coated DE for adsorption of negatively charged coliforms. A similar improvement in removal of bacteria was reported for the pilot study conducted by Lang et al. (1986). Alum coating of DE increased removal of total coliforms from 0.16 logs to 1.40 logs, and of HPC bacteria from 0.36 logs to 2.30 logs. Removal of viruses also increased with chemical pretreatment of filter cake (Brown, Malina & Moore, 1974). The removal of bacteriophage T2 and poliovirus was about 90% for an uncoated filter, but increased to more than 98% (1.7 logs) when the filter cake was coated with ferric hydrate or polyelectrolytes.

2.8 MEMBRANE FILTRATION

In membrane filtration, a thin semipermeable film (membrane) is used as a selective barrier to remove contaminants from water. There are very few contaminants that cannot be removed by membrane processes. For the past two decades, the use of membrane filtration in drinking-water treatment (including pathogen removal) has been growing, due to increasingly stringent drinking-water regulations and decreasing costs of purchasing and operating membrane filters.

The membrane processes most commonly used to remove microbes from drinking-water are microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO). Detailed descriptions of the fundamentals, design and operation of these processes are available (AWWA, 1996; Taylor & Wiesner, 1999). Table 2.7 summarizes these processes, including operating pressure, pore size, primary application and the type of microorganism that can be removed. Not all of these processes are used primarily for removal of pathogens. For example, RO is used mainly for desalination and NF for softening and for removal of precursors of disinfectant by-products. Nevertheless, the ability to remove pathogens broadens the application of these types of filter when used for these other treatment objectives.

Table 2.7 Major membrane filtration processes used in drinking-water treatment

Type	Operating pressure [a] (kPa)	Pore size [b] (μm)	Primary applications	Microbes removed
MF	30–50	≥ 0.1	Removal of particles and turbidity	Algae, protozoa and most bacteria
UF	30–50	≥ 0.01	Removal of dissolved nonionic solutes	Algae, protozoa, most bacteria and viruses
NF	500–1000	≥ 0.001	Removal of divalent ions (softening) and dissolved organic matter	Algae, protozoa, most bacteria and viruses
RO	1000–5000	≥ 0.0001	Removal of monovalent ions (desalination)	Algae, protozoa, most bacteria and viruses

MF = microfiltration; NF = nanofiltration; RO = reverse osmosis; UF = ultrafiltration

[a] All these are pressure-driven processes; the driving force is a pressure difference across the porous membranes.

[b] Pore size is sometimes described as molecular weight cut-off, which is the degree of exclusion of a known solute, determined under a given set of test conditions in the laboratory. Also see Figure 2.5 for pore size.

[c] These membranes are usually made from organic materials such as cellulose acetate and its derivatives, polyamides, polypropylene and other polymers.

[d] Membranes are assembled in different configurations, with hollow fibre and spiral wound the two most common.

Source: Adapted from AWWARF (1996), Taylor & Weisner (1999).

Membrane filtration removes microbial pathogens mainly by size exclusion; that is, microbes larger than the membrane pores are removed. Chemical coagulation is not usually needed before membrane treatment for the removal of microbes. However, some degree of pretreatment is usually employed to reduce membrane fouling (caused by accumulation of chemicals, particles and biological growth on membrane surfaces) and to avoid membrane degradation from chemical attack (caused by hydrolysis and oxidation). Fouling reduces membrane productivity, and membranes must be chemically cleaned to restore productivity.

Examples of pretreatment processes are microstraining, pH adjustment and addition of biocides (chlorine or copper sulfate). If the source water is of poor quality, advanced pretreatment systems (e.g. conventional coagulation–sedimentation–filtration or other membrane processes) may also be necessary.

Based on pore size, the order of effectiveness of microbial removal is RO, NF, UF and MF, with RO being the most effective. However, this is not always the case, because differences in membrane material or configuration, or failure

in the membrane can affect microbial removal. Discussion of the removal efficiency of these different membrane processes follows.

2.8.1 Microfiltration

MF membranes have pores of 0.1 µm or more (Table 2.7). Theoretically, MF can remove protozoa, algae and most bacteria very effectively, and this has been confirmed in a number of studies, some of which are discussed below. However, factors such as bacteria growing in the membrane systems can lead to poor removal of bacteria. Viruses, which are 0.01–0.1 µm in size, can generally pass through MF membranes, but may be removed by the membrane if they are associated with large particles.

Numerous pilot studies have directly evaluated the removal of *Giardia*, *Cryptosporidium* and other specific microbial pathogens by MF. For example, an extensive study using three MF membranes with pore sizes 0.08–0.22 µm found that *Giardia* and *Cryptosporidium* in the filtered water were below detection levels (< 1 cyst or oocyst/l) with two of the membranes (corresponding to log removals of > 4.7 to > 7.0 for *Giardia* and > 4.4 to > 6.9 for *Cryptosporidium*) (Jacangelo, Adham & Laîné, 1995). No cysts or oocysts were detected in the effluent, indicating that the difference in removal efficiency was a function of the feeding concentration. In the case of the membrane where cysts were detected in the filtered water, the membrane seal was defective, but even so it achieved removal of 4.6 logs for *Giardia* and 4.2 logs for *Cryptosporidium*. The authors concluded that MF could act as an absolute barrier to protozoan cysts, provided that the membrane remained intact. As expected, removal of MS2 bacteriophage by these MF membranes was less than 1 log, because the phage is 0.025 µm and the pore size of the membranes is 0.08–0.22 µm.

In another pilot study, MF membranes with an average pore size of 0.2 µm resulted in significant removal of cyst-sized particles (Karimi, Vickers & Harasick, 1999). The removal of *Giardia*-sized (5–15 µm) particles averaged 3.3–4.4 logs. The removal of *Cryptosporidium*-sized (2–5 µm) particles was lower, averaging 2.3–3.5 logs. These removals were a function of the spiking particle concentration and were independent of the membrane flux used (114–170 l/m^2 per hour). Algae were also effectively removed; the feed water contained 275–700 areal standard unit (asu) count of algae and 10–12.7 µg/l of chlorophyll-a, but these were reduced to below detection (< 25 asu and < 0.5 µg/l, respectively) in the filtered water. However, the HPCs from the filtered water exceeded those of the feed water, probably due to the growth of microorganisms in the pilot system.

A pilot study using two MF membranes with nominal sizes of 0.1 and 0.2 μm also confirmed the complete removal of *Giardia* and *Cryptosporidium* by MF, with neither of these organisms detected in the filtered water (States et al., 1999). Hollow fibre membranes with a nominal pore size of 0.2 μm were used in a pilot-scale study using MF membranes for treating filter backwash water (Parker et al., 1999), pretreated in sedimentation tanks. The MF membranes reliably produced water with turbidity below 0.1 NTU, compared to an influent turbidity of 12.4–88 NTU. Average removal efficiency was 99.46% (2.27 logs) for particles in the size range 3–15 μm and 91.3% (1.06 logs) for heterotrophic bacteria. The MF membranes were also challenged with different microbes. Removal efficiencies were 5.3 logs for *Cryptosporidium parvum*, 6.4 logs for algae, more than 4.3 logs for total coliforms, 3.3 logs for heterotrophic bacteria, more than 3.5 logs for aerobic spores, 2.7 logs for total culturable virus and 3.7 logs for male-specific coliphage.

Excellent removal of turbidity, oocyst-sized particles and indicator bacteria was found in a full-scale study of a 19 000 m^3/day MF plant using 0.2 μm MF membranes (Yoo et al., 1995a). Turbidity of up to 100 NTU was observed in the raw water; however, the finished water was always 0.05 NTU or less. Removal of oocyst-sized (4–10 μm) particles was greater than 3 logs. Neither total nor faecal coliforms were detected in any of the finished water samples. During a subsequent seeded challenge study, greater than 6-log removal was observed for both *Giardia* and *Cryptosporidium* at a flux rate of 0.94 gpm/m^2 (AWWARF, 1999).

2.8.2 Ultrafiltration

UF membranes have pores of 0.01 μm or more, small enough to remove some viruses in addition to bacteria and protozoa (Table 2.7). In the bench and pilot-scale studies discussed above, Jacangelo, Adham & Laîné (1995) found that UF, like MF, could act as an absolute barrier to protozoan cysts as long as membranes remained intact. Three UF membranes (with molecular weight cut-offs of 100 000–500 000 daltons, corresponding to pore sizes of 0.01–0.05 μm) were used in the studies. Neither *Giardia* nor *Cryptosporidium* were detected in the filtered water (corresponding to log removals of > 4.7 to > 7.0 for *Giardia* and > 4.4 to > 7.0 for *Cryptosporidium*). Removal of viruses by UF was significantly better than removal by MF, and depended essentially on the pore size of the membranes. The membranes with the lowest molecular weight cut-offs achieved the highest removal efficiency (6 log or higher) for MS2 bacteriophage in both bench and pilot-scale studies. The authors also concluded that, although physical sieving was the main mechanism for the removal of

protozoan pathogens by UF and MF, cake layer formation and changes in the fouling of the membrane also contributed to the removal of viruses.

A pilot study to investigate the removal of particles and indicator bacteria from two surface water supplies used a UF membrane with a molecular weight cut-off of 100 000 daltons (Jacangelo et al., 1989). The membrane effectively removed particles, turbidity, total coliforms and heterotrophic bacteria, and produced filtered water with turbidity less than 0.04 NTU. Particle removal was from 2.6 logs to greater than 4.6 logs, depending on influent particle concentration. No coliforms were detected in the finished water. Influent HPCs of 4–4500 cfu/ml were reduced to < 1–5 cfu/ml in the effluent. The authors concluded that the heterotrophic bacteria in the filtered water were due primarily to the regrowth of bacteria in the membrane system.

A systematic pilot study to evaluate the use of UF to remove microbial pathogens from four different source waters used membranes with a molecular weight cut-off of 100 000 daltons (Jacangelo et al., 1991). Removal efficiencies for *Giardia muris*, coliforms, heterotrophic bacteria and MS2 bacteriophage were determined. *Giardia muris*, total coliform bacteria and MS2 bacteriophage in the filtered water were below detection (corresponding to reduction efficiencies of > 4 logs, > 7 logs and > 6.5 logs respectively). Differences in water quality or changes in operating parameters did not affect the removal capabilities of the process, but maintenance of membrane integrity was critical to assuring process efficiency. Loss of membrane integrity (fibre breakage) was associated with the detection of both *Giardia muris* and MS2 bacteriophage in the permeate water. Heterotrophic bacteria were found in the permeate water, but this was due to colonization of a section of the sample tap piping rather than to penetration of the bacteria through the membrane.

2.8.3 Nanofiltration and reverse osmosis

The pore sizes of NF and RO membranes are smaller than those of UF membranes. However, NF and RO alone are seldom used to remove microbial pathogens because MF or UF are more cost-effective and can achieve a similar degree of microbial removal. Not surprisingly, there is far less literature on the removal of microbial pathogens by NF and RO than by MF and UF. Representative examples are discussed below.

Bench-scale study

A bench-scale study evaluated virus removal by five different RO membranes (Adham et al., 1998). MS2 bacteriophage was used as the model virus, seeded at concentrations of 10^3–10^8 plaque-forming units (pfu)/ml. Virus reduction was from 2.7 logs to more than 6.5 logs. For the membrane with the highest removal

efficiency, no MS2 was found in filtered water (detection limit < 1 pfu/ml). The authors concluded that an RO membrane was not always an absolute barrier to viruses, and that the levels of removal achieved by each membrane varied, depending on the membrane type and manufacturer.

Pilot-scale study

A pilot study to investigate the efficiency of integrated membrane systems used *Bacillus subtilis* endospores as a surrogate for *Cryptosporidium* and *Giardia* to challenge eight different integrated membrane systems (Owen et al., 1999). The systems included two different NF membranes with two different MF membranes as pretreatment, with and without in-line coagulation pretreatment. The systems did not completely remove spores, but gave overall cumulative removals of 8.0–11.0 logs. There was no difference in spore removal with or without in-line coagulation, but membrane configuration and membrane film significantly affected spore removal. The MF membranes, configured as hollow fibres, achieved 5.6–5.9 log removal of spores. The NF membranes, with an average pore size two orders of magnitude less than the MFs and a spiral wound configuration, achieved 2.2–4.5 logs removal.

The authors concluded that a hollow fibre configuration, which simply seals membrane fibres in a straight line, was unlikely to leak. In contrast, the spiral configurations crease membrane envelopes, and include feed stream and permeate stream spacers. The creases and spacers could compromise membrane integrity. Spore removal by the composite thin film NF membrane exceeded that of the cellulose acetate NF membrane by about 2 logs.

Full-scale studies

Full-scale studies to evaluate the removal of microbial pathogens by integrated membrane systems using NF as the major treatment unit have been reported by Lovins et al. (1999) and Gullick et al. (2000). Two composite thin film NF membranes and one cellulose acetate NF membrane with molecular weight cut-offs of 100–300 daltons were used. Protozoa (*Cryptosporidium* oocysts and *Giardia* cysts), bacteria (*Clostridium perfringens* spores) and bacteriophage (MS2 and PRD1) were used to challenge the different NF membranes. Similar to the finding by Owen et al. (1999), the two composite thin film NF membranes were significantly more effective than the cellulose acetate NF membrane at removing microbes. Removals of about 5.5 logs were achieved with the thin film membranes, with complete removal in more than half of the tests. This compared to removals of about 2 logs with the cellulose acetate membrane, which produced complete removal in less than 10% of the tests.

The efficiency of a UF membrane with molecular weight cut-off of 100 000 daltons was also investigated in this study. The observed microbial removal performance for the UF was similar to that of the two composite thin film NF membranes, and significantly higher than that of the cellulose acetate NF membrane. The authors suggested that this was due partly to the configurations of the membrane (hollow fibre for UF and spiral wound for NF). Integrated membrane systems with different configurations were tested in the study. Pretreatment (before NF) included conventional coagulation followed by sedimentation and sand filtration, hollow fibre MF with pore size 0.2 μm and hollow fibre UF with a molecular weight cut-off of 100 00 daltons. As expected, the highest pathogen removals were achieved by integrated membrane systems with composite thin film NF and UF pretreatment, with 6.3–11.0 log removals. However, some membranes did not remove microbes completely, even at relatively low feed concentrations, indicating that integrated membrane systems are not necessarily absolute barriers to pathogens.

2.9 BAG, CARTRIDGE AND FIBROUS FILTERS

A bag filter is one that has a non-rigid fabric medium for the filter. Water flow is usually pressure-driven from the inside of the filter bag to the outside. A cartridge filter is one that has a rigid fabric medium or membrane for the filter. In this type of filter, water flow is usually pressure-driven from the outside of the filter to the inside. Bag and cartridge filters are often developed for small systems and for point-of-use filtration applications. They are also sometimes applied as a pretreatment process for membrane filtration.

Bag filters and cartridge filters remove microorganisms by physical straining. The removal efficiency thus depends primarily on the pore size of the filter medium and on the size of the microbes. A typical pore size range is from 0.2 μm to about 10 μm. The pore size of the filter medium is usually designed to be small enough to remove protozoa such as *Cryptosporidium* and *Giardia*. Submicron particles, including viruses and most bacteria, can pass through the filters. As water passes through a bag or cartridge filter, pressure drop increases to a level impractical for operation. The bag or cartridge is then replaced by a clean one.

Since the removal mechanism is physical straining, chemical pretreatment is usually not required for bag filters and cartridge filters. Straining of large compressible particles can blind the filters and reduce filter life. High turbidity and algae can also clog these filters. These processes are therefore only appropriate for high-quality waters. A prefiltration process may be employed to remove large particles.

In principle, microbes larger than the pore size of the medium will be captured by the filters. The nominal size of a filter medium reported by the manufacturers represents an average size — there is often a pore size distribution, meaning that some pores will be larger than the nominal size. Furthermore, some biological particles such as cysts and oocysts do not have hard shells. These microbes may deform slightly, especially under pressure, allowing them to squeeze through small pores. Li et al. (1997) studied the removal of *Cryptosporidium* oocysts using field-scale bag filtration. A bag filter with a single layer of polypropylene fabric, with a nominal pore size of 1 μm, removed an average of 0.42 log of *Cryptosporidium*. This means that 38% of seeded *Cryptosporidium* oocysts passed through the filter, probably for the reasons mentioned above. When the bag filter was changed to multiple layers, the log removal increased to 1.41.

Arora, Di Giovanni & LeChevallier (1999) evaluated the performance of fibrous filters to treat recycled filter backwash water. Two filters, one made of polybutylene terephthalate with a nominal sizing of 5 μm and the other made of nylon with a nominal sizing of 2 μm, were used. The recycled backwash water was pretreated in a sedimentation tank. Pilot runs with the polybutylene terephthalate filter resulted in an average removal of 3 logs for *Cryptosporidium*, 0.5 logs for *Giardia* and 1 log for *Clostridium*. The nylon filter achieved an average removal of 3 logs for *Cryptosporidium*, 1.2 logs for *Giardia* and 1.5 logs for *Clostridium*.

3

Inactivation (disinfection) processes

This chapter covers the various disinfection processes used in drinking-water treatment to inactivate pathogenic microbes. It looks first at factors affecting the efficiency of disinfection process, and then goes on to consider the following disinfection processes:

- *pretreatment oxidation* — in which oxidants are added to water early in the treatment process.
- *primary disinfection* — a common component of primary treatment of drinking-water, and important because granular filter media do not remove all microbial pathogens from water
- *secondary disinfection* — used to maintain the water quality achieved at the treatment plant throughout the distribution system up to the tap.

3.1 FACTORS AFFECTING DISINFECTION

The principal factors that influence disinfection efficiency are disinfectant concentration, contact time, temperature and pH. Disinfectant concentration and

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contact time are integral to disinfection kinetics and the practical application of the CT concept (CT being the disinfectant concentration multiplied by the contact time). Development and derivations of this disinfection model are discussed in the modelling section below. Temperature, over the range appropriate for drinking-water, affects the rate of disinfection reactions according to the Arrhenius equation, although this may not hold for certain disinfectants at low temperatures. The pH of the disinfectant solution affects the reaction kinetics. For example, the disinfection efficiency of free chlorine is increased at lower pH values, whereas that of chlorine dioxide is greater at alkaline pH levels. Monochloramine is formed within seconds in the pH range 7–9, at chlorine to ammonia nitrogen ratios of less than 5:1 and at 25°C; it is also the predominant species when the pH is greater than 5.

Other factors that influence microbial sensitivity to disinfection include attachment to surfaces, encapsulation, aggregation and low-nutrient growth. Increased resistance to disinfection may result from attachment or association of microorganisms to various particulate surfaces, including:

- macroinvertebrates (*Crustacea, Nematoda, Platyhelminthes* and *Insecta*) (Tracy, Camarena & Wing, 1966; Levy, Cheetham & Hart, 1984);
- particles that cause turbidity (LeChevallier, Evans & Seidler, 1981; Ridgway & Olson, 1982);
- algae (Silverman, Nagy & Olson, 1983);
- carbon fines (LeChevallier et al., 1984; Camper et al., 1986);
- glass (Olivieri et al., 1985).

Ridgway & Olson (1982) showed that the majority of viable bacteria in chlorinated water were attached to particles. Stewart & Olson (1986) reported that aggregation of *Acinetobacter* strain EB22 increased its resistance to disinfection, making the bacteria 100-fold more resistant to hypochlorous acid (HOCl) and 2.3-fold more resistant to monochloramine. Several investigators have isolated encapsulated bacteria from chlorinated water (Reilly & Kippin, 1983; Clark, 1984) and concluded that production of the extracellular capsule helped protect bacteria from chlorine. Carson et al. (1972) reported that *Pseudomonas aeruginosa* grown in distilled water was markedly more resistant to acetic acid, glutaraldehyde, chlorine dioxide and a quaternary ammonium compound than cells cultured on tryptic soy agar. Similarly, Berg, Matin & Roberts (1981) and Harakeh et al. (1985) found that bacteria grown in a chemostat at low temperatures and submaximal growth rates caused by nutrient limitation (conditions thought to be similar to the natural aquatic environment) were resistant to several disinfectants.

3.2 PRETREATMENT OXIDATION

Water utilities often add oxidants early in the treatment process to:

- maximize the contact time with the oxidant;
- oxidize compounds for subsequent removal by the treatment process (e.g. iron or manganese);
- provide initial treatment in sufficient time for water to be further treated if necessary (e.g. oxidation of taste and odour compounds);
- control growth of microorganisms and higher organisms (e.g. zebra mussels) on intake structures and in treatment basins;
- improve particle removal in subsequent clarification and filtration processes.

There are a number of potential problems with pretreatment oxidation. Variable source water conditions mean that variable or high levels of oxidant may be needed. This may lead to overdosing of pre-oxidants, which can result in "pink coloured" water when potassium permanganate is misapplied. Also, the process can produce oxidation by-products such as trihalomethanes (THMs), haloacetic acids and bromate. For example, in using chlorine as a pretreatment oxidant, chlorinated by-products can form rapidly. This often limits the application of chlorine to a later stage of the treatment process, when precursor material has been removed. A further problem is that oxidants can lyse algal cells, releasing liver or nerve toxins, or creating objectionable tastes or odours. (Yoo et al., 1995b; Chorus & Bartram, 1999).

One concern with using pre-oxidants for disinfection is that particulate material may interfere with microbial inactivation. Such material protects bacteria and viruses from disinfectants by creating an instantaneous disinfectant demand (preventing the maintenance of a disinfectant residual in subsequent treatment steps) and by shielding the microbe from the oxidant (Hoff, 1978; LeChevallier, Evans & Seidler, 1981; Berman, Rice & Hoff, 1988).

The effect of particulate material on disinfection of cysts or oocysts has not been widely evaluated. Di Giovanni & LeChevallier (2000) studied the effect of turbidity on disinfection of *Cryptosporidium parvum* oocysts by chlorine dioxide or permanganate, and found that particulate material did not interfere with disinfection once the increase in oxidant demand had been satisfied (Figure 3.1). The authors hypothesized that protozoan cysts were too large to be completely shielded from the disinfectant.

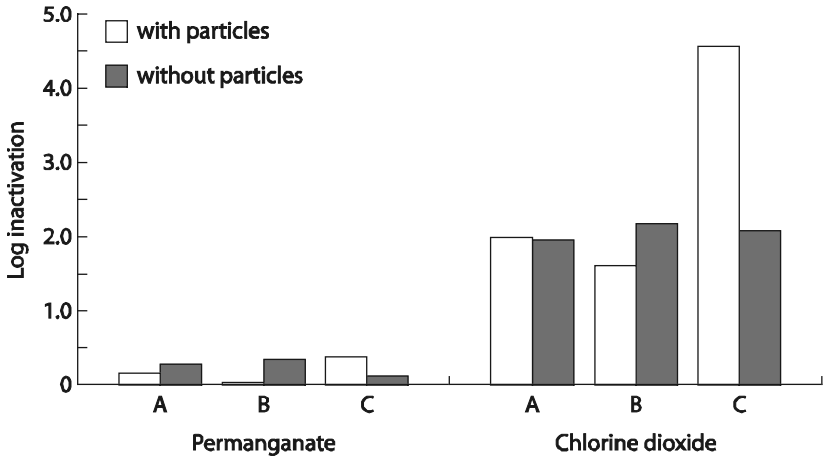

Figure 3.1 Effect of particulate material on disinfection of *Cryptosporidium*. Potassium permanganate applied at 2400 mg/min l^{-1}, chlorine dioxide applied at 120 mg/min l^{-1}. Source: Adapted from Di Giovanni & LeChevallier (2000).

3.3 PRIMARY DISINFECTION

A disinfection barrier is a common component of primary treatment of water. Primary disinfection is typically a chemical oxidation process, although ultraviolet (UV) irradiation and membrane treatment are gaining increased attention. This section looks at different types of disinfectant — chlorine, monochlorine, chlorine dioxide, ozone, UV light and mixed oxidants — in terms of their effectiveness against various pathogenic microorganisms. Further information on selecting a disinfection strategy for a piped distribution system can be found in the WHO publication *Safe piped water: Managing microbial water quality in piped distribution systems* (Ainsworth, 2004).

3.3.1 Chlorine

Mode of action

Chlorine gas and water react to form HOCl and hydrochloric acid (HCl). In turn, the HOCl dissociates into the hypochlorite ion (OCl^-) and the hydrogen ion (H^+), according to the following reactions:

(1) $Cl_2 + H_2O \Leftrightarrow HOCl + HCl$

(2) $HOCl \Leftrightarrow H^+ + OCl^-$

The reactions are reversible and pH dependent:

- between pH 3.5 and 5.5, HOCl is the predominant species
- between about pH 5.5 and 9.5, both HOCl and OCl^- species exist in various proportions
- above pH 8, OCl^- predominates.

The OCl^- and HOCl species are commonly referred to as free chlorine, which is extremely reactive with numerous components of the bacterial cell. HOCl can produce oxidation, hydrolysis and deamination reactions with a variety of chemical substrates, and produces physiological lesions that may affect several cellular processes. Baker (1926) theorized that chlorine destroys microorganisms by combining with proteins to form N-chloro compounds. Chlorine was later found to have powerful effects on sulfhydryl groups of proteins (Green & Stumpf, 1946, Knox et al., 1948; Venkobachar, Iyengar & Rao, 1977) and to convert several α-amino acids by oxidation into a mixture of corresponding nitriles and aldehydes (Patton et al., 1972). The exact product of the reaction depends on chlorine concentration and pH (Dakin 1916, 1917; Wright 1936).

Cytochromes, iron-sulfur proteins and nucleotides are highly vulnerable to oxidative degradation by HOCl, suggesting that chlorine causes physiological damage primarily to the bacterial cell membranes (Venkobachar, Iyengar & Rao, 1977; Camper & McFeters, 1979; Haas & Engelbrecht, 1980; Albrich, McCarthy & Hurst, 1981). Respiration, glucose transport and adenosine triphosphate levels all decrease in chlorine-treated bacteria (Venkobachar, Iyengar & Rao, 1977; Camper & McFeters, 1979; Haas & Engelbrecht, 1980). Electron microscopy of chlorinated bacteria has demonstrated morphological changes in the cell membrane (Zaske, Dockins & McFeters, 1980). In addition, chlorination can kill microbes by disrupting metabolism (Wyss, 1961) and protein synthesis (Pereira et al., 1973), or by modifying purine and pyrimidine bases and thus causing genetic defects (Patton et al., 1972; Hoyano et al., 1973; Haas & Engelbrecht, 1980).

Nearly 100 years of chlorination for disinfection of drinking-water has demonstrated the effectiveness of this process for inactivation of microbial pathogens, with the notable exception of *Cryptosporidium*.

Effectiveness of chlorine against bacteria and viruses

Table 3.1 shows CT values for 99% (2-log) inactivation of bacteria for various chlorine-based disinfectants. In general, the heterotrophic bacteria grown in

drinking-water were more resistant to disinfection than the laboratory-grown *Escherichia coli*.

Table 3.1 Comparative efficiency of disinfectants for the production of 99% bacterial inactivation in oxidant demand-free systems

	Escherichia coli			Heterotrophic bacteria		
Disinfectant	pH	Temp (°C)	CT mg/min l^{-1}	pH	Temp (°C)	CT mg/min l^{-1}
Hypochlorous acid	6.0	5	0.04	7.0	1–2	0.08 ± 0.02
Hypochlorite ion	10.0	5	0.92	8.5	1–2	3.3 ± 1.0
Chlorine dioxide	6.5	20	0.18	7.0	1–2	0.13 ± 0.02
	6.5	15	0.38	8.5	1–2	0.19 ± 0.06
	7.0	25	0.28			
Monochloramine	9.0	15	64	7.0	1–2	94.0 ± 7.0
				8.5	1–2	278 ± 46.0

Source: Adapted from LeChevallier, Cawthon & Lee (1988)

Certain bacteria show a high level of resistance to free chlorine. Spore-forming bacteria such as *Bacillus* or *Clostridium* are highly resistant when disseminated as spores. Acid-fast and partially acid-fast bacteria such as *Mycobacterium* and *Nocardia* can also be highly resistant to chlorine disinfection. One study showed that nearly all of the bacteria surviving chlorine disinfection were Gram positive or acid fast (Norton & LeChevallier, 2000), possibly because Gram-positive bacteria have thicker walls than Gram-negative ones.

Enteric viruses are generally more resistant to free chlorine than enteric bacteria, with CT values for 99% inactivation ranging from about 2 to more than 30 mg/min l^{-1} (Figure 3.2). Viruses associated with cellular debris or organic particles may require high levels of disinfection due to the protective nature of the particle surface (Akin & Hoff, 1986; Hoff, 1992). Chlorination effectively inactivates viruses if the turbidity of the water is less than or equal to 1.0 nephelometric turbidity unit (NTU). It requires a free chlorine residual of 1.0 or greater for 30 minutes, and a pH of less than 8.0. For groundwaters where turbidities are generally low, or for filtered surface water, White (1999) suggests the CT guidelines for the 99% virus inactivation shown in Table 3.2. These data are based on conservative interpretation of inactivation data for Coxsackie A2.

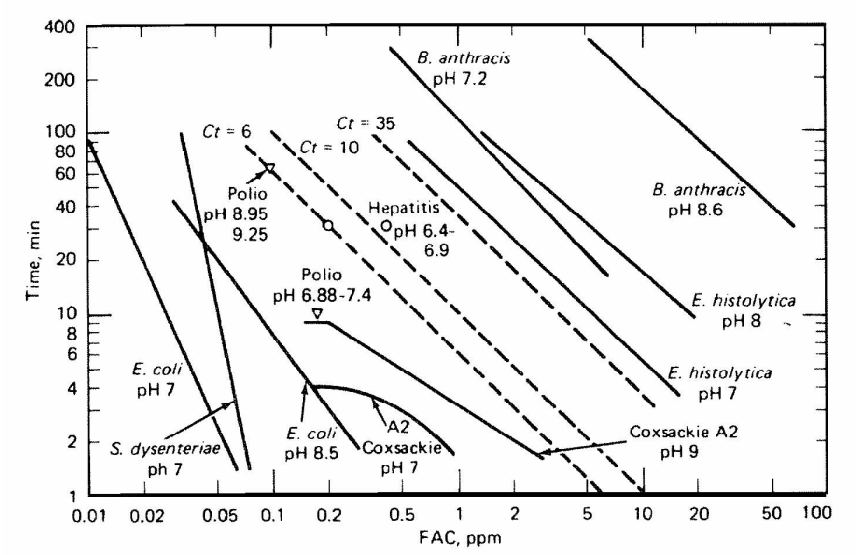

Figure 3.2 Disinfection (2-log) of microorganisms by free available chlorine (White, 1999).

Table 3.2. Disinfection time–chlorine concentration envelopes for 99% virus inactivation at 0–5°C and 10°C

	CT in mg/min l^{-1}	
pH range	0–5°C	10°C
7.0–7.5	12	8
7.5–8.0	20	15
8.0–8.5	30	20
8.5–9.0	35	22

Adapted from White (1999)

Effectiveness of chlorine against protozoa

Protozoan cysts such as *Entamoeba histolytica* and *Giardia lamblia* are highly resistant to chlorine disinfection and may require prolonged contact times at high chlorine residuals (2–3 mg/l) to achieve 99.9% (3-log) inactivation. Clark, Read & Hoff (1989) have described a mathematical model for *Giardia* inactivation that is based on the infectivity data:

$$CT = 0.9847\ C^{0.1758}\ pH^{2.7519}\ temp^{-0.1467}$$

where:

C = the disinfectant residual concentration
temp = the reaction temperature in degrees Celcius

The United States Environmental Protection Agency (USEPA) has published extensive CT tables for *Giardia* inactivation, for different temperature, pH, chlorine residual and other factors (USEPA, 1989b). For example, at a temperature of 25°C and pH 8.0, with a chlorine residual in the range of 1 to 2.6 mg/l, a contact time of 54–65 minutes is needed to achieve a 3-log reduction in *Giardia* (Table 3.3). If the temperature is reduced to 10°C, the contact time increases to 162–194 minutes (Table 3.4), and at 0.5°C it increases further, to 304– 368 minutes (Table 3.5).

Table 3.3. Estimated CT values for inactivation of *Giardia* cysts with free chlorine at 25°C

	pH 7 Log inactivation			pH 8 Log inactivation		
Chlorine (mg/l)	1	2	3	1	2	3
1	12	25	37	18	36	54
1.6	13	27	40	19	39	58
2	14	27	41	20	41	61
2.6	15	29	44	22	43	65

Source: Adapted from EPA, 1990.

Table 3.4. Estimated CT values for inactivation of *Giardia* cysts with free chlorine at 10°C

	pH 7 Log inactivation			pH 8 Log inactivation		
Chlorine mg/l	1	2	3	1	2	3
1	37	75	112	54	108	162
1.6	40	79	119	58	116	174
2	41	83	124	61	121	182
2.6	44	87	131	65	129	194

Source: Adapted from EPA, 1990.

Table 3.5. Estimated CT values for inactivation of *Giardia* cysts with free chlorine at 0.5°C

	pH 7 Log inactivation			pH 8 Log inactivation		
Chlorine mg/l	1	2	3	1	2	3
1	70	140	210	101	203	304
1.6	75	151	226	110	219	329
2	79	157	236	115	231	346
2.6	84	168	252	123	245	368

Source: Adapted from EPA, 1990.

E. histolytica cysts were inactivated at pH 7.0 in 10 minutes at 25°C with a residual of 3.5 mg/l (Chang, 1982). At pH 4, 30°C and 10 minutes of exposure, 2 mg/l of free chlorine produced a 99.9% reduction of cysts; however, if the pH was increased to 10, a chlorine concentration of 12 mg/l was needed to achieve the same 3-log reduction. Data on other emerging protozoan pathogens are lacking, although a recent report indicated that the microsporidian *Encephalitozoon* syn. *Septata intestinalis* was inactivated by more than 3 logs when exposed to 2 mg/l chlorine for 16 min at pH 7 and 25°C (Wolk et al. 2000).

Chlorine-based disinfectants are generally not effective at inactivation of *Cryptosporidium* (Table 3.6) and early studies found that *Cryptosporidium* oocysts were resistant to a variety of hospital disinfectants, including bleach (Campbell et al., 1982). Chlorine disinfection has not been effective in preventing outbreaks of cryptosporidiosis caused by *Cryptosporidium* in drinking-water and recreational water. Korich et al. (1990) reported that 80 mg/l of free chlorine or monochloramine required 90 minutes to achieve 90% inactivation of oocysts, and suggested that conventional disinfection practices would do little to inactivate waterborne *Cryptosporidium*. However, Rasmussen et al. (1994) examined the disinfection effectiveness of several biocides and found that inactivation of oocysts required an oxidation/reduction potential of about 800 mV, maintained for 30 minutes (Table 3.6). These authors suggest that oxidation/reduction potential is more important than CT for oocyst inactivation.

Table 3.6 Summary of free chlorine and monochloramine disinfection results for C*ryptosporidium*

Chlorine residual (mg/l)	Contact time (min)	CT product (mg/min.l^{-1})	Temp (°C)	pH	Per cent inactivation	Analytical method
Free chlorine						
80[a]	90	7200	25	7	> 99	Mouse infectivity
15[b]	240	3600	22	8	47	Mouse infectivity
968[c]	1440	1,393,920	10	7	85	Excystation
17[d,e]	30	510	NR	NR	99	Excystation
Monochloramine						
80[a]	90	7200	25	7	99	Mouse infectivity
15[b]	240	3600	22	8	99.6	Mouse infectivity
3.75[c]	1440	5400	10	7	80.5	Excystation

NR = not reported
[a] Korich et al. (1990)
[b] Finch, Kathleen & Gyurek (1994)
[c] Ransome, Whitmore & Carrington (1993)
[d] Rasmussen et al. (1994)
[e] Estimated chlorine residual to achieve an oxidation-reduction potential of 800 mV

By-products of disinfection with chlorine

THMs and other halogenated compounds are the main by-products of disinfection with chlorine. Factors affecting the formation of THMs are discussed in *Safe piped water: Managing microbial water quality in piped distribution systems* (Ainsworth, 2004).

3.3.2 Monochloramine

Mode of action

In dilute aqueous solutions (1–50 mg/l), chlorine reacts with ammonia in a series of bimolecular reactions:

$$HOCl + NH_3 \Rightarrow NH_2Cl \text{ (monochloramine)} + H_20$$
$$HOCl + NH_2Cl \Rightarrow NHCl_2 \text{ (dichloramine)} + H_20$$

$HOCl + NHCl_2 \Rightarrow NCl_3$ (trichloramine) + H_20

These competing reactions are dependent upon pH and the relative chlorine to nitrogen concentration (expressed as Cl_2:N). To a lesser degree they are also dependent upon temperature and contact time. The reaction of HOCl and ammonia will convert all the free chlorine to monochloramine at pH 7–8 when the Cl_2:N ratio is equimolar (5:1 by weight) or less.

Ingols (1958) examined the reaction of monochloramine with several amino acids and tripeptides. Exposure of alanine, tyrosine and glycylgylcylgylcine to the disinfectant for several hours at 25°C and pH 8.0 converted these compounds to organic chloramines. The sulfhydryl groups of cystine were oxidized to disulfides (by comparison, exposure of the same compounds to HOCl produced a variety of oxidized, hydrolysed or deaminated reactants). Reaction of monochloramine with hemin (an important component of enzymes such as cytochromes, catalases and peroxidases) resulted in products that could not be reactivated by reducing compounds. The author concluded that monochloramine may kill bacterial cells by reacting primarily with membrane-bound enzymes.

Jacangelo & Olivieri (1985) examined the reaction of monochloramine with amino acids, nucleic acids, nucleotides, nucleosides, purine and pyrimidine bases, and ribose sugars. Monochloramine was most reactive with sulfur-containing amino acids and tryptophan. When the sulfhydryl groups of cysteine were in excess, 1 mol of monochloramine reacted with 2 mol of cysteine to form 1 mol of the cystine disulfide. When monochloramine was in excess, the reaction proceeded beyond the disulfide state.

Watters et al. (1989) extended the observations of Jacangelo & Olivieri (1985) by examining whole cells. They found that *Enterobacter cloacae* could be reactivated after exposure to chloramine by addition of sodium sulfite, and hypothesized that sodium sulfite could reduce oxidized disulfides, or result in other types of oxidative injury. Interestingly, sodium sulfite had no effect on organisms exposed to free chlorine. The results suggest that free chlorine and chloramine react with different functional groups in the cell membrane.

Jacangelo & Olivieri (1985) found that monochloramine reacted more slowly with nucleic acids and free purine and pyrimidine bases than with amino acids. These results support the observation that many viruses are inactivated more slowly than bacterial cells. Berman & Hoff (1984) showed that simian rotavirus SA11 required more than 6 hours contact with 10 mg/l preformed monochloramine at pH 8.0 to achieve 99% inactivation. Shih & Lederberg (1976) found that exposure of *Bacillus subtilis* deoxyribonucleic acid (DNA) to monochloramine induced single and double stranded breaks, reduced the

transforming activity of DNA and enhanced the sensitivity of DNA to endonuclease cleavage.

Effectiveness of monochloramine

Monochloramine is not recommended as a primary disinfectant because of its weak disinfecting power (Table 3.1). This disinfectant is not effective for inactivation of *Cryptosporidium* (Table 3.6). In systems using monochloramine, free chlorine is usually applied for a short time before addition of ammonia, or an alternative primary disinfectant is used (e.g. ozone, chlorine dioxide).

By-products of disinfection with monochloramine

Treatment to produce a monochloramine residual poses the risk of nitrite formation in the distribution system, especially in low-flow stagnant areas, because bacteria on surfaces and in deposits may nitrify any slight excess of ammonia.

3.3.3 Chlorine dioxide

Chlorine dioxide is a strong oxidant that can be used to control iron, manganese and taste and odour causing compounds. It has also been used as a secondary disinfectant in many European countries.

Mode of action

Chlorine dioxide is highly soluble in water (particularly at low temperatures), and is effective over a range of pH values (pH 5–10). Theoretically, chlorine dioxide undergoes five valence changes in oxidation to chloride ion:

$$ClO_2 + 5e^- = Cl^- + 2O^{2-}$$

However, in practice, chlorine dioxide is rarely reduced completely to chloride ion (White 1999). Chlorine dioxide is thought to inactivate microorganisms through direct oxidation of tyrosine, methionyl, or cysteine-containing proteins, which interferes with important structural regions of metabolic enzymes or membrane proteins (Gates 1998). In water treatment, chlorine dioxide has the advantage of being a strong disinfectant, but not forming THMs or oxidizing bromide to bromate.

Effectiveness of chlorine dioxide against bacteria and viruses

Chlorine dioxide is roughly comparable to free chlorine for inactivation of bacteria and viruses at neutral pH (White, 1999), but is more effective than free chlorine at pH 8.5 (Hoff & Geldreich, 1981).

Effectiveness of chlorine dioxide against protozoa

Chlorine dioxide is an effective disinfectant for control of *Giardia lamblia*; the required CT values for 1-log inactivation (pH 6–9) range from 5 mg min/l at 20°C to 21 mg/min l^{-1} at 0.5°C (USEPA, 1989b; White, 1999). The 3-log inactivation CT values (pH 6–9) range from 19 mg/min l^{-1} at 15°C to 63 mg/min l^{-1} at 0.5°C. These values are 3–14 times less than those required for free chlorine, but approximately 20 times more than those required for ozone.

Figure 3.3 summarizes results from various studies of *Cryptosporidium* inactivation by chlorine dioxide. Peeters et al. (1989) reported 1.5 and 1.2-log inactivation of *Cryptosporidium*, using an animal infectivity method, for CT values of 3 and 9.8 mg/min l^{-1}, respectively (average of initial and final concentrations). Korich et al. (1990) reported a CT value of 78 mg/min l^{-1}, with an initial concentration of 1.3 mg/l and a contact time of 60 minutes, for a 90% (1-log) inactivation of *Cryptosporidium*, based on mouse infectivity. The CT for 1-log inactivation was calculated to be 51 mg/min l^{-1} (average of initial and final concentrations). Finch, Liyanage & Belosevic (1995) recalculated the Korich data using a dose–response model developed for CD-1 mice, and estimated a 99% (or 2-log) inactivation. Ransome, Whitmore & Carrington (1993), employing the excystation viability method, reported *Cryptosporidium* inactivation ranging from 0.14 to 1.4-log for average CT values ranging from 6.5 to 67.5 mg/min l^{-1}, respectively. Based on results from 12 animal infectivity experiments, Finch et al. (1997) reported *Cryptosporidium* inactivation ranging from 0 to greater than 3.2-log for average CT values ranging from 12.5 to 212 mg/min l^{-1}. Chlorine dioxide concentration decreased markedly at contact times of more than 30 minutes, a factor that could result in low CT values. LeChevallier et al. (1996) found that oocysts were more rapidly inactivated by chlorine dioxide at pH 8.0 than at pH 6.0, and that effectiveness was reduced by 40% when temperature was reduced from 20°C to 10°C. This finding is supported by other studies (Bernard et al., 1965; Owens et al., 1999; Ruffle, Rennecker & Marinas, 1998). Chlorine dioxide inactivation rates using a cell culture technique to determine infective oocysts were similar to rates generated using animal infectivity tests.

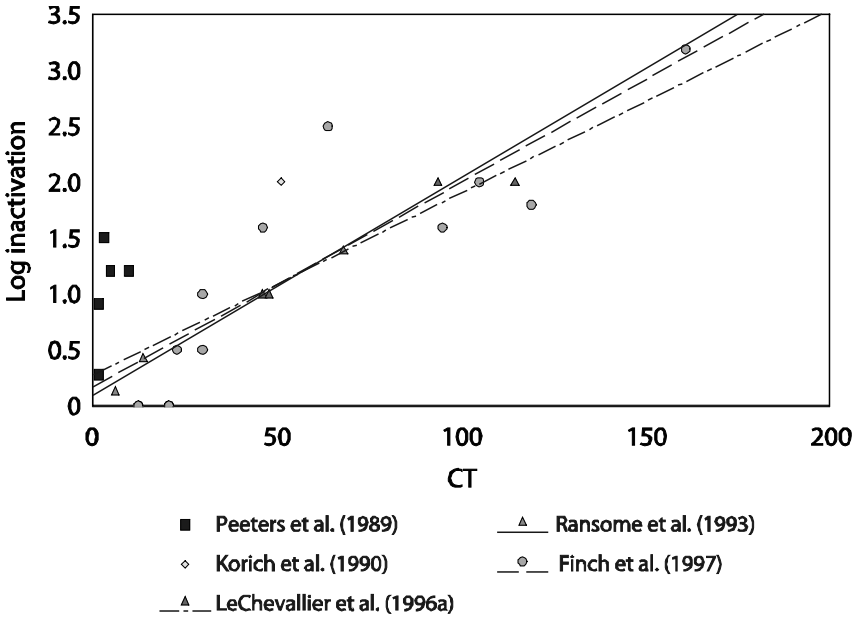

Figure 3.3 Summary of *Cryptosporidium* inactivation by chlorine dioxide

By-products of disinfection with chlorine dioxide

The chlorine in chlorine dioxide exists in a +4 oxidation state, compared to an oxidation state of +1 for chlorine in free chlorine (in hypochlorous and hypochlorite ions). This means that chlorine and chlorine dioxide have different pathways for disinfection and formation of by-products when used in drinking-water treatment. For example, chlorine dioxide does not produce significant levels of halogenated organic by-products.

Chlorine dioxide forms undesirable inorganic by-products (chlorite and chlorate ions) upon its reaction with constituents of water such as dissolved organic carbon, microbes and inorganic ions. Therefore, a water utility may need to provide additional treatment depending on the level of these inorganic by-products and their specific regulatory requirements (Gordon & Bubnis, 1995; WHO, 2000).

3.3.4 Ozone

Ozone has been used for more than a century for water treatment, mostly in Europe, although its use is now spreading to other countries.

Mode of action

The mechanism by which ozone inactivates microbes is not well understood. Ozone in aqueous solution may react with microbes either by direct reaction with molecular ozone or by indirect reaction with the radical species formed when ozone decomposes. Ozone is known to attack unsaturated bonds, forming aldehydes, ketones or carbonyl compounds (Langlais, Reckhow & Brink, 1991). Additionally, ozone can participate in electrophilic reactions, particularly with aromatic compounds, and in nucleophilic reactions with many of the components of the microbial cell. Carbohydrates and fatty acids react only slightly with ozone, but amino acids, proteins, protein functional groups (e.g. disulfide bonds) and nucleic acids all react very quickly with it (Langlais, Reckhow & Brink, 1991). It is likely, therefore, that microbes become inactivated through ozone acting on the cytoplasmic membrane (due to the large number of functional proteins), the protein structure of a virus capsid, or nucleic acids of microorganisms.

Free radicals formed by the decomposition of ozone are generally less effective for microbial inactivation than molecular ozone, because microbial cells contain a high concentration of bicarbonate ions that quench the free radical reaction, and many microbial cells also contain catalase, peroxidase, or superoxide dismutase to control the free radicals produced by aerobic respiration. In addition, some bacteria contain carotenoid and flavonoid pigments that protect them from ozone. These factors can account for reports that heterotrophic bacteria may be less susceptible to ozone inactivation than *Giardia* (Wolfe et al., 1989). Studies of peroxone (a mixture of ozone and hydrogen peroxide that promotes the generation of hydroxyl free radicals) showed that peroxone was comparable to ozone, or slightly more potent, when CTs were based on ozone residuals (Wolfe et al., 1989). These results suggest that free radicals provide little benefit in terms of microbial destruction.

Effectiveness of ozone against bacteria and viruses

Of the vegetative bacteria, *Escherichia coli* is one of the most sensitive (Table 3.7), while Gram-positive cocci (*Staphylococcus* and *Streptococcus*), Gram-positive bacilli (*Bacillus*) and mycobacteria are the most resistant (Langlais, Reckhow & Brink, 1991). *Mycobacterium avium* can be effectively controlled by low doses of ozone ($CT_{99.9}$ of 0.1–0.2 mg/min l^{-1}), whereas the

organism is highly resistant to free chlorine ($CT_{99.9}$ of 551–1552 mg/min l^{-1} for water-grown isolates) (Taylor et al., 2000).

Table 3.7 CT values (mg/min l^{-1}) for 99% inactivation at 5°C

Microorganism	Free chlorine (pH 6–7)	Preformed chloramines (pH 8– 9)	Chlorine dioxide (pH 6–7)	Ozone (pH 6–7)
E. coli	0.034–0.05	95–180	0.4–0.75	0.02
Poliovirus 1	1.1–2.5	770–3740	0.2–6.7	0.1–0.2
Rotavirus	0.01–0.05	3810–6480	0.2–2.1	0.006–0.06
Phage f2	0.08–0.18	–	–	–
G. lamblia cysts	47–>150	–	–	0.5–0.6
G. muris cysts	30–630	1400	7.2–18.5	1.8–2.0

Adapted from Hoff (1986)

Viruses are generally more resistant to ozone than vegetative bacteria, although phage appear to be more sensitive than human viruses (Langlais, Reckhow & Brink, 1991).

Effectiveness of ozone against protozoa

For the protozoa *Giardia lamblia* and *Naegleria gruberi*, ozone inactivation (Table 3.7) did not follow linear kinetics, due to an initial latent phase. However, CT products could be reasonably estimated with a CT_{99} (a CT for 99% inactivation) of 0.53 and 4.23 mg/min l^{-1}, respectively, at 5°C (Wickramamayake, Rubin & Sproul, 1984).

Ozone is effective for removal of *Cryptosporidium* (Table 3.8). Noticeable for *Cryptosporidium* is the impact of the analytical method on the CT values. Generally, excystation and vital staining are more conservative measures of oocyst inactivation than animal infectivity. Reliance on excystation and vital staining alone could greatly overestimate disinfection requirements for *Cryptosporidium*. On average, 4.5 mg/min l^{-1} CT was required for 99% oocyst inactivation (measured by mouse infectivity) by ozone at 20–25°C (Table 3.8). However, Finch et al. (1993) indicated that the conventional method of determining CT by using the final concentration of reactants at the end of the contact time overestimates the CT needed for disinfection and unduly increases treatment costs. The authors recommended the Holm disinfection model, which integrates the disinfectant concentration and time throughout the reactor. Using this alternative calculation, CT for *Cryptosporidium* inactivation were 6.9 mg/min l^{-1} at 7°C and 2.4 mg/min l^{-1} at 22°C.

Table 3.8 Summary of ozone disinfection results for *Cryptosporidium*

Ozone residual (mg/l)	Contact time (min)	CT product (mg/min l^{-1})	Temp °C	Per cent inactivation	Analytical method
1[a]	5	5	25	90–99	Mouse
1[a]	10	10	25	>99	infectivity
0.77[b]	6	4.6	'Room'	>99	Mouse
0.51[b]	8	4.1	'Room'	>99	infectivity
0.16–1.3[c]	5–15	7	7	99	Mouse
0.17–1.9[c]	5–15	3.5	22	99	infectivity
2.4 (avg)[d]	2.3	5.5	22–25	99	Mouse infectivity
1.25[e]	15	18.75	10	98.6	Excystation
4 (approx)[f]	2	8	'Room'	>90	Excystation
1–5[g]	10	10–50	5	18–39	Stain
1–5[g]	10	10–50	20	70–>99	
0.7–1.5[h]	14–25	9.8–27	8–10	42–84	Stain

[a] Korich et al. (1990)
[b] Peeters et al. (1989)
[c] Finch et al. (1993)
[d] Owens et al. (1994)
[e] Ransome, Whitmore & Carrington (1993)
[f] Armstrong et al. (1994)
[g] Parker, Greaves & Smith (1993)
[h] Hall, Pressdee & Carrington (1994)

To date, there are no accepted CT values for ozone for inactivation of *Cryptosporidium*, either for regulatory or operational application. Results of disinfection studies vary widely between studies and even between replicate trials. The USEPA is evaluating options for *Cryptosporidium* disinfection by ozone and, for developmental and cost-estimating purposes, is using values that encompass the range of experimental variability (Table 3.9). These values will probably be replaced with consensus values eventually, but are presented here to demonstrate the range of ozone CT values for different water temperatures and levels of inactivation.

Table 3.9 CT (mg/min l^{-1}) for *Cryptosporidium* inactivation by ozone

Log inactivation	Temperature		
	1°C	13°C	22°C
0.5	6	2	0.6
1.0	12	4	1.5
1.5	24	8	3.0
2.0	40	11	4.4
2.5	45	15	6.0
3.0	62	22	8.0

Source: Estimated based on preliminary data from G Finch (personal communication). For illustrative purposes only.

Effectiveness of ozone against algal toxins

Ozonation is an effective process for destruction of both intracellular and extracellular algal toxins. Essentially complete destruction of microcystins, nodularin and anatoxin-a can be achieved if the ozone demand of the water is satisfied (Yoo et al., 1995b; Chorus & Bartram 1999).

3.3.5 Ultraviolet light

Mode of action

UV light can be categorized as UV-A, UV-B, UV-C or vacuum-UV, with wavelengths ranging from about 40 to 400 nm. The UV light effective for inactivating microorganisms is in the UV-B and UV-C ranges of the spectrum (200–310 nm), with maximum effectiveness around 265 nm. Thymine bases on DNA and ribonucleic acid (RNA) are particularly reactive to UV light and form dimers (thymine–thymine double bonds) that inhibit transcription and replication of nucleic acids, thus rendering the organism sterile. Thymine dimers can be repaired in a process termed 'photoreactivation' in the presence of light, or 'dark repair' in the absence of light (Jagger, 1967). As a result, the strategy in UV disinfection has been to provide a sufficiently high dosage to ensure that nucleic acid is damaged beyond repair.

Effectiveness of UV against bacteria and viruses

Table 3.10 shows that UV is an effective disinfectant for bacteria and viruses (USEPA, 1986; Wolfe, 1990; Battigelli, Sobsey & Lobe, 1993). *Bacillus subtilis* spores are commonly used as a bioassay organism because of their resistance to inactivation, requiring about 31 mW-sec/cm^2 for a 4-log inactivation of spores (Qualls & Johnson, 1983). MS-2 is an F-specific single-stranded RNA virus

about 20 nm in diameter that can be used as a viral surrogate (Braunstein et al., 1996). Adenoviruses are double-stranded DNA viruses and are very resistant to UV inactivation. Typical doses used for drinking-water disinfection would not be effective for treatment of adenoviruses.

Table 3.10 Typical UV dosages required for 4-log inactivation of selected microbes

Organism	4-log inactivation dose range (mW-sec/cm^2)	Water source
Bacteria:		
Bacillus subtilis spores	31	Laboratory water
Escherichia coli	20	Laboratory water
S. faecalis		Laboratory water
Salmonella typhi	30	Laboratory water
Vibrio cholera	0.65	Laboratory water
Viruses:		
MS-2	50	Groundwater (1 source)
	64–93	Groundwater (11 sources)
	100	Laboratory water
Coxsackie AZ	30	Laboratory water
Hepatitis A	6–15	Groundwater (3 sources)
	16	Laboratory water
Poliovirus	23–29	Groundwater (8 sources)
	30	Laboratory water
Rotavirus — Wa	50	Laboratory water
Rotavirus SA11	40	Tap water
Adenovirus	186	Laboratory water (4 studies)

Adapted from Malley (2000) and USEPA (2003)

Effectiveness of UV against protozoa

Most of the early work on UV disinfection of *Giardia* (Rice & Hoff, 1981; Karanis et al., 1992) and *Cryptosporidium* (Lorenzo-Lorenzo et al., 1993; Ransome, Whitmore & Carrington, 1993; Campbell et al. 1995) relied upon excystation or vital staining to determine viability and found that UV inactivation was not effective for *Giardia* cysts or *Cryptosporidium* oocysts. However, more recent work (Clancy et al., 1998ab; Bukhari et al., 1999; USEPA, 2003) using mouse infectivity or cell culture showed that low or medium-pressure mercury vapour UV lamps, or pulsed UV technology can achieve 3-log inactivation of *Cryptosporidium* oocysts at UV doses less than 10 mW-sec/cm^2 (Figure 3.4). Similar sensitivities to UV inactivation have recently been shown for *Giardia* (Craik et al., 2000).

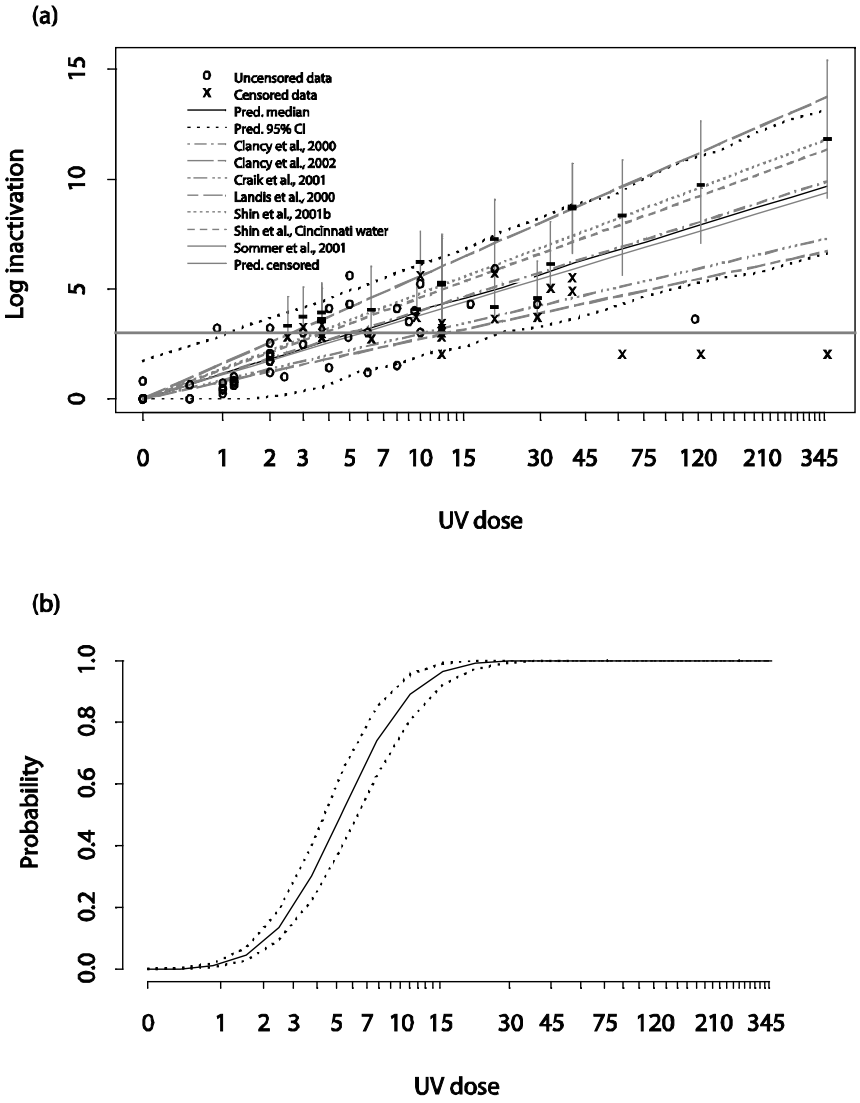

Figure 3.4 Summary of UV inactivation data for *Cryptosporidium*
(a) Scatter of individual study results.
(b) Probability plot for achieving a 3-log inactivation.
Source: USEPA (2003).

Guidelines and standards relating to the use of UV

Recently, guidelines have been developed to evaluate the effects of reactor design, selection of UV lamps, performance standards for lamp ageing and fouling, and the accuracy of UV sensors (DVGW, 1997; ÖNORM, 2001; USEPA, 2003). Standards for the installation and operation of UV systems are important because the effectiveness of UV disinfection can be impaired by the transmittance of the water, colour and the presence of particulate material.

3.3.6 Mixed oxidants

The use of mixtures of oxidants for microbial inactivation has gained attention as a way to maximize the efficiency of current disinfectants. The chemistry of mixed oxidant production is complex, resulting in a solution of free chlorine, chlorine dioxide, ozone and various oxidation states of chlorine. The oxidants can be produced from a sodium chloride brine in an electrolytically generated cell. Venczel et al. (1997) examined the inactivation of *Cryptosporidium* oocysts and *Clostridium perfringens* spores in oxidant demand-free water at pH 7 and 25°C using a disinfectant dose of 5 mg/l and contact times up to 24 hours. Free chlorine produced no measurable inactivation of *Cryptosporidium parvum* oocysts after exposure for 4–24 hours, although *Clostridium perfringens* spores were reduced by 1.4 logs after 4 hours. In contrast, a mixed oxidant solution resulted in more than 3-log inactivation of both oocysts and spores with 4 hours exposure. Other researchers, however, have found the mixed oxidant process equivalent to free chlorine for inactivation of biofilm samples (Crayton, Camper & Warwood, 1997). Additional research is needed to better understand the chemistry of seemingly incompatible oxidants within the mixed oxidant reaction.

Sequential disinfection

Other approaches to combining the advantages of various oxidants have used sequential disinfection. Finch, Kathleen & Gyurek (1994) reported that the sequential combination of free chlorination followed by monochloramination produced superior oocyst inactivation compared to the sum of both disinfectants examined separately. The combination of free chlorine (1 mg/l for 60 min) and chloramines (2 mg/l for 240 min) are typical values that might be found in conventional treatment plants. Similar synergies have been seen for ozone and chloramines, free chlorine and chlorine dioxide, and chlorine dioxide followed by free chlorine or chloramines (Liyanage, Finch & Belosevic, 1997; Corona-Vasquez, Rennecker & Marinas, 1999; Li, Finch & Belosevic, 1999). Sequential disinfection has been proposed, to lessen or eliminate the inactivation lag phase

(Corona-Vasquez, Rennecker & Marinas, 1999). Combinations of disinfectants require further investigation, and may provide important insights into inactivation mechanisms and disinfection theory.

3.4 SECONDARY DISINFECTION

This section looks at the use of secondary disinfection to maintain water quality in distribution systems. The publication *Safe piped water: Managing microbial water quality in piped distribution systems* (Ainsworth, 2004) provides more detail on this topic.

3.4.1 Maintenance of water quality in the distribution system

The purpose of a secondary disinfectant is to maintain the water quality achieved at the treatment plant throughout the distribution system up to the tap. Secondary disinfection provides a final partial barrier against microbial contamination and serves to control bacterial growth. The practice of residual disinfection has become controversial, with some opponents arguing that if biological stability is achieved and the system is well maintained, the disinfectant is unnecessary. These positions are presented in a series of papers published in *Water Supply* (Vol. 16(3/4), 1998).

3.4.2 Factors affecting microbial occurrence

Disinfectant residual and disinfectant level

The growth of bacteria and occurrence of coliforms depend on a complex interaction of many factors including water temperature, disinfectant type and residual, pipe material, corrosion and other engineering and operational parameters (Berger, LeChevallier & Reasoner, 1992; LeChevallier et al., 1991, 1993; LeChevallier, Welch & Smith, 1996). Recent research has indicated that various disinfectants differ in their ability to interact with biofilm bacteria (LeChevallier, 1991; De Beer, Srinivasan & Stewart, 1994). Monochloramine, although a much less reactive disinfectant than free chlorine, is more specific in the type of compounds that it will react with. Therefore, monochloramine can be more effective than free chlorine at penetrating and inactivating certain types of biofilm, particularly those containing corrosion products (LeChevallier, Lowry & Lee, 1990; LeChevallier et al., 1993; Norton & LeChevallier, 1997). A study of 30 distribution systems showed a difference in the density and occurrence of coliform bacteria between systems using free chlorine and those using chloramines (LeChevallier et al., 1996). Modelling indicates that the penetration

of free chlorine into a biofilm is limited by its fast reaction rate (De Beer, Srinivasan & Stewart, 1994). Free chlorine is essentially consumed before it can react with the bacterial components of the film (Chen & Stewart, 1996). Chloramines, on the other hand, are slower reacting; they can diffuse into the biofilm and eventually inactivate attached bacteria, a mechanism that has been demonstrated using an alginate bead model (Chen and Stewart, 1996). Stewart, McFeters & Huang (2000) showed that free chlorine did not effectively penetrate alginate beads containing bacterial cells, but chloramines did penetrate into the alginate material and reduced bacterial levels nearly one million-fold over a 60 minute interval (2.5 mg/l chloramines, pH 8.9). Kool, Carpenter & Fields (1999) reported that hospitals supplied with water containing a chloramine residual were 10 times less likely to have water-associated legionella infections. Similarly, Heffelfinger et al. (2003), in a study of 166 hospitals, found that nosocomial legionellosis was five times less likely in the hospitals served with chloraminated water. The authors attributed the effectiveness of chloramines for legionella control to the ability of the disinfectant to penetrate biofilms.

In addition to the type of disinfectant used, the residual maintained at the end of the distribution system was also related to coliform occurrences (LeChevallier, Welch & Smith, 1996). Systems that maintained dead-end free chlorine levels of less than 0.2 mg/l or monochloramine levels of less than 0.5 mg/l had substantially more coliform occurrences than systems maintaining higher disinfectant residuals. Systems with high assimilable organic carbon (AOC) levels needed to maintain high disinfectant residuals to control coliform occurrences. Therefore, maintenance of a disinfectant residual alone does not ensure that treated waters will be free of coliform bacteria.

Biostability

The presence of biodegradable organic matter in water will promote bacterial growth, and may be related to the occurrence of coliform bacteria in distribution systems (Bourbigot, Dodin & Lheritier, 1984; Camper et al., 1991; Geldreich & Stevens, 1987; LeChevallier, Babcock & Lee, 1987; LeChevallier et al., 1991). Biodegradable organic matter is commonly measured as AOC or biodegradable dissolved organic carbon (BDOC). Van der Kooij (1987) showed that AOC concentrations increased in water samples treated with increasing chlorine doses. Similarly, Hambsch & Werner (1993) reported higher biodegradability of humic substances after chlorination. LeChevallier et al. (1992) found that chlorination may increase AOC levels, depending on the point of chlorine application.

Corrosion control and pipe materials

Corrosion of iron pipes can influence the effectiveness of chlorine-based disinfectants for inactivation of biofilm bacteria (LeChevallier, Lowry & Lee, 1990; LeChevallier et al., 1993; Ainsworth, 2004). Free chlorine is affected to a greater extent than monochloramine, although the effectiveness of both disinfectants is impaired if corrosion rates are not controlled (LeChevallier, Lowry & Lee, 1990; LeChevallier et al., 1993). Improving corrosion control can improve the ability of residual disinfectants to control bacterial growth (Norton & LeChevallier, 1997).

The pipe surface itself can influence the composition and activity of biofilm populations. Biofilms develop more quickly and support a more diverse microbial population on iron pipe surfaces than on plastic polyvinylchloride (PVC) pipes, even with adequate corrosion control, biological treatment of water to reduce AOC levels and consistently maintained chlorine residuals (LeChevallier et al., 1993; Camper, 1996).

Pressure, cross-connection control and maintenance

Microbial quality of drinking-water cannot depend only on maintenance of a residual disinfectant. The extensive nature of the distribution system, with many kilometres of pipe, storage tanks, interconnections with industrial users and the potential for tampering and vandalism, provides opportunities for contamination. Cross-connections are a major risk to water quality. Although the risk can be reduced by vigilant control programs, complete control is difficult to achieve and water utilities worldwide face challenges in maintaining an effective cross-connection control program.

Despite the best efforts to repair main breaks using good sanitary procedures, main breaks provide an opportunity for contamination to enter the distribution system. Utilities typically isolate the affected section and repair, superchlorinate and flush the repaired pipe. However, it may be difficult to achieve flushing velocities sufficient to remove all contaminated debris; also, microbiological tests to check the final water quality may not detect contaminating organisms. McFeters, Kippin & LeChevallier (1986) reported high levels of damaged coliform bacteria, not detectable by standard coliform techniques, following the repair of a main break. Resampling of the site one week later showed persistence of high levels of the coliform bacteria, detectable only using m-T7 agar, a medium specially designed to recover chlorine-damaged coliforms.

Backflow devices to prevent the entry of contaminated water are important as a distribution system barrier. Because of high costs, backflow devices are installed mainly on service lines for facilities that use potentially hazardous substances (e.g. hospitals, mortuaries, dry cleaners and industrial users). It is not

common for all service connections to have backflow devices, so the possibility of back-siphonage exists at certain points. Also, installation of backflow devices for all service connections would make routine checking of the devices nearly impossible and, without routine inspection, the proper functioning of the units cannot be assured. Even when backflow devices have been installed, contamination events have occurred. For example, the failure of a backflow check valve allowed water stored for fire protection to enter the distribution system in Cabool, Missouri (USA) (Geldreich, 1996). A broken vent in the storage tank allowed birds to enter and contaminate the water with *Salmonella*. Three people died from *Salmonella* infection.

Recent research is focusing on transient pressure waves that can result in hydraulic surges in the distribution system (Kirmeyer et al., 2001). These waves have both a positive and negative amplitude, meaning that they can create transient negative pressures (lasting only a few seconds) in a distribution system, which may be missed by conventional pressure monitoring. Because these waves travel through the distribution system, any point where water is leaking out of the system is a potential entry point for microbes during the brief period of negative pressure.

3.4.3 Other non-chlorine disinfectants

Non-chlorine disinfectants include other halogens (iodine, bromine) and a variety of metals. Various authors (Hsu, 1964; Sharp, Floyd & Johnson, 1975; Alvarez & O'Brien, 1982; Pyle, Broadaway & McFeters, 1992) have proposed these alternative disinfectants for use in drinking-water supplies, although currently none have gained widespread acceptance. A combination of copper and silver ions can inactivate bacteria and viruses, although contact times may be long (hours to days) (Derby, 1947; Thurman and Gerba, 1989; Pyle, Broadaway & McFeters, 1992). Low levels of chlorine (0.1 mg/l) combined with silver (38 µg/l) and copper (380 µg/l) resulted in more than 5-log inactivation of *E. coli* in tap water within 120 seconds (Thurman and Gerba, 1989). Silver (30 µg/l) and hydrogen peroxide (30 µg/l) together provided a long-lasting residual effect capable of more than 5-log inactivation of *E. coli* in phosphate buffer (pH 6.8) after one hour exposure (Pedahzur et al., 1995). Photocatalytic titanium dioxide has also been examined for disinfection of water (Wel et al., 1994).

4

Performance models

This chapter describes two models for microbial removal or inactivation by water treatment processes. Section 4.2 describes a model for removal of particles by granular filtration, and shows how it can be used to predict the effect of process variables on the removal of microbial pathogens. Section 4.3 discusses a number of different models used to describe experimental disinfection data.

4.1 REMOVAL PROCESS MODELS

The removal process model described here is based on a mechanistic performance model that was first developed and applied in water filtration by O'Melia and co-workers (O'Melia & Stumm, 1967; Yao, Habibian & O'Melia, 1971). Substantial modifications have been made by Fitzpatrick & Spielman (1973), Rajagopalan & Tien (1976) and others. An extensive review of these theoretical models is available (Elimelech et al., 1995). The version described here is that of Rajagopalan & Tien (1976). It considers particle removal by

 Water Treatment and Pathogen Control: Process Efficiency in Achieving Safe Drinking Water. Edited by Mark W LeChevallier and Kwok-Keung Au. ISBN: 1 84339 069 8. Published by IWA Publishing, London, UK.

granular filters to involve two steps: transport and attachment. This section describes these two steps, and then considers each of the process variables in terms of their effect on the removal of microbial pathogens.

4.1.1 Transport

In the removal process, particles are first transported from suspension to a nearby media grain. The transport step, which is physical–hydrodynamic in nature, involves three main mechanisms:

- *interception* — particles following the streamline of fluid flow come into contact with a media grain (this mechanism is affected by the size of the particle);
- *sedimentation* — particles with density greater than that of water deviate from the streamline of fluid flow by gravity and come into contact with a media grain;
- *diffusion* — particles subjected to random motion by their thermal energy come into contact with a media grain.

Single collector efficiencies (defined as the ratio of the number of successful collisions between particles and a filter media grain to the total number of potential collisions in the projected cross-sectional area of the media grain) have been well developed to describe these transport mechanisms.

4.1.2 Attachment

To be removed, a particle must not only come into contact with a media grain, but must also attach to it. Not all contacts between particles and media lead to attachment; an attachment efficiency (α) is used to represent the fraction of successful contact. The value of α varies from one (all contact results in attachment) to zero (no contact results in attachment). In drinking-water treatment, chemical coagulation pretreatment promotes attachment efficiency, with optimized coagulation conditions increasing the value of α. A predictive equation for removal efficiency can be derived from single collector efficiency, attachment efficiency and the total number of media collectors.

4.1.3 Effects of process variables on removal efficiency

Variables that can affect the efficiency of removal of microbial contaminants by granular filtration include coagulation conditions, filtration rate, diameter of medium, filter depth and water temperature. Figure 4.1 illustrates the effects of these variables, as a function of particle size, and Table 4.1 shows the

parameters used in these simulations. The theoretical results give some indication of removal of microbial pathogens by granular filtration, but the model has limitations. First, it was developed for a clean bed and a monodisperse suspension, and thus does not take into consideration temporal variation in filter performance. Second, it was developed for passive (nonmotile) particles; however, some microbes (e.g. some species of coliform bacteria) are motile. Cell motility may change both transport mechanism and removal efficiency. Little is known of the effects of cell motility on filter performance, and the model does not take this factor into account. Finally, the model has been successfully tested for nonmicrobial particles but has yet to be systematically tested with microbes. Each of the process variables, and its effect on removal efficiencies, is considered in detail below.

Particle size

Model calculations indicate that particle diameter has a dramatic effect on removal mechanisms and efficiency (Figure 4.1a). Microbes that are submicron in size (e.g. viruses) are transported to media particles by molecular diffusion (Brownian motion). For such particles, removal efficiency decreases as particle size increases, because small particles diffuse faster than large ones. Assuming that microbes do not change in size before entering the filter, and that coagulation conditions are optimal ($\alpha = 1.0$), model predictions suggest the filter could remove 6.38 logs of MS2 bacteriophage (2.5×10^{-8} m diameter), 3.21 logs of rotavirus (7.0×10^{-8} m) and 2.53 logs of PRD1 bacteriophage (10^{-7} m).

Microbes with a diameter larger than about a few microns (e.g. protozoan cysts, algae and some bacteria) are removed by interception (Figure 4.1a). Removal efficiency of such particles increases as microbial size increases, because larger particles are more easily intercepted by the filter medium. When the filter is operated under optimal chemical coagulation, the predicted removal efficiencies are 1.44 logs for *Cryptosporidium* oocysts (5×10^{-6} m) and 4 logs for *Giardia* cysts (10×10^{-6} m). Numerous studies (e.g. Nieminski & Ongerth, 1995; Swertfeger et al., 1999) show that *Giardia* cysts are removed more efficiently than *Cryptosporidium* oocysts. Removal by gravity is never a dominant mechanism in these simulations; even for large microbes such as *Balantidium coli* cysts (6×10^{-5} m), the density of the particles (1.05 g/cm^3) is similar to that of water. The effect of gravity is insignificant for most microorganisms in the influent to filters, unless they are associated with dense particles.

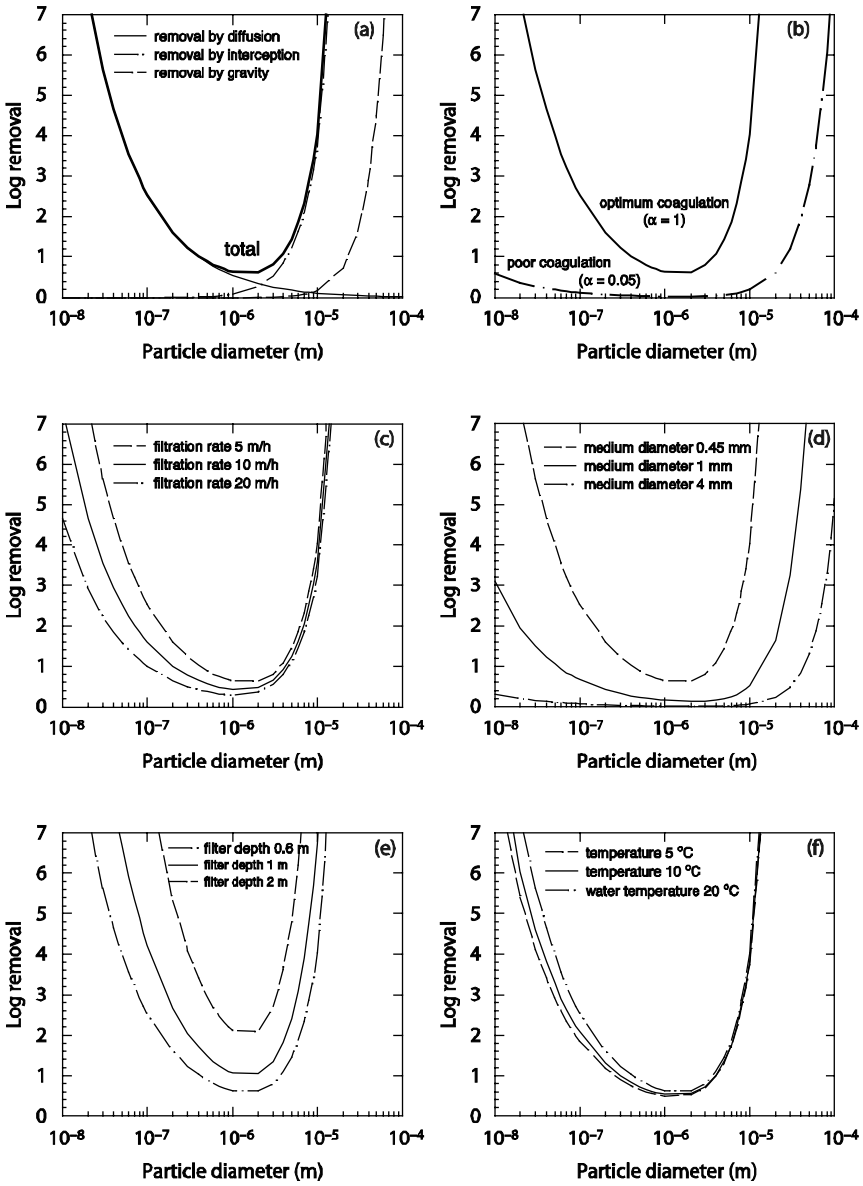

Figure 4.1 Effects of process variables on removal efficiency of granular filtration (simulation parameters are shown in Table 4.1).

Table 4.1 Parameters used in model calculations for Figure 4.1

Process variables	Values
Coagulation condition (α)	1.0 [a]
Filtration rate	5 m/h [b]
Medium diameter	0.45 mm [c]
Filter depth	0.6 m [d]
Water temperature	20°C [e]
Media configuration	monomedia
Particle density	1.05 g/cm^3
Hamaker constant	10^{-20} J
Filter porosity	0.4

[a] 1.0 and 0.05 in Figure 4.1b
[b] 5, 10 and 20 in Figure 4.1c
[c] 0.45, 1 and 4 mm in Figure 4.1d
[d] 0.6, 1 and 2 m in Figure 4.1e
[e] 5, 10 and 20°C in Figure 4.1f

Removal efficiency is lowest for microbes with a diameter of about 1 μm (Figure 4.1a). Particles of about this size are too large for diffusion to be effective and too small for interception to be effective. Thus, even with optimal coagulation conditions ($\alpha = 1.0$), the model predicts that the filter removes only 0.64 logs of coliform bacteria (1.0×10^{-6} m). Some bacteria in this size range will be motile, which will influence their removal, but the model does not take this into account.

Pretreatment with chemical coagulants

The calculations shown in Figure 4.1b illustrate the significant effect on filtration performance of pretreatment with chemical coagulants. When coagulation conditions change from optimal ($\alpha = 1.0$) to poor ($\alpha = 0.05$), log removals deteriorate from 6.38 to 0.20 for MS2 bacteriophage, from 0.64 to 0.03 for coliform bacteria, from 1.44 to 0.07 for *Cryptosporidium* oocysts, and from 4.00 to 0.20 for *Giardia* cysts. These simulated effects of chemical coagulation on filtration performance are qualitatively consistent with many experimental results (e.g. Al-Ani et al., 1986; Ongerth, 1990).

Filtration rate

The effect of filtration rate on filter performance depends on the size of the particle (Figure 4.1c). For microbes greater than a few microns in diameter, with a density close to that of water, removal is mainly by interception, and is therefore not strongly affected by filtration rate. Thus, increasing the filtration rate from 5 m/h to 20 m/h decreases the modelled removal efficiency only

slightly. For example, the removal of *Giardia* cysts reduces from 4.03 logs at a filtration rate of 5 m/h, to 3.58 logs at 10 m/h and to 3.22 logs at 20 m/h. A similar result has been observed experimentally by Al-Ani et al. (1986). However, the effects of filtration rate on removal efficiency are much more pronounced for submicron microbes, where removal is mainly due to diffusion, which is strongly affected by filtration rate. For example, the modelled removal of rotavirus decreases from 3.21 logs at a filtration rate of 5 m/h to 1.27 logs at 20 m/h.

Filter medium size and depth

Filter medium size (Figure 4.1d) and depth (Figure 4.1e) strongly affect microbial removal by filtration. Decreasing the size of the medium or increasing the depth of the filter increases the removal efficiency. This is in part because the number of filter media collectors increases, favouring the capture of particles. Decreasing the medium size also enhances the contact opportunity between particles and media grains due to diffusion and interception.

Temperature

Temperature has some effect on the removal of submicron microbes, but almost no effect on the removal of those larger than 1 μm (Figure 4.1f). When the modelled temperature was reduced from 20°C to 5°C, the removal efficiency of MS2 bacteriophage decreased from 6.38 logs to 4.66 logs, although the removal of *Cryptosporidium* oocysts was only reduced from 1.44 logs to 1.31 logs. This is partly because particle removal by diffusion (important for removal of submicron microbes) is strongly dependent on temperature, with an increase in temperature decreasing water viscosity and thus increasing the rate of diffusion. Particle removal by interception (important for larger microbes) is, on the other hand, not affected by temperature.

4.2 DISINFECTION MODELS

A number of researchers have used models to describe experimental disinfection data (Haas & Karra, 1984; Haas et al., 1995). The simplest disinfection model (Equation 1) is a combined one proposed by Chick (1908) and Watson (1908). In the Chick–Watson model, the rate of inactivation of a microorganism is dependent upon the concentration of the disinfectant and contact time. Equation 2 represents the integrated form of Equation 1, and simplifies to CT (the disinfectant concentration multiplied by contact time) when n (the coefficient of dilution) is equal to 1.

(1) $r = -kC^n N$

(2) $\ln (N/N_o) = -kC^n t$

where:
r = rate of microorganism inactivation
k, n = empirical constants
C = disinfectant concentration, M/V
N = microorganism concentration at time t, #/V
N_o = microorganism concentration at time t, #/V

Another disinfection model, represented in equations 3 and 4, was proposed by Hom (1972). It provides for a relationship between disinfectant concentration and contact time, and empirical constants m and n. The Hom model successfully described the disinfection of *Giardia* (Haas et al. 1995) and *Cryptosporidium* (Finch et al., 1993), and converts to the Chick-Watson model when m is equal to 1. In a typical disinfection experiment, disinfectant concentration decreases with time and a first order decay rate is generally assumed (Equation 5). Haas et al. (1995) presented the integrated form (Equation 6) after substitution of Equation 5 into Equation 3.

$$r = -kmNC^n t^{m-1} \quad (3)$$

$$\ln (N/N_o) = - kC^n t^m \quad (4)$$

$$C = C_o e^{-k't} \quad (5)$$

$$\ln(N/N_o) = -(m/nk')^m * kC_o^n * [1-e^{(-nk't/m)}]^m \quad (6)$$

where:
k' = first order decay rate of disinfectant, 1/t
C_o = initial disinfectant concentration, M/V
k, m, n : empirical constants for Hom model
t = contact time

Variations on these disinfection models are possible but are rarely used. The simple Chick-Watson model was the most appropriate model for comparing *Cryptosporidium* disinfection data from a number of research groups, because of the inherent variation in experimental data (unpublished data, International *Cryptosporidium* CT Workshop, Washington, DC, January 12–14, 1998).

4.2.1 Integrated disinfection design framework

The integrated disinfection design framework (IDDF) model incorporates disinfection kinetics into a hydraulic model of the treatment process (Bellamy, Finch & Haas, 1998). The four steps in implementing the framework are:

1. Determine the contactor hydraulics.
2. Determine the disinfectant characteristics.
3. Determine the inactivation kinetics.
4. Develop a disinfection model.

The advantage of the IDDF model is that it more accurately predicts microbial inactivation because it accounts for basin hydraulics, the decay of the disinfectant within the basin and non-linear disinfection kinetics. The model can be run as a spreadsheet calculation or with an easy-to-use operator interface. Because of the need to balance disinfection efficiency with disinfection by-product formation, a variation of the IDDF model will probably be used to estimate *Cryptosporidium* inactivation under future regulatory scenarios.

5

Treatment variability

Maintaining reliable treatment performance is critical for minimizing microbial risk, because health effects associated with microbial contaminants tend to be due to short-term, single dose exposure rather than long-term exposure. However, drinking-water treatment is a dynamic process and the treatment efficiency for removal or inactivation of microbial pathogens is variable. This is illustrated by an on-site survey of 100 water treatment plants across the USA, which found that the removal efficiency of particles greater than 2 μm ranged from 0.04 to 5.5 logs, with a median value of 2.8 logs (McTigue et al., 1998). The study also found significant variation in the removal efficiencies of *Cryptosporidium* oocysts and *Giardia* cysts, although the removal of these pathogens did not necessarily correlate directly with the removal of particles.

Some process variation is normal and expected; however, too much variability can result in treatment failures, leading to waterborne disease outbreaks. It is the objective of drinking-water standards, therefore, to keep process variability within acceptable limits.

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This chapter looks at the possible effects of treatment process variability, how changes in one unit process can affect the efficiency of other processes, the dynamic nature of treatment processes, the effects of changes in raw water quality and the variation that can arise from process measurements.

5.1 EFFECTS OF PROCESS VARIABILITY

Treatment efficiency for removal of microbes may vary between treatment plants, between unit treatment processes and between microbes. The net result is that removal efficiency may some sometimes be low. Figure 5.1 shows hypothetical log removals of a microbe by the conventional water treatment processes of coagulation and clarification, filtration and disinfection, as a cumulative frequency distribution function. An average removal of, for example, 1 log by coagulation and clarification, 2 logs by filtration and 3 logs by disinfection would result in an average removal of 6 logs for the combined processes. However, because of the variability associated with each unit process, the removal efficiency may be as low as 3.4 logs for 10% of the time.

Although it may be possible to offset the reduced performance of removal processes (e.g. coagulation and clarification, and filtration) with increased disinfection, often the failure of one process affects the performance of other processes. This is because unit processes in water treatment plants are interrelated, as described below.

5.2 RELATIONSHIPS BETWEEN TREATMENT PROCESSES

The performance of a treatment unit can affect the efficiency of downstream treatment units. For example, the presence of suspended solids increases the resistance of most microbes to disinfection (LeChevallier, Evans & Seidler, 1981). Therefore, a failure in the removal efficiency of turbidity or particles by granular filtration processes can decrease the inactivation efficiency of disinfection processes. Similarly, clarification affects filter performance. Clarification removes suspended solids, thus reducing the solid loading to the filters and improving filter performance. If an incorrect dose of coagulant is used and floc is carried over from a sedimentation tank, head loss develops more rapidly, shortening the filter run.

A further example of how treatment processes are related is the effect of pre-oxidation on the removal of particles and microbes by granular filtration. By affecting the surface properties of particles and microbes, pre-oxidation can improve the performance of granular filters (Au et al., 2002). However, many

water utilities are considering delaying or omitting the addition of oxidants such as chlorine and ozone before filtration, in an attempt to reduce the formation of disinfectant by-products. These strategies must be carefully considered because of possible adverse effects on filtration performance (Au & LeChevallier, 2000). Possible impacts (either positive or negative) on other unit processes must be evaluated when considering modification of any unit process to achieve a particular microbial goal.

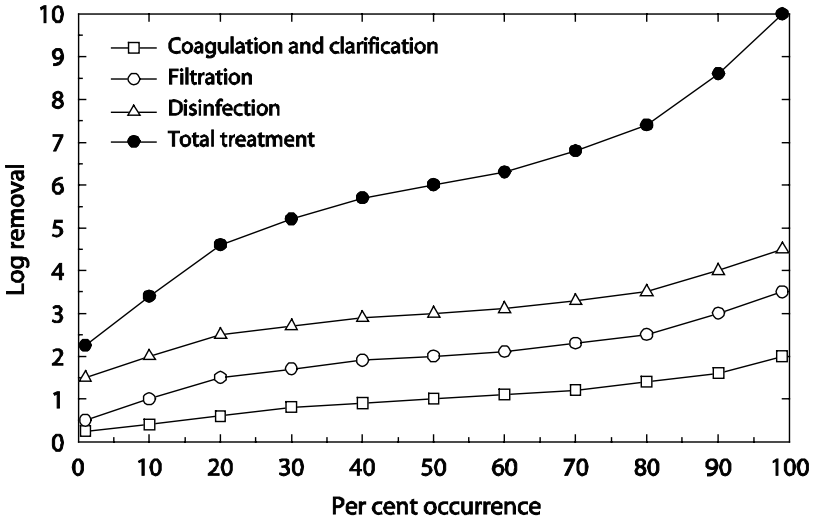

Figure 5.1 Hypothetical example of log removal as function of per cent occurrence.

5.3 DYNAMIC NATURE OF TREATMENT PROCESSES

Variation in the efficiency of water treatment processes can be due to the dynamic nature of the processes. For example, removal efficiency of granular media high-rate filtration varies throughout a filter run, which may last from a few hours to several days. As described in Chapter 2 (Section 2.5.1), after a filter is cleaned by backwashing, it performs poorly during the ripening period, before achieving a stable level of performance, which will eventually be followed by degradation and breakthrough of microbes at the end of the run. The effect of the variable performance of an individual filter on the final quality of the filtered water can be reduced by using multiple filters. This concept is

similar to that of multiple barriers, where sufficient overlap in treatment systems ensures a reliable finished water quality.

5.4 EFFECTS OF CHANGES IN RAW WATER QUALITY

Changes in raw water quality can affect the efficiency of treatment processes. Depending on local and seasonal situations, each water treatment plant encounters different ranges of raw water quality. Data from 67 surface water treatment plants in the USA showed that the variation in particles greater than 3 μm in raw water followed a log-normal distribution pattern; particle concentrations ranged from 28/ml to 11×10^7/ml, with a geometric mean of 22 800/ml (Arora et al., 1998). Factors influencing raw water quality are discussed in Chapter 6 (Section 6.2). A change of any water quality parameter in the source water may affect apparent treatment efficiency, as discussed in Section 5.3. For example, in their study of 67 surface water treatment plants in the USA, Arora et al. (1998) found that the removal efficiency (based on the difference in particle concentrations between raw and filtered waters) increased with increasing particle concentration in raw water. For raw water particle concentrations from 10^3–25×10^3/ml, the median removal efficiency was 2.08 logs; whereas, when concentrations increased to 10^6–10^7/ml, the median removal efficiency increased to 3.2 logs. The greater removal efficiency at higher particle concentrations was due primarily to more efficient clarification. This is to be expected because removal of particles by clarification depends significantly on aggregation efficiency, which is a second-order process with respect to particle concentration (i.e. a higher particle concentration means that particles will collide more frequently and thus be more likely to aggregate).

5.5 VARIABILITY DUE TO PROCESS MEASUREMENTS

With respect to removal of microbes, treatment reliability relates to the expected variation in treatment performance. Monitoring of process performance must include assessment of variability. However, uncertainties in analytical measurements may make this process more complex, particularly when few analyses are performed (Frey et al., 1998). Direct measurements of treatment performance for microbial removal may be difficult due to the time it takes to perform the analysis and the low concentrations of microbes in raw waters. Surrogate measures such as turbidity, particle counts or total coliforms may be used, but these also have limitations (Nieminski & Bellamy, 2000).

These difficulties and limitations create uncertainties in estimating treatment performance, meaning that observed (apparent) treatment performance and variability may not reflect the actual (intrinsic) performance. For example, a

treatment plant may show 2 logs of particle count removal, resulting in an effluent count of 10 particles/ml. If the source water particle count increases due to a storm event, so that the difference between the source water level and the treated count (which is still 10 particles/ml) is now 4 logs, without any change in treatment parameters, has treatment improved? Using the apparent measure of performance (i.e. particle counts), the conclusion would be that it has improved. However, the apparent improvement of the performance of the treatment process was influenced by changes in the source water; the intrinsic capability of the plant to provide 4 logs of microbial protection may have been present all along!

A study by McTigue et al. (1998) illustrates this point (Table 5.1). In pilot plant experiments, the level of *Cryptosporidium* was varied from 26 to 4610 oocysts/l. Monitoring of the plant effluent showed a consistent removal of approximately 4 logs. Turbidity and particle count data, which were limited because of relatively low levels in source water, showed an apparent removal of 1.0–1.6 logs. A plot of *Cryptosporidium* levels in raw water versus detection of oocyst in filtered effluent suggests that breakthrough occurs at a treatment plant performance level of approximately 4–5 logs (Figure 5.2).

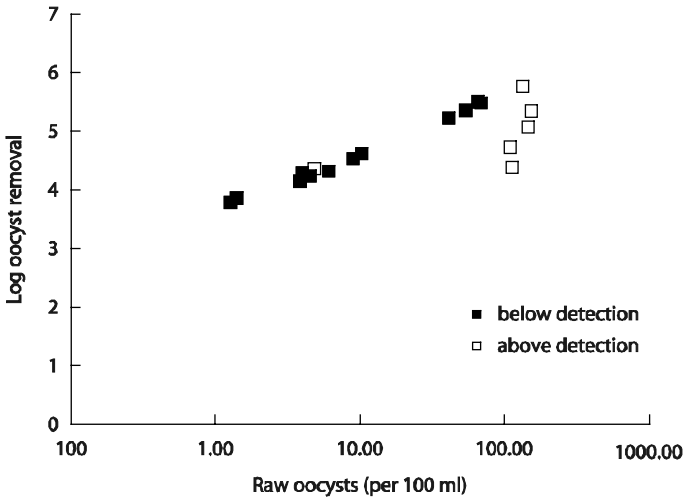

Figure 5.2 Evaluation of a pilot treatment plant performance for removal of *Cryptosporidium* oocysts. Data demonstrate performance of 4–5 log removal. Source: D Cornwell, personal communication (2001).

Table 5.1 Impact of source water concentration on apparent treatment performance (results of three trials)

Oocysts/l			Turbidity (NTU)			Particles > 3 µm/ml		
Raw	Effluent	Log removal	Raw	Effluent	Log removal	Raw	Effluent	Log removal
26	0.0017	4.2	2.5	0.07	1.6	7000	350	1.3
688	0.041	4.2	2.0	0.07	1.5	7700	530	1.2
4610	0.214	4.3	1.3	0.07	1.3	4700	480	1.0

NTU = nephelometric turbidity unit
Source: McTigue et al. (1998)

6

Process control

To assure optimal finished water quality, control programs should be comprehensive, and should include multiple barriers and adequate process design and operation. As discussed in Chapter 1, the third edition of the World Health Organization's (WHO) *Guidelines for Drinking-water Quality* provides guidance on the development of a water safety plan, based on a water safety framework (WHO, 2004). Such control programs are the basis for maintaining reliable treatment performance. Multiple barriers should provide a consistent level of protection, and adequate design and operation should ensure that performance meets specifications. Even so, treatment performance is likely to vary, mainly because of the dynamic nature of each individual treatment process, the interrelationship between different processes and changes in raw water quality, as discussed in Chapter 5.

This chapter looks at process control, in the context of a risk management approach, from source water to the distribution system.

 Water Treatment and Pathogen Control: Process Efficiency in Achieving Safe Drinking Water. Edited by Mark W LeChevallier and Kwok-Keung Au. ISBN: 1 84339 069 8. Published by IWA Publishing, London, UK.

6.1 RISK ASSESSMENT AND PROCESS CONTROL

Use of risk assessment techniques as a tool for process control in the water industry has received increased attention in the past decade. Examples of such techniques include "hazard analysis critical control point" (HACCP) and "failure mode and effects analysis" (FMEA) (Hall, Watts & Egerton, 2000). In FMEA, risk is quantified (ranked) in terms of the frequency of specific failure events and the consequences of those failures, as illustrated in Table 6.1. Selection of appropriate failure events, such as a high concentration of particles or coliforms, is directly correlated with microbial risk. Using the rankings, risk can be quantified for individual elements of the treatment process or for whole treatment works.

Table 6.1 Example of ranking of frequencies and consequences for failure mode and effects analysis

Rank	Frequency of failure
1	Very unlikely (< 1/100 years)
2	Unlikely (> 1/100 years)
3	Moderate (> 1/10 years)
4	Frequent (> 1/year)
5	Very frequent (>1/month)
Rank	**Consequence of failure**
1	No impact on operation (increased operator effort only)
2	Limited impact (minor deterioration in output quality, internal incident report)
3	Moderate (customer awareness, increased pressure group activity)
4	Severe (regulatory exceedance, adverse publicity)
5	Catastrophic (life or health threatening, environmental damage)

Source: Hall, Watts & Egerton, 2000.

The water safety plan described in the WHO *Guidelines for Drinking-water Quality* (WHO, 2004) provides a common framework for applying risk management techniques in the water industry. The procedure has three main components:

- *System assessment* to determine whether the drinking-water supply chain as a whole (up to the point of consumption) can deliver water of a quality that meets identified targets. It includes assessment of design criteria for new systems.
- Identification of control measures in a drinking-water system that will collectively control identified risks and ensure that the health-based targets are met. For each control measure identified, an appropriate means of

monitoring should be defined, to ensure that any deviation from required performance is rapidly detected.

- *Management* plans describing actions to be taken during normal operation or incident conditions, and documenting the system assessment (including upgrade and improvement), monitoring and communication plans, and supporting programmes.

This common framework quantifies hazards or risks within the whole treatment process, and identifies important monitoring and remedial actions at designated hazard control points. The rest of this chapter discusses hazard control process in a conventional water treatment plant.

6.2 SOURCE WATER PROTECTION

Table 6.2 summarizes the main elements of a hazard control strategy for source waters. Understanding variations in raw water quality is important because such variations affect the treatment efficiency (as discussed in Chapter 5, Section 5.4) and thus the health risk associated with the finished water. In general, raw water quality is influenced by both natural and human factors. Important natural factors include wildlife, climate, topography, geology and vegetation. For example, beavers and other mammals are potential sources of *Giardia intestinalis*, and migratory geese have caused seasonal increases in coliform bacteria in some north-eastern water supply watersheds in the USA (Robbins et al., 1991). Human factors include point sources (e.g. discharges of municipal wastewater and industrial wastewater) and nonpoint sources (e.g. urban runoff, livestock or recreational activities). Municipal wastewater can be a major source of microbial pathogens, urban runoff and livestock can contribute a substantial load of coliform bacteria, and recreational activities involving body contact can be a source of faecal contamination.

Table 6.2 Hazard control elements for source waters

Potential control strategies	Control measures
Assessment of pollution sources	Wildlife Agriculture Sewage treatment plants
Watershed protection	Land acquisition, riparian barriers Land or water use restrictions
Hydrological conditions	Rainfall, flow, monitoring Changes in source water quality Reservoir destratification
Watershed networks	Reporting network Identification and prosecution of violators

Protection of source water can help to minimize microbial risk associated with the water entering a drinking-water treatment plant. Possible control measures to protect source water include land acquisition, watershed inspection programmes, reservoir-use restrictions and riparian buffers. Few water utilities own all or even most of the land within their watersheds; thus, it may be difficult for water utilities alone to control or reduce the risk from identified hazards. Competition for water and pressure for increased development in a catchment may appear to limit the extent to which potentially polluting activities can be reduced. However, it is often possible to contain hazards without substantially restricting activities. Collaboration between stakeholders can allow pollution to be reduced without reducing beneficial development.

From a water utility perspective, developing a monitoring programme and carrying out corresponding actions at the early stage of the treatment process are sometimes the most effective ways to minimize microbial risk from raw water. Examples of methods to reduce the risk include determining the vulnerability of the intake to microbial contaminants, managing the raw water pumping schedule and applying pretreatment oxidants.

Hydrological events can increase microbial levels in source water. For example, rainfall can wash microbes into receiving streams and increased stream flow can resuspend microbes settled in streambed sediments. Figure 6.1 demonstrates a peak in *Cryptosporidium* levels associated with rainfall, and subsequent increases in river flow and turbidity levels (Atherholt et al., 1998). Similar increases in total coliforms, faecal coliforms, faecal streptococci and staphylococci have been seen in other studies (e.g. Davis, Casserly & Moore, 1977). Changes in flow may also be due to release of water from upstream dams, reservoirs or other impoundments. Turnover of lakes and reservoirs following seasonal stratification can release microbes or other factors that increase disinfectant demand, which may interfere with treatment operations.

A watershed-monitoring network can be useful for detecting contamination events and alerting downstream users of the pending plume. For example, the Ohio River Valley Water Sanitation Commission (ORSANCO[1]) is a network of users of water from the Ohio River. ORSANCO monitors daily for a variety of contaminants, and serves as a centralized clearing house for data analysis and interpretation. By pinpointing the source of the contamination, violators can be identified and prosecuted. The result is a greater attention to minimizing contamination reaching the river and an overall improvement in water quality. Similar networks are present for some of the major rivers in Europe and Asia (e.g. the Rhine River in Germany, Holland and Switzerland; the Llobregat River in Spain; and the River Han in Korea) (Grayman, Deininger & Males, 2000).

6.3 COAGULATION, FLOCCULATION AND CLARIFICATION

Coagulation

Chemical coagulation pretreatment is the most important factor in ensuring efficient removal of microbes by coagulation, flocculation and clarification and by granular media filtration. It also indirectly affects the efficiency of the disinfection process. Although the coagulation process itself is unlikely to cause any microbial hazard or risk to finished water, a failure or inefficiency in the coagulation process could result in a high microbial risk to drinking-water consumers. Hazard control strategies for the coagulation process are outlined in Table 6.3.

[1] www.orsanco.org

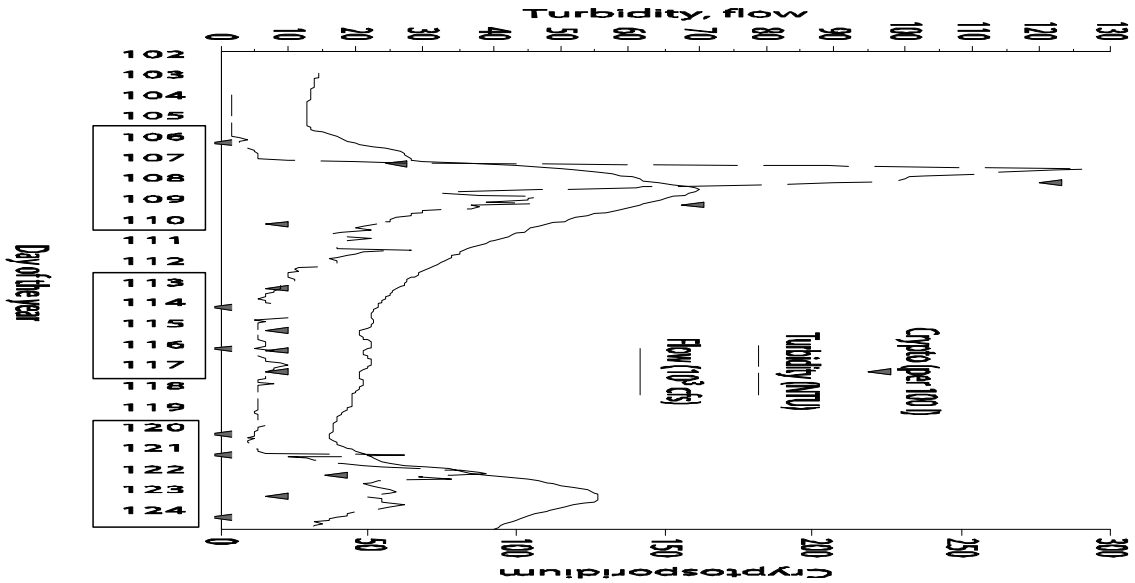

Figure 6.1 Hydrological event showing increases in *Cryptosporidium* oocyst levels accompanying increases in source water turbidity and river flow rate (Atherholt et al., 1998).

Table 6.3 Hazard control elements for coagulation, flocculation and clarification

Potential control strategies	Control measures
Chemical coagulation	Selection of appropriate primary coagulant and coagulant aid (if required) Dose (determined by testing) Coagulant feed paced to flow rate pH and alkalinity adjustment Appropriate mixing Temperature
Flocculation	Gentle mixing to maximize flocculation Flow rate Flocculant aids
Clarification	Surface loading rate Effective sludge removal Settled water turbidity or other process indicator

The first step is to choose an appropriate coagulant (and coagulant aid if necessary) and dose. Next, it is important to ensure that the chemical feed rate is appropriate for the plant flow, because changes in flow rate could result in an over or under-dose of coagulant, impairing performance. Water chemistry and temperature can affect the performance of many coagulants, and adjustment of pH may be necessary for optimal performance.

Commonly used approaches for determining appropriate coagulation chemistry and for monitoring coagulation include jar tests, streaming current detectors, zeta potential, pilot filters, historical dosage charts, visual observations, pH, alkalinity, temperature and ultraviolet (UV) absorbance (for a review, see Logsdon, Hess & Chipps, 2000). Typically, coagulation efficiency is evaluated using one or more of these approaches and the process parameters adjusted accordingly. The choice of approach depends on the site. For example, streaming current potential may be appropriate when charge neutralization is the main destabilization mechanism, but may not be suitable when enmeshment in a precipitate (sweep-floc) is the main mechanism.

Flocculation

Critical to the performance of effective flocculation is gentle mixing to promote particle aggregation. The calculation of the velocity gradient necessary for proper flocculation can be estimated by the G value, as shown below:

$$G = \left(\frac{P}{V\mu}\right)^{1/2}$$

where:
P = the power input to the fluid
V = the volume of the flocculator
μ = the absolute viscosity of the water

If the G value is too high, the floc may be sheared; if it is too low, sedimentation may occur within the flocculation basin. In water treatment, typical G values are 10–100/sec (Weber 1972). Although this calculation has many flaws, it is useful for flocculator design and scale-up (Letterman, Amirtharajah & O'Melia, 1999).

Clarification

Factors influencing clarification performance include the surface loading rate (expressed as flow rate per unit surface area of the clarification basin), the size and shape of the tank, flow velocity, adequate sludge removal and physicochemical characteristics of the water (USEPA, 1991; Gregory, Zabel & Edzwald, 1999). Recommended surface loading rates vary widely, depending on the type of clarification process. Conventional sedimentation basins may have surface loading rates of 0.6–2.0 m/h (0.25–0.8 gpm/ft^2). High-rate clarifiers may have surface loading rates of 7 m/h (2.9 gpm/ft^2) or greater. Adequate sludge removal is important because sludge accumulation reduces the volume of the clarification basin and can increase the velocity of the flow through the basin. To prevent the formation of currents and breakage of floc, the overflow rate should be as low as possible. Adjustable V-notched weirs provide operational flexibility.

6.4 FILTRATION

Granular media filtration is an important barrier to microbes (see Chapter 2, Section 2.5). It may be the only barrier in some cases; for example, for removing *Cryptosporidium* oocysts by direct filtration when chlorine is used as the sole disinfectant. Hazard control elements for operating granular media filters to reduce microbial risk are outlined in Table 6.4.

Filter performance can be evaluated by various methods, such as on-line measurement of effluent turbidity (from individual and combined filters) and counting of particles or other surrogates for microbes. To provide

comprehensive process control for filtration, it may be useful to measure other operational parameters related to filter performance (e.g. rate of head loss).

The riskiest operations to perform with a granular media filter are starting a filter after backwashing, and increasing the filtration rate (Logsdon, Hess & Chipps, 2000). Filters typically perform poorly at first after backwashing, and passage of microbial pathogens during this ripening period can be formidable. Methods to minimize such problems include various start-up strategies (e.g. filtering to waste, allowing the filters to settle after backwashing and starting slowly) and adding a filter aid to the backwash water supply during the final minutes of the backwash process. Passage of particles and microbes at the end of a filter run can be avoided by taking the filters out of service before head loss becomes terminal, or turbidity or particle counts increase. In many systems, backwash of filters is simply based on the filter run time, to avoid any decrease in water quality at the end of a filter cycle.

Table 6.4 Hazard control elements for granular media filtration

Potential control strategies	Control measures
Monitoring of process control	On-line turbidity or particle counting Flow rate Head loss rate
Minimize filter ripening, breakthrough	Filter to waste Slow start of filters Allowing filters to settle after backwash Add coagulant to wash water supply Avoid terminal head loss
Minimize changes in filtration rate	Surface loading rate Effective sludge removal Settled water turbidity or other process indicator
Effective backwash cleaning	Proper bed expansion Media agitation by air or mechanical washers

Changes in the filtration rate are often unavoidable; for example, when one filter is taken off-line for backwashing. An increase in filtration rate can be detrimental to filtered water quality (Cleasby, Williamson & Baumann, 1963; Fitzpatrick, Campbell & Cable, 1999). The impact can be minimized in various ways; for example, by slowly increasing the filtration rates for the filters remaining in service, or by decreasing plant production while a filter is temporarily out of service.

Also critical to the functioning of granular media filters is the cleaning of the filters during backwash. If not performed correctly, cleaning can lead to

clumping of the filter media (formation of mud balls), improper distribution of the media (formation of mounds), destratification of the multimedia layers or media washout. To clean the media grains properly, the filter bed must be fluidized as well as scoured (either mechanically or by air).

6.5 DISINFECTION

In most conventional treatment processes, an adequate level of disinfection is critical for reducing microbial risk to acceptable levels (Table 6.5). Microbial pathogens include highly diverse groups and it is impossible to monitor the survival of all pathogens. Estimating the level of inactivation of more resistant microbial pathogens, by applying the CT concept (disinfectant concentration and contact time) for a particular pH and temperature, ensures that more sensitive microbes are also effectively controlled. The CT concept can sometimes be as simple as providing a certain disinfectant residual for a prescribed contact time.

Table 6.5 Hazard control elements for disinfection

Potential control strategies	Control measures
Indirect monitoring	Disinfectant dose and/or residual Contact time pH, temperature
Direct monitoring	Coliform, *Escherichia coli* and/or other treatment indicators Surrogates: bacteriophage, spore-forming bacteria

The use of indirect monitoring methods depends on the type of disinfectant used (Haas, 1999). For example, control of chlorination systems is often based on measurements of residual chlorine; control of systems using ozone can be based on off-gas ozone monitors or measurements of the dissolved ozone residual; and control of UV systems can be based on continuous monitoring of light absorption and control of lamps to deliver a particular energy intensity.

To assess the inactivation efficiency of the disinfection process, indicator organisms are often used. Typical indicators include total coliforms, faecal coliforms and heterotrophic bacteria, as measured by heterotrophic plate count (HPC). Other indicators may include bacteriophage, aerobic spore-forming bacteria or *Clostridium* oocysts.

6.6 DISTRIBUTION SYSTEM

Protection of the distribution system is the last and one of the most important of the multiple barriers necessary for provision of safe drinking-water. Any microbial contamination at this point has a high probability of resulting in public health risk, even if previous control steps have been applied effectively. Because of the extensive nature of the distribution system, with many kilometres of pipe, storage tanks, interconnections with industrial users and the potential for tampering and vandalism, opportunities for microbial risk do occur (Geldreich 1996; Geldreich & LeChevallier, 1999; Ainsworth, 2004). Hazard control strategies should focus on three essential elements:

- maintaining the quality of the treated water by adequate maintenance of the distribution system;
- minimizing bacterial growth;
- preventing recontamination of the water during distribution (Table 6.6).

Fundamental to the quality of the treated water is the proper operation and maintenance of the pipe system. The WHO publication *Safe piped water: Managing microbial water quality in piped distribution systems* (Ainsworth, 2004) provides comprehensive guidance on the management of distribution system operation and maintenance. It includes guidance on development of a monitoring program for water quality and other parameters, such as pressure in the distribution system. Control measures include using a more stable secondary disinfecting chemical than is used in primary treatment (e.g. chloramines instead of free chlorine), reducing the time that water spends in the system (e.g. avoiding stagnation in storage tanks and looping dead-end sections), replacing pipes, flushing and relining, and maintaining positive pressure in the distribution system.

Critical factors for controlling the replication of bacteria in finished drinking-water are:

- maintenance of a disinfectant residual
- limitation of biodegradable organic material
- control of corrosion.

Other parameters, such as temperature, construction materials and detention time are also important, but may not be easily controlled. In the absence of a disinfectant residual, the permissible level of biodegradable organic carbon may be very low.

Preventing recontamination of the treated water is the primary focus of a cross-connection control program. Devices to control backflow and back-siphonage should be installed at any location that may pose a risk to the treated

water (e.g. industrial users, mortuaries, hospitals, tanker trucks and street cleaners). Hydraulic surges caused by rapid changes in pump or valve operations may cause transient negative pressures that are not recorded by conventional pressure monitors (LeChevallier, 1999). Detecting and controlling leaks can limit the opportunities for entry of microbes during negative pressure events.

Table 6.6 Hazard control elements for distribution system protection

Potential control strategies	Control measures
Distribution system maintenance	Flush and clean tanks regularly Minimize stagnation Maintain and replace infrastructure Monitor to detect areas of water quality degradation
Control of bacterial growth	Maintain an effective disinfectant residual Reduce biodegradable organic carbon Control corrosion
Cross-connection control and avoidance of transient pressure	Institute a cross-connection control programme Maintain positive distribution water pressure Avoid hydraulic surges that may create transient negative pressures Control leakage

7

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