2

Minimizing potential for changes in microbial quality of treated water

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2.1 INTRODUCTION

The microbial quality of drinking-water can change as it travels from the treatment plant to the extremities of the distribution network. Microbial proliferation will depend on the:

- transit times
- system condition
- construction materials
- water temperature
- disinfectant residual
- hydraulic conditions
- initial physical, chemical and microbial characteristics of the treated water.

It is not meaningful or practicable to strive for a sterile drinking-water network devoid of all microorganisms. The principal objective is to remove pathogenic organisms from the water supply and prevent contamination during distribution. This requires effective management of the operation, maintenance and cleanliness of the distribution network. The management process should include optimization of treatment to minimize the entry of microbial nutrients and deposit-forming components into the network. This will help to prevent water discolouration, tastes, odours and the proliferation of microorganisms (which may create a food-chain leading to the appearance of animals such as crustaceans). The potential health significance of microorganisms growing in piped supply systems is discussed in Chapter 1. The presence of large numbers of bacteria in the conveyed water may make it difficult to identify serious contamination from outside the pipework and reservoir structures. Finally, the proliferation of nonpathogenic organisms may make water unpalatable and encourage recipients to consume an alternative, and possibly less safe, source of water.

This chapter looks at how treatment can be optimized to control microbial growth, corrosion of pipe materials and the formation of deposits such as biofilms and sediments. It is not a general guide to water treatment.

2.2 MICROBIAL GROWTH FACTORS

Biological activity in a distribution system is normally most intense at the interface between the water and structural materials (in formations generally described as biofilms), and within deposits formed by particulate matter and corrosion.

The growth of biofilms depends on the nature of the material, the hydraulic conditions and the physical and chemical characteristics of the water (Camper et al., 2000). Colonisation occurs from the first contact between certain microorganisms (mainly bacteria) and a new material. It then evolves through the integration of various levels and species that can cohabit and exchange nutrients by reacting to external conditions.

When a microbial biofilm has formed, or a deposit containing organic matter has precipitated, it can serve as a food source for predators such as protozoa, which may themselves be consumed by higher animals such as *Asellus aquaticus* (see Chapter 6).

The factors controlling microbial growth and development in distribution systems are shown in Figure 2.1. Some of these are discussed below.

- **Temperature** — If nutrients are available, the microbial activity (as measured by HPC) increases significantly at water temperatures above 15ºC, in the absence of a disinfectant residual.

- **pH** — Most microorganisms survive at the pH values normally found in drinking-water.
- **Oxygen** — Water supplies are normally well aerated, which reduces the risk of microbially-induced corrosion, denitrification, sulfide production and other consequences of anaerobic stagnation. However, oxygen may not penetrate to the bottom layers of biofilms, corrosion tubercles, and other pipe deposits and reservoir sediments where anaerobes such as sulfate-reducing bacteria may proliferate.
- **Nutrients** — Although some microorganisms can survive on mineral elements, they are of little significance in distribution networks. However, many microorganisms can proliferate if there is sufficient dissolved or particulate organic matter containing carbon, nitrogen or phosphorus.
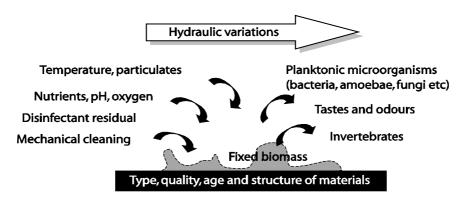

Figure 2.1. The factors influencing microbial change in water distribution systems.

2.3 TREATED WATER QUALITY OBJECTIVES

To prevent or minimize bacterial proliferation in distribution systems, the following general guidance should be followed:
- minimize particles leaving the treatment works;
- minimize the amount of particulate, colloidal and dissolved iron, manganese and aluminium compounds leaving the treatment works;
- restrict the biodegradable organic content leaving the works;
- control the corrosion potential of the water with respect to distribution system materials;
- minimize the factors causing the consumption of a residual disinfectant;
- adapt the disinfectant residual to the local conditions and climatic conditions;
- introduce a monitoring policy that can identify failures in achieving the above.

2.3.1 Disinfection strategy and the distribution system

The disinfectant concentration and contact time applied must be appropriate for the prevailing water temperature and pH, and for the target microorganisms. This is best achieved within a contact tank at the treatment plant where the hydraulics are designed to ensure effective contact between microorganisms and disinfectant. Details may be found in the companion volume addressing drinking-water treatment (LeChevallier et al., 2003). In some countries, transmission mains may be used for ensuring such contact. If so, it is important to ensure that there are no side connections or branches before full disinfection has been achieved under well-controlled contact conditions. In reality, this may be difficult to achieve.

Residual disinfectant will be consumed by corrosion products on the internal surface of metal piping, by iron and manganese deposits from corrosion and carry-over from the plant, by organic biomasses and by part of the organic matter circulating in the water. Therefore, maintenance of a residual is assisted by control of corrosion, treatment that minimizes carry-over of particulates and coagulant, low levels of dissolved organic matter, a system that is free of all types of deposit and short water-transmission times between the treatment plant and the consumer.

Chlorine, chloramines and chlorine dioxide are the three disinfectants that have been used to maintain a residual in distribution. Ozone decays too rapidly for this application. The selection of the most appropriate chemical and its dose depends on the microbial water quality targets or performance targets (see WHO, 2004). Selection also depends on the risks of developing tastes, odours and disinfection by-products such as trihalomethanes (THMs) at the point of disinfection and within the network. This decision can be complex. In many situations, the choice is also dependent on the performance of available dosing and monitoring equipment.

Chlorine

Chlorine is commonly used to maintain a residual. Its disinfecting power is a function of pH, because when chlorine is added to water it reacts to produce hypochlorous acid (HOCl):

$$Cl_2 + H_2O \rightarrow HOCl + H^+ + Cl^-$$

The hypochlorous acid will dissociate to the hypochlorite ion (OCl$^-$) as the water increases in pH:

$$HOCl \rightarrow H^+ + OCl^-$$

The hypochlorite ion is a less powerful disinfectant than hypochlorous acid. Table 2.1 shows how the proportion of the hypochlorite ion increases with pH.

Table 2.1. Dissociation of hypochlorous acid as a function of pH

pH	% HOCl	% OCl⁻
7	78	22
8	28	72
9	4	96
10	0	100

Source: Snoeyink & Jenkins (1980)

The correct selection and control of pH is therefore crucial for effective disinfection at the treatment works when using chlorine. This is less important when seeking to maintain a residual because the hypochlorite ion, although less effective, will decay more slowly and will thus persist further into the network for a particular dose. In Europe, common target concentrations for free chlorine residual at the tap are 0.1–0.3 mg/l. At the higher value, consumers commonly detect the taste and odour of chlorine. However, detection levels vary widely between people and some can detect chlorine at much lower levels. (Complaints generally occur in response to changes in concentration rather than to consistent values, whether high or low.) In some countries, much higher concentrations have been employed to maintain a residual (UKWIR, 1998a).

The formation of THMs must be considered when maintaining chlorine residual. Applying adequate treatment before distribution minimizes organic precursor compounds (Carlson, 1991) and is thus important in controlling THMs. If organic precursors to THMs remain, further chlorination may create THMs. The factors affecting the formation of THMs (UKWIR, 2000c) are:

- *pH* — about 10–20% more THMs will form at pH 9 than at pH 7;
- *time* — the rate of formation of THMs is greatest during the first 2–20 hours;
- *temperature* — at $< 10^O C$, THM concentrations do not increase significantly;
- *total organic carbon (TOC)* — at values of TOC above 4 mg/l it will be difficult to prevent THMs exceeding 100 µg/l if free chlorine is maintained to the tap for travel times of 2–3 days.

Monochloramine

A monochloramine residual may have advantages over a free chlorine residual (for a health-related benefit, see Box 2.1). In water, ammonia and chlorine react to form monochloramine (NH_2Cl), dichloramine ($NHCl_2$) and nitrogen trichloride (NCl_3). The chloramines are less powerful disinfectants than free chlorine, and are therefore often used as a secondary rather than a primary disinfectant within the treatment plant. However, they do persist in distribution

(decay rates can be up to 20 times slower than free chlorine). Nitrogen trichloride produces a strong taste and odour at concentrations above 0.02 mg/l, whereas taste and odour thresholds for monochloramine are between 0.48 and 0.65 mg/l. High concentrations of dichloramine (> 0.15 mg/l) may produce tastes and odours. It is, therefore, important to control the disinfection process to produce a stable residual that is predominantly monochloramine. This requires evaluation of the water in question as a function of temperature, but normally a molar ratio of chlorine to ammonia of one and a pH value above seven is required (Snoeyink & Jenkins, 1980; Palin, 1975).

Monochloramine residuals in distribution will not increase THMs, although the process of chloramination can give rise to these chemicals, because free chlorine will be present at some point in the process. However, if this step is well managed it will be short lived, and any THMs formed will be at low concentrations. Monochloramine is more effective than free chlorine at penetrating and inactivating organisms within biofilms, especially where corrosion products are present (LeChevallier et al., 1993; Norton, 1995). Chloramination has also been found to be effective in controlling *Naegleria fowleri* in Australian water supplies (Christy & Robinson, 1984; UWRAA, 1990).

Treatment to produce a monochloramine residual does pose the risk of nitrite formation in the distribution system, especially in low-flow stagnant areas. Bacteria on surfaces and in deposits may nitrify any slight excess of ammonia. However, careful control of the chloramination process will prevent most nitrite problems (Williams, Andrews & Wakeford, 2001). If nitrite does occur at certain locations, despite good control of the chloramination process, then the presence of internal pipe deposits at these locations should be investigated.

Chlorine dioxide

Chlorine dioxide is a powerful biocide and is used in treatment works, especially where there is a problem with THM production. However, chlorine dioxide in water produces inorganic breakdown products, chlorite and chlorate. The health significance of chlorite and chlorate in drinking-water is discussed in the third edition of WHO *Guidelines for Drinking-water Quality* (WHO, 2004). The persistence of chlorine dioxide in distribution is unclear. Microbial aftergrowth in the presence of chlorine dioxide has occurred in some systems. This has been explained by rapid conversion of chlorine dioxide to chlorite in distribution systems and the subsequent measurement of chlorite rather than the dioxide (Brett & Ridgway, 1981). Therefore, any attempt to maintain a chlorine dioxide residual would require careful investigation.

Box 2.1. Disinfection and the risk of legionnaires' disease.

The authors conducted a retrospective case-control study to identify risk factors for hospital outbreaks of legionnaires' disease. They identified 32 hospitals where one or more outbreaks had been identified between 1979 and 1997. In addition, 48 control hospitals, matched for size and whether they had transplant programmes, were identified.

Case-hospitals were far more likely to be supplied with water that contained free chlorine (rather than monochloramine) as a residual disinfectant than were control hospitals (adjusted odds ratio 10.2, 95% confidence intervals 1.4 to 460). The authors estimated that about 90% of all hospital outbreaks of legionnaires' disease could be prevented if all water utilities in the USA switched to chloramination.

The suggestion is that monochloramine residual disinfection is more effective at inhibiting the growth of biofilm in water distribution systems, and that this in turn affects the risk of legionnaires' disease. This is an example of how disinfection practice can affect the quality and safety of water in distribution.
Source: Kool, Carpenter & Fields (1999).

Management of disinfectant residuals

Not all countries have the same attitude to the maintenance of disinfectant residuals in distribution systems. Water suppliers in many European countries, for example, are seeking to reduce the use of chlorine and its derivatives wherever it is feasible to do so, by optimising treatment to remove organic material.

Hydraulics simulation models and results of microbiological analysis can be used to optimize the management of disinfectant residuals (Heraud et al., 1997; Dukan et al., 1996; Piriou et al., 1997). Controlling disinfectant residuals by booster dosing (re-dosing with a disinfectant at strategic points in the network) is described in Chapter 3. Modelling the concentration of a disinfectant in distribution requires knowledge of its reactivity with:

- the treated water (e.g. chlorine is consumed rapidly at first, followed by an on-going but slow process);
- the pipe deposits and biofilms present in the system;
- the pipeline and network construction materials (information on the chlorine and monochloramine demand of common distribution materials is available (UKWIR, 1998b; AWWARF, 1998)).

Such modelling requires specialist knowledge, but can assist in optimising the treatment process or identifying the network locations where booster disinfection will be most effective.

2.3.2 Particulate content, turbidity and coagulant residual

Particles capable of surviving the various phases of drinking-water treatment can transport microorganisms adsorbed on their surface, fixed as biofilm or integrated into the porous mass. They may be protected from an oxidizing disinfectant if the particles contain reducing compounds, such as iron oxides or organic matter. If ultraviolet (UV) light irradiation is used, the shadow cones projected by the particle mass can limit the effectiveness of this disinfecting procedure. Thus, achieving turbidities of less than 1.0 nephelometric turbidity unit (NTU) in waters entering distribution will significantly reduce the risk of breakthrough of pathogenic microorganisms, many of which may be resistant to disinfection.

The particles that settle in the network may eventually form adhesive deposits and sediments in reservoirs and pipes where microorganisms will be protected. This causes a secondary problem if changes in flow direction and velocity resuspend these deposits and associated microorganisms, contaminating the water supply.

Another common cause of particulate formation in distribution arises from the by-products of water treatment processes (e.g. where iron and aluminium compounds are used as flocculants). Water that complies with recommended metal concentrations on health and aesthetic grounds may contain sufficient material to precipitate as deposits in the distribution system (UKWIR, 2000a). Post-treatment deposition of iron and aluminium coagulant, manganese and silica has been observed (WRc, 1981). There is no generally applicable guidance for the residual concentrations of these components of treated water to avoid such deposition problems. It is therefore prudent to routinely monitor not only the composition of water leaving a treatment works but also the composition during passage to the extremities of the network to reveal deposition processes. In the United Kingdom, the recommendations shown in Table 2.2 have been made for water leaving treatment works.

Information is available on procedures for identifying and rectifying process conditions that lead to such problems, and for identifying where such post-treatment works precipitation is occurring (WRc, 1990).

Table 2.2. Recommended values for UK final waters to reduce deposition in distribution.

Determinand	Suggested maximum[a]	Suggested target[b]
Iron (mg/l)	0.10	0.05
Aluminium (mg/l)	0.10	0.05
Manganese (mg/l)	0.05	0.025
Turbidity (NTU)	0.50	1.00

NTU = nephelometric turbidity unit
Source: a, WRc (1990); b, UKWIR (2000a)

2.3.3 Organic matter

In recent years, attention has focused on the carbonaceous organic matter that can be used by microorganisms as a source of nutrients. Among the various parameters published by research teams, two have been made into international standards: biodegradable dissolved organic carbon (BDOC) and assimilable organic carbon (AOC). The purpose of these parameters is to measure the nutritional potential of water directly or indirectly in terms of carbonaceous organic compounds (Kaplan, Reasoner & Rice, 1994); they are not intended for routine monitoring. In the terminology of water safety plans, these parameters are used for "process validation" (see Section 7.3.3). They must be studied in depth during an investigational stage in order to understand fully how they develop in the source water and evolve throughout the treatment process. Thus, they serve to optimize the process and provide information during pilot testing, with a view to modifying or designing the treatment plant. The parameters will then be checked only periodically to verify the performance of the treatment works.

Assimilable organic carbon

This parameter was developed by Van der Kooij (1992). It is based on culturing two bacterial strains in the water under investigation and matching the maximum number of cells obtained with a calibration curve produced by using an easily assimilated nutrient such as sodium acetate (APHA-AWWA-WEF, 1995). After many years of experience, Van der Kooij has been able to establish an AOC scale that allows waters to be classified in terms of bacterial regrowth potential. A value of no more than 10 µg/l of AOC from the *Pseudomonas* p17 strain is recommended for biologically stable water. The AOC level is considered to indicate the quantity of carbon in a test water that can easily be assimilated by bacteria.

Biodegradable dissolved organic carbon

The sample to be analysed is placed in contact with a native mixed biomass. Monitoring of the dissolved organic carbon (DOC) enables the degradation of the organic matter to be observed, with the corresponding increase in carbon dioxide and bacterial cells (Figure 2.2). When the degradation has reached a plateau, the value obtained is described as refractory dissolved organic carbon (RDOC). The difference between the initial DOC and the RDOC enables the BDOC to be calculated in milligrams per litre (Block et al., 1992). A 30-day incubation method has been published (Servais, Anzil & Ventresque, 1989). For a faster result, Joret & Levi (1986) incubated the sample on a mass of colonised sand, generally taken from the sand filters of treatment plants. This sand contains a native biomass that is well adapted to the water under investigation

and can completely degrade the BDOC in 5–7 days. These two methods have been compared by Volk et al. (1994).
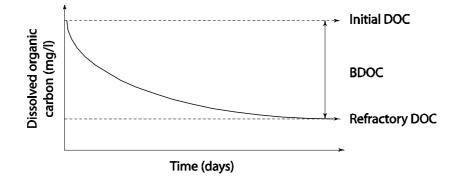

Figure 2.2.   An example of a typical biodegradation curve for dissolved organic carbon (DOC).

The method allows quantification not only of the easily assimilable carbon but also of the carbon that will be degraded by bacteria more slowly during distribution. It has been found that waters that are biologically stable in distribution have BDOC values of 0.2 mg/l or less. Comparisons of the methods have shown that AOC is the most easily assimilated fraction of the BDOC. Whichever method is used, ozonation has been found to lead to conversion of refractory TOC into biodegradable TOC. Consequently, the use of ozone is not recommended during the final treatment stages before distribution (Joret, Levi & Volk, 1991; Volk & LeChevallier, 1999).

Biofilm formation potential

Methods for assessing biofilm formation potential combine information about the dissolved organic content with an evaluation of its potential to promote the proliferation of fixed biomasses.

One of these methods determines the biofilm formation potential (BFP) parameter (Van der Kooij & Veenendal, 1992). The water under investigation is percolated continuously through a device containing glass cylinders. At regular intervals, a cylinder is sampled and the fixed biomass estimated by calculating the metabolic activity. This is achieved by measuring the level of adenosine triphosphate (ATP, a component of all living microbial cells) using luminescence. Plotting the kinetics allows different waters to be compared, and investigators in the Netherlands have used this technique to identify water resources that are more likely to support biofilm formation in distribution systems (Van den Hoven et al., 1996).

A European task force has worked on the development of a similar method that is simple, easy and cheap, and does not require complex equipment (AGHTM, 1999). The water is percolated through a bed of glass beads and the biomass is assessed by

bacterial counts on R2A agar or by measuring ATP, total protein or TOC. The protocol has been validated but results vary between laboratories.

Although these methods are not the subject of international standardization, they can be used to select water resources and treatment options for minimizing biodegradable organic matter entering distribution systems. However, due to difficulties with interlaboratory comparability, the analysis must be done systematically by the same individuals using the same methods, to limit the variability of results.

2.3.4 Limiting the potential for corrosion and scale

Internal corrosion of iron pipes reduces their structural strength and may create leaks and bursts, as does external corrosion. Internal corrosion also increases the consumption of disinfectant residual, decreases the water-carrying capacity of the pipe and creates deposits that are undesirable in terms of maintaining high microbial quality (LeChevallier et al., 1993). The internal corrosion of the traditional cast and ductile iron pipes, protected by a paint layer of coal-tar (no longer recommended due to leaching of polycyclic aromatic hydrocarbons), often produces hard adhesive tubercles, as shown in Plate 2.1.

In unprotected steel pipes, the corrosion products tend to be more uniform and less adhesive. Other scales based on adhesive and layered calcium carbonate deposits can form in mains when the conveyed water is excessively supersaturated with calcium carbonate. These may also be found in association with corrosion products and biofilms (Lu, Kiene & Levi, 1998). Calcium carbonate deposits and ferrous corrosion products usually require mechanical action or an acid chemical process for their removal.
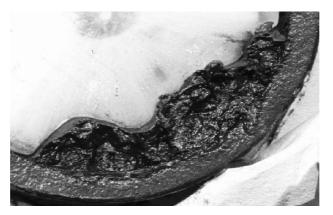

Plate 2.1. Corrosion tubercles in 40 year old cast iron main (100 mm diameter).

To prevent corrosion and scale, the water should be saturated or slightly over-saturated with calcium carbonate (Legrand & Leroy, 1995).

Several options are available for controlling the corrosivity of treated water towards the range of materials used in the distribution network. These should be considered on a case-by-case basis. It is also important to consider how changes in water composition may affect the corrosion of all distribution and plumbing materials. Guidance is available for minimizing corrosion in networks containing pipes made from iron, lead, copper, galvanised steel and cement-based materials (UKWIR, 2000b).

2.4 OPTIMIZATION OF TREATMENT

2.4.1 Water sources

Where there is a choice of water source, content of biodegradable organic matter should be considered. Protected underground resources or sites, where there can be groundwater replenishment or bank infiltration, are preferred. In most circumstances, the quality of groundwater improves during its passage through subterranean rocks and subsoils, resulting in more biologically stable water entering the distribution system. In contrast, surface waters containing a lot of humus-based material, urban or industrial effluents, or a proliferation of algae are more difficult to treat and are more likely to contain a high proportion of undesirable nutrients.

For small, community-managed systems, the selection of source waters with lower potential for promoting regrowth is preferred, given the usually limited availability of water-quality testing equipment and lack of skills in interpreting the results of microbiological analysis. However, the overriding requirement should still be the selection of sources with the lowest risk of contamination with pathogens.

2.4.2 Drinking-water treatment plant

The water treatment plant should be designed and operated to minimize dissolved and particulate nutrient entering the network. The processes include biodegradation, retention and adsorption (Jacangelo et al., 1995), summarized below.

A preliminary stage of biological nitrification will remove ammonia without using chlorine, thus limiting the formation of unwanted by-products. However, the biodegradation will not be effective below 5°C, and if nitrification stops during prolonged periods of low temperatures, it will be necessary to use a stand-by chemical method.

Flocculation must be optimized by selecting the best flocculant and the best pH, not just to reduce turbidity but also to reduce both colloidal and dissolved organic matter. If this stage is well managed, it can also reduce some undesirable organic

micropollutants and disinfection by-product precursors (Crozes, White & Marshall, 1995). The chemistry of the coagulation process must also be controlled to minimize carry-over of dissolved and colloidal coagulant. This is especially important in waters of low calcium hardness (WRc, 1992).

Some clarifiers, like the floc blanket reactors that promote prolonged contact between the water and the microorganisms held in the sludge, allow a biodegradation stage to be introduced at the beginning of the treatment cycle (Campos et al., 1999a). The addition of powdered activated carbon may be used to reduce the DOC content.

Biological sand filters have a biodegrading effect because of the biomass that develops in the first few tens of centimetres. The particle size and contact time must promote good contact between the organic carbon and the biomass. Generally, the bacteria must not come into contact with disinfectants used during water pretreatment or in the filter cleaning water, because this would slow down the biological activity in the filters (Laurent et al., 1999a). Monitoring is essential to identify any risk of excessive growth, which could cause blockages in the filter mass and thereby limit the effectiveness of filter backwashing (Croue et al., 1997). Rapid filtration usually results in a BDOC reduction of 20–30%, as long as the water does not contain disinfectant residual.

Dune, riverbank and soil filtration are very effective methods if the residence times are long enough. In addition to the removal of particulate matter and microorganisms, BDOC may be considerably reduced, ensuring stability in the distribution system.

The biocidal and oxidizing effects of ozonation can be highly beneficial. However, ozonation converts refractory organic matter into biodegradable organic matter (Ribas et al., 1997). This effect can neutralize the BDOC improvement produced by a preliminary sand filtration stage, creating a need for further biological filtration downstream (Van der Kooij, Hijnen & Kruithof, 1989; Volk et al., 1993).

Filtration through granular activated carbon (GAC) in the adsorption mode can reduce organic carbon in a controlled way. However, the GAC will become saturated with organic compounds quite quickly, which means that frequent regeneration will be required, according to the type of water. The presence of an active microbial biomass can slow the saturation of the GAC due to breakdown of the adsorbed organic materials. The choice of the type and brand of carbon must be based on a preliminary pilot study over several months on-site, to observe the adsorption decline pattern and the progressive emergence of the degrading biomass. The process cannot be simulated in the laboratory or by short-term experiments (Bablon, Ventresque & Benaïm, 1988). Low temperatures do not favour the generation of active biomass.

Recent studies have shown that the combined use of powdered carbon and ultrafiltration can eliminate not only the finest particles, bacteria and protozoa, but

also a large proportion of the TOC (Clark, Baudin & Anselme, 1996; Campos et al., 1999b).

Membrane retention of TOC will also remove a large proportion of mineral salts, without the use of activated carbon. Some form of remineralisation or treatment with corrosion inhibitor may then be required to prevent corrosion in the distribution system (Agbekodo, Legube & Cote, 1996; Laurent et al., 1999b).

Box 2.2. Water quality deterioration associated with a change in disinfection practice.

In the late summer of 2000, the water supply in the City of Coquitlam, Canada, had high total coliform counts, though indicators of faecal contamination had been negative.

The supply came from surface water in mountainous areas and was treated by newly introduced coarse screening and chlorination. In May 2000, an ozone treatment facility had been commissioned to treat water before chlorination and distribution. With the onset of ozonation there was an increase in assimilable organic carbon (AOC) and biodegradable dissolved organic carbon (BDOC) in water entering the distribution system.

The supply area varied in elevation from 64 to 390 m. Water for consumers at the highest point travelled through a series of service reservoirs until it reached the highest storage tank (summit), some 420 m above sea level. Chlorine levels declined through each reservoir, from 0.7 to 0.8 mg/l at the treatment works to about 0.4 mg/l in the lowest reservoir, and less than 0.05 mg/l at the summit. In the autumn of 2000, there had been several high total coliform counts in the area of distribution served by the two highest reservoirs. High heterotrophic plate counts were common in these parts of the system as well. The positive total coliform counts led to the issue of advice to boil drinking-water.

The cause of this coliform bloom was a combination of the increase in AOC and BDOC (produced by ozonation), combined with low chlorine levels in the distal part of the system. The two highest reservoirs had particularly large storage in relation to demand, with the summit reservoir holding almost seven days supply and the next highest four days. The problem was resolved by regular cleaning of reservoirs and flushing of mains, reducing residence times in the service reservoirs and installing additional chlorination equipment.

Although coliform blooms as reported here do not represent a risk to human health, advice to boil water was issued. Boil water advice may carry its own risks to public health from scalds, anxiety and increased costs.
Source: Gehlen et al. (2002)

2.4.3 Decentralized treatment

The development of larger urban areas and the difficulties of building large treatment works in cities leads to more extensive distribution systems. In these situations, it may be necessary to adjust water quality in the distribution network. For example, recent investigations have demonstrated the feasibility of using membrane technology for treating water in an urban distribution network (Levi et al., 1997).

Chlorination booster stations (also known as relay stations) are commonly used in networks to manage chlorine residuals. They avoid the need for excessively high doses of disinfectant at treatment works to ensure that residuals reach the extremities of a network. The locations of rechlorination stations can be optimized using hydraulic models to simulate the residence times and the disinfectant residual (see Chapter 3). The effects of water temperature should be included, to avoid overdosing and prevent excessive formation of by-products in certain seasons of the year. However, booster chlorination should not be regarded as a means of preventing contamination of the distribution system (Snead et al., 1980) or as an alternative to monitoring system performance.

2.5 SUMMARY

It is not meaningful or practicable to strive for a sterile drinking-water network devoid of all microorganisms. Although the presence of nonpathogenic organisms does not directly threaten public health, their proliferation may make water unpalatable and encourage recipients to consume an alternative, and possibly less safe, source of water. In addition, the presence of large numbers of bacteria in the conveyed water, or in biofilms and deposits, may compromise the identification of serious contamination from outside the pipework and reservoir structures.

The optimization of treatment to minimize the amount of microbial nutrients and deposit-forming components entering the network will help to prevent water discolouration, tastes, odours and the proliferation of microorganisms. Optimization of treatment should be the first stage in any plan to ensure the microbial quality of water during distribution. Once deposits and biofilms have formed in a system, they are difficult to remove.

Water treatment can be optimized to prevent microbial growth, corrosion of pipe materials and the formation of deposits by adopting the following practices:
- continuous and reliable elimination of particles and the production of water of low turbidity;
- precipitation and removal of dissolved (and particulate) iron and manganese;

- minimization of the carry-over of residual coagulant (either dissolved, colloidal or particulate) that may precipitate in reservoirs and pipework;
- reduction (as far as possible) of dissolved organic matter, especially AOC and BDOC, that provides carbon-based nutrients for microorganisms;
- maintenance of the corrosion potential within limits that avoid damage to the structural materials and consumption of disinfectant;
- production of water with a low disinfectant demand, enabling disinfectant residuals to be maintained throughout the network without giving rise to unwanted by-products;
- adaptation of the disinfectant residual and its control to local conditions and climatic variation.

Predicting the effects of treatment options to minimize biological degradation in the network is not readily achieved using laboratory simulations. However, measurements of operationally-defined parameters such as AOC or BDOC can be used to evaluate the relative effects of different treatment processes designed to remove organic nutrients.

2.6 REFERENCES

AWWARF (1998). *Development of disinfection guidelines for the installation and replacement of water mains*. American Water Works Association Research Foundation, Denver, USA, 55.

Agbekodo KM, Legube B, Cote P (1996). Nanofiltration yields high quality drinking water in Paris suburb. *Journal of the American Water Works Association*, 88:67–74

AGHTM Biofilm Group (1999) Standard method to evaluate aquatic biofilms. In: Keevil CW, ed. *Biofilm in the aquatic environment*. Royal Society of Chemistry, 210–219.

APHA–AWWA–WEF (1995) *Standard methods for the examination of water and wastewater*, 19th ed. American Public Health Association, Washington DC.

Armon R et al. (1997). Survival of *Legionella pneumophila* and *Salmonella typhimurium* in biofilm systems. *Water Science and Technology*, 35(11–12):293–300.

Bablon GP, Ventresque C, Benaïm R (1988). Developing a sand GAC filter to achieve high rate biological filtration. *Journal of the American Water Works Association*, 80:47–53.

Block JC et al. (1992). Indigenous bacterial inocula for measuring the biodegradable dissolved organic carbon in waters. *Water Research*, 26:481–486

Brett RW, Ridgway JW (1981). Experiences with chlorine dioxide in Southern Water Authority and Water Research Centre. *Journal of the Institution of Water Engineers and Scientists,* 35:135–142.

Camper AK et al. (2000). *Investigation of the biological stability of water in treatment plants and distribution systems*. American Water Works Association Research Foundation.

Campos C et al. (1999a). Application of powdered activated carbon to floc blanket reactors for the removal of trace organic compounds. *Proceedings of the American Water Works Association Annual Conference*, Chicago.

Campos C et al. (1999b). Effect of process design and operating procedures on the performance of PAC/UF systems for the removal of organic compounds. *Proceedings of the AWWA Membrane Technology Conference*, Long Beach, California.

Carlson M (1991). Disinfection by-products precursor removal. *Proceedings of the 1991 AWWA Annual Conference — Water research for the new decade*, 151–191.

Christy P and Robinson B (1984). Disinfection of water for control of amoebae. *Water* 11, 21–24.

Clark MM, Baudin I, Anselme C (1996). Membrane powdered activated carbon reactors. In: *Water treatment membrane processes*, McGraw-Hill, 15-1–15-22.

Croue JP et al. (1997). Impact des traitements de potabilisation sur le CODB et la distribution des substances humiques et non humiques de la matière organique naturelle. *Revue des Sciences de l'Eau*, 1:63–81.

Crozes G, White P, Marshal M (1995). Enhanced coagulation: its effect on NOM removal and chemical costs. *Journal of the American Water Works Association*, 87(1):78–89

Dukan S et al. (1996) Dynamic modelling of bacterial growth in drinking water networks. *Water Research*, 30(9):1991–2002.

Gehlen IJ et al. (2002). Addressing heterotrophic bacteria regrowth in the city of Coquitlam water distribution system. In: *Proceedings of the NSF International / World Health Organization Symposium on HPC Bacteria in Drinking water*, April 22–24, 2002, Geneva, Switzerland, 479–500.

Heraud J et al. (1997). Optimised modelling of chlorine residual in a drinking water distribution system with a combination of on-line sensors. *Journal of Water Supply: Research and Technology — Aqua*, 46:1–12.

Jacangelo JG et al. (1995). Selected processes for removing NOM : an overview. *Journal of the American Water Works Association*, 87:64–77.

Joret JC, Levi Y (1986). Méthode rapide d'évaluation du carbone éliminable des eaux par voie biologique. *Tribune du Cebedeau*, 510(39):3–9.

Joret JC, Levi Y, Volk C (1991). Biodegradable dissolved organic carbon (BDOC) content of drinking water and potential regrowth of bacteria. *Water Science and Technology*, 24(2):95–101.

Kaplan LA, Reasoner DJ, Rice EW (1994). A survey of BOM in US drinking waters. *Journal of the American Water Works Association*, 86:121–132.

Kool JL, Carpenter JC, Fields BS (1999). Effect of monochloramine disinfection of municipal drinking water on risk of nosocomial Legionnaires' disease. *Lancet*, 353:272–77.

Laurent P et al. (1999a) Biodegradable organic matter removal in biological filters: evaluation of the chabrol model. *Water Research*, 33(6):1387–1398.

Laurent P et al. (1999b). Microbiological quality before and after nanofiltration. *Journal of the American Water Works Association*, 91(10)62–72.

LeChevallier MW et al. (1993) Examining the relationship between iron corrosion and the disinfection of biofilm bacteria. *Journal of the American Water Works Association*, 85:111–123.

LeChevallier MW et al. (2003). *Drinking-water treatment and water quality: the impact of treatment on microbial water quality and occurrence of pathogens and indicators in surface water*. WHO, Geneva and IWA Publishing, London.

Legrand L, Leroy P (1995). *Prevention de la corrosion et de l'entartrage dans les réseaux de distribution d'eau*. Ed. CIFEC, France.

Levi Y et al. (1997). Demonstration unit of satellite treatment in distribution system using ultrafiltration and nanofiltration. *Proceedings of the American Water Works Association Membrane Technology Conference*, New Orleans, 1997, 581–595.

Lu W, Kiene L, Levi Y (1998). Chlorine demand of biofilms in water distribution systems. *Water Research*, 33(3):827–835.

Norton CD (1995). *Implementation of chloramination and corrosion control to limit microbial activity in the distribution system.* American Water Works Service Company Report.

Palin AT (1975). Water disinfection — chemical aspects and analytical control. In: Johnson JD, ed. *Disinfection — water and wastewater.* Ann Arbor Science, Ann Arbor, Michigan, USA.

Piriou P et al. (1997). Prevention of bacterial growth in drinking water distribution systems. *Water Science and Technology*, 35(11–12):283–287.

Ribas F et al. (1997). Efficiency of various water treatment processes in the removal of biodegradable and refractory organic matter. *Water Research*, 31(3):639-649.

Servais P, Anzil A, Ventresque C (1989). Simple method for the determination of biodegradable dissolved organic carbon in water. *Applied and Environmental Microbiology*, 55(10):2732–2734.

Snead MC et al. (1980) The effectiveness of chlorine residuals in inactivation of bacteria and viruses introduced by post-treatment contamination. *Water Research*, 14:403–408.

Snoeyink VL, Jenkins D (1980). *Water chemistry.* Wiley, New York.

UKWIR (1998a). *Impact of service pipes on the bacteriological quality of water supplies.* Report 98/DW/02/15, UK Water Industry Research Limited, London.

UKWIR (1998b). *Chlorine and monochloramine demand of materials.* Report 98/DW/03/9, UK Water Industry Research Limited, London.

UKWIR (2000a). *Toolboxes for maintaining and improving drinking water quality.* Report 00/DW/03/11, Section 4, Toolbox for Iron Discolouration and Particulates, UK Water Industry Research Limited, London.

UKWIR (2000b). *Toolboxes for maintaining and improving drinking water quality.* Report 00/DW/03/11, Section 8, Toolbox for pH, UK Water Industry Research Limited, London.

UKWIR (2000c). *Toolboxes for maintaining and improving drinking water quality.* Report 00/DW/03/11, Section 3, Toolbox for Trihalomethane Formation, UK Water Industry Research Limited, London.

UWRAA (Urban Water Research Association of Australia) (1990) Report No 15 Chloramination of Water Supplies. Water Services Association of Australia.

Van den Hoven T et al. (1996). Simplified methods to monitor water quality changes in distribution systems. *Water Supply*, 14(3/4):515–533.

Van der Kooij D, Hijnen WAM, Kruithof JC (1989). The effects of ozonation, biological filtration and distribution on the concentration of easily assimilable organic carbon (AOC) in drinking water. *Ozone: Science and Engineering*, 11:297–311.

Van der Kooij D (1992). Assimilable organic carbon as an indicator of bacterial regrowth. *Journal of the American Water Works Association*, 84:57–65.

Van der Kooij D, Veenendaal HR (1992). Assessment of the biofilm formation characteristics of drinking water. *Proceedings of the American Water Works Association Water Quality Technology Conference*, Toronto.

Volk C et al. (1993). Effects of ozone on the production of biodegradable dissolved organic carbon during water treatment. *Ozone: Science and Engineering*, 15:389–404.

Volk C et al. (1994). Comparison of two techniques for measuring biodegradable dissolved organic carbon in water. *Environmental Technology*, 15:546–556.

Volk C, LeChevallier MW (1999). Impacts of the reduction of nutrient levels on bacterial water quality in distribution systems. *Applied and Environmental Microbiology*, 65(11):4957–4966.

WHO (2004) *Guidelines for drinking-water quality*, 3rd ed. World Health Organization, Geneva.

Williams AM, Andrews S, Wakeford P (2001). The control of nitrite formation in London's distribution system. *Journal of the Chartered Institution of Water and Environmental Management*, 15:162–166.

WRc (1981). *A guide to solving water quality problems in distribution systems.* Technical Report TR167, Water Research Centre, Swindon, UK, 31.

WRc (1990). *Aesthetic water quality problems in distribution systems. A source document for the water mains rehabilitation manual*. Water Research Centre, Swindon, UK, 115.

WRc (1992). *Coagulant residuals in soft water.* Report no. 1330 UM, Water Research Centre, Swindon, UK.