Chapter 5

TREATMENT EFFICIENCY

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5.1 Introduction

The primary purpose of water treatment is to provide drinking water to consumers that is free of waterborne pathogens. Because no single treatment process can be expected to remove all of the different types of pathogens that can be found in water (under all conditions), multiple barriers are desirable. Multiple barriers will also ensure additional safety in the case that a single treatment step is not working optimally. The number of treatment processes (technical barriers) required is influenced by the quality of the source water (see Chapter 4). Groundwaters that are protected from surface influence are usually of relatively good quality and so traditionally few, if any, treatment processes are required. Lowland surface water sources are usually of much poorer quality and more treatment processes are needed to provide an acceptable level of safety.

A number of treatment processes are also designed to modify the chemical and physical properties of the water (rather than pathogen elimination). State of the art treatment includes techniques to reduce AOC and reducing matter, so that on the one hand the regrowth of the pathogens in the distribution system is low and on the other hand the disinfection is more effective. This chapter, however, does not detail such processes but concentrates solely on the reduction of faecal-oral infection risk.

A wide spectrum of pathogenic agents can be found in water and monitoring for their presence on a routine basis is impracticable. Traditionally (as outlined in Chapter 1) microbial safety of drinking water has been confirmed

by monitoring for the absence of microorganisms of faecal origin. Bacteria such as *E. coli*, faecal streptococci and *Clostridia* have been used for this purpose, because they are consistently present in high numbers in the faeces of warm-blooded animals and are relatively easy to detect in water. These bacteria and groups of bacteria are microbial indices of faecal pollution and form the basis of guidelines and national standards.

It has been recognised that the microbial indicator parameters do not necessarily behave in the same way as certain pathogens in water treatment processes. The ability of treatment processes to remove specific pathogens has been directly measured, with such studies typically conducted at bench or pilot scale some of them using water spiked with pathogens (Sommer and Cabaj, 1993; Jacangelo *et al*., 1995; Bellamy *et al*., 1985; Hunt and Mariñas, 1999). The potential removal determined in such pilot studies will, however, not necessarily be achieved in full-scale treatment. Therefore, there is a need for alternative parameters that correlate more closely with the behaviour of specific pathogens both to assess the disinfection potential of full-scale treatment and to measure process performance during treatment.

Safe drinking water is the result of careful evaluation of source water quality and variation (as outlined in Chapter 4) and adequate, reliable treatment processes combined with performance monitoring to assure that treatment is within operating parameters. The focus for the control of process operation should be put on simple measurements, which can be done on-line. If the input to the system and its normal performance is known, the on-line measurement will be a perfect indication of disturbances and changes in the water quality. This shifts the emphasis of quality control of drinking water from end product testing (*i.e.* testing for failure) to the testing and control of treatment processes (*i.e.* preventing failure). Current treatment processes and appropriate indicators of performance are discussed below.

A verification of the quality at the end of the treatment chain is necessary. For this purpose non microbial parameters like flow, colour and disinfectant residual (where appropriate) are suitable (see also Chapter 2, Table 2.4). Microbial parameters for the validation of the treatment process include *E. coli*, total and thermotolerant coliforms, heterotrophic bacteria and aerobic spore-forming bacteria. However, it should be stressed that this verification should not be mistaken as a determination of the safety of the drinking water.

5.2 Microbial treatment efficiency

A review of the available data on treatment efficiencies has been published by LeChevallier and Au (2002). Disinfection can be achieved in two ways:

- The physical removal of the pathogens.
- The inactivation (death) of the pathogen.

Apart from careful characterisation of the disinfection potential of a given treatment process (which in many cases involves experimental determinations) it is also important to identify simple measurements that give information rapidly on whether the treatment process is working properly. For the latter, physical and chemical measurements (preferably on-line measurements) are often better than microbial determinations.

A review of potential inactivation rates for different disinfection treatments has been published by Sobsey (1989). More recently, the United States Environmental Protection Agency (USEPA, 1999) has compiled data on achievable disinfection efficiencies for various processes and combinations of treatment steps. Although the actual inactivation will be influenced by many factors (including the ability of many of the microbial parameters to remain viable while becoming non-culturable), the following subsections list typical ranges reported for each treatment process. Although retention of water in reservoirs and impoundments can bring about significant improvements in quality as a result of inactivation, sedimentation and predation this process is not discussed here. For more details the reader is referred to the review by LeChevallier and Au (2002). For more precise evaluation of the reduction of individual pathogens by a treatment process, specific experimental determination is necessary.

5.2.1 Coagulation and sedimentation

The most common coagulants in use throughout the world are aluminium sulphate, ferric sulphate, ferric chloride and poly-aluminium chloride. These coagulants are mixed into the water where they produce hydroxide precipitates that are ‘fluffy’ and enmesh particles and microbes along with some of the dissolved organic carbon. In some circumstances, flocs generated by aluminium and ferric salts can be strengthened by the addition of coagulant aids such as long chain organic polymers. The flocs formed by this process must be removed. This can be achieved by sedimentation or, if the flocs are very light,

fine air bubbles may be used to carry them to the surface (air flotation) where they are skimmed off. They can also be removed by direct filtration.

Various forms of coagulation and sedimentation are used in water treatment and there are differences in general practices between countries, which makes the comparison of data difficult. However, published data indicate that this process may remove between 40% and 99% of bacteria, which translates into 0.2 and 2 logs of removal. Removal of viruses is rather poor, below 1 log, whereas for parasites such as *Cryptosporidium* removal of up to 2 logs has been reported.

The retention of the formed flocs is very important because of the accumulation of pathogens, since even single flocs may contain sufficient numbers of pathogens to be of hygienic importance (Gale *et al*., 1997). Continuous measurements of turbidity or particle counts are useful for monitoring the efficiency of this process.

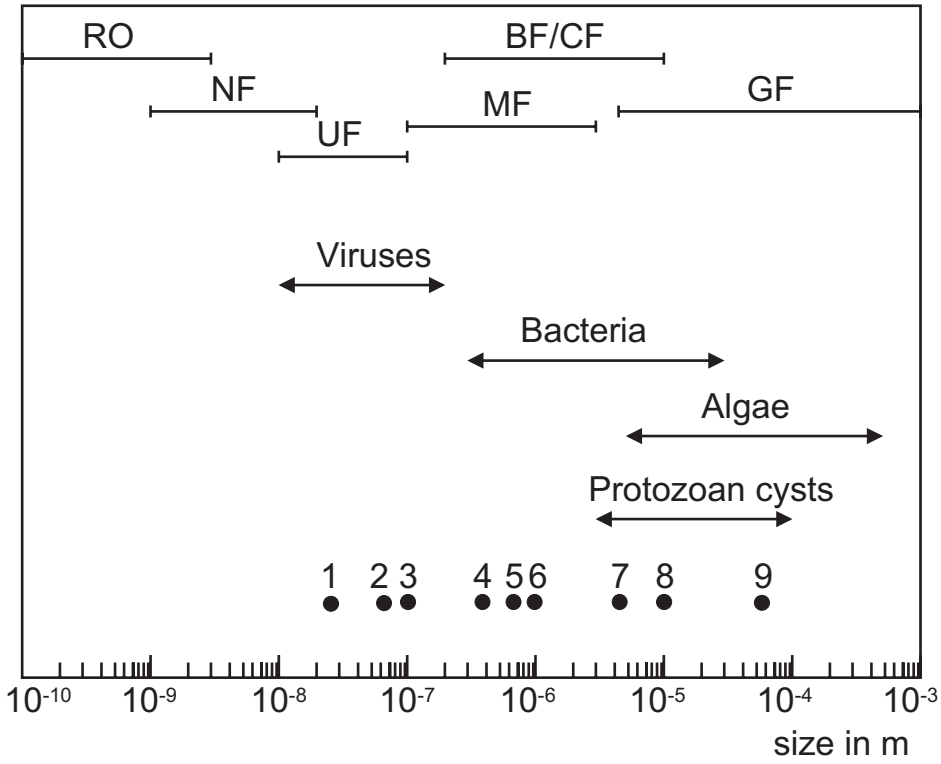
5.2.2 Filtration

Various filtration processes are used in drinking water treatment. Used with proper design and operation, filtration can act as a consistent and effective barrier against microbial pathogens. Filtration processes that are used in potable water treatment and the pore size of filter medium are shown in Figure 5.1, along with the sizes of selected microbial particles. This provides an insight into the removal mechanisms and likely efficiencies of the different filtration processes.

Filtration is a physical removal of organisms together with other particulate matter. On-line measurements of turbidity or particle counts, as well as determination of particle size distribution are excellent control parameters for this process. If parallel filtration units are operated, it is essential that each unit is measured separately in order to ensure the recognition of poor performance in an individual filter unit.

Figure 5.1. Filter medium pore sizes and the size of microbial particles (with selected microorganisms marked with numbers)

(Adapted from LeChevallier and Au, 2002)



Key:
RO: reverse osmosis. NF: nanofiltration. UF: ultrafiltration. MF: microfiltration. BF/CF: bag and cartridge filters. GF: granular filtration including slow sand filtration (slow sand filters have lower pore sizes than rapid-rate filters)
1. MS2 bacteriophage. 2. Rotavirus. 3. PRDI bacteriophage. 4. *Mycobacterium avium* complex (represents smallest size). 5. *Yersinia* spp. 6. Coliform bacteria. 7. *Cryptosporidium* oocysts. 8. *Giardia* cysts. 9. *Balanthidium coli* cysts.

5.2.2.1 Rapid filtration

Rapid filters are deep beds (0.6-1.0 metres) of sand, anthracite and sand or granular activated carbon. The particle size of the medium is usually about 1 mm. They are operated at flow velocities of about 5-15 metres per hour. Rapid filters retain most of the flocs and other particles that escape chemical coagulation and sedimentation. The size of particles that can be removed in deep-bed filtration can be much smaller than the pore size of the filter (Hall,

1998). This is due to electrostatic adhesion causing adsorption of particles that are in close proximity to the filter medium. Rapid filters are stopped and backwashed according to a time cycle (usually 24 hours), when flow becomes excessively restricted due to clogging or when the turbidity or particle counts of the filtrate become unacceptably high.

The microbial removal efficiency of rapid filters can be influenced by a number of factors. Correct operation and maintenance of rapid filters is essential otherwise performance may be lost. In poorly maintained filters, cracks have been observed particularly near the walls, which allow unfiltered material to pass through, decreasing the bacteriological quality of the filtrate. Changes in the flow rate can dislodge deposits containing microorganisms causing them to pass into the filtrate. When a filter is put back into service after backwashing, the initial filtrate is of poor quality in terms of turbidity and bacterial numbers. This is due to displacement of residual backwash water, and the lower efficiency of the clean filter media, compared with a partly used (ripened) filter (Amirtharajah and Wetstein, 1980). For this reason the initial filtrate may be run to waste or returned to the start of the treatment processes for a period of up to 30 minutes. Alternatively a 'slow start' procedure may be used in which the flow rate through the filter is restricted until the filtrate becomes of acceptable quality. Additionally, backwash water should not be recycled within the treatment plant.

Published data indicate that coagulation combined with rapid filtration may remove between 2 and 3 logs of bacteria, while reported removal of viruses range from 1 to 3 logs and for parasites such as *Cryptosporidium* 2 to 3 logs. Continuous measurements of turbidity and/or particle counts are important for monitoring.

5.2.2.2 Slow sand filtration

Slow sand filtration is a biological treatment process, which has to be used without coagulation pre-treatment. Other pre-treatment, particularly rapid filtration, may be used to remove high particle loads. Typically, a slow sand filter has a depth of about 0.7 metres and is operated at flow rates of 0.1 to 0.3 metres/hour compared to 5-15 metres/hour in rapid filters. The sand is mixed in size ranging from 0.15 to 0.35 mm. The pores are still quite large at about 60 μm. Although there is some filtration in depth, as in rapid sand filtration, the vital process is the formation of a biologically active layer (the Schmutzdecke) in the top 20 mm. Optimum treatment performance is dependent on a well-established Schmutzdecke. This provides an effective surface filtration of very small particles, including bacteria, parasites and viruses.

Any particles that pass through the Schmutzdecke may be retained in the remaining depth of the sand by the same mechanisms as exist in rapid filtration. The growth of the Schmutzdecke and its retention of particles cause a loss of permeability in the top layer of sand so that after some weeks of operation, flow rates decline. When this occurs, the filter is taken out of service and the top 20-30 mm removed by skimming. Slow sand filters are known for their high efficiency in removing bacteria and parasites, but small channels can occur in the filter if not properly operated and maintained which influence performance. In well-maintained systems with slow sand filtration it is possible to achieve a performance similar to a combination of coagulation and filtration. Continuous measurements of turbidity and/or particle counts are important for monitoring.

5.2.2.3 Activated carbon filtration

Activated carbon filters are predominantly used to remove organic compounds. However, they may also affect counts of microbial organisms including reduction of viruses and parasites. Due to growth in the filters, increased heterotrophic plate counts and total coliform counts can sometimes be observed.

5.2.2.4 Membrane filtration

In membrane filtration water is passed through a thin film, which retains contaminants according to their size. Membrane filtration has been playing an increasing role in drinking water treatment, including pathogen removal. The most commonly used membrane processes in drinking water treatment for microbial removal are microfiltration (MF) and ultrafiltration (UF) (see Figure 5.1). Detailed description of the fundamentals, design and operation of these processes are available in the literature (AWWARF, 1996; Taylor and Wiesner, 1999). Other membrane processes such as reverse osmosis (RO) and nanofiltration (NF), which are used primarily for other purposes, also remove pathogens.

Membrane filtration removes microbial pathogens primarily by size exclusion; microbes with sizes greater than the membrane pore size are removed. Chemical coagulation prior to the membrane is not a requirement for microbe removal. However, some degree of pre-treatment must be employed to reduce membrane fouling. Fouling arises from accumulation of chemicals, particles and the growth of organisms on membrane surfaces, resulting in reduced membrane productivity. Once fouling accumulates to such a level that the productivity of the system is unacceptable, the membranes must be

chemically cleaned to restore productivity. Advanced pre-treatment systems such as conventional coagulation-sedimentation-filtration or other membrane processes may also be considered, depending on the quality of the source water.

Published data indicate that membrane filtration may remove up to 6 logs of bacteria, viruses or parasites. Process performance is generally monitored by measurement of physical parameters such as pressure drops across the membrane.

5.2.3 Chemical inactivation

Chemical disinfection to inactivate pathogens is an important treatment barrier. Chemicals used include chlorine, chloramine, chlorine dioxide and ozone. Treatment effectiveness is a function of dose, contact time, temperature and sometimes pH. Chemical disinfection can be placed at different positions in the treatment train and more than one disinfectant can be used, however it is important to note that organisms entrapped in particles may be shielded from the action of the chemicals. Primary disinfection is the process by which microorganisms are inactivated during the treatment process, while a secondary disinfectant can be added prior to distribution to maintain the water quality within the distribution system. Secondary disinfection provides a final barrier against bacterial contamination and regrowth within the distribution system. The practice of residual disinfection is, however, controversial (IWSA, 1998). It has been suggested that if biological stability is achieved and the system is well maintained then the disinfectant is unnecessary and may mask ingress into the distribution system by killing the bacterial indicators (but not the more robust pathogen microorganisms).

The concept of disinfectant concentration and contact time is integral to the understanding of disinfection kinetics and the practical application of the CT concept (which is defined as the product of the residual disinfectant concentration [C in mg/l] and the contact time [T in minutes], that residual disinfectant is in contact with water – USEPA, 1999) is important. Allowance must be made for the decline in concentration over time and in measuring time it is important to take account of the hydraulic behaviour of the treatment plant (in particular any short-circuiting). Temperature, over the range appropriate for drinking water, affects the rate of disinfection reactions according to the Arrhenius Law, although some deviations have been noted for certain disinfectants at low temperatures. The pH of the disinfectant solution also affects reaction kinetics. Table 5.1 outlines CT values for inactivation of viruses.

Table 5.1. CT values for virus inactivation

(USEPA, 1999)

		Inactivation		
Disinfectant	Units	2-log	3-log	4-log
Chlorine[1]	mg min/l	3	4	6
Chloramine[2]	mg min/l	643	1 067	1 491
Chlorine dioxide[3]	mg min/l	4.2	12.8	25.1
Ozone	mg min/l	0.5	0.8	1.0
UV	mW s/cm^2	21	36	not available

1. Values based on a temperature of 10 °C, pH range 6 to 9, and a free chlorine residual of 0.2 to 0.5 mg/l.
2. Values based on a temperature of 10 °C and a pH of 8.
3. Values based on a temperature of 10 °C and a pH range of 6 to 9.

5.2.3.1 Chlorination

Chlorination can take a number of forms including the use of chlorine, chloramines and chlorine dioxide. Each chemical has different disinfecting properties. Monochloramine (formed by the combination of chlorine with nitrogenous compounds) has a lower disinfection activity than chlorine but is more stable. Chlorine dioxide may be chosen because of its greater effectiveness against parasites.

Nearly 100 years of drinking water chlorination has demonstrated its effectiveness in the inactivation of microbial pathogens and the benefits of chlorination out-weigh any disadvantages, such as production of trihalomethanes. Enteric viruses are generally more resistant to chlorine than enteric bacteria, and viruses associated with cellular debris or organic particles may require high levels of disinfection due to the protective nature of the particle surface. Chlorination is considered to be highly effective for virus inactivation if the water has a turbidity of ≤ 1.0 nephelometric turbidity units (NTU), a free chlorine residual of 1.0 or greater for at least 30 minutes, and a pH of < 8.0. Protozoan cysts such as those of *Cryptosporidium* and *Giardia lamblia*, however, are highly resistant to chlorine disinfection (USEPA, 1989). Other factors that influence microbial sensitivity to chlorine include surface attachment, encapsulation, aggregation and low-nutrient growth.

Chlorine is a strong disinfectant that is effective at inactivating bacteria and viruses and, under certain circumstances, Giardia. CT values for 2 log inactivation of vegetative bacteria may vary between 0.02 and 200 mg min/l

(Grohmann, A; 2002) This wide range depends on a number of factors particularly the presence of reducing matter. One purpose of water treatment, therefore, is to eliminate such matter from water prior to chlorination. Residual levels of reducing matter can be determined by electrochemical methods such as oxidation-reduction-potential (ORP) measurements. The use of high dosage of chlorine is, therefore, by itself not a guarantee of safe drinking water as the presence of reducing matter may result in high concentrations of disinfection by-products (DBP), such as trihalomethanes (THM), which are toxic.

No significant reduction of Cryptosporidium is achieved with conventional CT values. Since pH, temperature and chemical composition will influence the disinfection potential they need to be monitored together with the CT measurements.

Because of the weak disinfecting power of monochloramine, it is not recommended as a primary disinfectant and it is ineffective in the inactivation of *Cryptosporidium*. Most systems using monochloramine apply a short period of free chlorine prior to ammonia addition or use an alternative (*e.g.* ozone, chlorine dioxide) primary disinfectant. Chloramines have CT values of more than 80 mg min/l for a 2 log inactivation of bacteria; values for the same inactivation of viruses are above 600 mg min/l and, therefore, they are only suitable for the inactivation of bacteria.

Chlorine dioxide is a strong oxidant as well as a powerful disinfectant and, therefore, can be used for the control of iron, manganese and taste and odour causing compounds as well as a primary disinfectant. It has also been used as a secondary disinfectant in many European countries. However, chlorine dioxide forms inorganic by-products (chlorite and chlorate ions) upon reaction with water constituents, and a water supplier may need to provide additional treatment depending on the level of these inorganic by-products and specific regulatory requirements. Chlorine dioxide is roughly comparable to free chlorine for inactivation of bacteria and viruses at neutral pH (), but it is more effective than free chlorine at an alkaline pH of 8.5 (Hoff and Geldreich, 1981). CT values for chlorine dioxide resulting in a 2 log inactivation of vegetative bacteria are less than 1 mg min/l. While values around 4 mg min/l have been reported for viruses and those for *Giardia* inactivation are around 15 mg min/l. Temperature and chemical composition need to be monitored together with the CT measurements (or calculations) and chlorine residual.

Chlorination usually takes place at a central treatment point but, particularly in developing countries, there is growing interest in applying it at household level. Sachets or tablets of a chlorine compound (sometimes together with a coagulant to remove turbidity) are sometimes used. Decentralised

production of sodium hypochlorite is now possible from the electrolysis of a solution of common salt and this may provide a cost-effective source of chlorine solution. Combined coagulant-disinfectant tablets or powders or use of a solution of sodium hypochlorite are available for household water treatment (Sobsey, 2002).

5.2.3.2 Ozonation

Ozone has been used for more than a century for water treatment, mostly in Europe, but this usage is spreading to other areas. Despite this long use, the exact mechanism of how ozone inactivates microbes is not well understood, although it is known that ozone in aqueous solutions may react with microbes by direct reaction with the molecular ozone, or via reaction with the radical species formed on ozone decomposition.

Of the vegetative bacteria, *E. coli* is one of the most sensitive to ozone disinfection, while Gram-positive cocci (*Staphylococcus* and *Streptococcus*), the Gram-positive bacilli (*Bacillus*) and the mycobacteria are the most resistant. *Mycobacterium avium* can be effectively controlled by low doses of ozone, whereas the organism is highly resistant to free chlorine. It has been reported that heterotrophic plate count bacteria may be less susceptible to ozone inactivation than other indicator organisms. Viruses are generally more resistant to ozone than vegetative bacteria, although phages appear to be more sensitive than human viruses. Ozone is effective against *Giardia* and to a lesser extent *Cryptosporidium*. Because ozone does not produce a stable residual it is frequently followed by chlorination to produce a residual disinfectant for distribution. Due to the relatively fast decay of ozone even in pure water, hydraulics of the ozonation reactor are very important (see below).

Ozone will oxidise organic components present in the water, such as natural organic matter to produce smaller organic substances. Since these are usually more biodegradable, ozonation will increase bacterial growth after treatment. To prevent this, post ozonation removal of the oxidation products is necessary.

Ozone is a very powerful disinfectant for inactivation of vegetative bacteria. CT values below 0.5 mg min/l are reported for 2 log reduction of bacteria. CT values between 0.5 and 1 mg min/l are required for a 2 log inactivation of viruses. Inactivation of protozoa like *Giardia* is possible at temperatures above 15°C with CT values of 0.7 mg min/l for 2 log inactivation, while at 5°C the CT value increases to 1.3 mg min/l. For the same inactivation of *Cryptosporidium* the CT values required are about ten times higher. Content

of organic carbon will also influence the disinfection efficiency. Therefore the measurement of CT values needs to include control of temperature and quality of water entering the ozonation reactor.

Case study: Hydraulics of an ozonation reactor

In Switzerland, food-related laws and regulations require health risks assessment and the evaluation of critical treatment steps in drinking water production. In the city of Zurich, a considerable fraction of drinking water is produced from lake water following a multistage procedure. During a health risk assessment the hydraulics of the ozonation reactor were evaluated by addition of a concentrated sodium chloride solution to the water inlet of the reactor for a period of two hours. Five sampling points along the water flow allowed the spread of the addition through the reactor to be followed (Kaiser *et al.*, 2000). Modelling of the experimental data showed that the reactor was best described by a series of four mixed reactors followed by a plug-flow reactor with considerable back flow. The model was confirmed by the comparison of modelled and measured ozone profiles and atrazine concentrations.

Modelled inactivation of microorganisms showed a remarkable difference between a single plug-flow model and the model derived from the experimental measurements. According to the model, the ozonation should reduce vegetative bacteria and viruses by more than 6 logs, spores of *Bacillus subtilis* will be inactivated by 1.5 logs, whereas the inactivation of *Cryptosporidium* is less than 1 log.

5.2.3.3 UV disinfection

UV action results from absorption by nucleic acids (DNA and RNA), leading to the dimerisation of pyrimidine bases, and all organisms are susceptible to UV light. Exposure to UV results in reduced viability of the treated cells. However, most bacteria have evolved different repair systems to cope efficiently with UV damage to their genetic material, for example, thymine dimers can be repaired both in the presence ('photoreactivation') or absence of light ('dark repair') (Jagger, 1967). Thus, UV doses in a certain range will only transiently reduce the ability of bacteria to form colonies without having a long-term effect on their survival (Mechsner *et al.*, 1991). Therefore, for the UV disinfection of drinking water it is essential to treat each volume part with a sufficient light dose to kill the bacteria. Usually a dose of 400 J/m^2 (40 mW s/cm^2) is accepted as being sufficient for efficient treatment.

Three types of light source are used for UV disinfection, namely:

- Low-pressure mercury lamp.
- Medium-pressure mercury lamp.
- Pulsed lasers.

The most popular so far is the low-pressure mercury lamp, which emits light at the wavelength of 254.7 nm, almost exclusively. Due to the rather low light intensity of such lamps, radiation times required for efficient disinfection are substantially higher than those for the second type, the medium pressure mercury lamp, which emits light of higher intensity and also of longer wavelength. It is sometimes claimed that the medium-pressure lamps have a better performance, because they may act in a dual way, damaging both DNA and proteins, some of which might be involved in the DNA-repair process. On the other hand, due to the much higher light intensity of medium-pressure lamps, the required contact time is much shorter with a concomitant risk of volume parts not being treated sufficiently. Recently, the use of pulsed UV lasers has been suggested. It is claimed that the same extent of cell inactivation can be achieved with this light source at less than one tenth of the dose of low-pressure mercury lamps. Rubin *et al.* (1982) showed a dependence of photoinactivation of yeast cells on the UV light intensity at the same dose. A similar dependence was observed for the photoprotection. Therefore, at high light intensity more dead cells were found at lower doses.

Another factor interfering with this type of disinfection is the UV transmission of the water. For treatment process evaluation the minimal UV dose for water with different UV transmission characteristics must be known. Biodosimetric determination of the UV dose under production conditions has been proposed as the best method for determining efficiency (Sommer and Cabaj, 1993). This procedure includes the addition of spores of *Bacillus subtilis* to the water before treatment; from the difference between the colony counts before and after treatment the UV dose in the reactor can be inferred from a dose-response curve determined in the lab. Similar dose-response curves can be determined for other organisms of interest (*e.g.* pathogens) and the reduction potential of the treatment system can be evaluated.

The transmission of the water should be monitored on-line with the help of an UV detector. The determination of colony forming units of coliform bacteria is not a satisfactory measure of UV inactivation because of the possibility of repair mechanisms coming into play (Mechsner *et a*l., 1991). If a microbial indicator parameter is required, the reduction of spores should be measured since they are easy to measure and at the same time quite resistant to UV light.

UV disinfection has been proven to be adequate for inactivating bacteria and viruses. UV doses of 400 J/m^2 will reduce vegetative bacteria by 4 to 8 logs. Virus inactivation is by 3 to 6 logs. Protozoa are more resistant to UV disinfection, but newer studies showed that in neonatal mouse infection studies with UV treated *Cryptosporidium oocysts* at a UV dose of 410 J/m^2 a 4 log reduction in infection occurred. Similar UV doses are required for a 4 log reduction in spores of *Bacillus subtilis*.

UV disinfection case study

In Austria, Germany and Switzerland certification requirements have been established for the UV disinfection of drinking water, which typically require biodosimetric determination of the disinfection efficiency under production conditions (Snozzi *et al*., 1999). Spores of *Bacillus subtilis* are used for this process since repair mechanisms are not important and can be neglected. The water entering the UV plant is inoculated with the spores and their concentration is determined before and after UV treatment. The UV dose can be calculated from the reduction of viable spores and a dose-response curve measured previously in the lab. Variation of the light intensity and the flow rate allows the definition of the range of flow rate with turbulent mixing within the reactor.

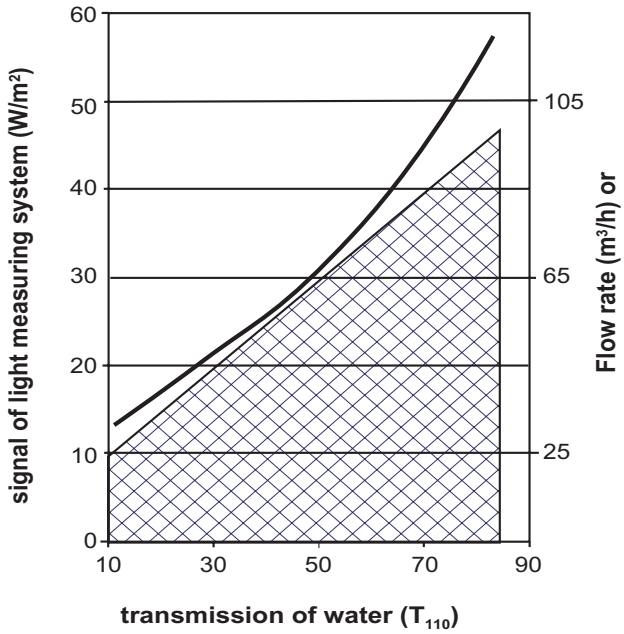
The result of the experimental determination of the disinfection efficiency can be represented in a graph (Figure 5.2) showing the maximal flow rate as a function of the UV transmissions of the water, which will ensure a minimal radiation dose of 400 J/m^2. If operation remains within these limits, the predetermined reduction of the number of viable pathogens can be ensured.

This experimental determination of the UV disinfection potential of a given reactor is very reproducible. Deviations between different determinations several months apart were found to be less than 2% (Snozzi, 2000).

Measurements of UV light intensity in the reactor serves as a control for process performance (it is important that the measuring point should be positioned such that changes in the UV transmission of the water will influence the reading of the light meter).

Figure 5.2. Measured UV light intensity as a function of UV transmission of the water

(Adapted from Snozzi, 2000)



The dashed area represents combinations of UV transmissions and flow rates, which result in reliable disinfection. The solid curve represents the signal of the light measuring device as a function of the UV transmission of the water. UV transmission (T_{100}) is given as percent transmission using a 100 mm light pass.

5.2.3.4 Solar water disinfection

Solar panels can be used to generate electricity to power the UV lamps mentioned in the previous section but in low-income countries the sunlight alone can be used to kill or inactivate many, if not all, of the pathogens found in water. Solar water disinfection is a method of treating relatively small amounts of water at the point of use. There are three ways in which solar radiation can be used to eliminate pathogens. The first is through heating, the second through the effect of the natural UV radiation and the third through a mixture of both thermal and UV effects. None of these methods is yet widely used but laboratory experiments and field programmes show that some systems have good potential to produce potable water. Solar disinfection is included in the

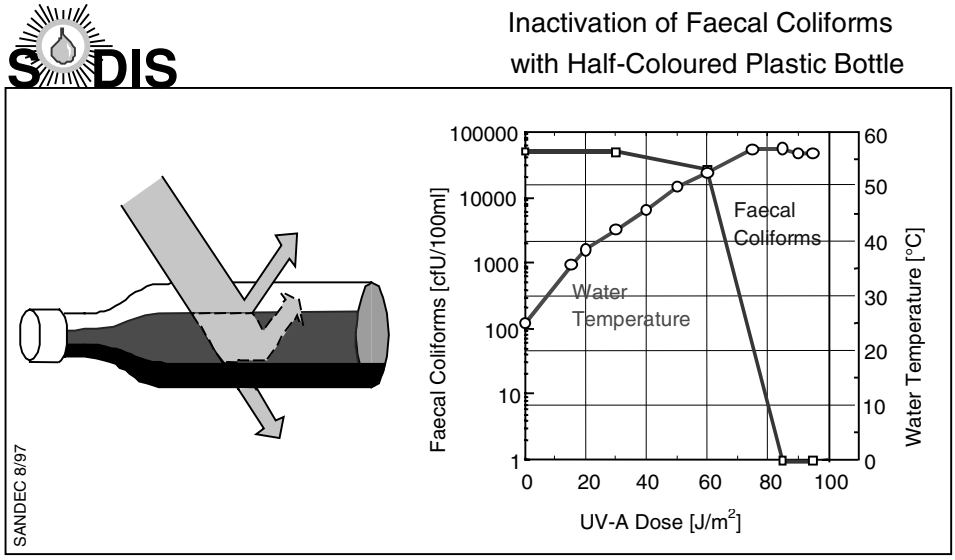
technologies reviewed by WHO for household water treatment and storage (Sobsey, 2002).

Thermal heating from the sun can be via solar cookers (which concentrate the rays of the sun with reflectors) or from simply exposing black-painted containers to the sun. In many systems temperatures can reliably reach over 55ºC killing many pathogens. With the cookers and some of the other systems the temperature of the water can easily exceed 65ºC, a pasteurisation temperature capable of inactivating nearly all enteric pathogens. Achievement of specific temperatures can be monitored using simple low-cost re-usable water pasteurisation indicators, based on the visible melting of wax in a clear plastic tube.

The use of heating and UV radiation to simultaneously disinfect water is used by a number of different solar treatment systems. The widest known is the SODIS system (Figure 5.3), which is suitable for low-income countries. The only equipment required is locally available bottles to contain the water (which needs to have a turbidity <30 NTU). This technique is now being field tested in various parts of the world and increasing amounts of data are becoming available on its effectiveness. Obviously for the UV to be effective the bottle material needs to be transparent to the useful wavelengths of the UV rays. The promoters of SODIS suggest the use of thin PET plastic bottles rather than PVC ones because the former material is more chemically stable. The half of the bottle furthest from the sun should be painted with black paint to improve the heat gain from the absorption of thermal radiation, and the bottle can be laid on a dark roof to further increase the potential temperature rise in the water. Shaking a partly filled bottle to aerate the water before filling it completely has been found to give a faster pathogen kill rate (Reed, 1997). The water requires several hours of exposure to strong sunlight to obtain the advantageous synergy between UV dosage and temperature rise (Wegelin *et al*., 1994, Sommer *et al*., 1997). In cloudy weather a much longer period (such as two days or more) is required because of the lower level of UV radiation and the reduced likelihood of the temperature of the water ever exceeding 50ºC.

Figure 5.3. Schematic representation of solar water disinfection and the influence of the water temperature on the UV-inactivation of bacterial cells

(Printed with permission of M. Wegelin)



5.3 Summary

This chapter reviews the different treatment barriers available to ensure the production of safe drinking water. The choice of which barriers to implement depends on a number of considerations including the source water quality. Non-microbial indicator parameters that can be measured on-line are most useful for assessing process performance and such monitoring is important within the total system approach to risk management. Treatment steps with relevant pathogen removal or inactivation are described together with possible indicators for the measurement of process performance.

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