MICRORISK

Microbiological risk assessment: a scientific basis for managing drinking water safety from source to tap

Efficacy of water treatment processes

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4.1 RATIONALE

Drinking water treatment of surface water was originally started to improve the aesthetic properties of drinking water. By the time of the Egyptians (15^{th}-13^{th} century BC) and Romans (300 BC-200 AC) settling was applied to reduce turbidity and in the 5^{th} century B.C. Hippocrates, the Father of Medicine, invented the "Hippocrates Sleeve", a cloth bag to strain rainwater. Supply of settled and filtered water in modern times started in 1804 (Scotland) and 1806 (Paris). Initially slow sand filters were used to provide a more aesthetic product and soon filtration was recognised to reduce outbreaks of typhoid and cholera. In the 1870's Robert Koch demonstrated that bacteria existing in water supplies could cause disease and he studied water filtration systems that were effective in removal of bacteria after the Hamburg cholera outbreak of 1892. In his biography of Koch's work, Brock [1988] states that "water filtration has probably saved more lives than immunization and chemotherapy combined". In 1906 the first ozonation plant for disinfection was started in France, and chlorination became common practice around the same time, although promoted by John Snow after his pioneering epidemiologic studies during London's cholera outbreaks of the 1850's. From 1920 the combination of sedimentation, filtration and chlorination virtually eliminated epidemics of classical waterborne diseases, such as cholera and typhoid, in areas so supplied [AWWA, 2006]**.**

However, outbreaks of waterborne disease due to poor drinking water quality still occur today, even when treatment is in place. Chapter 1 provides an overview of 84 outbreaks in Europe between 1990-2004. Further fault tree analysis indicated 18/24 groundwater and 17/22 surface water supply outbreaks were, at least partially, caused by insufficient treatment or treatment failure. Both chronic and temporary filtration failures were identified as major contributors in the fault tree analysis. When contamination of groundwater supplies goes unnoticed, the need for additional treatment is not recognised and treatment can be chronically insufficient. Within an additional 30 outbreaks, studied in Sweden, 57% were due to faecally contaminated raw water receiving insufficient treatment.

From 1974 to 2002, 26 out of 35 outbreaks in the USA and Canada, as reported by Hrudey and Hrudey [2004], were due to surface water treatment failure or inadequate treatment to deal with sudden peak increases of pathogen concentrations in source water. Some major outbreaks like that of cryptosporidiosis in Milwaukee where treatment efficiency was compromised, would have been prevented or the impact on human health reduced, by adequate treatment. Some outbreaks resulted from a combination of increased source contamination (mostly due to rainfall) and treatment failure. This illustrates that treatment needs to be able to deal with peak events in source water that are not prevented by source protection. The outbreaks due to peak events also showed that recognising events and taking corrective actions is essential to prevent outbreaks. Treatment is often the critical control step in the chain of barriers where sufficient (real-time) monitoring can recognise events and where corrective

action can take place. Rapid changes in water quality should always be considered as indicators of events.

Although counteracting peak events is necessary to prevent outbreaks, sufficient treatment during baseline (normal) conditions is also required to safeguard public health. In specific situations the sporadic cases (during baseline conditions) appear to represent a greater proportion of waterborne disease than outbreaks [Nichols, 2003]. This was also a conclusion reached for a water supply system in Gothenburg, based on failure reporting and quantitative risk assessment [Westrell *et al.*, 2003]. Pathogens are likely to be present in most surface waters (possibly below detection limits) and sufficient drinking water treatment is required to eliminate them. The required treatment depends on the level of source water contamination and relates to the health-based target for the population that drinking water is provided for. WHO [2004] states "Performance targets are most frequently applied to treatment performance – i.e., to determine the microbial reduction necessary to ensure water safety". The health based target suggested by WHO is 10^{-6} DALY per person per year. For a given source water quality, the treatment performance target (log-removal) can be calculated using the health target and translating this back, using dose-response data to maximum exposure levels of pathogens through drinking water, as is illustrated in Figure 4.1, taken from the WHO "Drinking-water Guidelines" [WHO, 2004].

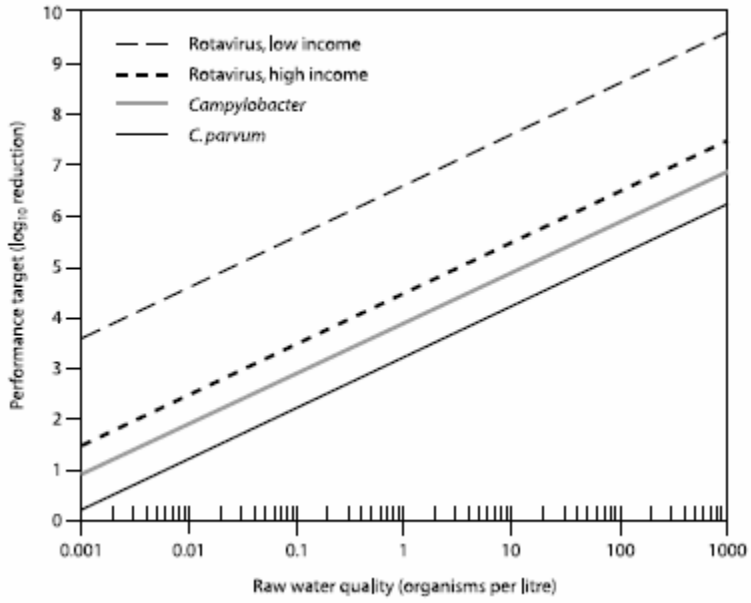


Figure 4.1 Performance targets for selected bacterial, viral and protozoan pathogens in relation to source water pathogen concentration (to achieve 10^{-6} DALYs per person per year) [WHO, 2004].

As an example a river water in The Netherlands used as source that contains up to 10 *Cryptosporidium* oocysts per litre and needs to be treated to not exceed the maximum tolerable oocysts concentration in drinking water of $6.3*10^{-4}$ oocysts per litre. Thus, such water requires treatment to provide 4.2 log reduction of *Cryptosporidium*.

Similarly required reduction can be calculated for the other chosen pathogens: *Giardia*, *Campylobacter*, *E. coli* O157, Enterovirus and *Norovirus* in the MicroRisk CTSs. The example uses point estimates but as discussed in Chapters 7 & 8 the assessments attempt to account for variation and uncertainty.

All current drinking water treatments processes show variations in treatment efficacy. Source water quality changes with time resulting in different performance of the treatment process. Temperatures influence the efficacy of disinfection processes if dosing regimes are not adapted accordingly. Varying amounts of suspended solids can impact on sedimentation and filtration as well as disinfection. Furthermore treatment efficacy varies due to production flows, filtration backwash cycles and chemical dosing control loops. These known regular variations should always be accounted for in design and operation. In addition to these regular variations, hazardous events can occur. Pathogen concentration peaks in source water due to heavy rainfall have been recognised as causes of outbreaks [Hrudey and Hrudey, 2004], in addition to equipment failures, power failures or human errors. The resulting short periods of reduced efficacy can have a large impact by leading to a local outbreak, but may also affect the total yearly average health risk. For example: if a treatment normally provides 4 log inactivation but during 1 day in a year the removal is reduced to 1 log; this results in an increase of the yearly average number of organisms in the treated water, which becomes 4 times higher. Therefore an assessment of the frequency, duration and magnitude of hazardous events are all essential for properly undertaking a quantitative risk assessment. Special attention needs to be paid to simultaneous effects of hazardous events on source water concentrations and the different treatment processes.

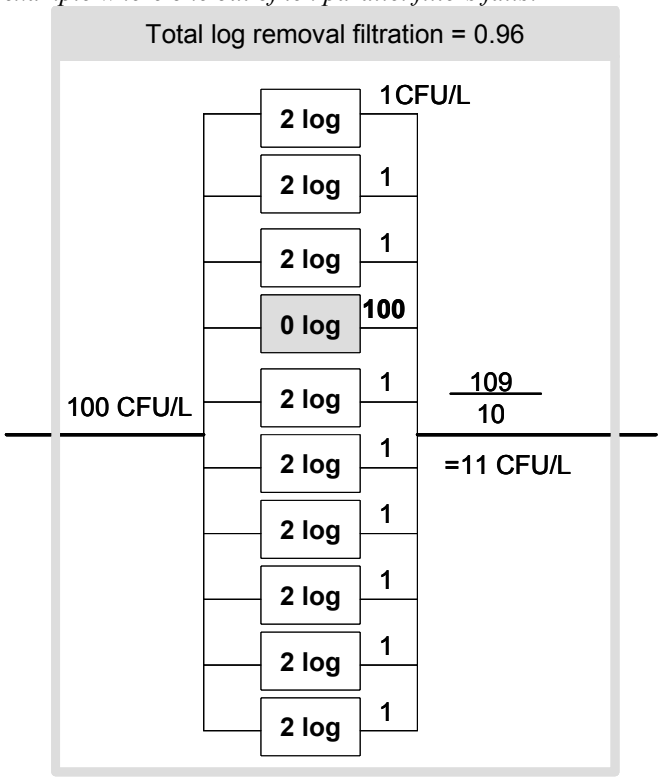
Hence, data collected for treatment assessment, as well as other parts of the QMRA model (Chapter 7), needs to reflect normal process variations and hazardous events. Treatment assessment relies on microbial as well as operational data. Monitoring data for pathogens is generally unavailable and indicators are seldom monitored throughout treatment and often at best weekly. Short-term events are therefore unlikely to be picked up. Statistical models based on operational data, therefore, generally provide the best estimate of regular variations in treatment efficacy and can be applied to extrapolate the occurrence of rare events. Hazardous events are however not likely to be part of regular variations and might require different monitoring approaches. A statistical assessment normally relies on monitoring and most treatment plants only perform monitoring of raw and treated water in order to comply with legislation. Monitoring after treatment will generally result in large datasets of non-detect samples. Probabilistic assessment can then provide the likelihood that treatment reduction was above a certain limit, but seldom provides the most likely reduction. One cause of uncertainty is that the concentrations in raw water vary with time and pre- and post-treatment (step) samples are seldom representative of the total volume being treated. In addition, the treatment can vary between parallel units and in time, adding to sample variability. Microbiological analysis also includes a number of uncertainties of which variable recovery of pathogens is well known (Chapter 7). For example, culture-based techniques do not pick up injured but still infectious pathogens [Villarino *et al.*, 2003] and such disinfected organisms may undergo repair mechanisms [von Sonntag *et al.*,

2004]. When data on process conditions is used to model pathogen reduction, errors in the process models further add to uncertainty. Model parameters such as disinfection kinetics or log reduction for specific process conditions are often inconclusive or simply not available for some pathogens. Also model assumptions on hydraulic characteristics or other affecting mechanisms can cause considerable uncertainties. So despite extensive monitoring, the present approach with grab samples cannot fully account for all the produced water variabilities and uncertainties. Nonetheless, using the probabilistic methods presented here and in Chapter 7, factors related to variability and uncertainty are partly accounted for and are propagated throughout the risk assessment.

In risk assessment a full-scale system is represented by a model. Since a model is a simplification of reality, the amount of detail applied needs to match the purpose of the assessment (Chapter 7). In the exposure assessment (Figure 7.1) treatment is represented as a single barrier, but the multiple processes in most treatment systems, sometimes referred to as barriers, make the treatment a multiple barrier system in itself [WHO, 2003]. By combining different types of processes in series, failure of one barrier can be partially compensated by others. In practice each treatment barrier consists of several parallel process units, like filters or sedimentation tanks. The poorest performing unit then determines the total efficacy of the barrier [Gale, 2002]. This is illustrated by an example for filtration, which normally provided two log *Cryptosporidium* removal (see Box I). When one out of ten parallel filters failed completely (no removal) the total efficacy of the filtration step was reduced to 1 log removal. The barrier efficacy is thus determined by its 'weakest link'. In this chapter a further focus is on how units within one process work together to safeguard against pathogens in source water.

BOX I: Weakest link

When a treatment step consists of several parallel treatment units, the poorest performing unit will dominate the pathogen removal. This is illustrated in the following example where one out of ten parallel filters fails:



So why model reduction by treatment instead of monitoring the produced drinking water? Chapter 3 showed that the level of presence (or absence) of pathogens in faecal

polluted source water can be determined by water quality monitoring with reasonable effort. Chapter 5 showed that intensive monitoring of faecal indicators in finished water still left a major uncertainty about the presence of pathogens in the treated water since the ratio between indicators and pathogens is unknown. Table 5.10 in Chapter 5 also shows that the percentage of positive indicator samples leaving the plant is of the same order of magnitude as the percentage of positives in the distribution system. Although this does not indicate that no contamination takes place during distribution, it does show that the water quality leaving the plant is relevant for risk assessments. The WHO guidelines for drinking-water quality state "Water Quality Targets are typically not developed for pathogens, because monitoring finished water for pathogens is not considered a feasible or cost effective option" and prefer the application of performance targets (Figure 4.1). The statutory *Cryptosporidium* monitoring in the UK exemplify a dataset of finished water pathogen monitoring. Analysis in Paragraph 4.7.4 illustrates the limitations of this data for quantitative treatment assessment. Hrudey and Hrudey [2004] showed that the occurrence of false positives makes it virtually impossible to estimate indicator bacteria concentrations in drinking water by monitoring at the observed low level. By quantifying the pathogen reductions in treatment, indicator and pathogen concentrations entering the distribution network can be estimated even at levels far below current detection limits. By applying probabilistic methods the probability of pathogens occurring in a certain volume of produced drinking water can be assessed. This probability can then be used in the total QMRA chain [Haas *et al.*, 1999; Chapter 7]. Several studies on quantitative risk assessment have pointed out that the main uncertainty lies in estimating pathogen concentrations in drinking water leaving the treatment plant [Teunis *et al.*, 1997; Haas and Eisenberg, 2001]. This chapter aims towards providing data and approaches to reduce that uncertainty.

Apart from providing an estimate of pathogens in drinking water over the past period, the treatment assessment also provides valuable information for preventing future risks. Firstly it increases the knowledge of the treatment system where the approach to quantify treatment efficacy raises new questions. What is the hydrology in my disinfection contact tank or (how) do we adjust the addition of coagulants when the water temperature changes? What is the impact of a break-through in one of a series of filters? Sadly a huge amount of (operational) data is collected at most treatment plants, without being used. Analysing some of these datasets can reveal the actual performance of the plant, the challenges it faces and the way operators react to these challenges. So the treatment assessment helps you to 'know your system', a key component to risk assessment mentioned in Chapter 2. By using operational data and monitoring outcomes the evaluation of pathogen reduction and the effect of changes on the quantified risk end-point will be refined. Based on these outcomes critical limits can be set on monitoring in relation to operational parameters to keep the risk at the baseline level and prevent events. When treatment moves beyond critical limits at one point and cannot be corrected at its source, corrective action needs to be taken at another point in the treatment chain in order to maintain the efficiency of the overall treatment barrier. Undertaking a treatment assessment also provides information for where monitoring and additional data collection will reduce treatment data uncertainty. Thus resource use can be focussed at the most appropriate points.

This chapter discusses the common drinking water treatment processes in relation to quantitative risk assessment for the MicroRisk project. A short overview of treatment principals and its capacity to eliminate pathogens is provided. Next, the chapter introduces the twelve systems that have been assessed within the MicroRisk project. A framework for interpreting the data collected follows and is then applied to the systems. Rather than presenting all assessments in full, examples from these are used to illustrate the treatment assessment approaches that ranged from simple to complex. Also a full assessment of one system is presented. Finally the experiences from performing the assessments are discussed and conclusions drawn. Chapter 7 provides more background on the applied statistical approaches, and integration of the treatment data into each catchment-to-customer (CTS) QMRA is addressed in Chapter 8.

4.2 TREATMENT PROCESSES THAT ELIMINATE PATHOGENS

4.2.1 Pathogen reduction in drinking water treatment

Drinking water treatment strives to provide safe and aesthetic drinking water in sufficient quantities. Chemical flocculation and filtration reduce pathogens as well as turbidity and dissolved organic carbon (DOC). Oxidation processes like ozonation disinfect the water, reduce colour, taste and odour, break down micro-pollutants and can have a positive effect on subsequent clarification processes. Disinfection can also have adverse health effects by producing disinfection by-products (DBPs) [USEPA, 2003].

Outbreak surveys [Hrudey and Hrudey, 2004; Chapter 1] show that contaminated groundwater is a major source. The treatment of groundwater is generally less extensive than for surface water and is not primarily aimed to eliminate faecal pathogens. In some cases disinfection will take place partly when the groundwater is known to be of variable quality and partly to reduce bacterial regrowth potential in the distribution system. When contamination of a groundwater source is suspected, the approaches for surface water treatment presented in this chapter can also be applied to groundwater treatment.

When assessing pathogen reduction by treatment, all treatment processes that are known to affect pathogens are addressed, even if they were not originally aimed at reducing pathogens, such as pre-oxidation. The assessment strives to provide a best estimate of pathogen reductions, which includes an assessment of variability and uncertainty. Minimal reduction is typically the only data considered when supplied information consist of ‘removal greater than …’, which is often the case when monitoring results in only non-detects. Potential reduction is the reduction that might be expected when data is insufficient to set an upper limit (‘removal smaller than …’). For the assessment itself a conservative safe approach is taken, with minimal and potential

reductions estimated or derived from expert judgement. Sensitivity analysis utilising these estimates, as discussed in Chapter 7 can provide insight into the uncertainty and importance of these treatment reduction estimates. It also provides a rationale to support decisions for further data collection or treatment adaptation.

Treatment can be divided in two reduction principles; particle removal and disinfection. In order to compile current knowledge on pathogen reduction by particle removal processes, a literature research was conducted based on lab-scale, pilot-scale and full-scale conditions [Hijnen *et al.*, 2005a]. The resulting dataset of microbial reduction provides reference values for full-scale treatment assessment, referred to as Mean Elimination Capacity (MEC[a]). The full-scale conditions were given higher weight-of-evidence in the MEC estimation than laboratory assessments. To illustrate the variation observed in literature, minimum and maximum reported reductions were also presented. Effects of specific design parameters, operation and water quality are discussed separately and only quantified in the range of "better" or "worse" than average.

Chemical disinfection is typically related to process conditions like disinfectant concentration, contact time, temperature and other associated water quality characteristics. Disinfection tables or kinetic parameters as reported in guidelines [USEPA, 2003] were used for some of the MicroRisk index pathogens. For the remaining index pathogens, disinfection kinetic parameters were obtained from a limited literature review.

Several sources in literature and guidelines provide an overview of reported removal or inactivation of typical treatment processes for a range of organism [Sobsey, 1989; LeChevallier and Au, 2004; USEPA, 2003; Hijnen *et al.*, 2005a, b]. The sections below on process characteristics and reduction principles are not intended as a handbook but rather to provide 'ballpark figures' from literature and are illustrated in the examples in subsequent sections of the chapter. The information provided in Sections 4.2.2 to 4.2.6 is based on [Hijnen *et al.*, 2005a, b] except when stated otherwise.

4.2.2 Coagulation-flocculation, sedimentation, flotation

Principle
Chemical coagulants added to water result in floc formation. These flocs partly capture dissolved and colloidal matter, and particulate materials, and are subsequently removed

[a] Each experiment in the review was awarded a weight according to how closely it resembled reality. Experiments conducted in the laboratory with cultured micro-organisms resulted in a weight of 1, an experiment conducted at full-scale using environmental micro-organisms already present in the water resulted in the maximum weight of 5. The MEC was calculated as the weighted average of all experiments for one type of treatment process and micro-organism (sum of log removal*weight per experiment, divided by the total sum of weights). The FS-score is the average of all weights used to calculate the MEC. A FS-index of 5 means all studies were performed at full scale with environmental micro-organisms.

by clarification processes. The temperature, dose of coagulants, pH and the characteristics of the organic matter govern the overall efficiency of chemical treatments. Limitations may be that small flocs partly remain in the water after clarification, protecting micro-organisms within the floc structure. Because of their size (0.02-10 μm) free suspended micro-organisms can be regarded as colloids, which under normal conditions do not settle at a significant rate. Settling is enhanced by coagulation-flocculation. The negative surface charges of the colloids are neutralized, enhancing the formation of larger particle aggregates during mixing. Dosed polymer aids may be used to further enhance floc formation. Processes like, settling, floatation, clarification or filtration will remove the flocs. For all floc removal processes, stable conditions are essential, hence sudden flow changes should be regarded as hazardous events and minimised. Design, operation and water quality all influence the efficacy of flocculation. To assess the local operational conditions, regular (jar)tests are required to optimize the system and adapt it to changing conditions. Factors influencing pathogen removal are briefly discussed.

Clarifier types are chosen based on local conditions including, costs, maintenance and local suitability. Hijnen [2005a] however reported that variation in pathogen removal within one type of clarifier was similar to variation between different types. Most systems are able to provide adequate removal if properly optimised and operated for the local situation.

The coagulant and polymer type need to be chosen in relation to the water quality and process design. Disease outbreaks following conventional treatment (Chapter 1) illustrate the need for frequent optimisation. For example, preceding the Milwaukee outbreak, the coagulant was changed from alum to alum chloride resulting in smaller flocs and thus, less entrapment of smaller particles like *Cryptosporidium* oocysts (4-5 μm dia), which eventually contributed to the outbreak [Hrudey and Hrudey, 2004].

Coagulant and coagulant aids (polymer) dose and mixing energy are the main operational parameters for conventional treatment. The dose of coagulant and polymer needs to be adjusted to the amount and type of particles in the water. For example, algal blooms are known to have an adverse effect on clarification. The coagulant is rapidly mixed during dosing, followed by gentle mixing in order to facilitate the flocculation, without short-circuiting. Sometimes pre-oxidation is applied before or in combination with coagulation to enhance process efficiency. A optimisation of dosing and settling can improve *Cryptosporidium* removal from less than two to almost five log units while total particle removal can be improved from 2 to 3 log [Dugan *et al.*, 2001].

Water quality: A pH between 5.8 and 8.5 is necessary when using alum as coagulant, otherwise poor coagulation is likely. Sufficient alkalinity is also required when using Alum. Cold water temperature will interfere with coagulation and sedimentation by decreasing the settling rate of the flocs. At low turbidity (< 1-5 NTU), insufficient material can be present to form suitable flocs. Thus co-entrapment of pathogens with flocs, whose presence in not always related to turbidity, is also reduced. High turbidity

may require an increase of coagulant dose to effectively flocculate particulates and pathogens.

Operation
Settling is generally operated as a continuous process, which requires skilled operation, backed up with regular jar-tests to provide optimal treatment under all conditions. The operation includes flow control, chemical dosing (coagulant, coagulant aid, pre-oxidation, pH adjustment), mixing conditions, sludge blanket height, sludge removal rate, air flow rate (if flotation used) and other conditions, depending on the process type. All these can have a significant impact on pathogen removal.

Indicators
Surface properties of micro-organisms may impact on their removal, however, that is largely reduced by the presence of sufficient coagulant. Inactivation can play a role when pre-oxidation is practiced, especially for bacteria and viruses. This is discussed in more detail in Section 4.2.5 (disinfection). Other studies have shown that some process indicator micro-organisms like MS2 phage can be very sensitive to chemical dosing (see experiments in Paragraph 4.7). Hence the zone of surrogate micro-organisms introduction (before or after chemical dosing) may have significant impact on the assessed removal in challenge tests. Experimental studies have shown that removal of most micro-organism types is quite similar (Table 4.1).

Surrogates
Turbidity is traditionally used to monitor clarification and in some cases particle removal is used. The latter cannot be used as an index of pathogen removal in the chemical treatment, since new particles are formed during the process. Auto fluorescent micro-algae from surface waters, in the size fractions between 1-20 µm, however, seem to account for this and may potentially be used as an in-line surrogate for particle removal. Stable low turbidity does indicate good performance, but there is no direct relationship with the removal of pathogens. Sudden peaks in turbidity can indicate events of poor performance, but conversely, peaks in pathogens may not be associated with turbidity peaks.

Critical limits
Critical limits can be set to ensure proper functioning. High turbidity or low particle removal is an obvious trigger for corrective actions. Use of streaming current detectors and zeta potential monitors to measure net surface charge of particles is currently studied and could lead to operational guidelines and limits based on local jar tests.

Table 4.1 The MEC (mean elimination capacity) of micro-organisms by coagulation/floc-removal

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	5	12	3.5	1.8	1.7	0.2 – 4.3

Bacteria[b]	6	9	4.7	1.5	1.4	0.6 – 3.7
Bacterial spores	6	11	4.7	1.4	1.2	0.8 – 3.2
Cryptosporidium	8	13	3.9	1.9	1.8	0.4 – 3.8
Giardia	8	12	4.3	1.6	1.3	0.0 – 2.9

[a] the higher the number, the more experiments resembled full scale and environmental organisms
indicator bacteria (*E. coli,* coliforms, faecal streptococci)

Other types of particle formation and removal processes in addition to conventional treatment (see further 4.2.4), includes direct filtration and natural sedimentation. Natural sedimentation, without addition of chemicals, generally takes place in (storage) reservoirs before treatment and is not controlled.

4.2.3 Granular media filtration

Principle
The reduction mechanisms of micro-organisms during filtration through porous media are a combination of size exclusion, physical/chemical adsorption/desorption and biological competition/predation. In the process, water is directed through a bed of granular media, generally a range of fine sands. Particles are entrapped between or attach to the filter grains. During backwash particles are partly rinsed off and disposed of with the backwash water. Micro-organisms that remain attached either die-off or detach and thereby penetrate the barrier. The attachment processes of small particles are generally described by colloid filtration theory [Yao *et al.*, 1971]. The strength of attachment to the filter grain surface is determined by particle and grain surface characteristics (electric charge, pre-coating, hydrophobic/hydrophilic interactions) and water characteristics (salt content, pH, DOC). The adsorption-efficiency varies substantially between different types of organisms and even between different strains of the same species [Gerba *et al.*, 1980]. Efforts to predict filtration efficiency incorporating these effects have not yet lead to satisfactory results due to the large number of variables to account for, but generally the following relations have been observed:
Bi- and trivalent cations increase the adsorption efficiency by reducing electrostatic repulsion. At pH 5-7 the adsorption efficiency is higher than at higher pH, although the effect depends on the iso-electric points of the particles and the filter grains. Organic compounds compete for attachment sites to filter grains, but since attachment does not appear to be site-limited, DOC may have a minor effect [Lo and Sproul, 1977; Gerba, 1984; Dizer, 1988]. Surface structure and ionic exchange capabilities of filter material can increase adsorption, although this is less well quantified [Elimelech and O'Melia, 1990]. Changes in flow can cause attached organisms to detach by sheer forces. Detachment is a goal during backwash, but sudden changes in filtration velocity can also lead to increased turbidity or particle counts post filtration. Apart from the initial effect of straining, the log removal of micro-organisms is expected to relate to filter bed depth. In theory removal of micro-organisms is higher at lower filtration rates. Pilot studies have supported this relationship, although not as strongly as predicted. Table 4.2 illustrates that the observed removal of different types of micro-organisms in experiments is not as large as theoretically expected.

Operation
Filters are operated in cycles (between backwashing) of generally one to several days. After backwash turbidity is generally higher, but normally stabilises at low levels in the filtrate after 15 to 45 minutes. The filter resistance (pressure drop across the filter) slowly increases during filtration when the removed particles are building up in the filter. By the end of the filter cycle an increase of turbidity can sometimes be observed again, indicating breakthrough of particles. Huck *et al.* [2002] showed that removal of *Cryptosporidium* can be severely impaired at this stage, while Emelko *et al.* [1999] observed breakthrough already before turbidity increase was observed. Other studies have shown a gradual reduction in removal efficiency down to 50% of the initial log removal during the filter cycle. The filter is backwashed by reversing the flow at an increased velocity, often combined with air bubbles. The accumulated matter is disposed of with the backwash water and the filter cycle starts again. Many variations of filter operations are possible and filter operation requires skilled personnel. More detail on the operation and maintenance of filters can be found in [Hrudey and Hrudey, 2004]. Significant to pathogens is that, settled backwash water is in some waterworks returned to the head of the plant to reduce water losses. From a risk perspective this is not recommended, since this increases the potential pathogen concentration at the head of the plant, combined with a period of less efficient filter performance during the new cycle, and has though to be responsible for the prolonged plant challenge during the Swindon cryptosporidiosis outbreak [Badenoch *et al.*, 1995]. Filter cycle management in relation to pathogen reduction is further discussed under critical limits.

Pathogen indicators or model micro-organisms
Since filtration relates to the size of particles, the size of the pathogen indicator (called a model micro-organism [Ashbolt *et al.*, 2001]) needs to be the same as the size of the pathogen. Phages can be used as models for viruses, bacteria for bacterial pathogens and bacterial spores model protozoan oo/cyst removal. Surface properties differ between strains and also within a population of the same strain. Hijnen [2005b] reported that the surface properties of *Cryptosporidium* change with age. While there is no perfect model for any pathogen, as even pathogens of the one group vary greatly, for risk assessment the above proposed model micro-organisms were applied from a practical point of view (it's the best we got).

Surrogates
Traditionally turbidity has been used as a surrogate for filter performance. A direct relation between turbidity and micro-organism removal does not exist. However a stable and low turbidity of the filtrate does indicate that a filter is working properly, and based on this it can be assumed that the filtration removal is efficient as a barrier step. The USEPA "recognizes that turbidity reduction is not a direct indication of pathogen removal, but it is an effective indicator of process control" [USEPA, 2003]. Rather, it is the variation in drinking water turbidity that should be targeted and this has also been shown as an association to sporadic disease (see Chapter 1). Particle counting at different size fractions is a more sensitive technique than turbidity, yet more costly. Reduction of different sized particles enables to differentiate between different groups

of pathogens, although it is still not a direct measurement of pathogen reduction. This is exemplified in Section 4.7 where reduced particle removal at the start of a filter cycle, did not correspond with the *E. coli* reduction.

Critical limits

Once a filter has been put into operation, the effect relies on the continuous operation and avoiding changes in flow rate or water quality. Periods of poor removal can be circumvented by applying filtrate to waste after backwashing and backwashing before breakthrough occurs, based on appropriate operational limits. This is currently assessed through turbidity measurement or particle counting after filtration with a high-enough resolution to account for short-time variations. Prior to the Milwaukee outbreak treated water turbidity increased from 0.1 to 1.5 NTU, and similar turbidity peaks were observed at other outbreaks [Hrudey and Hrudey, 2004]. As a control measure, the USEPA [2003] states that 95% of time 0.3 NTU must not be exceeded in filtered water, and turbidity must never exceed 1 NTU when monitoring each individual filter with a frequency of 1/15 minute or higher in order to apply 0.5 log removal credit for *Cryptosporidium*. When 0.1 NTU is maintained a 1 log removal filtration credit can be applied. Remarkably these restrictions are not required during the 15 minute period after backwash when the highest turbidity is expected. The USEPA [2003] applies 0.5 log removal credits for a second stage filtration for *Cryptosporidium.* WHO [2004] indicates < 1 log removal for most micro-organisms under baseline conditions and a maximum of 2-4 logs.

Other sudden changes in water quality (pH, electrical conductivity) are also precursors of events and, based on their operational identification, this is corrected either upstream (improve clarification) or downstream (increase disinfection dose). When a single filter in an array of parallel filters fails, this filter may be taken out of production, since it will have a large impact on the overall performance (see also example in Box I).

Table 4.2 The MEC (mean elimination capacity) for micro-organisms by rapid sand filtration

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	12	63	3.2	0.8	0.6	0.1 – 3.8
Bacteria[b]	12	109	3.2	0.6	0.6	0.1 – 1.5
Bacterial spores	11	102	4.2	1.3	1.4	0.0 – 2.9
Cryptosporidium	15	151	3.3	2.0	1.8	0.0 – 3.1
Giardia	10	124	3.3	1.7	1.6	0.0 – 6.5

[a] the higher the number (1-5), the more experiments resembled full scale and environmental organisms; [b] indicator bacteria (*E. coli,* coliforms, faecal streptococci)

Other types of granular media filtration

Granular Activated Carbon (GAC) filtration

GAC filtration uses granular activated carbon as filter material and is generally applied as second stage filtration to remove taste and soluble micro-pollutants but will also account for particle removal (Table 4.3). GAC filters are sometimes operated up to several months without backwashing when the inflow contains little suspended material. The filter material is relatively coarse, but its surface characteristics can improve attachment. Relatively few well-performed studies on GAC filtration are available, with partly contradictory results, both indicating better and worse removal of micro-organisms as compared to rapid sand filters [Hijnen *et al.*, 2005b].

Table 4.3 The MEC (mean elimination capacity) for micro-organisms by granular activated carbon (GAC) filtration

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	2	10	2	0.4	-	0.2 – 0.7
Bacteria[b]	3	16	3	1.4	-	0.9 – 2.9
Bacterial spores	4	8	5	0.8	-	0.4 – 1.2
Cryptosporidium	1	12	3	0.9	-	0.7 – 1.1
Giardia	2	16	3	1.7	-	0.4 – 3.3

[a] the higher the number (1-5), the more experiments resembled full scale and environmental organisms; [b] indicator bacteria (*E. coli,* coliforms, faecal streptococci)

Slow sand filtration

Slow sand filters are operated at low filtration rates (0.1-1.0 $m.h^{-1}$) and contain fine sand (0.2-1.0 mm) with a filter bed depth of 0.5 to 1 m. Headloss slowly increases due to filter ripening and build up of the 'smutzdecke' in the top layer, where the main reduction of micro-organisms appears to occur, partly due to biological processes like predation and partly due to adsorption and size exclusion in the pore matrix. After the filter bed surface is scraped off (months –year interval) when headloss becomes too great, filter to waste is applied and several weeks need to be allowed for filter ripening before it goes back in production. The fine filter material, low filtration rate, no backwash and biological activity in the filter make it a potentially effective barrier against pathogens and will level out peaks from the inlet water (Table 4.4). Due to retardation (attachment and detachment) a long-term, gradual increase of micro-organisms breakthrough may be observed over time.

Table 4.4 The MEC (mean elimination capacity) for micro-organisms by slow sand filtration

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	10	13	3.3	2.2	2.1	0.6 – 4.0
Bacteria	9	17	3.4	2.7	2.4	1.2 – 4.8
Bacterial spores	5	9	4.4	1.5	1.3	0.0 – 4.0
Cryptosporidium	6	8	3.4	3.8	nd	0.3 – >6.5
Giardia	3	3	3.7	3.3	nd	1.2 – 6

[a] the higher the number (1-5), the more experiments resembled full scale and environmental organisms;

Bank filtration and artificial recharge

The processes of bank filtration and artificial recharge are similar to slow sand filtration. Since filter bed composition is less well controlled, there is a potential for the

presence of macro-pores reducing efficiency. Due to the natural processes in the soil, the water characteristics can change (redox potential, salt content) which in turn influences pathogen attachment efficacy. Bank filtration is particularly vulnerable to high water levels, which reduce the travel distance between the surface water and the abstraction well. Ultimately the abstraction well can be submerged during floods. Adequate sealing of the well-head is then required. The LT2ESWTR [USEPA, 2003] gives only a 0.5 to 1 log *Cryptosporidium* removal credit for bank infiltration, depending on the distance between well and surface water. WHO [2004] suggests a 4 log removal after 4 m transport for all pathogens, which appears too high compared to some studies [Schijven et. al, 1998]. For MicroRisk the treatment assessment values for slow sand filtration in Table 4.4 were used for bank filtration.

Other granular media filtration
A whole range of different filter materials are used in drinking water treatment, where the removal of micro-organisms are poorly documented. Precoat types of filtration, including diatomaceous earth and perlite are mentioned in an overview of treatment reduction efficacy [WHO, 2004]; where they were accredited a baseline removal of 50%, 90% and 99.9% for bacteria, viruses and protozoa respectively. For all filter materials, the use of sand filtration reference values in Table 4.2 are suggested when they are supported by actual measured turbidity, particle or indicator removal (see Section 4.5.1).

4.2.4 Conventional treatment

Principle
The term 'Conventional treatment' is used here to address the combination of coagulation-flocculation-floc removal including granular media filtration. This combination of processes is very common for surface water treatment and the total reduction by the combined processes has been studied extensively. The results are summarized in the Table 4.5. The principle, operation, indicators and the control points have been discussed in the former 'coagulation' and 'filtration' Sections 4.2.2 and 4.2.3.

Surrogates
Log removal of turbidity has been recognised as a generally conservative surrogate for *Cryptosporidium* oocyst removal by conventional treatment (in contrast to the individual processes). Particles (in the size fractions of 7-11 µm and 4-7 µm respectively) were found to be a more accurate and still conservative surrogate for *Cryptosporidium* and *Giardia* oo/cysts..

Table 4.5 The MEC (mean elimination capacity) for micro-organisms by conventional treatment (coagulation-flocculation-sedimentation-filtration)

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	7	69	3.6	3.0	2.5	1.2 – 5.3
Bacteria	7	54	3.1	2.1	2.1	1.0 – 3.4

Bacterial spores	11	62	4.7	2.4	2.1	1.4 – 4.7
Cryptosporidium	15	162	3.7	3.2	2.9	1.4 – 5.5
Giardia	8	67	4.3	3.4	3.3	2.1 – 5.1

[a] the higher the number (1-5), the more experiments resembled full scale and environmental organisms; [b] indicator bacteria (*E. coli,* coliforms, faecal streptococci)

Direct filtration

Direct-filtration or in-line filtration is when coagulation takes place directly before granular media filtration (no separate floc removal stage). Depending on the design of the system, some sedimentation will take place on top of the filter, while the main reduction occurs within the filter, increasing the filter efficacy and resistance (Table 4.6).

Table 4.6 The MEC (mean elimination capacity) for micro-organisms by direct filtration

Organisms	**Data characteristics**			**MEC**		
	Studies	Data	FS-index[a]	MEC	50%ile	Range
Viruses	7	48	2.9	0.9	0.5	0.1 – 3.9
Bacteria[b]	4	35	3.0	1.4	1.5	0.8 – 3.3
Bacterial spores	5	31	2.6	2.2	2.2	1.5 – 3.9
Cryptosporidium	11	244	2.6	3.0	2.9	0.8 – 5.4
Giardia	9	115	3.1	2.5	2.8	0.8 – 3.9

[a] the higher the number (1-5), the more experiments resembled full scale and environmental organisms; [b] indicator bacteria (*E. coli,* coliforms, faecal streptococci)

4.2.5 Chemical disinfection

Principle

A disinfectant (a strong oxidant) is added to the water and through chemical changes yields non-infectious pathogens. Commonly used oxidants in drinking water production and distribution are (in increasing oxidant strength) chloramines, chlorine, chlorine dioxide and ozone. The disinfectants also form a range of reactive decomposition products, depending on water composition. At higher pH chlorine and ozone are less effective because the decomposition equilibrium shifts from highly reactive to less reactive compounds. Protozoa are most resistant against oxidants, followed by viruses and bacteria. The level of inactivation depends on the concentration (C) of the disinfectant and the contact time (t), expressed as Ct (concentration in $mg.L^{-1}$ x contact time in min.). Special contact tanks or clear water tanks are used to provide sufficient contact time. The concentration of oxidants decreases due to reaction with organic compounds, organisms and auto-decomposition. At low temperatures both oxidant decay and inactivation of micro-organisms slows down. After sub-optimal exposure to oxidants some pathogens/indicators may become non-culturable but still infectious (after time for repair). The oxidants also form Disinfection-By-Products (DBP's) which may have adverse health effects. This limits the level of oxidation that can be applied and requires a balance between level of disinfection and formation of DBP's [Havelaar *et al*., 2000].

The presence of particulate matter will also affect the disinfection efficacy. Micro-organisms attached to particles are more likely to be resistant to oxidation than when

freely suspended [Marin *et al.*, 1996]. The choice of disinfectant depends on several factors including:

- Efficacy against pathogens (bacteria, viruses and protozoa);
- Ability for accurate monitoring and control of the process;
- Ability to maintain a residual in the distribution system;
- Water composition and formation of DBPs; and
- They should not comprise the aesthetic quality of the drinking water.

Operation
The oxidant can be dosed as a liquid (chlorine, chlorine dioxide) or as a gas (chlorine, ozone). Sequential dosing of chlorine and ammonium compounds forms chloramines. The level of dosing is generally proportional to flow, without adjustment to temperature or contact time.

Indicators
Each species and strains of micro-organism can differ in their susceptibility to oxidants, but this is often generalised based on the inactivation of bacterial indicators. Even the state of the organisms can result in different susceptibilities. An indicator with an inactivation rate constant close to that of the pathogen should be selected to reduce uncertainties.

Surrogates
A surrogate needs to reflect the amount of oxidant exposure during a process. Some suggested DBPs as a surrogate [Gunten *et al*., 2001], however, generally a very low level of DBPs is strived for, making it a relatively insensitive measure. Monitoring for DBPs is generally expensive. Further, operators will strive to reduce DBPs while maintaining disinfection, thus breaking the link between DBPs and pathogens. Overall, DBPs are not considered a suitable surrogate for chemical disinfection, and modelling Ct from *in situ* sensors provides the preferred surrogate measure.

Critical limits
Disinfection is an actively controlled process and operational and critical limits can be directly related to the functioning of equipment (dosing pump operation, dosing flow, chemical concentration) or measured process condition (water flow, temperature, measured residual). In the future a critical limit could be set for the inactivation of the target organism(s) by applying ‘soft-sensors’, which combine signals and translate these into disinfection efficacy by modelling. With the disinfection model proposed below, the measured flow, temperature, and oxidant residual can be combined into a ‘disinfection soft-sensor’.

Chemical disinfection calculations:
Inactivation is usually determined in laboratory systems with high numbers of cultured micro-organisms spiked into the system under well-controlled Ct conditions. In practice several factors cause a deviation from the laboratory conditions.

Water will pass a full-scale reservoir with varying contact times since it is not a plug-flow or batch reactor. A small percentage of the total flow will receive little disinfection, thus having a large impact on the average concentration after disinfection, similar to the weakest link example of filtration in Box 1. In disinfection modelling the continuously stirred tank reactor (CSTR) model is applied to account for hydraulics [USEPA, 2003], assuming that a disinfection contactor consists of a number of CSTRs in series. The applicable number for a full-scale system can be determined with tracer tests or CFD modelling. Do Quang *et al.* [2000] showed how the number of CSTRs can be estimated from the contactor geometry. Basically a reservoir without baffles acts as a single CSTR and (ozone) contactors with baffles can be modelled by one CSTR per contact chamber. These assumptions are applied in treatment assessment unless full-scale tracer tests show different hydraulics. Inactivation in a single CSTR is calculated as:

$$\frac{N}{N_0} = \frac{1}{1 + k_e \bullet c \ \bullet t_h}$$

where N_0 and N (micro-organisms L^{-1}) are the concentrations before and after the CSTR respectively, k_e ($L.mg^{-1}.min^{-1}$) is the natural logarithm based inactivation rate constant, c ($mg.L^{-1}$) is the disinfectant concentration at the outlet of the CSTR and t_h (min) is the hydraulic residence time in the CSTR. The total inactivation by a number of CSTRs is calculated as the product of the inactivation in each of the CSTRs. Since full-scale installations are characterized by a few CSTRs, the achievable inactivation is strongly reduced. Do Quang *et al.* [2000b] provide more advanced CSTR models to include different types of baffles and disinfectant decay.

An alternative to CSTR modelling is the Ct10 calculation [USEPA, 2003] where the Ct in reservoirs is corrected by applying a baffling factor to account for short-circuiting. A baffling factor of 0.1 is applied to an unbaffled reservoir (which corresponds to the T10 of a single CSTR). Figure 4.2 shows that the models can provide different inactivation results when they use the same measurements of C and t. At low Ct values the CSTR model predicts higher inactivation that the Ct10 method. At higher Ct values, the Ct10 approach predicts high levels of inactivation, whereas the CSTR approach strongly reduces the effect of high Ct values. Although it is unlikely that full-scale contactors really are completely mixed, the CSTR seems to provide a more realistic estimate of inactivation in full-scale reactors than the Ct10 method. This was partly verified by Smeets *et al.* [2006] when large volume samples of ozonated water at full scale still showed the presence of *E. coli*, limiting the inactivation to an estimated 2.6 logs; while the Ct10 models for this situation predicted over 40 log inactivation versus 5 log by the CSTR model. The CSTR model is recommended for treatment assessment in QMRA to prevent unrealisticly high estimates of inactivation. Based on such modelling, guidelines for disinfectant concentration in relation to flow and temperature can be set for operational purposes. The example also illustrates that although inactivation models are helpful for treatment assessment, their results still need to be critically judged.

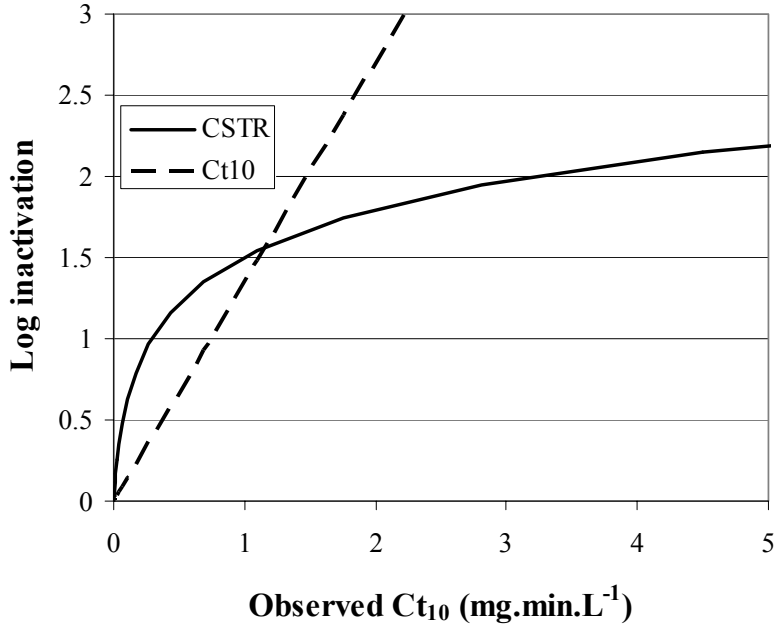


Figure 4.2 *E. coli* inactivation by chlorine in a reservoir (one CSTR, baffling factor 0.1) at 5°C modelled by CSTR and Ct10 methods

The inactivation rate constant k_e is specific for the assessed micro-organism and disinfectant conditions used. Values reported here for viruses and *Giardia* were taken from the SWTR Ct tables [AWWA, 1991], for *Cryptosporidium* from [USEPA, 2003] and for *E. coli* from experiments described in USEPA [2003]. Studies can show considerable differences in disinfection kinetics under similar conditions. Most kinetics have been determined with cultured micro-organisms in artificial, disinfectant demand free water, which does not reflect the reduced inactivation in natural waters [Sobsey, 1989; Thurston Enriquez *et al.,* 2003; Hijnen *et al.*, 2004; Smeets *et al.*, 2005]. Possible explanations are the heterogeneous susceptibility of environmental *E. coli* [Hom, 1972], starvation status of bacteria [Lisle *et al.*, 1998] or protection by encapsulation in aggregates of micro-organisms or particles [Sobsey, 1989; Hijnen *et al.*, 2004]. Haas and Kaymak [2003] suggest that high initial microbiological densities in disinfection experiments result in higher inactivation rate constants. These findings stress the need to determine inactivation rates under full-scale conditions with environmental and not laboratory organisms. Conservative k_e values from literature were used in treatment assessments (Tables 4.7, 4.8, 4.9). Sensitivity analysis showed that uncertainty about full-scale hydraulics has a much stronger effect on the assessed inactivation than uncertainty about k_e.

Inactivation is temperature dependant, so k_e is temperature specific. The Arrhenius equation was used to determine k_e at different temperatures:

$$k_e = A * e^{\left(\frac{-E_a}{RT}\right)}$$

A is the frequency factor in $l.mg^{-1}.min^{-1}$, E_a is the activation energy ($J.mol^{-1}$), R=8.314 ($J.mol^{-1}.K^{-1}$) is the ideal gas constant and T is the absolute temperature (K). In general the inactivation rate doubles with each 10 °C increase in temperature. The effect of pH was only taken into account for chlorine disinfection of *Giardia*. Information was insufficient to account for pH effects with other micro-organisms.

When exposure increases the inactivation rate decreases at a certain point, which is referred to as tailing, due to reduced susceptibility of environmental organisms. Also a 'threshold' exposure is sometimes observed below which no inactivation takes place, referred to as 'shoulder'. Several alternative models have been suggested to account for these effects [Finch *et al.*, 2001]. However these models require a number of additional parameters, which are generally not available since most inactivation experiments have less than 7 log dynamic range, hence the maximum model output was set to 7 log units.

Tables 4.7, 4.8 and 4.9 provide an overview of the disinfection kinetic rate constants used as 'ballpark figures' for different groups of micro-organisms. The values for *Cryptosporidium*, *Giardia* and (entero)viruses were based on USEPA Ct tables [USEPA, 2003]. These only included pH dependency for *Giardia* and chlorine, while virus figures are valid for pH 6-9. *E. coli* inactivation by chlorine and chlorine dioxide was based on literature values from [Narkis *et al.*, 1995; Rice *et al.*, 1999], while *E. coli* and *Campylobacter* inactivation by ozone was a combination of literature values and experimental results (see Section 4.7 and Smeets *et al.* [2006]). Smeets *et al.* [2006] demonstrated that environmental bacteria were less susceptible to ozone than cultured bacteria, hence both inactivation rate constants are presented here. Inactivation of *Campylobacter* and *E. coli* O157 were considered to be similar to the *E. coli* inactivation reported for environmental organisms (ozone).

Table 4.7 Chlorine inactivation rate constants by micro-organisms

Chlorine inactivation rate constants				Calculated log inactivation for observed Ct of 1 $mg.min.L^{-1}$	
Micro-organism	Temperature dependency (Arrhenius)		k_e	plug flow	single
	E $J.mol^{-1}$ $*10^3$	A $l.mg^{-1}.min^{-1}$	$L.mg^{-1}$ $.min^{-1}$ 10°C		CSTR
Cryptosporidium	-	-	-	-	-
Giardia pH 7	47	7.5	0.062	0.027	0.026
Virus	50	9.8	4.58	2.0	0.75
Ct<0.5 *E. coli*	46	9.8	19.6	[a]	[a]
Ct>0.5 *E. coli*	49	9.8	6.67	2.9	0.88

[a] Not valid under these conditions

Table 4.8 Chlorine dioxide inactivation rate constants by micro-organisms

Chlorine dioxide inactivation rate constants	Calculated log inactivation for observed Ct of 1 $mg.min.L^{-1}$

Micro-organism	Temperature dependency (Arrhenius) E $J.mol^{-1}$ $*10^3$	A $l.mg^{-1}.min^{-1}$	k_e $L.mg^{-1}.min^{-1}$ 10°C	plug flow	single CSTR
Cryptosporidium	59	8.8	0.0054	0.0023	0.0023
Giardia <5°	138	25.3	[a]	[a]	[a]
Giardia >5°	30	5.0	0.24	0.10	0.093
Virus	48	8.8	0.47	0.20	0.17
E. coli	23	5.7	16.4	7.1	1.24

[a] Not valid under these conditions

Table 4.9 Ozone inactivation rate constants by micro-organisms

	Ozone inactivation rate constants			Calculated log inactivation for observed Ct of 1 $mg.min.L^{-1}$	
Micro-organism	Temperature dependency (Arrhenius) E $J.mol^{-1}$ $*10^3$	A $l.mg^{-1}.min^{-1}$	k_e $L.mg^{-1}.min^{-1}$ 10°C	plug flow	single CSTR
Cryptosporidium	63	11.0	0.24	0.10	0.093
Giardia	49	9.8	4.9	2.14	0.77
Virus	47	9.7	10.0	4.35	1.04
E. coli lab	48	11.6	[a] 576	[a] 250	[a] 2.76
E. coli environ	48	11.1	174	75.6	2.24

[a] Included to illustrate the difference between cultured and environmental *E. coli*, use of inactivation rate constants for environmental *E. coli* is preferred when assessing full scale treatment.

4.2.5 UV disinfection

Principle

Ultra violet (UV) disinfection relies on UV transmission through water. Lamps are placed in a flow-through contact compartment that is continuously operated. The inactivation of pathogens is related to the UV fluence, which is the product of UV light intensity and contact time. Contact time depends on rector volume and flow and the number of lamps, and the lamp power determines the intensity. The intensity decreases with the distance from the lamp surface due to radial expansion and the UV absorption of water and dissolved organic substances therein. Therefore the flow pattern inside the UV reactor has a high impact on reactor efficacy and considerable effort has been put in optimising reactor configurations for disinfection. In order to test this, dosimeter experiments are carried out which result in a Reduction Equivalent Fluence (REF). High concentrations of micro-organisms are treated, and simultaneously the UV intensity is measured inside the reactor. The measured inactivation is then related to the measured intensity providing guidelines for operation [Sommer *et al.*, 2000].

Low-pressure or medium pressure mercury lamps are generally used. Low-pressure lamps emit a major peak at a narrow spectrum around 256 nm (monochromatic light), at which frequency thymine dimers are formed within microbial DNA or RNA, hampering reproduction. Medium pressure lamps emit a much broader spectrum of UV light (polychromatic UV), which also affects proteins and membrane transport.

Polychromatic light can be twice as effective as monochromatic light at the same fluence. In some cases repair mechanisms of bacteria either in light or dark conditions can make an inactivated organism infectious again and several viruses have been shown to use repair enzymes in the host cell [von Sonntag *et al.*, 2004]. Although protozoa repair was demonstrated they appear not to regain infectivity. Turbidity will have a negative effect on UV disinfection by shielding pathogens from UV light. Temperature, pH, electric conductivity and alkalinity seem to have no effect on inactivation. Hijnen *et al.* [2006] has recently published a detailed literature review of UV disinfection including these observations.

Operation
Operation consists of maintaining flow and lamp intensity within given specifications. Operating a number of reactors in parallel and bringing units on- or off-line controls flow and in some units lamp power can be varied. During stable operation, UV intensity decreases due to fouling of the lamp sleeve and ageing of the lamp. When UV intensity drops below a threshold value, the sleeve is cleaned, the lamp power increased or the lamps are replaced. UV disinfection requires regular maintenance and testing of equipment.

Indicators
Indicators can be used to determine fluence, and predict pathogen inactivation. Table 4.10 shows the susceptibility of pathogens and potential indicator organisms. MS2 is a conservative modelr for most viruses although a single study on *Norovirus* inactivation suggests that it may be relatively UV resistant. *Clostridium perfringens* and the aerobic spore-former *Bacillus subtilis* are conservative indicators for bacteria. Some studies show that environmental populations of bacteria are less susceptible to UV than cultured organisms. In contrast, cultured Adenovirus were more UV resistant [Hijnen *et al.* 2006].

Surrogates
A surrogate needs to represent the fluence that pathogens are exposed to with sufficient sensitivity to identify short-circuiting. Currently several substances are tested for suitability, but so far no definite applicable surrogate has been reported [Kiwa, 2006].

Critical limits
As described under operations, critical limits are set to the desired level of inactivation of target pathogens. Through the REF, this is translated to the measured UV intensity. Selecting the appropriate number of reactors, the number of UV lamps and their intensity/power sets the required dose.

Table 4.10 UV inactivation dose (fluence) for a required log inactivation by micro-organism*

	Required fluence ($mJ.cm^{-2}$)			
MIC required (log):	1	2	3	4

Adenovirus type 2,15,40,41	42	83	125	167
Adenovirus type 40	56	111	167	-[d]
Adenovirus[a] (not type 40)	25	50	-[d]	-[d]
Calicivirus canine	10	21	31	41
Calicivirus bovine	5	11	16	21
Calicivirus feline	9	19	28	38
Coxsackie virus B5	8	17	25	34
Hepatitis A	6	11	17	22
Poliovirus type 1	7	15	22	30
Rotavirus SA-11	10	20	29	39
Bacillus subtilis[a]	56	111	167	222
Campylobacter jejuni[b]	3	7	10	14
Clostridium perfringens[a]	45	95	145	-[d]
E. coli O157[b]	5	9	14	19
E. coli[a]	5	9	14	18
Legionella pneumophila[b]	8	15	23	30
Salmonella typhi[a]	6	12	17	51
Enterococcus faecalis[a]	9	16	23	30
Shigella dysenteriae[b]	3	5	8	11
Shigella sonnei[b]	6	13	19	26
Vibrio cholerae[b]	2	4	7	9
Yersinia enterocolitica[b]	3	7	10	13
Cryptosporidium USEPA[c]	3	6	12	-[e]
Giardia USEPA[c]	2	5	11	-[e]
Acanthamoeba[c]	40	71	119	167

*UV fluence ($mJ.cm^{-2}$) requirements for a MIC (Mean Inactivation Capacity) of 1 up to 4 log by monochromatic UV radiation for viruses, bacteria, bacterial spores and protozoan (oo)-cysts based on the k-values with or without correction for environmental species; for bacteria in wastewater a higher correction for environmental species is needed and further research has to clarify the need for a higher fluence to account for photoreactivation; for *Giardia* increased fluence requirement because of dark repair is a factor for further research. [a] environmental spp. ; [b] corrected for environmental spp. ; [c] no correction for environmental spp. (research needed) ; [d] MICmax < 4 log ; [e] no value due to tailing [Hijnen *et al.*, 2006].

4.2.6 Membrane filtration

Membrane filtration has become a cost-effective alternative to conventional separation processes. Common applications include low-pressure membranes, microfiltration and ultrafiltration as particle barriers, and high-pressure membranes, nanofiltration and reverse osmosis (RO) to remove colloidal and dissolved compounds.

Principle

Membranes are used to remove particulate matter by size exclusion (microfiltration [MF] and ultrafiltration [UF]) or dissolved substances by a combination of size exclusion and more complex process (Nanofiltration [NF] and reverse osmosis [RO]). Although these terms are used to indicate the size range that is removed by these membranes, there actually is a gradual change in pore size and molecular weight cut-off due to the large choice of available membranes. Microfiltration removes protozoan oo/cysts and most bacterial pathogens. Ultrafiltration removes bacteria and most viruses, depending on the chosen membrane type. Nanofiltration and reverse osmosis theoretically remove all organisms; since their pores are small enough to also account for dissolved molecules and salt. Jacangelo [2005] showed that the molecular weight cut-off for membranes is not always a good predictor of virus removal, since a few large pores have a limited effect on removal of dissolved molecules (typically less than 1 log) but a large effect on removal of pathogens (over 6 log units). Removal of pathogens then not only relies on pore size, but also on surface properties of the organisms in relation to the membrane material, similar to granular filtration, as well as loss of membrane integrity with age. Three membrane forms are most common; spiral wound (RO and NF), tubular (NF, UF, MF), ceramic (all, but expensive, so hardly applied in drinking water).

Membranes can be operated as 'dead end', where all the water goes through or with cross flow operation, which drains water from the feed-side of the membrane to keep a sufficiently high flow at the membrane surface to reduce fouling. Sometimes multiple membrane 'inserts' are placed in a pressure vessel which are combined in a 'stack'. This is the operational unit that is generally backwashed or cleaned. Several stacks are operated in parallel so one or two can be cleaned while the others maintain production.

Operation

The water production increases with increasing pressure across the membrane. At high pressure less membranes are needed, but energy costs and fowling of membranes increases. Therefore the pressure (and flow or flux) across the membrane is kept between strict operational limits. Membranes are operated in cycles, micro- and ultra filters are backwashed frequently to remove solids accumulated at the feed side of the membrane. Backwashing is not possible with nanofiltration and reverse osmosis. All membranes require regular chemical cleaning during which they are taken off-line. Monitoring particle removal when sufficient particles are present in the feed water tests membrane integrity. Additionally bubble or acoustic testing can be used *in situ.* NF and RO are typically applied to waters with low particle content to prevent fouling, and particle removal can only provide limited verification of membrane integrity. Therefore other substances, like sulphate are monitored to verify integrity of RO membranes. Membrane integrity can be compromised by rupture due to physical stress, damaging by sharp particles, membrane decomposition or leakage of seals and fittings. In practice loss of membrane integrity occurs frequently and this is accepted up to a point out of practical considerations. When integrity monitoring reaches the intervention level integrity is restored by repair or replacement of membranes. The varying removal of pathogens in practice is illustrated in Figure 4.3. Table 4.11 provides some information

on the potential removal by an intact membrane and the practical removal due to monitoring and leak repair limitations.

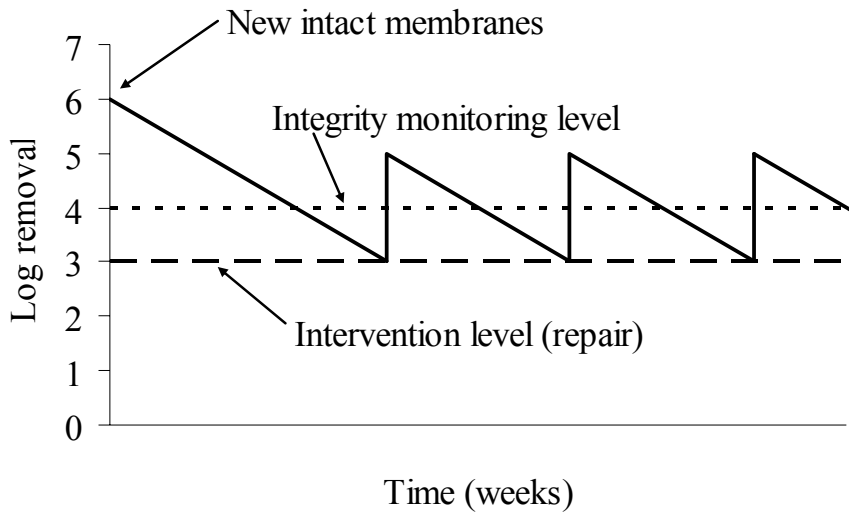


Figure 4.3 Theoretical variation of pathogen removal by membranes at full-scale due to integrity failure and repair in relation to monitoring

Indicators
The size of the indicator needs to represent pathogen size. Therefore phages and indicator bacteria are used for viruses and bacteria respectively.

Surrogates
Particle counting is generally used for MF and UF integrity monitoring since it is more sensitive than turbidity monitoring. It also provides information on the cut-off of different sizes of the particles, which can then be related to groups of pathogens. However particle counting does not account for surface characteristics. Kruithof *et al.* [2001] used powdered activated carbon as challenge particles to verify membrane integrity for up to 5.8 log units of virus removal. Conductivity or sulphate reduction is used for integrity testing of RO membranes up to 2 and 3 logs respectively, representing the measured removal of MS2 phages in challenge tests. The assumed (linear) relationship between sulphate removal and pathogen removal was not investigated [Kruithof *et al.*, 2001].

Critical limits
The integrity monitoring (by particle counting or sulphate measurements) is the most important in verifying sufficient removal. By monitoring individual membranes, a smaller leak can be detected than at the level of total flow. However, this is seldom practical to perform.

Table 4.11 MEC values for pathogen removal by intact membranes and monitored removal in practice due to leakage and monitoring limits. (ranges found in [Kruithf et al., 2001; Jacangelo, 2005])

	pore size µm	Log removal Viruses	Log removal Bacteria	Log removal Protozoa	Operational level (log)[c]
Microfiltration	0.1-1	0 – 3.7	0-4.3	2.3 - >7	
Ultrafiltration	0.01-0.1	>6.5	>7	>7	4
Nanofiltration[a]	0.001-0.01			2.2	
Nanofiltration[b]	0.001-0.01			5.5	
Reverse Osmosis	0.0001-0.001	2.7-6.5			2-3

[a] Cellulose acetate membrane, [b] composite thin film membrane

[c] Operational removal level is limited by source water particle or sulphate concentration and the intensity of product water monitoring (total flow or per stack) based on practical examples.

4.3 TREATMENT ASSESSMENT FRAMEWORK

This paragraph introduces the method that was used to assess the reduction of pathogens by a treatment system. The treatment schemes of the twelve Catchment-to-Tap Systems (CTS) that were assessed in the MicroRisk project (Chapter 3) are described and the available data at these sites is discussed. A framework is presented to combine this data for input into a QMRA model.

4.3.1 Treatment assessment approach

The goal of the treatment assessment is to describe pathogen reduction performance for input into a QMRA model. By combining this information with the pathogen concentration in the source water, the number of pathogens in the treated water can be estimated. Chapters 7 & 8 describes how this is combined with consumption and distribution data to calculate the health risk in a QMRA. Treatment performance for the twelve CTS are presented in Section 4.3.2 as case studies. Examples from these case studies are used in Sections 4.4, 4.5 and 4.6 to illustrate the methodology.

Different pathogens pose varying challenges to water treatment. Bacteria are less well removed by filtration than other micro-organisms but are readily inactivated by disinfection. Protozoa are relatively insensitive to chlorine disinfection but better removed by filtration than bacteria and readily inactivated by UV, whereas viruses are somewhere in between for both types of processes. By assessing the reduction of the selected suite of pathogens and indicators, the challenges posed by other (unknown) pathogens is probably covered. The treatment assessment does not differentiate between different species of micro-organisms unless specifically discussed in Section 4.2.

In order to assess a treatment, specific data that provides information on the performance of that system is needed. Frequent monitoring of pathogens at different stages of treatment would provide the ideal dataset to determine the barrier efficiency,

but pathogen concentrations are often too low for ready diction and especially monitoring after the first treatment steps results in mostly non-detects. Various indicators, surrogates or process conditions that can be used to estimate pathogen treatment efficacy were described in Section 4.2. Many water utilities already collect some of this data either for compliance to drinking water laws (indicator sampling) or for operational purposes (residual chlorine measurement to control chlorine dosing). The available data was therefore compiled from the Microrisk CTSs to provide treatment performance data.

Reduction at a single site varies in time and this variation was accounted for by collecting data over a period of several years. In order to do risk calculations these variations were expressed as Probability Density Function (PDF) (see Chapter 7). The quality of the data varied between the CTSs, and between treatment processes within a single CTS. This chapter describes how reduction by each treatment barrier can be estimated based on commonly available data, as illustrated in Figure 4.4.

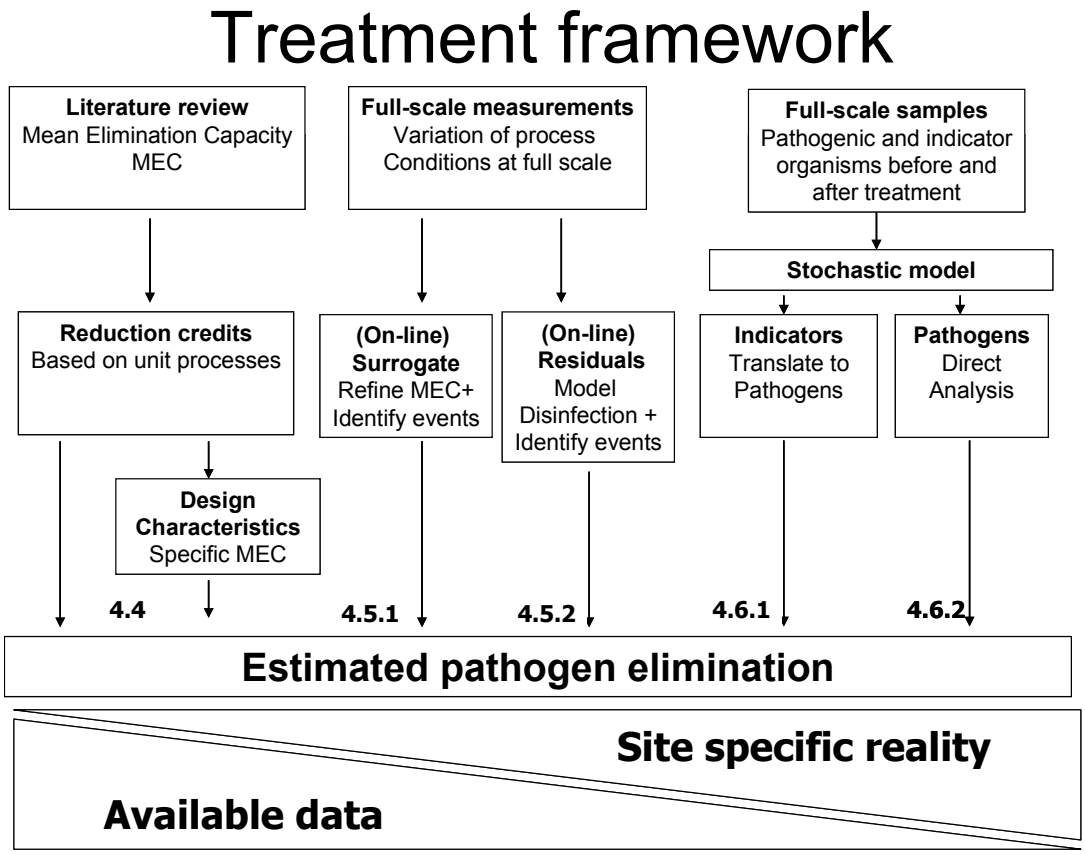


Figure 4.4 MicroRisk treatment performance data framework. The numbers correspond to the paragraphs that discuss the use of this type of data

Other types of data that can be used for treatment performance assessment include results from (site specific) pilot test and failure reports. Such tests are more applicable to the local situation than general literature values. Other data like operational diaries or

failure reports can provide information about the frequency and duration of events in treatment.

4.3.2 Treatments assessed in the MicroRisk project

Within the MicroRisk project the twelve systems (as in Chapters 3, 5, 7 and 8) were used. These treatment schemes are briefly outlined in Table 4.12.

Table 4.12 Treatment schemes of the catchment-to-tap systems (CTSs) in the MicroRisk project

CTS	Treatment scheme [1]
1	PreO_3 (Cl_2 in summer) - Coa - Sed - RF - O_3 - GAC - super Cl_2-deCl_2
2	Coa - Sed - RF - infiltration - reservoir - RF - O_3 - GAC - SSF
3	PreO_3 - Coa - Sed - RF - O_3 - GAC - Cl_2 (ClO_2 until July 2003)
4	Coa - Sed - O_3 -GAC - ClO_2
5	Coa - Sed - O_3 or Cl_2 - GAC - Cl_2
6	Reservoir - Cl_2 (summer) - Coa - Sed - GAC - Cl_2
7	Bank filtration - O_3 - GAC
8	Pre-Cl_2-Coa - DAF - RF - GAC - Cl_2
9	RSF - O_3 - GAC - SSF
10	PreCl_2 - Coa - Sed - RF - O_3 - Cl_2
11	DF - RF - Cl_2 + ClO_2
12	RF - Cl_2

[1]Coa = coagulation, Sed = sedimentation, DAF = dissolved air flotation, RF = rapid (granular) filtration, DF = direct filtration, O_3 = ozonation, GAC = granulated activated carbon filtration, Cl_2 = chlorination, ClO_2 = chlorine dioxide dosing, SSF = slow sand filtration.

Collected data included pathogen and indicator monitoring results, measured surrogates like turbidity and particle counting, process conditions like temperature, pH, DOC concentrations, disinfectant residual measurement and operational data like flows, disinfectant dose failure reports and supporting pilot information as exemplified here.

Pathogen monitoring

Most systems only monitored pathogens in the source water (Chapter 3). CTS 1 monitored Enterovirus, *Campylobacter*, *E. coli* O157, *Giardia* and *Cryptosporidium* monthly after each treatment step for a year. The majority of these samples resulted in non-detects (see Section 4.9). CTS 2 and 9 provided approximately 25 results of *Campylobacter* monitoring both after filtration and ozonation with a certain percentage of positives due to the large analysed volume. CTS 10 monitored *Giardia* both in raw and treated water resulting in approximately 50% positive samples (see analysis in Chapter 7).

Indicator monitoring

All systems provided their results of indicator motoring. Raw water was typically monitored daily to weekly and treated water was monitored daily for the presence of *E. coli* and/or different types of coliforms. Some sites also monitored *Enterococci*/ *faecal streptococci* or other faecal indicators. Spores of sulphite-reducing *clostridia* (SSRC) were monitored daily to monthly in treated water at several sites (1, 2, 3, 4, 9, 10, 12) but only four had corresponding source water data. Some large treatment plants also monitored *E. coli* at several points during the process (CTS 1, 2, 5, 6, 9) and sometimes also SSRC (CTS 1, 2, 9).

Surrogates
Turbidity of the treated water was monitored at most sites although the frequency varies between once per week to every twenty seconds. Most CTSs also monitored turbidity after filtration, generally on the combined flow of all filters. Some sites reported that turbidity was monitored on-line but data were unavailable and only used for alarm purposes. Particle counting was only applied over a longer period at CTS 5 and 6.

Process conditions
General process conditions like temperature and pH were provided for all CTSs in sufficient numbers to describe variation over the year. Specific process conditions like disinfectant residuals showed some shortcomings. Some CTSs (2, 9) only measure ozone residual once per week. Since this process is very variable, such a sampling strategy will miss short periods of poor removal. The on-line ozone measurements at these sites were deemed unreliable, and were placed at a point where generally no ozone residual was present. Others were unsure of the point of residual measurement in a contactor or the contactor volume.

Operational data
Most operational data provided was from on-line measurements of water treatment equipment, like dosing pump flows. This helped to estimate frequency of equipment failure, although failure could not always be separated from intentional shutdown of equipment for maintenance. This required diary reports or failure reports, which most CTSs couldn't provide in computer format. Only CTS 5 provided detailed diary and failure reports which are analysed in Chapter 8.
Collection of flow data was essential for data analysis and process modelling, since it provided information whether a line was actually in operation during a measurement. CTS 3 provided on-line disinfectant residual data which showed regular periods of low residual. Presumably this occurred during periodic shut-down of the plant (daily). However, no data was available to indicate when the plant was actually producing water.

Conclusions data collection
The data collection made clear that most data was not collected with the aim to assess treatment efficacy. As a result, no 'ideal' CTS was identified, which had high quality data for all treatment steps. In general the large surface water treatment plants monitor more rigorously than the smaller plants. Still, a lot of potentially useful data could not be used because some information, which could easily be recorded or monitored, was missing. The use and shortcomings of the data is illustrated in the examples in this chapter. Based on the first iteration of the treatment assessment of the CTSs, general guidelines on additional data collection are given in Section 4.10 to address the shortcomings.

4.4 POINT ESTIMATES

4.4.1 Point estimates

The first step in assessing treatment efficacy was the collection of relevant literature and local data. Basic information about the local treatment design is required; the unit processes, how these are connected and their design specifications. Relevant information like design Ct values for chemical disinfection processes or UV dose are generally available for a given treatment plant. For physical removal processes like coagulation-sedimentation and filtration basic information like the application of coagulants, filtration rates and filter material and size provide additional information to better estimate the efficacy.

The following example illustrates a basic assessment from unit processes for CTS 11. Treatment consisted of direct filtration (in-line coagulant dosing before the rapid sand filter), a second rapid sand filtration step and disinfection with chlorine and chlorine-dioxide. Filter specifications like bed height, filtration rate, filter material and size at CTS 11 corresponded to the studied filters described in Section 4.2, so the removal efficacies presented in Tables 4.2 and 4.6 were used here. At CTS 11 the Ct for chlorine and chlorine dioxide were 7.5 and 4.5 $mg.min.l^{-1}$ respectively based on design flow, reservoir size and average disinfectant residual reported by the water company. Inactivation by disinfection was calculated with the single CSTR model (Section 4.2.5). Table 4.13 illustrates the estimated pathogen reductions for the unit processes based on the MEC values reported in Section 4.2.

Table 4.13 Log inactivation at CTS 11 based on unit processes

	Direct filtration	**Rapid filtration**	**Disinfection Chlorine**	**Disinfection Chlorine Dioxide**	**Total treatment**
Virus	0.9	0.9	1.5	0.6	3.8
Bacteria	1.4	0.6	1.7	1.9	5.6
Cryptosporidium	3.0	2.0	0.0	0.0	5
Giardia	2.5	1.7	0.1	0.4	4.7

From Table 4.13 it is clear that direct filtration performs better than rapid filtration only. Filtration is an important barrier for the protozoa and disinfection for the bacteria and viruses. The inactivation reported in Table 4.13 is a mean estimate of the potential inactivation at full-scale. Since conditions can vary substantially, a more sophisticated estimate of inactivation is needed and is discussed in the next paragraph.

4.4.2 Uncertainty of point estimates

The basic assessment described in Table 4.13 for pathogen removals at CTS 11 lack any interpretation of the expected variations described in Section 4.2. Table 4.14 shows the best estimate again, but now with the minimum and maximum removal reported for the processes at CTS 11. A worst case assumption is that minimum disinfection could be 0 log due to periodic under-dosing of disinfectant. Maximum possible inactivation is calculated assuming ideal plug flow (Ct calculation), highest estimate of disinfectant concentration and temperature. Inactivation is unlikely to exceed this maximum. Thus the range of all possible inactivation outcomes in Table 4.14 was established for CTS 11.

Table 4.14 Log reduction at CTS 11 based on unit processes, including maximum likely uncertainty

	Viruses min-MEC-max	**Bacteria min-MEC-max**	***Cryptosporidium* min-MEC-max**	***Giardia* min-MEC-max**
Direct Filtration	0.1-0.9-3.9	0.8-1.4-3.3	0.8-3-5.4	0.8-2.5-3.9
Rapid Filtration	0.1-0.8-3.8	0.1-0.6-1.5	0.0-2-3.1	0.0-1.7-6.5
Disinfection Chlorine	0-1.5- >7	0-1.7- >7	0-0-0	0-0.1-0.3
Disinfection Chlorine-Dioxide	0-0.6-3.5	0-1.9- >7	0-0-0	0-0.4-1
Minimal total reduction	0.2	0.9	0.8	0.8
Best estimate reduction	**3.8**	**5.6**	**5.0**	**4.7**
Maximum total reduction	>18.2	>18.8	8.6	11.7

Table 4.14 and the case study in Table 4.23 illustrate that without site-specific verification of the performance, the range in possible treatment performance is large, and needs to be narrowed down to better describe what may actually be occuring.

4.4.3 Modelling variation in point estimates

The minimal estimate of reduction by the total treatment in Table 4.14 was a 'worst case' assumption in which every treatment step performs at its worst. Most systems are expected to have some treatment steps perform better. The way variation in the data described in Table 4.14 has been modelled in the MicroRisk project is to describe each treatment step by a triangular PDF with the parameters minimum, MEC and maximum to represent 'expert knowledge' as discussed in Chapter 7. The best estimate is now not necessarily the sum of the best estimates of each unit processes; rather, each triangular distribution provides insight into which step and pathogen has the most uncertainty/variability associated with it (see Figure 4.5 for direct filtration).

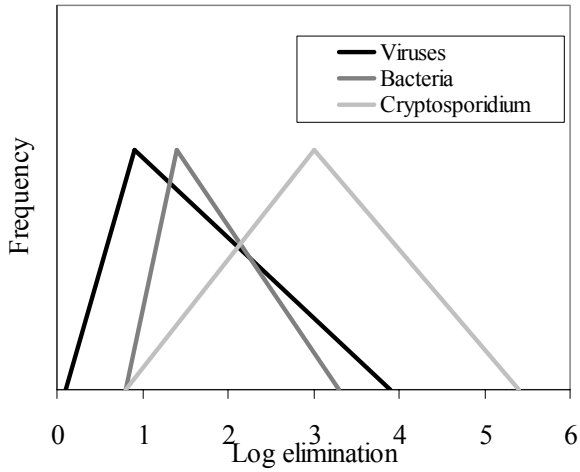


Figure 4.5 Triangular PDF (Probability Density Function) of virus, bacteria and *Cryptosporidium* removal by direct filtration

The triangular distribution is used as a first pass representation of process variability and uncertainty.

BOX II Over-all treatment performance
Pathogen reduction by treatment varies over time. The total effect of treatment over a given period is expressed by the 'over-all' log reduction. The average log removal does not represent the actual fraction of pathogens that pass treatment. The average fraction over the period is therefore used to calculate over-all reduction (over-all reduction = -log (average fraction)). This is illustrated by the following example for a period of three days. The average log reduction was 2.3 but the over-all reduction was 1.5 log, so the number of pathogens (load) after treatment is 1.5 log lower than in source

This is especially useful when assessing several treatment steps in series to provide an estimate of the combined reduction. When one process is performing poorly, the other processes are still providing reduction. By combining the triangular distributions of the individual processes in a Monte Carlo analysis, the total reduction with its variability can be simulated. The result is a description of how the total treatment efficacy is likely to vary. This is explained in more detail in Chapter 7. The overall reduction represents the pathogen removal over the assessed period (see Box II).

4.4.4 Results for MicroRisk Systems

Initial treatment performance assessments for all systems (CTS) in MicroRisk are summarised in Tables 4.15 to 4.18. The first observation for all systems and organisms is that the minimum removal is generally low, whereas the potential (maximum) removal can be extremely high. This shows that although point estimates are often used to estimate removal, their uncertainty generally ranges between zero and >10 log removal. This large uncertainty is partly caused by lack of information on system operation, especially disinfection processes can have a large impact.
The point estimate of the reduction is generally slightly higher than the estimated over-all reduction based on triangular distributions. The 2.5% percentile of the estimated variation is generally substantially higher than the minimum estimate. The extremely low minimum estimates are unlikely to occur due to the multiple barriers in the treatment systems.

Table 4.15 Model process schemes used for initial assessment of over-all virus reduction by treatment of the catchment-to-tap systems (CTS) in the MicroRisk project

CTS	Viruses Treatment model	Point estimates Best estimate	Min	Max	Triangular distributions Best estimate	2.5%	97.5%
1	O_3 - Coa/Sed - RF - O_3 - GAC - Cl_2 - Cl_2	12.2	1.4	> 34	8.1	10.1	22.0
2	RF - O_3 - GAC – SSF	4.7	0.9	> 15.5	4.7	3.8	11.0
3	O_3 - Conv - O_3 - GAC - Cl_2	6.5	1.4	> 22.1	6.4	5.8	14.8
4	Coa/Sed - O_3 - GAC – ClO_2	4.3	0.4	> 18.8	4.7	3.9	12.9
5	Coa/Sed - O_3 - GAC - Cl_2	2.2	0.4	5	1.9	1.0	4.3
6	Coa/Sed - GAC - Cl_2	2.2	0.4	5	1.9	1.1	4.3
7	SSF - O_3 – GAC	3.8	0.8	> 11.7	3.6	2.7	8.9
8	Cl_2 - Conv - GAC - Cl_2	6.9	1.4	> 13	6.5	6.1	14.6
9	RSF - O_3 - GAC – SSF	4.5	0.9	> 15.5	4.6	3.7	10.8
10	Cl_2 – Conv - O_3 - Cl_2	6.9	1.2	> 26.3	6.0	5.6	14.5
11	DF - RF - Cl_2 - ClO_2	3.8	0.2	> 18.2	5.2	4.6	14.0

12	RF - Cl_2	2.2	0.1	>	10.8	2.5	1.5	8.0

Table 4.16 Model process schemes used for initial assessment of over-all reduction of bacteria by treatment of the catchment-to-tap systems (CTS) in the MicroRisk project

CTS	Bacteria Treatment model	Point estimates Best estimate	Min		Max	Triangular distributions Best estimate	2.5%	97.5%
1	O_3 - Coa/Sed - RF - O_3 - GAC - Cl_2 - Cl_2	15.1	1.9	>	34.3	10.2	11.7	23.2
2	RF - O_3 - GAC - SSF	7.3	2.2	>	16.2	6.3	5.5	12.0
3	O_3 - Conv - O_3 - GAC - Cl_2	9.0	1.9	>	27.3	8.0	8.1	18.4
4	Coa/Sed - O_3 - GAC – ClO_2	7.7	1.5	>	20.6	6.5	6.0	14.5
5	Coa/Sed - O_3 - GAC - Cl_2	2.9	1.5		6.6	3.1	2.3	5.4
6	Coa/Sed - GAC - Cl_2	2.9	1.5		6.6	3.2	2.3	5.5
7	SSF - O_3 – GAC	6.5	2.1	>	14.7	5.6	4.8	11.2
8	Cl_2 - Conv - GAC - Cl_2	7.1	1.9	>	13.3	7.2	6.7	15.1
9	RSF - O_3 - GAC – SSF	7.0	2.2	>	16.2	6.3	5.4	11.9
10	Cl_2 – Conv - O_3 - Cl_2	7.1	1.0	>	24.4	6.2	6.3	16.6
11	DF - RF - Cl_2 - ClO_2	5.6	0.9	>	18.8	5.6	4.9	13.2
12	RF - Cl_2	2.1	0.1	>	8.5	2.1	1.2	6.8

Table 4.17 Model process schemes used for the initial assessment of over-all *Cryptosporidium* reduction by the treatment of catchment-to tap-systems (CTS) in the MicroRisk project

CTS	*Cryptosporidium* Treatment model	Point estimates Best estimate	Min		Max	Triangular distributions Best estimate	2.5%	97.5%
1	O_3 - Coa/Sed - RF - O_3 - GAC - Cl_2 - Cl_2	4.8	2.1		10.8	4.8	3.9	8.1
2	RF - O_3 - GAC - SSF	6.9	1.0	>	11.4	4.3	3.6	9.0
3	O_3 - Conv - O_3 - GAC - Cl_2	4.2	2.1		7.7	4.0	3.1	6.4
4	Coa/Sed - O_3 - GAC – ClO_2	3.0	1.1	>	6.4	2.6	1.8	4.5
5	Coa/Sed - O_3 - GAC - Cl_2	2.8	1.1		4.9	2.4	1.6	4.2
6	Coa/Sed - GAC - Cl_2	2.8	1.1		4.9	2.4	1.6	4.2
7	SSF - O_3 – GAC	4.8	1.0	>	7.9	3.1	2.1	6.9
8	Cl_2 - Conv - GAC - Cl_2	4.1	2.1		6.6	3.6	2.7	5.9
9	RSF - O_3 - GAC – SSF	6.8	1.0	>	11.3	4.4	3.5	9.0
10	Cl_2 – Conv - O_3 - Cl_2	3.9	1.4		8.5	3.7	2.9	6.5
11	DF - RF - Cl_2 - ClO_2	5.0	0.8		8.6	4.0	3.1	7.7
12	RF - Cl_2	2.0	0.0		3.1	1.2	0.4	2.8

Table 4.18 Model process schemes used for the initial assessment of over-all *Giardia* reduction by the catchment-to-tap systems (CTS) in the MicroRisk project

CTS	*Giardia* Treatment model	Point estimates Best estimate	Min		Max	Triangular distributions Best estimate	2.5%	97.5%
1	O_3 - Coa/Sed - RF - O_3 - GAC - Cl_2 - Cl_2	9.6	2.5	>	31.2	9.1	9.4	20.1
2	RF - O_3 - GAC - SSF	7.8	1.6	>	22.8	7.0	6.5	15.6
3	O_3 - Conv - O_3 - GAC - Cl_2	6.4	2.5	>	17.5	7.4	6.6	15.9
4	Coa/Sed - O_3 - GAC – ClO_2	4.9	0.7	>	15.2	5.1	4.4	13.6
5	Coa/Sed - O_3 - GAC - Cl_2	3.3	0.7		6.2	2.6	1.7	5.1
6	Coa/Sed - GAC - Cl_2	3.3	0.7		6.2	2.6	1.7	5.1
7	SSF - O_3 – GAC	5.9	1.6		15.3	5.3	4.5	11.2
8	Cl_2 - Conv - GAC - Cl_2	6.6	2.5	>	15.4	7.1	6.5	15.6
9	RSF - O_3 - GAC – SSF	7.5	1.6	>	22.8	7.0	6.4	15.5
10	Cl_2 – Conv - O_3 - Cl_2	5.5	2.1	>	19.5	5.9	5.1	13.9
11	DF - RF - Cl_2 - ClO_2	4.7	0.8	>	11.7	5.1	4.3	12.5

12	RF - Cl_2	1.8	0.0	>	8.5	2.0	1.0	6.4

4.5 USE OF SURROGATES AND PROCESS MODELLING

4.5.1 Validating physical processes with measured surrogates

The large ranges in estimated log removals described in the previous section indicated the need to reduce the uncertainty at specific sites. Many sites apply (on-line) turbidity measurement to monitor filter performance. As stated in Section 4.2, a universal relation between turbidity and pathogen reduction does not exist, yet a good working filter is able to produce water with a turbidity constantly below 0.1 NTU.

Site 10 provides an example where turbidity was measured before and after conventional treatment (coagulation-sedimentation and filtration). The filtrate turbidity varies between 0.1 and 1 NTU, indicating that filtration does not work effectively. The triangular PDF for a poor performing filter applied the MEC as maximum and the mean of the minimum and the MEC as most likely removal (see Figure 4.6).

Turbidity at site 11 is recorded daily before and after direct filtration. Turbidity is consistently reduced from >1 NTU to <0.06 NTU thus verifying that the filter is working well. The triangular PDF for a well performing filter applied the MEC as minimum and the mean of the MEC and the maximum as most likely removal. Figure 4.6 illustrates how the triangular PDF is adapted for these CTSs based on the recorded turbidity.

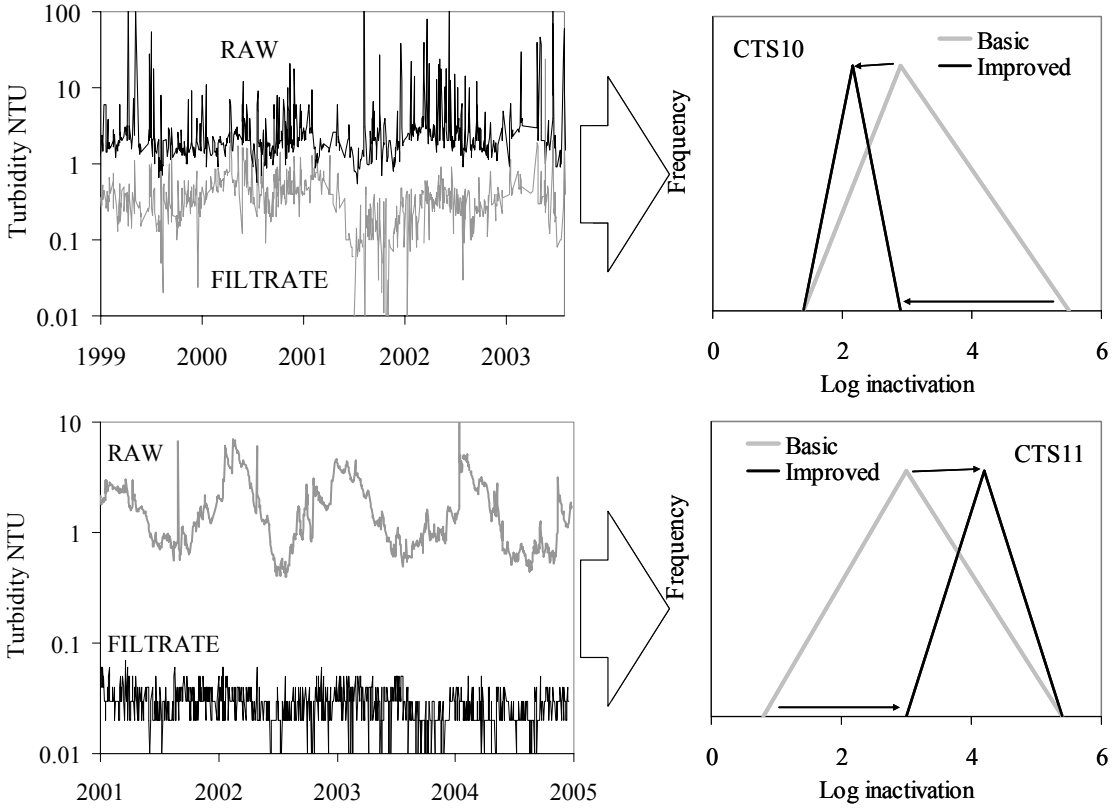


Figure 4.6 Turbidity before and after conventional treatment (CTS 10) and direct filtration (CTS 11) is used to refine the triangular PDF from point estimates to use of turbidity as a surrogate for "good" or "poor" performing filters

Particle removal based on particle counts before and after treatment has a better relationship with micro-organism removal than turbidity removal [Hijnen *et al.*, 2005a]. Changing process monitoring from turbidity to particle counting should lead to a more accurate assessment of treatment efficacy.

4.5.2 Modelling disinfection processes

Initial point estimates for disinfection were modelled based on general operational information, which generally resulted in a very wide range of possible inactivation. By including all available information about the variation of the conditions during disinfection, uncertainty can be reduced. Disinfectant concentration monitoring at the outlet of a contactor, or at several points when the contactor is baffled verifies the presence of disinfectant which is the primary requirement for disinfection. Continuous monitoring (recorded every 1 to15 minutes) is preferred to account for short failure of dosing which results in no inactivation. Flow monitoring verifies the contact time in the contactor and should be recorded at least hourly, since flow variations are generally gradual., Daily recording of temperature is sufficient since it varies gradually.
The effect of variation in disinfection conditions can be verified by modelling based on these input data (see 4.2.5). Figure 4.7 illustrates the result of this approach to better

model *E. coli* O157 inactivation at site 10. Flow variations were not recorded, so design flow was applied in the calculations. The clear water reservoirs (no baffles) were used as contactors, therefore a single CSTR model was applied assuming 30% filling of the reservoir resulting in a conservative estimate of the inactivation.

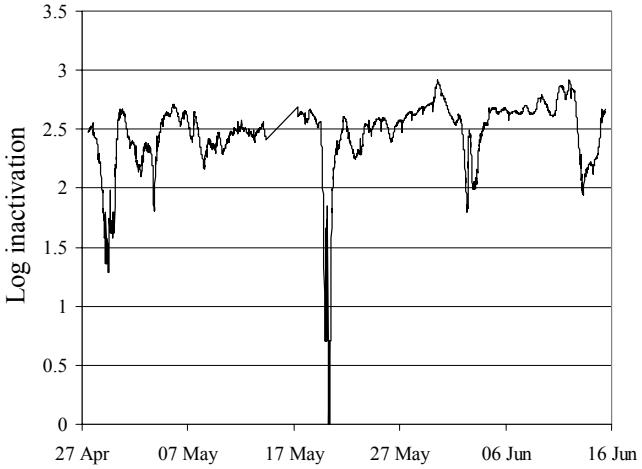


Figure 4.7 Modelled variation of *E. coli* O157 inactivation by chlorination at CTS 10

The estimated mean inactivation of *E. coli* was approximately 2.5 log. On four occasions, however, the chlorine concentration was reduced for a period of two days (not specifically weekends), resulting in 1.5 to 2 log inactivation. The event on 20/5/04 lasted for 12 hours with practically complete failure of disinfection (no chlorine residual). Since no records of flow, operational diary or failure report was available for this site, it was assumed that the treatment was operating and supplying water during this 'event'. As a result the over-all inactivation of *E. coli* by chlorine disinfection in this period was reduced from 2.5 to 2.1 log by this one event. The initial point estimates for disinfection at CTS 10 resulted in a triangular PDF with parameters 0, 3 and 7. Modelling disinfection verified that inactivation appears to vary between 2 and 3 log 95% of the time and also showed that some events of less inactivation occurred. A beta distribution was fitted to the calculated inactivation (expressed as fraction, see Chapter 7) to provide more information in the QMRA. The same approach was applied to determine the PDF of virus and *Giardia* inactivation. For *Cryptosporidium* no further assessment was necessary as the point estimates described previous had showed that even under ideal conditions at CTS 10, no inactivation of *Cryptosporidium* by chlorine is expected.

The largest uncertainty in this more detailed modelling was in the hydraulic characteristics of the clear water reservoirs for different fill levels. If a 2 or 10 CSTRs in series hydraulic model was verified by tracer tests, calculated inactivation of *E. coli* would be 4 or >7 log respectively. Additional monitoring of flow and the variation of the reservoir levels could improve the assessment of inactivation by chlorine disinfection at CTS 10. Temperature should be monitored daily instead of monthly.

Disinfection assessments at other CTSs indicated some general shortcomings of available data to model disinfection. Only CTS 2 provided results of tracer tests of the ozone contactors. Despite baffling, the hydraulics were characterised by only 4 CSTRs. For the other CTSs a single CSTR model was used for modelling. Some sites only monitored disinfectant dose so disinfectant decay was not included in the models, thus conservative estimates of disinfectant concentrations were applied for modelling disinfection.

Further uncertainties arise from the limited knowledge of inactivation kinetics for (environmental) pathogens (Section 4.2.5). For CTS 2 special pilot tests (see 4.7.1) were performed to determine kinetics of *Campylobacter* inactivation by ozone, since information was not available in the literature.

The measurement of residuals at low concentrations can cause analytical errors due to the cross sensitivity to disinfection by-products [USEPA, 1999] and can result in an over-estimation of the achieved inactivation.

Most CTSs operate their disinfection processes at a fixed set-point for disinfectant residual. Thus seasonal variation of inactivation due to temperature fluctuations and hourly variation due to flow fluctuation occured. This would generally be expected to result in reduced inactivation at lower winter temperatures, when the source water contains the highest number of pathogens (Chapter 3). This provides opportunities for improved operation by adapting operation to changing conditions.

4.6 INDICATOR AND PATHOGEN MONITORING

4.6.1 Indicator monitoring

Compliance with drinking water legislation is partly based on monitoring for indicators like *E. coli* and sulphite-reducing clostridia. In general, most samples were non-detects, which widens the uncertainty in estimating their actual values. Conclusions from mainly non-detect samples results in an estimate of an “upper limit”, for example the average concentration in treated water is below X organisms per litre with a 95% confidence. The uncertainties when translated to pathogen concentrations are very large (see 5.4.2).

If indicators are also monitored in the raw water, their reduction by treatment can be assessed. *E. coli* removal can than be related to removal of bacterial pathogens with similar characteristics, which supports their expected removal but the non-detects again increase uncertainties.
When indicators are present and monitored before and after a single process, its efficacy can be assessed and described by a PDF. The number of positive samples may decrease in the subsequent treatment steps and will be of less value for the QMRA unless a reasonable percentage of the samples remain positive (say 50%).

When possible, 'paired' measurements before and after treatment on the same day should be used to calculate log removals. Differences in measured concentrations will however occur due to the variations in time and "space" (see Chapter 7). Indicators are generally monitored 12 to 52 times per year. It is unlikely that rare events would be captured in such a dataset. Despite these shortcomings, the removal of indicators can support the risk assessment and reduce the uncertainty on the overall (baseline) reduction of pathogens, i.e. for non-event periods. Further advise on appropriate statistical methods are described in Chapter 7.

Section 4.9.3.1 provides an example of how *E. coli* monitoring before and after pre-ozonation at CTS 1 was used to determine the PDF for inactivation of the bacterial pathogens *E. coli* O157 and *Campylobacter*.

The following example illustrates how analysing sufficient sampling volume can be used to refine the assessment of log removals. At CTS 2 *E. coli* is monitored in the raw water, filtered water and ozonated water. In raw water 100 mL samples were sufficient to monitor *E. coli* concentrations, in the range of 10-10,000 $CFU.L^{-1}$, while one-litre samples after filtration gave some 85% positives at concentrations between 1 and 100 $CFU.L^{-1}$. A PDF of bacterial removal was based on the 400 data pairs with the method described in Section 4.9.3.1.

After ozonation only 3% of the 1-L samples were positive and thus provided limited information on variation of *E. coli* inactivation. The inactivation was "larger than 1.2 log units" although several log units of inactivation by ozonation were expected. By increasing the sample volume to 25-100 L after ozonation, 44% were positive for *E. coli*. The concentrations ranged from 0.02 to 0.15 $CFU.L^{-1}$, indicating 2.2 to 3 log inactivation. In this case the 3% positive 1 L samples apparently represented the occasional positives of a low baseline concentration rather than events.

In these examples, *E. coli* reduction was used as an indicator of *E. coli* 0157 and *Campylobacter*. Alternative model micro-organisms are discussed in Section 4.2, and include the use of spores of sulphite-reducing clostridia or aerobic spores as indicators for protozoa reduction and phages for enteric viruses. Monitoring of these indicators is less common and their applicability needs to be considered based on the local situation before starting a monitoring programme.

Figure 4.8 shows that under certain conditions indicator organisms can be measured all the way through a multiple barrier treatment system such as CTS 1. In this case aerobic spores are present in very high numbers in the raw water (10,000 $CFU.100\ mL^{-1}$). Since aerobic spores have a high ozone resistance, both pre-ozonation and intermediate-ozonation have little effect on their concentrations. Sedimentation, rapid gravity filtration and granular activated carbon filtration will more effectively remove spores. The final disinfection by super-chlorination and de-chlorination achieves substantial reduction of aerobic spores. Remarkably 5 out of 15 samples of the raw water were negative for aerobic spores (<100 $cfu.100\ mL^{-1}$). This either indicates high variability of the aerobic spore concentration or problems at the laboratory.

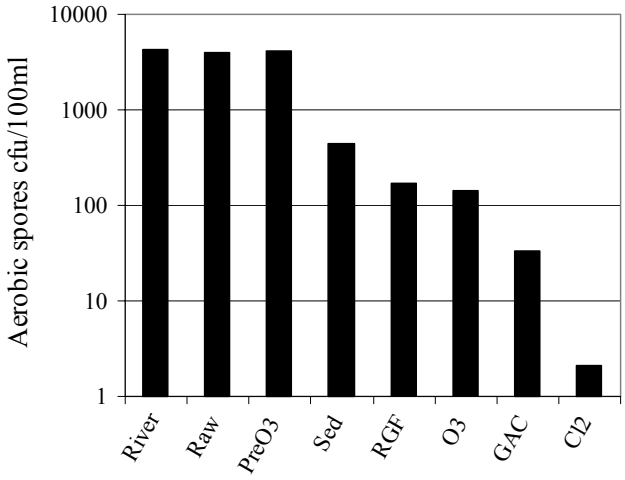


Figure 4.8 Average aerobic spore concentrations at a multiple barrier treatment plant (CTS 1)

Other organisms monitored at CTS 1 showed that removal of *E. coli,* Enterococci and Total Coliforms mainly occurred at the pre-oxidation stage, while Aerobic Spores, Sulphite-reducing clostridia and *Staphylococcus aureus* were mainly removed during the coagulation-clarification step.

4.6.2 Pathogen monitoring

The most direct determination of the pathogen risks and treatment barrier efficacy is by sampling for pathogens at full-scale, but is costly and long-term investigation with frequent sampling is seldom feasible.

At CTS 1, *Campylobacter*, *E. coli* O157, Enterovirus, *Giardia*, *Cryptosporidium* and *Pseudomonas aeruginosa* were monitored monthly in the raw water and after each treatment step for one year. However, only *Pseudomonas* and Enterovirus were detected in the raw water. The former was detected three times after pre-ozonation and once after sedimentation, while infectious Enterovirus was reduced strongly by pre-ozonation but still detected twice after inter-ozonation. Since many pathogens were absent in the highly polluted source water, the monitoring within treatment provided high uncertainty on the efficacy of pathogen removal at CTS 1.

At CTS 2 *Campylobacter* was monitored in the raw water, after filtration and after ozonation. All raw water samples were positive, in concentrations of 10 to 1,100 $MPN.L^{-1}$ (mean 211 $MPN.L^{-1}$). The corresponding values after filtration were 0.4 and 110 $MPN.L^{-1}$ (mean 11 $MPN.L^{-1}$) and after ozonation 39% were positive, between 0.04 and 0.4 $MPN.L^{-1}$ (mean 0.05 $MPN.L^{-1}$). Thus, filtration provided 1.3 log removal, which confirms 'good' performance of the filter compared to the initial estimate of 0.1 to 1.5 log removal and a MEC of 0.6. Ozonation provided 2.3 log inactivation which is in the same range as the initial conservative estimated inactivation of 2.6 (as part of calculations that resulted in table 4.16).

When samples before and after filtration were paired, the inactivation seemed to vary between 0 and 2.5. This large observed variation may be caused by the MPN method used (three dilutions in tripled), which resulted in a most probable number with a 95% confidence interval of 0.4 to 1 log depending on the number of positive tubes. When pairing MPN numbers before and after filtration the calculated reduction had a confidence interval of approximately 1.6 log. Advanced statistical methods as described in Chapter 7 are required to estimate the PDF parameters directly from the presence-absence results of the MPN method.

4.7 COLLECTING ADDITIONAL INFORMATION

4.7.1 Pilot-scale experiments

Pilot-scale experiments may supply additional information where the monitoring or operational full-scale installations give insufficient information about treatment performance for different organisms. This may be due to the absence or low numbers of organisms in the final stages of treatment or limitations of monitoring to assess the reduction efficiency for certain groups, like viruses. Pilot experiments can help determine the applicable (local) reduction values from the wide range found in the literature When additional treatment is considered on-site pilot testing will provide site specific performance indication of the process under consideration.

Pilot challenge tests under full-scale conditions: SSF
Slow Sand Filtration (SSF) was one of the major barriers against pathogens at CTS 2, and data indicated that *Campylobacter* reduction might be insufficient to reach health based targets. As described in Section 4.2 SSF can reduce bacteria by 1.2 to 4.8 log. This could not be verified by indicator monitoring at full-scale since most samples before and all samples after SSF were negative for indicator organisms. Long term pilot tests were conducted at the treatment site where the pilot filters were run parallel to the full-scale filters under the same conditions but with additional indicator micro-organisms dosed directly before the SSF pilot filter. The results indicated that over 2 log removal occurred shortly after scraping off the “schmutzdecke” up to 4 log removal for a fully ripened filter. At CTS 2 only 2 out of 26 filters are scraped at the same time, and filter to waste is applied for several weeks. Hence, the point estimate for the SSF treatment step of 3 logs removal was applied [Hijnen *et al.,* 2004; Dullemont *et al.*, 2006].

Bench-scale ozone disinfection tests with natural water
At CTS 2 *E. coli* was occasionally detected after ozonation, which overall indicated some 2.2 log inactivation. Further, *E. coli* was regarded as a potential index for *Campylobacter* inactivation by ozonation. This was suggested as the literature provided no references on *Campylobacter* inactivation by ozone and only four publications on *E. coli* inactivation were identified. Bench-scale ozone inactivation tests of laboratory-cultured and environmental *E. coli* and *Campylobacter* were performed using water from CTS 2. Results showed that inactivation of both organisms was similar and that

environmental *E. coli* and *Campylobacter* were more resistant to ozone than cultured organisms. The resulting inactivation kinetic parameters are presented in Table 4.9 and were also used for modelling ozone disinfection of bacteria at other CTSs [Smeets *et al.*, 2005].

Pilot challenge tests under full-scale conditions: Conventional treatment
The reduction efficiency of enteric viruses by chemical flocculation in drinking water treatment was assessed in a pilot-plant under similar conditions to full-scale. The relative reductions of index bacteriophages φX174 and MS-2 by different sub-treatment steps was assessed. The phages were added either before the addition of chemicals or in the first flocculation chamber and the relative cumulative recovery was measured at subsequent steps. Results are summarized in Figure 4.9, but in general φX174 was not reduced as well as MS-2 and thus φX174 was a more conservative indicator of the barrier function and the preferred model organism (surrogate) for human enteric viruses. It is evident in Figure 4.9 that the chemical addition alone provided more than 1 log reduction, and flocculation and sedimentation/filtration each an additional log reduction. The total reduction by conventional treatment was in the range of 3.8 logs. The finding that chemical addition can may lead to substantial virus inactivation needs to be demonstrated with human enteric viruses, as phages are known to be more sensitive to some chemical effects, such as pH shock, and factors other than the current Derjaguin-Landau-Verwey-Overbeek (DLVO) model of virus attachment are important and even differ between MS2 and φX174 [Song *et al.*, 2005].

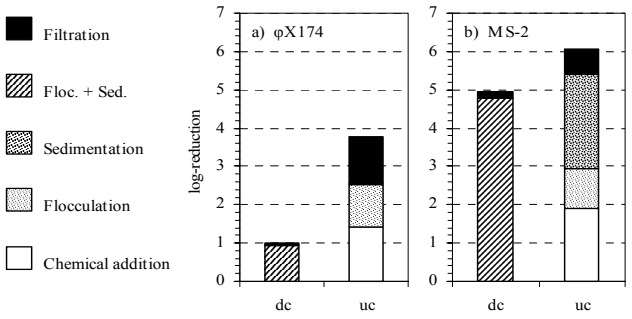


Figure 4.9 Log-reduction of a) φX174 and b) MS-2 bacteriophages by conventional treatment.
dc= downstream case, bacteriophage addition into first flocculation chamber.
uc=upstream case, bacteriophage addition upstream of chemical addition, differentiated between floccuation/sedimentation**.**

Occurrence of retardation in conventional treatment
Retardation during conventional processes is suspected. A peak concentration of pathogens appear to be removed substantially when comparing samples shortly after spiking the source water. However, the removed pathogens are slowly released over time, so a substantial part of the total pathogen load does pass treatment. This was tested by spiking the water with both phage surrogates and a salt solution. The passage of the phage spike and the conductivity increase was followed through the system after sedimentation (Figure 4.10a) and filtration (Figure 4.10b). The peak of both

bacteriophages coincides with the conductivity peak which showed that no apparent retardation through sedimentation and filtration occurred (Figure 4.10) although phages φX174 were released in low concentrations from the filter after 12h of operation [Heinicke *et al*., 2004]. This study supports that measurements before and after treatment can be paired when sufficient hydraulic residence time is accounted for.

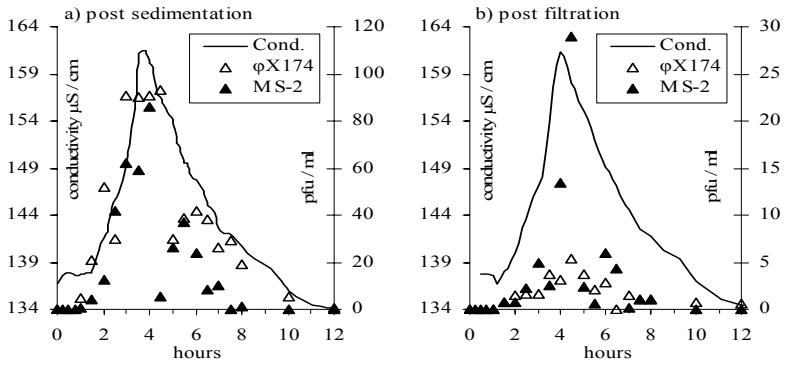


Figure 4.10 Peaks of conductivity and bacteriophages in the upstream case (uc). A) post sedimentation and B) post filtration. Note difference in scale on secondary y-axis.

4.7.2 Full-scale indicator spiking

Naturally occurring pathogens in source waters can be so low that removal at full-scale cannot be determined from ordinary sampling. In specific cases it may be possible to challenge an operational full-scale plant with model micro-organisms. The model micro-organism needs to be safe for the consumers and should not lead to a deterioration of water quality. At CTS 8 sufficient reduction of *Cryptosporidium* is essential, and performance of the full-scale treatment needed to be verified. An underlying difficulty in simply measuring naturally occurring *Cryptosporidium* densities before and after DAF (or some other) treatment to determine the treatment efficacy is that oocysts typically exist in low numbers that are difficult to detect. Dosing full-scale plants with oocysts to attain high enough densities to obtain detects is simply not an option due to associated health concerns. Chung *et al.* [2004] demonstrated that Baker's yeast (*Saccharomyces cerevisiae*) is a good model for *Cryptosporidium* removal by water treatment processes and is a substantially safer organism to work with than *Cryptosporidium*. The functioning full-scale water treatment plant in CTS 8 was challenged with *S. cerevisiae* to gain an initial assessment of the *Cryptosporidium* removal capabilities of the DAF and rapid sand filtration (RSF) water treatment plant.

Two out of three parallel DAF units studied showed a mean *S. cerevisiae* removal of 1.5 log, whereas the third line only provided 1 log removal. Combined removal by the three parallel DAF units was therefore 1.3 log. The lower saturated air flow to one unit was considered as the cause for reduced removal.

RSF provided 2.0, 2.3 and 2.6 log removals for three parallel filters respectively, providing a combined removal of 2.2 log. After six hours of operation the removal by

one filter was reduced from 2.0 to 1.4 log for approximately 10 minutes, while the other filters maintained their removal capacity. The combined removal was reduced to 1.8 log removal during this event for which no clear cause was found.

The effect of the ripening period after backwashing was also studied at one filter at CTS 8. The mean removal during the first hour after the filter was brought back on-line (2.4-$\log_{10}$) was not significantly different (5% level) to that observed during other periods. Although no filter-ripening period was observed in the microbiological data, concurrent particle count data collected during the trial did display a definite 'ripening-period' involving *Cryptosporidium* and *S. cerevisiae* sized particles. Backwash water recirculation resulted in increased numbers before DAF treatment, but had little effect on the removal efficiency of the treatment.

When using these results in QMRA to assess *Cryptosporidium* risks, the uncertainty concerning how closely *S. cerevisiae* model naturally-occurring oocysts needs to be accounted for. Regardless, such work does provide the opportunity to identify weaknesses in a system, such as poorly performing components of the treatment system, or the magnitude of inherent fluctuations in treatment performance. Combination with surrogate measurements such as particle counting demonstrates potential limitations of such surrogates for quantifying treatment efficacy for pathogen removal. It also shows which issues are irrelevant to microbiological risk, such as backwash water recirculation at this site. Such details may otherwise go unnoticed when comparing pre- and post-treatment water quality samples of pathogens or model micro-organisms that do not exist in large enough quantities to be detected in numbers sufficient to ensure relatively accurate representation of actual conditions [Signor *et al.*, submitted].

4.7.3 Failure reports

Although short periods of treatment failure can have a large impact on average risk of infection, water companies generally do not record failure events in treatment systematically. A study of collected treatment failure reports aimed to examine the impact of incidents on the annual risk of infection [Westrell *et al.*, 2003]. It accounted for the type, frequency and magnitude of failures in treatment and distribution in a microbial risk assessment that included *Cryptosporidium*, rotavirus and *Campylobacter jejuni*. The main risk incidents in water treatment were associated with sub-optimal particle removal or malfunction in disinfection. The majority of the potential annual infections were likely to be due to the pathogens passing treatment, due to its variability, and not due to the actual failures thus adding to the background endemic rate. However, the failures represent short periods of malfunctioning and may potentially be linked with outbreaks as well as to exposure of sensitive sub-groups of the population. Among the modelled pathogens, viruses appear to pose the largest risk of infection. The simulated total number of annual infections was within the equivalent range estimated from epidemiological data, accounting for the fraction of gastroenteritis attributable to tap water.

4.7.4 Direct monitoring of pathogens in drinking water: UK *Cryptosporidium* survey

Data from UK water companies on the outcome of their statutory monitoring for *Cryptosporidium* was obtained for 216 sites. For 8 sites data has also been provided on the numbers of *Cryptosporidium* in the source waters. The overall removal at these sites was calculated from the average concentration before and after sampling. Table 4.19 shows the treatment processes used at each site and the average overall *Cryptosporidium* oocyst removal. A variation of one log was observed among similar treatments.

Table 4.19 *Cryptosporidium* reduction based on UK regulatory monitoring

Treatment processes	**log removal**
A. Coagulation-Polyelectrolyte-Sedimentation-Filtration-GAC filtration-Chlorination	4.3
B. Coagulation-Polyelectrolyte-Sedimentation-Filtration-GACfiltration-Ozone-Chlorination	4.3
C. Coagulation-Polyelectrolyte-Sedimentation-Filtration-GAC filtration-Chlorination	3.4
D. Coagulation-Polyelectrolyte-Sedimentation-Filtration-GAC filtration-Chlorination	3.3
E. Impoundment-Coagulation-Polyelectrolyte-Sedimentation-Dissolved Air Flotation-Filtration-GAC filtration-Chlorination-	3.2
F. Impoundment-Coagulation-Polyelectrolyte-Sedimentation-Filtration-GAC filtration-Chlorination-	3.2
G. Coagulation-Sedimentation-GAC filtration-Ozone-Chlorination-	3.1
H. Coagulation-Sedimentation-Filtration-GAC filtration-Chlorination-	2.6

The typical minimum-mean-maximum treatment performance criteria for a *conventional treatment-GAC filtration-chlorination* plant is presented in Table 4.20. Compared to CTS 8 (Table 4.17), which applies a similar process scheme, the estimated 3.6 log over-all reduction of *Cryptosporidium* (95% confidence interval: 2.7 to 5.9 log) is similar.

Table 4.20 Minimum, Mean Elimination Capacity (MEC) and maximum log reduction of *Cryptosporidium* oocysts by processes and worst case, estimated and best case log reduction by a typical surface water treatment system

	***Cryptosporidium* min-MEC-max**
Conventional treatment (Coag.-Sed.-Filtr.)	1.4-3.2-5.5
GAC filtration	0.7-0.9-1.1
Disinfection Cl_2	0-0-0
Estimated total log reduction	**2.1-4.1-6.6**

4.8 INFLUECE OF SYSTEM CONFIGURATIONS

4.8.1 Assessment of combined processes

The application of multiple barriers in treatment provides process performance stability, as failure of one barrier does not directly result in the failure of the whole treatment. However, the barriers need to be assessed for their combined effect on drinking water safety. Both positive and negative interaction between treatment steps can be observed.

Positive interaction occurs when the failure of a preceding treatment step leads to increased efficacy of the following treatment step. When coagulation-sedimentation is sub-optimal, rapid sand filters typically show an increased performance. The resulting combined removal efficacy is much more stable that the individual removal efficacies.

Negative interaction can also occur. When particle removal before a disinfection process is reduced, the efficacy of the disinfection can also be reduced due to protection of organisms in aggregates. This can occur during both chemical and UV disinfection. A failing pre-oxidation step can also result in an increase of disinfectant demand later in treatment. This leads to lower Ct values and thus to less disinfection. This is illustrated by the on-line data analysis of CTS 1 in Chapter 8 (SCADA monitoring data analysis). Failing coagulation-sedimentation can also lead to increased oxidant demand with the same result.

Both positive and negative interaction can be observed with conventional treatment. Table 4.21 shows the range of organism removal by coagulation-sedimentation and filtration separately and combined found in literature. When one treatment fails the other one stays intact or even increases removal. So the observed minimum for combined process in Table 4.21 is higher than the added minimal removal of both separate processes. On the other hand, when the first process is very effective in removing pathogens, the second step can be less effective, therefore maximum removal of the combined process is lower than the individual process maxima added.

Table 4.21 The range of *Cryptosporidium* removal found in conventional treatment experiments is smaller than when both processes are studied separately (Section 4.2), showing that positive interaction between processes has occurred.

	min	MEC	max
A Coagulation-sedimentation experiments	0.4	1.9	3.8
B Filtration experiments	0	2	3.1
A + B	0.4	3.9	6.9
Conventional treatment (coag-sed-filt.) experiments	1.4	3.2	5.5

The combination of physical removal and inactivation has proved to be effective in maintaining high removal for different organisms. Studies on removal of indicators at full-scale suggest that where prechlorination is used there would appear to be only a low risk of microbial breakthrough. Micro-organisms surviving the initial impact of prechlorination were removed efficiently by the combined effects of chlorine continuing to act and the subsequent treatment processes used. GAC filtration enhances

the survival and even proliferation of micro-organisms including those of sanitary significance such as members of the coliform group. Final disinfection would act as an effective final barrier and there is no evidence to suggest that, even in the case of GAC effluents, this is being unduly challenged because of any inadequacies of previous treatment stages. A two-stage treatment by rapid and slow sand filtration plus post chlorination removed >99.99 % of *E. coli*, faecal streptococci, *C. perfringens* and coliphages, as did treatment by pre-chlorination, coagulation, sedimentation, rapid sand filtration and post chlorination.

Careful selection of data is needed in a treatment assessment, as now illustrated. At CTS 3 turbidity was monitored in the raw water, after coagulation-sedimentation and after rapid sand filtration-softening-GAC filtration. Figure 4.11 shows how turbidity decreased after settling. This can be used to verify settling efficacy with greater detail. After the two filtration processes turbidity frequently exceeded the settled water turbidity up to the level of raw water turbidity. This turbidity increase may be caused by particles formed during softening or by particle release by the GAC filters. Comparing filtrate turbidity to raw water turbidity without considering settled water turbidity would erratically suggest poor particle removal by treatment. This example illustrates the importance of monitoring each process individually for interpretation in a risk assessment.

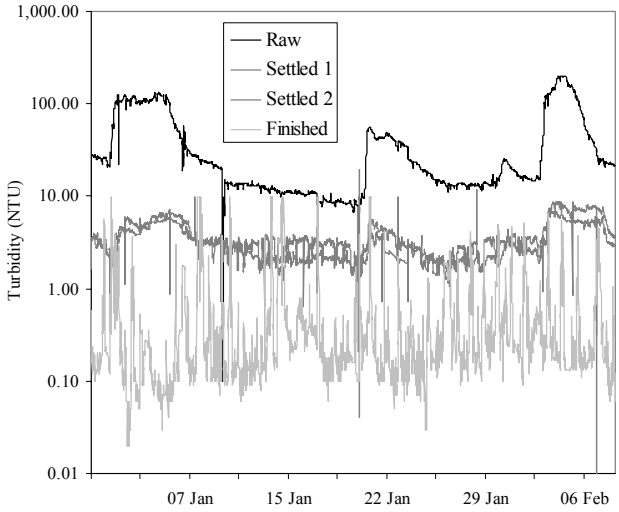


Figure 4.11 Turbidity removal at CTS 3, settled 1 and settled 2 are two parallel lines. Turbidity decreased by settling and then increased periodically after filtration due to other treatment processes.

4.8.2 Effect of treatment configuration and parallel lines

Full-scale treatments generally consist of two or more lines and multiple parallel process units for each process step. In general, other units compensate for a single failing unit. However, this is not the case for pathogen removal. When a process consisting of 10 parallel units has a typical removal of 3 log units, the failure of one unit will reduce the effect of the whole process to one log removal (see Box I – weakest link).

At CTS 2 the ozonation takes place in five parallel contactors each treating one fifth of the total flow. All lines are operated in the same way. Still ozone measurements in all contactors show differences in performance of each line. *E. coli* disinfection was modelled based on these measurements. Figure 4.12 shows the relative contribution of each line to the total amount of *E. coli* in the treated water on one day that ozone was measured. In Figure 4.2 line 3 performs relatively poorly, on other days this was observed for line 1. Depending on the period either one of these lines was responsible for 70%-90% of the total load of organisms in the treated water. Line 2 performed the best and was only responsible for a fraction of the organisms after ozonation.

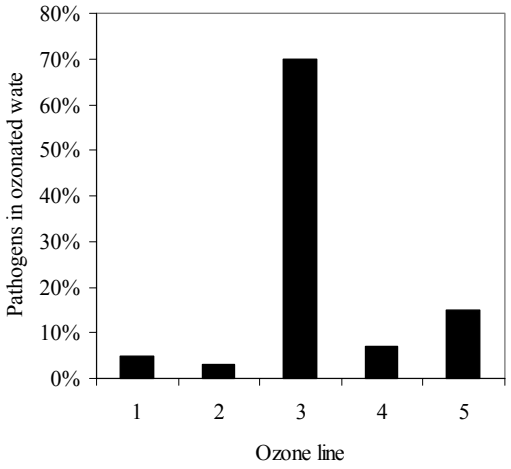


Figure 4.12 Relative contribution of each of 5 ozone lines tot the total number of bacteria after ozonation modelled based on measured ozone concentrations on one day at CTS 2

4.9 EXAMPLE OF A FULL TREATMENT ASSESSMENT

A full assessment of the treatment of CTS 1 is presented as an example. Figure 4.13 shows the treatment scheme. Background information on the reduction of pathogens by these treatment processes is given in Paragraph 4.2. To keep the example comprehensive it is focussed on bacteria. The same methods can be applied to other pathogens.

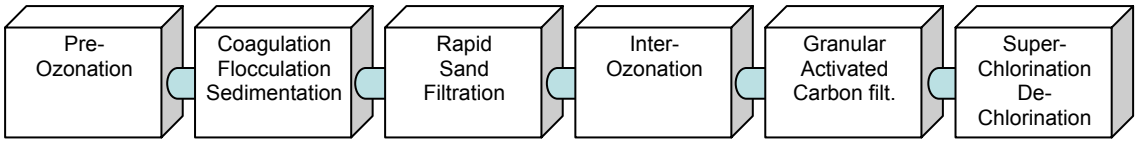


Figure 4.13 Treatment scheme at CTS 1

4.9.1 CTS 1 point estimate assessment

The efficacy of the physical treatment steps was estimated from values reported in the literature in Section 4.2. Coagulation, sedimentation and filtration at CTS 1 are considered as conventional treatment. Disinfection takes place at three stages in the plant. For a first estimate of the inactivation by disinfection, general information about the treatment processes at CTS 1 was collected (Table 4.22).

Table 4.22 Disinfection process characteristics for CTS 1

	Contact time	**Disinfectant residual**	**CT**	**Temperature**
	min	**mg.L^{-1}**	**mg.min.L^{-1}**	**°C**
Pre-oxidation Cl_2+O_3	4	0-2	0-8	5-23
Oxidation O_3	15	0.3	4.5	5-23
Disinfection Cl_2	120	2-3	240-360	5-23

Based on the specified conditions in Table 4.22, the expected range of inactivation was estimated by calculating minimal, likely and maximum inactivation respectively. Data analysis for this as well as the other CTSs showed that most systems do experience short periods of no disinfectant residual. Therefore as a first approximation minimum inactivation was set to 0, since a continuous presence of disinfectant needs to be verified by further data analysis. The likely reduction was calculated with the CSTR model, assuming 1 CSTR and 'mean' disinfectant residual and representative contact time and temperature values as a conservative estimate to calculate the assumed reduction. The more detailed analysis that follows was used to refine the calculations. The maximum inactivation was calculated based on Ct assuming plug-flow conditions (no correction for t10) during 'high' disinfectant concentration and temperature and longest contact time. Disinfection is unlikely to exceed this level. Table 4.23 summarizes the resulting reductions from these disinfection calculations in combination with the literature values of the physical processes for the treatment steps at CTS 1.

Table 4.23 Minimum, Mean Elimination capacity (MEC) and maximum log reduction of pathogens by processes and worst case, estimated and best case log reduction by total treatment at CTS 1

	Viruses min-MEC-max	**Bacteria min-MEC-max**	***Cryptosporidium* min-MEC-max**	***Giardia* min-MEC-max**
Pre-oxidation Cl+O3	0-1.6->7	0-2.9->7	0-0.3-2.7	0-1.3->7
Conventional treatment (Coag.-Sed.-Filtr.)	1.2-3.0-5.3	1.0-2.1-3.4	1.4-3.2-5.5	2.1-3.4-5.1
Oxidation O3	0-1.7->7	0-2.9->7	0-0.4-1.5	0-1.4->7
GAC filtration	0.2-0.4-0.7	0.9-1.4-2.9	0.7-0.9-1.1	0.4-1.7-3.3
Super chlorination	0-3.2->7	0-3.4->7	0	0-1.3->7
Disinfection Cl	0-2.3->7	0-2.4->7	0	0-0.5-1.8
Estimated total log reduction	**1.4-12->34**	**1.9-15->34**	**2.1-4.8-11**	**2.5-9.6->31**

Table 4.23 illustrates that the point estimates result in a broad range between the minimum and maximum removal. The estimates for the single processes represent the variability of pathogen reduction by these processes based on the variables for the disinfection conditions or by reported values in literature. The minimum estimate of the

total reduction actually represents a 'worst case' in which all treatment steps perform at their worst or fail, which has a extremely low likelihood of occurrence. Usually other processes will continue to function when one process fails. This combined effect was simulated by applying a triangular distributed PDF for each treatment step and combining the treatments steps in a Monte Carlo simulation, as described in Chapter 7. The min, MEC and max values in Table 4.23 are applied as the minimum, most likely and maximum parameters for the PDF. By randomly drawing reduction values (generally 10,000 draws or more) from each PDF of the respective treatment steps, reduction by the total treatment was calculated. The result represents the expected variation of reduction by the total treatment in time. This variation is presented by three values in Table 4.24. The over-all reduction is the total reduction of pathogens over the period (see Box II over-all reduction). The reduction varies between the 5% and 95% percentiles for 90% of the time.

Table 4.24 Pathogen log removals at CTS 1 based on a Monte Carlo assessment of treatment performance

	Viruses	**Bacteria**	***Cryptosporidium***	***Giardia***
Over-all log reduction	8.1	10.2	4.8	9.1
5th percentile	11.0	11.8	4.2	10.0
50th percentile	15.7	16.6	5.9	14.3
95th percentile	21.0	20.6	7.8	19.2
Minimal log reduction in 10 000 simulations	4.1	6.6	2.7	5.3

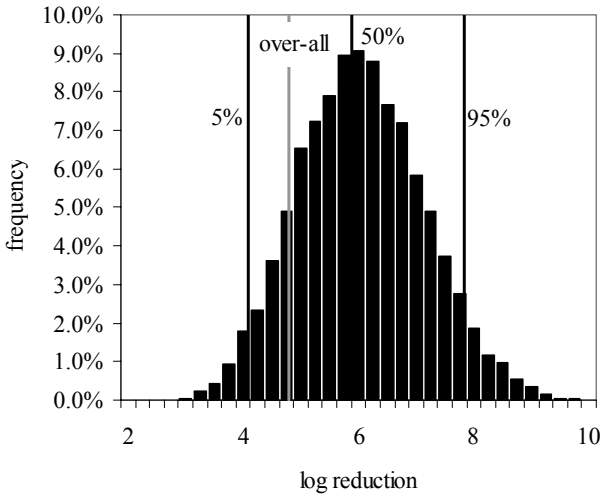


Figure 4.14 Monte Carlo frequency distribution of *Cryptosporidium* reduction at CTS 1. Over-all reduction is the resulting reduction of all the treated water according to this frequency distribution.

Figure 4.14 and Table 4.24 show that without further information, we would expect that *Cryptosporidium* reduction by CTS 1 varies between 4.2 and 7.8 log for 90% of the time resulting in over-all reduction of *Cryptosporidium* of 4.8 log.
The corresponding values for virus removal were between 11 and 21 log reduction 90% of the time, with an over-all removal of 8.1 log. The calculations illustrate that the

expected poor removal during 2.5% of the time has a high impact on the overall removal of viruses. So if the rare occasions of poor removal would not occur, the overall reduction would be much higher. From Table 4.23 it is clear that no removal may occur at the disinfection processes part of the time. Therefore disinfection is an important control point in drinking water treatment, as described in Chapter 2. Detailed data analysis using additional data is needed to determine the frequency of "no inactivation".

The catchment of CTS 1 is a highly polluted river and the expected reductions provided in Table 4.24 and Figure 4.14 are likely to be sufficient to reach health based targets as presented in Section 4.1. The minimal reduction in Table 4.23 represents performance that will not reach a health based target. The expected reduction in Table 4.24 should therefore be verified with additional data.

4.9.2 CTS 1 assessment using surrogates and process conditions

For each individual treatment step the available local surrogate and process data was investigated to reduce the uncertainty about pathogen reduction by the CTS 1 water treatment. Each treatment step is now discussed in further detail..

4.9.2.1 CTS 1 Modelling pre-oxidation with ozone

The inactivation by a disinfection process is modelled to verify inactivation based on the measured process conditions and design characteristics at full scale. At CTS 1 only the general values on process conditions in Table 4.22 were provided for pre-ozonation. No data on ozone concentrations or dosage was available to provide information on variation of inactivation in time, only the variation of water temperature was provided. To better account for variation the inactivation was calculated at minimum and maximum temperature to replace the single most likely value of the triangular PDF by 1.5 and 2.0 log (Figure 4.15).
The design of the system was next taken into account. At CTS1 the contact time was very short and no baffles were applied in the contact chamber, therefore plug-flow was unlikely. The maximum estimate of inactivation was adjusted by performing a single CSTR calculation under 'high' conditions instead of a Ct calculation, which resulted in 2.3 log inactivation (Figure 4.15).
Ozone gas flow was recorded every 20 seconds, but since the ozone in gas concentration was unknown, it could not be used to calculate ozone dose. The on-line records of ozone gas flow were only used for event analysis. Periods of zero gas flow represent the absence of ozone. During some of these periods, chlorine may have been used for pre-oxidation, but no records on that were provided. The event analysis provides an estimate of 'how bad it could be'. Ozone gas flow normally varies between 60% and 200% of the mean flow. The data shows that ozone gas flow is less than 5% of the mean flow for 1% of the time, indicating no ozonation. Therefore the frequency of no inactivation was increased to 1%. Figure 4.15 shows how the triangular PDF was refined.

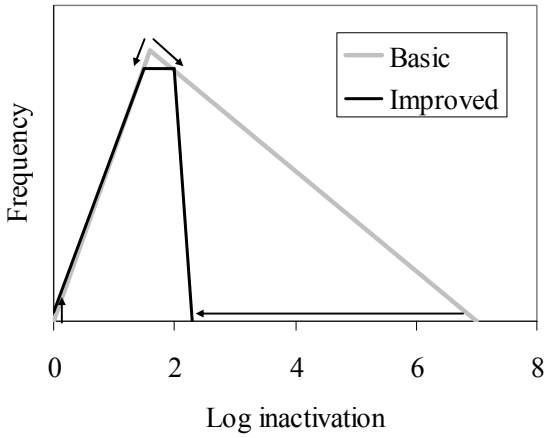


Figure 4.15 Refining the triangular PDF of log inactivation of bacteria by pre-ozonation at CTS 1 using local process data.

The PDF could be further refined by determining the residence time distribution in the contactor, monitoring disinfectant concentration and recording temperature at a higher frequency.

Two other CTSs have a pre-oxidation step. CTS 3 also used ozone and provided similar information on disinfection conditions, apart from the ozone gas flow. Therefore the same approach was applied there. CTS 10 used Cl_2 but provided no information on the disinfection conditions, therefore disinfection could not be modelled.

4.9.2.2 CTS 1: Surrogates for conventional treatment (coag.-sed.-filtr.)

On-line measurements of turbidity at CTS 1 were available for the raw water, after the flat bed clarifiers (two lines) and after a single filter (out of 20 parallel filters). Turbidity in raw water was assumed to be unaffected by pre-ozonation so it was representative of conventional treatment influent. The turbidity measurement for the single filter was regarded to be representative for all individual filters (not for the combined flow). Since conventional treatment (coagulation-sedimentation-filtration) is regarded as a combined process, only the raw water and the filtrate turbidity were analyzed.

The literature review in Section 4.2 made clear that a good working filter (as part of conventional treatment) would provide a constant turbidity below 0.1 NTU. The turbidity after the filter in Figure 4.16 was >0.1 NTU for 50% of the time reaching up to 2 NTU. Turbidity after filtration showed a clear daily variation due to the filter backwash cycle (see magnification in Figure 4.16). Apparently filter to waste was not applied, increasing the chance of pathogen breakthrough. The triangular distribution from the basic assessment was therefore adjusted to represent a poorly working filter as explained in Section 4.5.1. The maximum removal was set to the MEC value, the minimum value was not changed. The most likely reduction was arbitrarily chosen as the mean of the minimum and MEC. Table 4.25 shows the improved triangular PDF parameters based on turbidity monitoring.

Table 4.25 Improved parameter values using triangular distributions to describe pathogen removal by conventional treatment at CTS 1

Organisms	**PDF parameters**		
	Min	Most likely	Max
Viruses	1.2	2.1	3.0
Bacteria	1.0	1.8	2.1
Cryptosporidium	1.4	2.3	3.2
Giardia	2.1	2.7	3.4

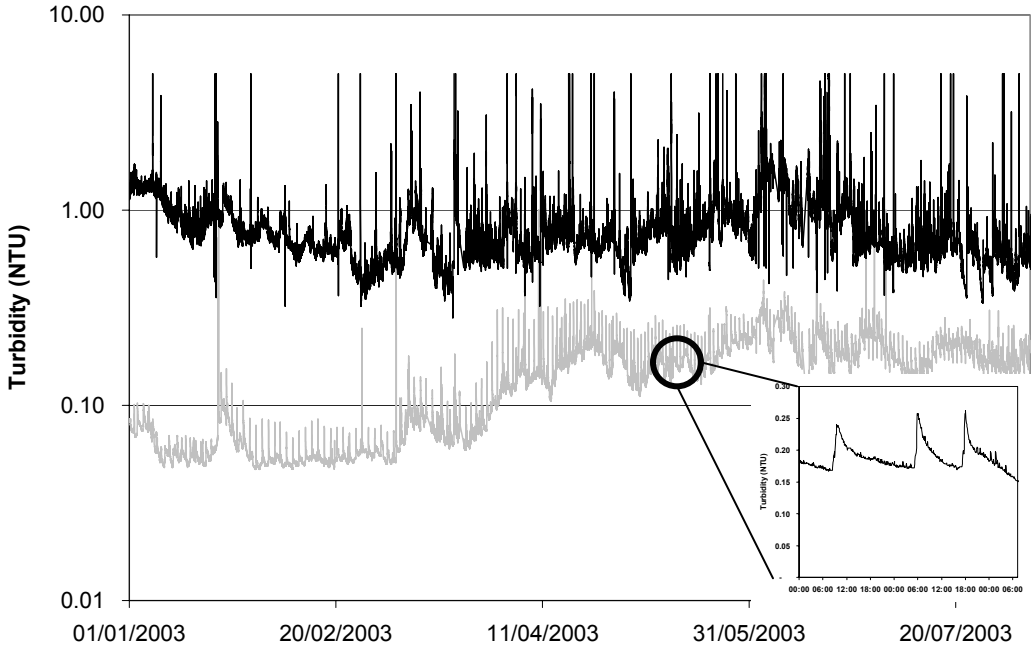


Figure 4.16 Turbidity of raw water (black) and after filter 9 (grey), on-line measurement recorded every 5 minutes and magnification of 3 filter cycles with increased turbidity after backwash.

Including turbidity monitoring of all filters could refine the assessment. Examples of using surrogate monitoring at CTS 10 and 11 were given in Section 4.5.1.

4.9.2.3 CTS 1: Modelling oxidation with ozone

Ozone concentration during main disinfection was recorded every 25 seconds at two parallel lines at CTS 1. Temperature was recorded 49 times in two and a half years, flow was recorded every 5 minutes. Apart from volume no information about the hydraulic characteristics of the ozone contactor was provided.

MEC and over-all inactivation

A single CSTR model was used to calculate a conservative estimate of inactivation. The average measured ozone concentration was 0.48 $mg.L^{-1}$ which is higher than the set point of 0.3 $mg.L^{-1}$. At 0.48 $mg.L^{-1}$ and 10°C a MEC of 3.1 log inactivation of bacteria was calculated, which is slightly higher than first estimated (2.9 log Section

4.9.1). Disinfection was calculated for every single ozone residual record, corresponding temperature and flow were determined by interpolation. Due to variation of the (measured) ozone concentration the overall inactivation of bacteria was slightly lower; 2.8 log for line 1 and 3.0 for line 2 and 2.9 for the combined flow.

Event analysis and PDF
Event analysis at line 1 showed 9 measurements with no ozone residual and 154 measurements that lead to less than 1 log inactivation (0.07 %). Line 2 only contained one measurement leading to less than 1 log inactivation. The results indicated that the main disinfection at CTS 1 was controlled very well, and the minimum inactivation of 0 was too pessimistic. Since the data accurately describes variation of inactivation in time, the baseline variation was described by a beta distribution (of π). The parameters of the PDF are based on the calculated inactivation by maximum likelihood estimation as described in Chapter 7. Statistical uncertainty of the PDF parameters was nil due to the high number of data points. The baseline PDF did not account for the events of less than one log removal. When 0.3 log removal was assumed 0.07% of time (representing no inactivation by one process line so π of the total flow is 0.5) this has no impact on the over-all inactivation by ozonation. Therefore the baseline PDF was applied.

Determining residence time distribution in the contactor and monitoring temperature more frequently (daily) at CTS 1 could further reduce the uncertainty. To reduce remaining uncertainty about inactivation kinetics of environmental pathogens pilot research is needed.

Other CTSs applied only intermediate disinfection (2, 7, 9), only final disinfection (8, 11, 12) or both (3, 4, 5, 6, 10) either with ozone or chlorine. Many sites provided on-line monitoring data but CTS 1 provided the most detailed data on disinfectant concentration and flow. Often a clear water reservoir was used for the final disinfection, so the water level was likely to vary during the day, thus impacting on residence time (distribution). None of the CTSs provided data on the water level and most did not provide the flow data, leaving unnecessary uncertainty in the disinfection calculations.

4.9.2.3 CTS 1: Surrogates for GAC filtration

Turbidity was monitored after rapid sand filtration. The ozonation has no effect on turbidity so this turbidity was considered to be representative for GAC filtration influent. GAC was followed by super chlorination and de-chlorination, which have no effect on turbidity. The monitored turbidity of the treated water was considered representative for GAC filtration effluent. Comparing these monitoring data shows that the GAC filtration has very little effect on the turbidity. Since the GAC filters were not effective in removing turbidity, the triangular PDF was adjusted for a poorly performing filter as described in Section 4.5.1. The MEC was applied as the maximum, while the minimum was maintained. The most likely reduction was arbitrarily chosen as the mean of the MEC and the maximum. Table 4.26 shows the adjusted parameters of the triangular PDF for GAC filtration.

Table 4.26 Improved parameter estimates using a triangular distribution to describe pathogen removal by GAC filtration at CTS 1

Organisms	**PDF parameters**		
	Min	Most likely	Max
Viruses	0.2	0.3	0.4
Bacteria	0.9	1.1	1.4
Cryptosporidium	0.7	0.8	0.9
Giardia	0.4	1.0	1.7

4.9.2.4 CTS 1: Super-chlorination

Chlorine residuals were measured on-line at several points:

1 post super-chlorination
2 pre de-chlorination
3 post de-chlorination
4 post reservoir (drinking water)

The inactivation of *E. coli* by super-chlorination was modelled based on on-line measurements (one per minute) of the residual chlorine before de-chlorination, monitored flow (every 5 minutes) and records of river water temperature. No hydraulic characteristics of the contactor were provided, therefore it was modelled conservatively as a single CSTR.

This resulted in 2.7 log inactivation of bacteria in winter up to 2.9 log in summer. In 2003 only 440 out of 525600 measurements (0.08%) indicated less than 1 log inactivation of bacteria. These short periods of recorded low residual concentrations might be due to failure of chlorine measurement or signal processing since they abruptly drop from normal level to 0 while a gradual decrease of concentration is expected when dosing is interrupted. The modelling predicted that sufficient residual was maintained 99.92% of the time and that the moments of possible loss of residual have had no significant impact on the over-all inactivation by super-chlorination. Since the data accurately describes variation of inactivation in time, the baseline variation was described by a beta distribution (of π). The parameters of the PDF were based on the calculated inactivation by maximum likelihood estimation as described in Chapter 7. Statistical uncertainty of the PDF parameters was nil due to the high number of data points.

By determining residence time distribution in the contactor and monitoring temperature more frequently (daily) at CTS 1 one could further reduce the uncertainty. To reduce remaining uncertainty about inactivation kinetics of environmental pathogens pilot research is needed. CTS 1 was the only system where super-chlorination was applied.

4.9.2.5 CTS 1: Modelling Post-chlorination

After de-chlorination approximately 0.5 $mg.L^{-1}$ chlorine residual is present in the water, which is monitored every 5 minutes. The residence time in the completely filled 5 000 m^3 treated water reservoir is approximately 75 minutes. However, the reservoir was not

always 100% full, especially during peak demand the level dropped. Therefore a residence time of 30 minutes was applied in the CSTR model (40% full). Disinfection ranged from 1.7 log in winter to 2.5 log in summer. Approximately 0.1% of the measurements resulted in less than 1 log disinfection of bacteria.

Determining residence time distribution in the reservoir at different fill levels and monitoring reservoir levels could further reduce the uncertainty. Again monitoring temperature more frequently (daily) at CTS 1 would also reduce uncertainty. To reduce remaining uncertainty about inactivation kinetics of environmental pathogens pilot research is needed.

Most CTSs applied post-disinfection (CTS 1, 3, 4, 5, 6, 8, 10, 11, 12) and these all provided data on disinfectant concentration monitored daily to every minute. None of the sites provided records of reservoir levels, thus leaving uncertainty in the assessment.

Apart from the contact time in the reservoir, a significant amount of disinfection can take place during transport and distribution. This was regarded part of distribution and was not assessed.

4.9.2.5 CTS 1 Improved treatment assessment results

Combining the described PDFs for each treatment step in a Monte Carlo analysis assessed the total treatment. Table 4.27 show the results of this improved assessment for bacteria, compared to the point estimate results. The estimated over-all log reduction is very similar for both approaches. The 90% confidence interval is much smaller using local data. The largest difference is in the minimum log reduction in 10.000 simulations. In the basic assessment, relatively low removal of bacteria of 6.6 log could still occur, although not very frequent. The local information actually indicated that this very low reduction is very unlikely, and that 9.5 log reduction is always maintained, thus reaching the health targets. Analyzing micro-biological data would then not be necessary but has been included for completeness below.

Table 4.27 Results of the improved treatment assessment of bacteria reduction at CTS 1 using surrogate monitoring and process modelling.

	Triangular PDF from literature	**Improved PDF surrogates/modelling**
Over-all log reduction	10.2	11.8
p5	11.8	11.3
p50	16.6	13.2
p95	20.6	15.0
Minimal log reduction in 10.000 simulations	6.6	9.5

4.9.3 CTS 1: Utilization of micro-biological data

The results of microbiological samples represent the actual reduction by treatment. Since they are generally monitored less frequently than surrogates or process conditions, they provide less detailed information on variation. Issues concerning

recovery of methods, (human) infectivity and the translation of indicators reduction to pathogens require more advanced statistical approaches to construct PDFs from this data. The statistical methods are described Chapter 7. This paragraph describes how the micro biological data from CTS 1 was used to improve the treatment assessment. Individual processes were assessed based on available microbiological data.

4.9.3.1 CTS 1: Indicator inactivation by pre-oxidation

The indicator organisms *E. coli* was measured (twice) monthly before and after pre-oxidation resulting in 122 and 59 records respectively. Results for *E. coli* are presented in Figure 4.17. *E. coli* concentrations in raw water varied between 100 and 10,000 MPN.100mL^{-1}. After pre-oxidation the concentration varied between 1 and 100 MPN.100 mL^{-1}, so approximately 2 log inactivation of *E. coli* was achieved. Breakthrough occured on two dates in January and February 2004 when the post pre-ozonation *E. coli* concentration showed peaks up to raw water concentrations. This illustrated that the process was susceptible to breakthrough. The main goal of pre-oxidation at CTS 1 was improvement of coagulation and not directed at inactivation. Nonetheless, the analysis shows that pre-oxidation does have potential to provide direct inactivation. Section 4.9.4.3 discusses the need to use this potential to reach health based targets.

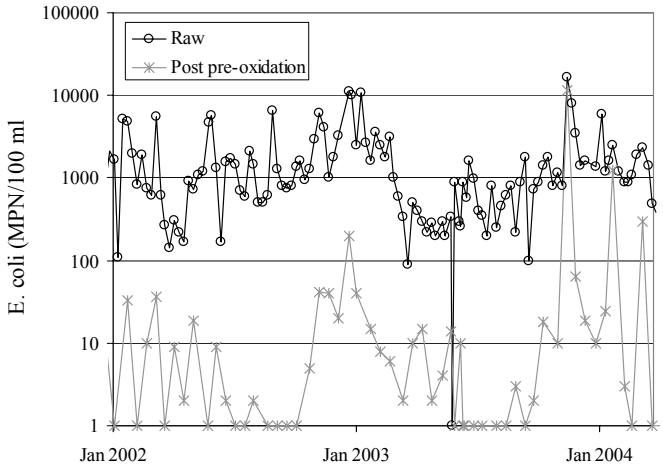


Figure 4.17 *E. coli* monitoring before and after pre-ozonation generally show 2 log inactivation of *E. coli*, but some events of no inactivation occur.

When measurements before and after pre-oxidation were paired by date the resulting dataset of *E. coli* inactivation, presented in Figure 4.18, shows how frequently a level of inactivation is reached. Inactivation of *E. coli* was generally 2 to 3.5 log. This is slightly higher than the inactivation assessed by modelling (1.5-2 log, Section 4.9.2.1). This suggests that the model may be too conservative.
The inactivation of bacteria was described by a beta distributed PDF. The parameters of the beta distribution were estimated from the measured *E. coli* concentrations using the Bayesian MCMC framework described in Chapter 7.

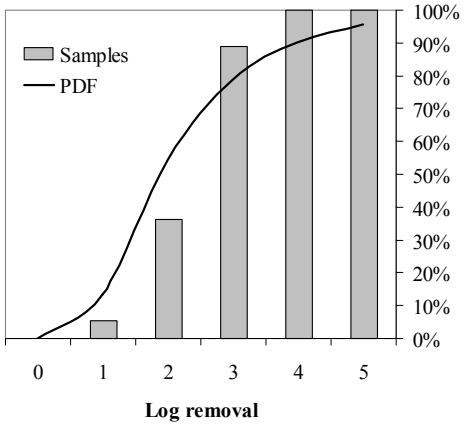


Figure 4.18 Cumulative frequency distribution and CDF (Cumulative probability Density Function) of *E. coli* inactivation by pre-oxidation at CTS 1

4.9.3.2 CTS 1: Indicator removal by conventional treatment

Indicator organisms were measured historically (twice) monthly before and after coagulation-sedimentation and filtration and monthly during the monitoring program. Four of the *E. coli* samples after filtration (12%) were positive showing 1-1.3 log removal. Seven of the negative samples after filtration indicated 1.3 log up to 3 log removal. Both surrogate and indicator monitoring indicate the same level of removal of bacteria (1-3 log). The 11 removal values do not provide additional information about treatment variation, therefore the triangular distribution determined from surrogates (Section 4.9.2.2) is maintained for removal of bacteria. In order to refine this PDF, substantial monitoring of filtered water would be required, preferably in larger sample volumes.

Some 38% of the samples after conventional treatment were positive for *Clostridium perfringens* at CTS 1. Only 4 positive samples and 13 negative samples out of 29 could be paired with samples before conventional treatment. All 13 negative samples indicated more removal than the positive samples. Figure 4.19 shows the frequency distribution. Removal of *Clostridium perfringens* was regarded a model for protozoan (*Cryptosporidium* and *Giardia*) removal, and resulted in 1.4-3.2 and 2.1-3.4 log removal estimates for *Cryptosporidium* and *Giardia* respectively (Section 4.9.2.2). The only exception (or possible event) was a single record of over 100 CFU.100 mL^{-1} of *Clostridium perfringens* after conventional treatment. This indicated that conventional treatment may be susceptible to events but the data is insufficient to estimate frequency, duration and magnitude of such an event. The triangular PDF of protozoan removal from surrogate monitoring (Table 4.25) was used in the treatment assessment.

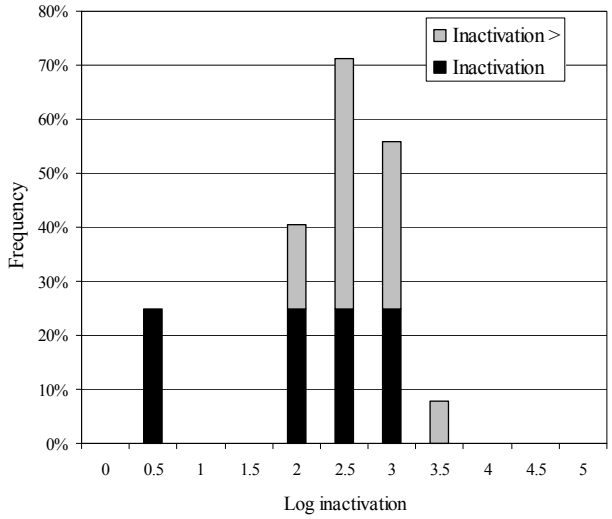


Figure 4.19 Frequency distribution of *Clostridium perfringens* removal by conventional treatment

More intensive monitoring is required to better assess the possibility of breakthrough of the conventional treatment, since the applied PDF may be too optimistic. Other CTSs generally did not monitor indicators after the first clarification steps, apart from CTS 2, which is discussed in Chapter 7.

4.9.3.3 CTS 1: Indicator inactivation by inter-ozonation

Indicator organisms were measured historically (twice) monthly before and after ozonation. No sample after ozonation was positive for *E. coli*, indicating that over-all inactivation of *E. coli* was approximately 2.7 logs. About 38% of the samples before and 24% of the samples after ozonation were positive for *Clostridium perfringens*. Most positive samples could not be paired so the dataset was not analysed as described above, but could have been modelled based on the fraction removed (Chapter 7). Due to a single 100 $CFU.L^{-1}$ result, an over-all inactivation of *C. perfringens* spores of 0.9 log was indicated. These observations support estimates of inactivation efficacy based on modelling (Section 4.9.2.3), but provide no additional information on the level and variation of the inactivation by ozonation.

4.9.3.4 CTS 1: Indicator removal by GAC filtration

Indicator organisms were measured historically (twice) monthly before and after GAC filtration. No *E. coli* was found before or after GAC. One sample after GAC (6.25%) was positive for *Clostridium perfringens*. So the number of data that could be paired was insufficient to provide such a PDF of removal. Approximately 0.9 log over-all removal of *C. perfringens* spores by GAC was shown by the data. The *Clostridium* removal was similar to the MEC reported in Table 4.3. This spore reduction was more optimistic than the turbidity assessment of 'poor' removal of *Cryptosporidium* and *Giardia*. However, the data was so scarce it provided too little information on the variation of removal, so the triangular PDF based on turbidity (Table 4.26) was applied in the treatment assessment.

4.9.3.5 CTS1: Indicator inactivation by Super-chlorination and de-chlorination

All indicator monitoring was negative at this point in treatment, therefore no assessment of indicator inactivation is possible for these treatment steps.

4.9.4 CTS 1: Total treatment assessment results

4.9.4.1 Results treatment assessment

At CTS 1 the micro-biological data that was available had little influence on the PDFs of the improved assessment. Only the PDF for the first treatment step of pre-ozonation was changed based on the *E. coli* monitoring data. The results of the total assessment including all available data in Table 4.28 are very similar to the improved assessment.

Table 4.28 Results of the total treatment assessment of bacteria reduction at CTS 1 using all available data.

	Triangular PDF from literature	**Improved PDF surrogates/modelling**	**Best PDF all data**
Over-all log reduction	10.2	11.8	11.8
p5	11.8	11.3	11.2
p50	16.6	13.2	12.9
p95	20.6	15.0	16.1
Minimal log reduction in 10.000 simulations	6.6	9.5	9.7

4.9.4.2 Comparing results to health basted performance targets

The source water for CTS 1 was a heavily polluted river, for which *Campylobacter* concentrations of 100 $CFU.L^{-1}$ were likely (see Chapter 3). Following the WHO example of a health based target at 10^{-6} DALY (see Section 4.1) a *Campylobacter* removal performance target of 5.9 log inactivation was required. The results in Table 4.28 verify that the treatment potentially achieved over 9.7 log reduction of bacteria in the assessed period. Since sufficient reduction was verified already using surrogates and process monitoring, the extra monitoring of micro-organisms that was performed for this assessment was not required to verify *Campylobacter* reduction. Using only triangular distributions based on literature, rare occasions of non-compliance to the health based performance target were already unlikely to occur (6.6 log reduction during 1/10,000 of time). The surrogate and process monitoring was valuable to verify that the minimum reduction actually was much higher than required by the health based performance target. Some suggestions on how to further improve the surrogate and process monitoring have been provided.

Only a basic assessment using point estimates and triangular distributions was presented for other pathogens. For *Cryptosporidium* with an assumed source water concentration of 10 oocysts per litre a performance target of 4.2 log units can be derived from Figure 4.1. The over-all reduction of 4.8 log (Table 4.28) was just sufficient to reach the yearly target. Reduction was sufficient 95% of the time (p5 = 4.2) but on rare events (1/10,000) it may be as low as 2.7 log. An improved assessment for *Cryptosporidium* would result in less removal by conventional treatment and GAC filtration, same as for bacteria. The disinfection steps were unlikely to result in significant *Cryptosporidium* reduction. As a result *Cryptosporidium* reduction might be

critical at CTS 1. Depending on the interpretation of the assessment results by the health inspectorate, additional data collection, improving operation of current treatment or even additional treatment could be necessary. When the hydraulic characteristics of the ozonation processes actually better resemble plug-flow, this would lead to a substantially higher level of *Cryptosporidium* inactivation. Applying a higher ozone dose would also lead to more inactivation. Monitoring conventional treatment with frequent (daily to weekly) clostridial spore analysis in sufficiently large volumes (1 L) could verify that the treatment is currently providing more removal than expected from the turbidity measurements. Improving the performance of conventional treatment could improve the actual *Cryptosporidium* reduction. Optimization could be studied in a pilot plant.

A (Rota)virus reduction in the order of 5.5 log units may be required based on the example in Figure 4.1 at a source water concentration of 10 viral particles.L^{-1}. Over-all virus removal appears to be sufficient based on the basic assessment, although rare occasions (1/10,000) of low reduction (4.4 log) might occur. Viruses, like bacteria, are susceptible to disinfection processes. Modelling these processes for bacteria showed that they perform consistently well, and that the basic assessment is too conservative. It is expected that an improved assessment for viruses will verify that current treatment performance is sufficient to reach health based targets.

4.9.4.3 Setting of critical limits and corrective actions

Each treatment process at CTS 1 contributes to the reduction of one or more pathogens, and thus can be considered as a control point. For each treatment process, critical limits can be defined to either set the proper conditions for the process and/or to verify that a process is working properly.

The goal of the pre-ozonation is improvement of the conventional treatment, and is primarily operated for that cause. The rest of the treatment sufficiently reduces bacteria and viruses under base-line conditions but *Cryptosporidium* inactivation by pre-ozonation could be enhanced. During events, like failure of the other disinfection steps, the ozone dose at pre-ozonation can be increased as a corrective action to maintain sufficient disinfection.

Conventional treatment could potentially provide more *Cryptosporidium* removal. Optimizing the coagulation-sedimentation process with jar tests and applying filter to waste could be considered. These would lead to new set points for operation and critical limits for turbidity.

The disinfection processes at CTS 1 (inter-ozonation, super-chlorination and de-chlorination) are currently operated at a fixed disinfectant concentration. By measuring this concentration at the contactor outlet, proper functioning of the dosing equipment and control loops are currently verified. This could be improved by varying the set-point for the disinfectant concentration with changing process conditions, like temperature and flow, to provide sufficient inactivation of target pathogens. When the source water is sufficiently characterized, the target level of disinfection could be

adapted to address the variations in source water. Thus process operation can react to seasonality and peak events with sufficient disinfection and without substantial over-dosing.

4.9.4.4 Monitoring plan

Some possible improvements of monitoring were suggested in Sections 4.9.2 and 4.9.3. In general monitoring of indicator organisms is only useful for risk assessment after the first treatment steps when most samples are still positive. Each step in the assessment needs to be monitored separately (inflow and outflow) in order to translate the indicator reduction to pathogens. Indicator monitoring needs to be applied frequently (daily to weekly) to provide sufficient information on variation and limit statistical uncertainty. In the example the on-line monitoring of surrogates and process conditions proved to be more effective to verify treatment efficacy than micro-biological monitoring. Some additional monitoring of reservoir levels and flow would further refine this assessment. Determining flow conditions for the disinfection processes would greatly improve the interpretation of the monitored process conditions.

4.10 DISCUSSION AND CONCLUSIONS

4.10.1 General findings in CTS treatment assessments

Within the MicroRisk project twelve treatment system throughout Europe and including Australia were assessed as a pilot for QMRA. Some general lessons were learned from this. Microbiological data that was collected according to current practice proved to be insufficient to quantify treatment efficacy without large uncertainty. End-product testing generally resulted in close to 100% non-detects. This approach only provided an estimate of minimal pathogen reduction by treatment. The reduction that could thus be indicated was generally insufficient to provide safe drinking water. Microbiological samples after the first treatment steps provided additional information on the efficacy of these steps. These were almost in the same order of magnitude as those found for the total treatment. Additional treatment steps, such as disinfection, were quantified by various indirect means. Process models used measured disinfectant residuals, temperature, hydraulic characteristics of contact chambers and inactivation kinetics from literature to calculate inactivation of different pathogens at full-scale. Physical removal processes were quantified based on reported efficacy in literature and monitored removal of surrogates such as turbidity or particles. Combining the results of the different treatment steps in a stochastic model led to a decrease in uncertainty related to pathogen reduction by water treatment. Further, the results led to a change in focus on what types of data need to be collected for quantitative assessment of pathogen reduction by treatment. Full-scale hydraulics plays a major role in disinfection and is the main cause of uncertainty in disinfection processes. Recommendations for improvement of treatment configuration or operation could be made while providing a quantified estimate of the effect of such an improvement on microbiological safety.

4.10.2 Conclusions and recommendations

Despite the level of data available on treatment, there is always a degree of uncertainty on the efficacy of pathogen reduction. The amount of relevant data available for a treatment system will have a large impact on the certainty that can be reached. Still some uncertainty will remain in the outcome of the treatment assessment. The impact of the uncertainty on the performance of the treatment component of a full risk assessment model (Chapter 7) therefore also needs to be assessed. In the end a best estimate with confidence intervals is required to apply legislation, and to focus attention on where to best manage water treatment.

The intention of the treatment work package in the MicroRisk project was to provide a protocol on how to assess treatment. Working through the CTSs it became clear that each system is so specific that a generally applicable protocol is not possible at this stage. Therefore the presented approach was applied to provide guidelines on how to assess a treatment in a systematic way. When going through this in cycles of data collection, treatment assessment and adapted monitoring the assessment will reach a satisfactory level of verified water safety. A guideline for the first steps is:

1 Collect all relevant data that is available for the treatment studied and assess the treatment as presented in this chapter (first iteration).
2 Collect additional data by monitoring or pilot experiments. From the CTSs in MicroRisk the following prioritisation was suggested:
   - perform additional pathogen sampling to determine base-line and peak (event) pathogen concentrations in the source to better establish health based performance targets;
   - monitor relevant indicators frequently (daily to weekly) after the first (physical) treatment steps at the point where more than 50% of the sample are positive;
   - use on-line measurements, such as turbidity or particle monitoring for physical processes to verify that events are rare (and their duration);
   - monitor disinfectant residual, reservoir levels, flow and temperature on-line to model disinfection processes; and
   - improve hydraulic characterization of these disinfection processes by tracer test or CFD modelling to refine these models.
3 When uncertainty remains too large, more drastic measures like large volume sampling, pilot studies or full-scale studies with model micro-organisms should be considered.

Recommendations for scientific research include:

- Setting up of a common database for treatment efficacy (along the lines of Hijnen *et al.* [2005a]) to keep up to date removal credits and inactivation rate constants accessible;
- Improved disinfection modelling to verify performance of disinfection processes since micro-biological sampling at the last stages of treatment provides minimal quantitative data. Knowledge about disinfection kinetics of environmental pathogens is still limited and more research on topics like kinetic parameters and tailing is required;

- The applicability of surrogates and indicator organisms is still inconclusive and requires more research under full-scale conditions; and
- Improve detection and recovery techniques for pathogens.

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