CHALMERS

Applicability of QMRA on Artificial Groundwater Recharge

Master of Science Thesis in the Master's Programme Geo and Water Engineering

RICKARD DAHLSTRÖM

Department of Civil and Environmental Engineering
Division of GeoEngineering

Engineering Geology Research Group

CHALMERS UNIVERSITY OF TECHNOLOGY
Göteborg, Sweden 2011
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Department of Civil and Environmental Engineering
Division of GeoEngineering
Engineering Geology Research Group
Chalmers University of Technology
SE-412 96 Göteborg
Sweden
Telephone: + 46 (0)31-772 1000

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ABSTRACT

To assess the microbial risk of groundwater works the capacity of natural as well as artificial groundwater recharge to reduce microorganisms needs to be estimated. There are many parameters affecting the reduction of microorganisms during the infiltration process, which need to be assessed.

The aim of this study was to analyse occurred events related to microbial contamination of groundwater supplies and to evaluate the applicability of a model for quantitative microbial risk assessment (QMRA) on artificial groundwater recharge. Artificial groundwater recharge is commonly applied in, for example, Sweden and additional studies on microbial risks are needed to better understand the processes determining the reduction of microorganisms. News articles describing microbial contamination events were reviewed to identify the causes and the magnitude of the problem. A clear correlation was observed between contamination events and rainfall events. This emphasises the importance of preventing surface water flows reaching groundwater wells and causing contamination.

For the QMRA part, the parameters affecting the microbial pathogen reduction in artificial groundwater recharge were analysed and literature data for the reduction potential was compared with the default values used in the evaluated model by the Swedish Water and Wastewater Association. The default values for the reduction are only valid for slow sand filtration with filtration depth less than 1.5 meters. The data found in the literature is valid for larger filtration depths and thus show a higher reduction potential. In the evaluated QMRA model it is not possible to define the filtration depth or other parameters in order to adjust the level of microbial pathogen reduction. The knowledge about microbial pathogen reduction in artificial groundwater recharge is limited today and to enable more reliable risk assessments site investigations are needed. Guidelines on what important parameters to consider and the reasonable levels of reduction would be of great help in order to assess the risk when no site investigation has been made.

Key words: quantitative microbial risk assessment (QMRA), artificial groundwater recharge, unsaturated zone, granular filtration, microbial pathogens

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Preface

In this study, the waterborne outbreaks related to groundwater sources were analysed, together with an evaluation of the applicability of quantitative microbial risk assessment (QMRA) on artificial groundwater recharge. The study was performed in the spring 2011.

The work on this thesis has been carried out at the Department of Civil and Environmental engineering, Division of GeoEngineering, Engineering Geology Research group at Chalmers University of Technology, Sweden.

The study has been conducted by Rickard Dahlström, student at the master's programme Geo and Water Engineering, with Professor Lars Rosén as examiner, Assistant Professor Andreas Lindhe and Annika Malm as supervisors.

First I would like to thank Andreas Lindhe for excellent supervision and support. Furthermore, I would like to thank Lars Rosén, Annika Malm, Olof Bergstedt and Thomas Pettersson for great assistance and interesting feedback.

Finally, I would like to thank my family and my friends for a wonderful support through my time at Chalmers.

Göteborg September 2011

Rickard Dahlström

1 Introduction

1.1 Background

About half of the population in Sweden is supplied with drinking water from surface water sources. The other half is supplied with water from groundwater sources. Groundwater has been used for a long time and is typically considered to be of good quality. For many groundwater supplies, however, the treatment steps (if there is any) may not be sufficient if the water quality for some reason is reduced (e.g. a microbial contamination event). An investigation of waterborne outbreaks from 1995 to 2003 showed that 43 % of the infected persons were supplied with water from groundwater sources (Lindberg and Lindqvist 2005). Groundwater works are generally smaller and does not supply as many people as surface water works. When comparing the number of waterborne outbreaks the groundwater works were causing 80 % of the events. The groundwater can be either natural groundwater or artificial groundwater, which means that when water is infiltrated through the ground using e.g. infiltration dams.

Artificial groundwater recharge may be an adequate microbial barrier by itself or in combination with other barriers. However, due to the increased knowledge concerning microbial pollution of groundwater, there is an increasing demand for assessing microbial risks for water supplies based on naturally and/or artificial groundwater recharge. This means that useful evaluation criteria need to be developed for this type of barrier. The process for natural and artificial groundwater recharge is more complex and less understood than the treatment processes typically used in conventional water works for surface water.

When waterborne outbreak event occurs the pathogen(s) causing the problem is typically not detected. End-product testing, i.e. analyses of the treated water, is important but it has to be combined with a proactive approach in order to prevent an unacceptable drinking water quality.

After the hazards that in different ways can harm the water supply and the consumers have been identified, it is important to identify the pathways, i.e. how the contaminants may reach the water source and in the end the consumers. This knowledge is of importance in order to reduce or cut off pathways, including the work of constructing water protection areas. Several parameters have to be considered to properly describe the transport of contaminants. Dilution, retention and die off of the pathogens are some examples of relevant parameters. The soil material, the topography and overflows are other important factors affecting the possibility of pathogens reaching water sources.

A model for quantitative microbial risk assessment (QMRA) has been developed within a project financed by the Swedish Water and Wastewater Association (Lundberg et al. 2009). As typically done in QMRA the model is based on the raw water quality and the ability to reduce the level of pathogens in different treatment steps. The model provides a structured way of analysing and quantifying the microbial risks that drinking water consumers are exposed to. The results are expressed as the probability of infection and also as DALYs (Disability Adjusted Life Years). The QMRA model is an important tool for determining if the microbial risk is acceptable or not.

The model is mainly developed to be used for drinking water systems based on surface water as the raw water source. The treatment steps included in the model are

the ones commonly used in Sweden. In order to better understand and properly model groundwater supplies using a QMRA model, the parameters affecting the pathogen reduction need to be understood.

1.2 Aim

The overall aim of this work is to analyse waterborne outbreaks related to groundwater sources and to see how a quantitative microbial risk assessment (QMRA) could be performed to properly acknowledge artificial groundwater recharge as a microbial barrier. Specific objectives are:

- To analyse occurred events related to microbial contamination of drinking water supplies and to see if there are any differences between systems based on surface water and groundwater.
- To analyse the aspects/parameters affecting the reduction of microbial contaminants in artificial groundwater recharge and to see how this can be quantified.
- To apply a QMRA model at a system using artificial groundwater recharge to evaluate the need of data, uncertainties in input data and the applicability of the model.
- To suggest possible improvements of the QMRA model so that groundwater supplies can be better analysed. This is done based on a literature review and an application of the QMRA model.

1.3 Method

Part 1 – News articles on 300 indications of microbial contamination were investigated. The events considered in this thesis were those related to microbial pathogens and groundwater, the other events were left out.

Part 2 – A literature review were performed in order to identify and analyse the parameters affecting the pathogen reduction in sand filters and soil infiltration. From the literature, relevant reduction studies were compared and a proposal of a new general reduction values was presented and compared with the default values from the QMRA model by Lundberg et al. (2009).

Part 3 – Identified data on possible reductions levels were used in the QMRA model in order to assess the relevance of the data and the difficulties of using QMRA on artificial groundwater recharge.

1.4 Delimitations

The microbial pathogens are considered to be the highest risk in drinking water supplies according to the World Health Organization (WHO), see e.g. Svenskt Vatten (2008). The events of interest for this thesis are the cases with microbial risk of faecal origin, the other microbial risk and chemical risks are not considered. The surface water works and the distribution networks are also not considered in this thesis since the protective barriers have no effect after the water has left the waterworks.

2 Drinking water supplies

Usually groundwater is of good quality and therefore no advanced treatment methods are typically needed. In many of the small groundwater works there is no protective barrier for pathogens other than chlorination as in case of emergency. Nowadays more and more small waterworks install UV disinfection both as a first disinfection step or to reduce the amount of chlorination.

In Sweden, 80 % of the waterworks are small groundwater works. Many groundwater works distribute water without any disinfection. In 1996 the drinking water production based on groundwater sources constituted 40 % of the total production (Lindberg and Lindqvist 2005).

2.1 Raw water source and pathogen transportation

The first step in the assessment of the raw water source is to find the possible sources of pathogens. It can be point sources such as wastewater treatment plants, pipe leakage, and combined sewer overflow discharge or on site sewer. It may also be diffuse sources from animals near by the water source. Many of the events occurred after excessive rainfall. It seems that some of the groundwater wells are sensitive for leakage into the well from surrounding area.

It can be possible to identify the pathways of pathogens to the raw water source using digital elevation models (3D representation of a terrain surface) and site and soil investigation.

2.2 Chlorination

Drinking water chlorination started in order to stop typhoid fever spreading via the drinking water. Today, chlorination is widely used for chemical disinfection. Chlorination is often used as the final treatment barrier in a water treatment plan and since there are different disinfection agents that can be used the efficiency varies. The laboratory analysis of the disinfectant efficiency for chlorination is adequate, however the analysis is not made under similar circumstances as the water in the supply system. It is easy to calculate the amount of chlorine needed but the uncertainties may be high. In the analysis all the microbial aggregates are eliminated and the microorganisms are homogeneously distributed. However, many pathogens in the drinking water system are surrounded by a protective substance that protects them against the disinfectant (Schoenen 2002). It is thus important to make analysis of the level of chlorine in the treatment plant. Some pathogens such as *Giardia lamblia* and *Cryptosporidium parvum* cannot be killed with traditional disinfectants. Thermal disinfection (boiling) kills or inactivates all faecal pathogens like virus, bacteria and protozoa (Schoenen 2002).

2.3 UV disinfection

UV radiation is effective to inactivate all waterborne pathogens. The UV radiation damage the nucleic acids of the cell or virus (Hijnen et al. 2006).

Viruses and bacterial spores are the most persistent organisms to UV radiation and *Adenovirus* is the most resistant pathogen known. It has been demonstrated that UV radiation is very effective of inactivation of *Cryptosporidium* and *Giardia*. These two pathogens are of major concern for the drinking water quality, since they are resistant to chlorination. As a disinfection barrier UV has no negative side effects of importance and is not affected by temperature, pH or reactive organic matter (Hijnen et al. 2006).

2.4 Granular media filtration

The history of sand filtration as a treatment processes for drinking water reach back to the year of 1804 in Scotland (Hijnen et al. 2006). Back then the primary object was to reduce the bad smell and taste. Shortly it was recognized that the sand filtration reduced the cholera and typhoid outbreaks. However, the disinfection treatment as ozonation and chlorination become the common practice.

Soil filtration is the nature's own way to treat the water from pathogens. Sand filtration systems can reduce many pathogens and provide microbiologically safe drinking water. It is as stated earlier one of the oldest processes used to produce microbiologically safe drinking water.

The fundamental processes that control the transport of biological contaminants in subsurface waters are important to understand in order to be able make a predictive model for the removal of pathogens. Today, the fundamental processes for the transports of bacteria, viruses and protozoa are still limited (Schinner et al. 2010).

When the water is filtrated through the granular media there are several parameters that need to be considered. Inactivation of pathogens will occur, which is depending on time. The reduction of pathogens is caused by disintegration and predation. Some zooplanktons are known for predation of bacteria and C*ryptosporidium* oocysts. The extent of this activity differs in saturated and unsaturated zone, see Figure 2.1.

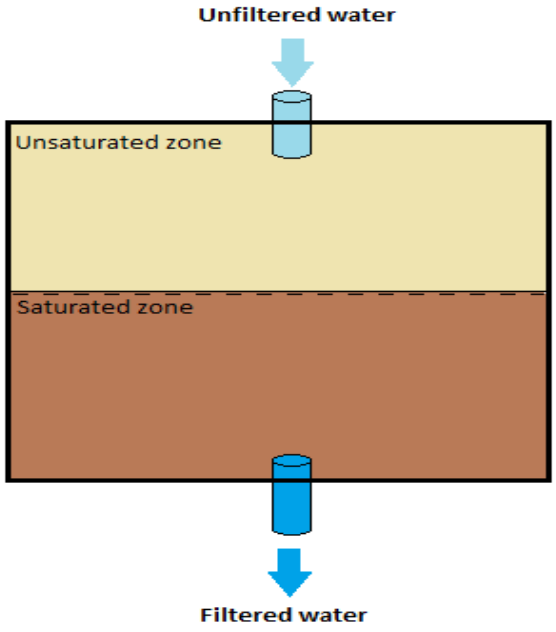

Figure 2.1 A conceptual model of a filter with an unsaturated and a saturated zone.

The natural and engineered filtration systems are far more complex environments than the laboratory experiments and the theoretical models. The heterogeneity in physical, chemical and biological properties may affect the relevance of the laboratory analysis, causing unfavourable conditions. However, the laboratory studies are important in understanding the behaviour for the microorganisms. The colloid filtration theory can be used as a theoretical model for removal efficiency in laboratory experiments due to the attachment (Schinner et al. 2010).

The hydraulic conductivity is another one important parameter for the infiltration rate (Zagerholm et al. 2007). Depending on the media used in the infiltration plant and the porosity, the hydraulic conductivity can be estimated. During excessive loads the filter capacity may be decreased due to more rapid flow through the filter.

If the filtration is adjusted to the raw water contamination and if the filtration is combined with flocculation, then the filtered water can be free from any faecal contaminants (Schoenen 2002).

2.5 Artificial groundwater recharge

The access to water sources for drinking water production are decreasing in the world, the increased population, drought, increased use of chemicals and global warming further stresses these vulnerable sources of water. Artificial groundwater recharge may be a solution to some of those problems.

In Sweden, the infiltrated surface water will be considered as natural groundwater if the residence time is 14 days or longer (Svenskt Vatten 2008). The depth to the water table is one important factor in pathogen reduction for artificial groundwater recharge. Increased depth to the water table will increase the capacity of the unsaturated zone to reduce pathogens. The unsaturated zone is more effective in removing and reduce the amount of pathogens than the saturated zone (Jin et al. 2000). The temperature has a large influence on the reduction rate of microorganisms, most likely due to the increased activity of indigenous groundwater microorganisms (Toze 2003).

Adsorption to surfaces in the aquifer reduce the inactivation rate for both viruses and bacteria (Oliver et al. 2006). The predation rate can also be reduced with clay materials present in the aquifer by coating and protect the pathogens and increased organic matter in the soil will also reduce the removal of pathogens (Toze 2003).

In colder climate it is important that the open sand filter do not freeze during the winter. If it does the water pressure may be increased, to ensure enough water through the filter. This can result in water passing the filter without sufficient filtration (Schoenen 2002).

3 Pathogens

The ingestion of faecal contaminated water is one of the greatest microbial risks of this century (Cabral 2010). The major pathogens of concern are bacteria, virus and protozoa. Some of these pathogens are difficult to analyse and even to identify; therefore the common way to assess the risk is to measure indication organisms in the water. The infectious dose varies between the pathogens and it also varies between the different species within the pathogens. Most enteric viruses and protozoa may only require ten or less infectious particles to cause an infection while bacteria usually need more than 1,000 infectious cells (EPA 1992).

3.1 Sources of pathogens

Animal production is a major source of potential pathogens (Goss and Richards 2008). The sources of pathogens are infected humans and animals where the pathogens are excreted in the faeces. When the faeces influence the water the pathogens may travel for long distances and are a threat to human health. Most microbial organisms are heterogeneous distributed in the water or attached to suspended particles, which make it difficult to measure the actual concentration. In difference to chemical exposure microbial organisms does not accumulate the dose over time but it may give rise to immunity to the specific organism (Lindberg and Lindqvist 2005).

3.2 Bacteria

Bacteria are a microorganism that consists of one single cell, without a cell nucleus. The life and eco system are dependent on the bacteria and they exists almost everywhere. However, pathogenic bacteria have caused severe diseases around the world, and in many cases the drinking water has been the source (Schoenen 2002).

3.2.1 Campylobacter

Campylobacter are a major cause of diarrhoeal illness, gastroenteritis, in the world. They cause more cases of diarrhoea than *salmonella* in developed countries. The main reservoir for *Campylobacter* is animals (WHO 2000). *Campylobacter* can survive up to 4 months in fresh waters, with longest survival at 4°C (Rollins and Colwell 1986); (Thomas et al. 1999). The infectious dose is as few as 1,000 organisms. The incubation period is usually 1-3 days and the illness last for 3-6 days (Svenskt Vatten 2008). About 7,000 cases in Sweden are estimated every year, where 40 % of those cases were infected in Sweden (SMI 2010c). The knowledge about the spreading of *Campylobacter* is limited. However, the presence of *Campylobacter* in surface water is strongly dependent on rainfall. *Campylobacter* are not resistant to disinfection and therefore easy to eliminate (WHO 2008).

3.2.2 Coliforms

Most of the coliform bacteria are harmless bacteria that live in the soil, participating in the degradation of plants and animals (Svenskt Vatten 2008). Within the group of

coliform bacteria is *Escherichia coli*. Most strains of *E. coli* are harmless but some *E. coli* strains cause acute diarrhoeas and gastroenteritis. The *Enterohemorrhagic E. coli* (EHEC) can cause severe diseases and the strains of particular interest are O148, O157, O111 and O124 (Cabral 2010). *E. coli* indicate faecal contamination of the water since it is found in the intestines of warm blooded animals and humans. The main reservoir for *E. coli* are humans and cattle to a lesser extent (WHO 2008). The incubation period is 3-8 days and the symptoms occur for about ten days. *E. coli* can survive for months in water troughs and sediments and the infectious dose is as few as 100 organisms. (WHO 2005)

3.3 Virus

Viruses are biological agents consisting of a protein coat with nucleic acids (DNA or RNA) (Sen 2011). The viruses of interests are enteric viruses. The enteric viruses are excreted in faeces of infected humans and can infect the gastrointestinal tract. They may cause disease and have a short incubation period (WHO 2008). Viruses are more resistant to treatment processes than bacteria. Viruses are difficult to filter due to their small size. As for all viruses they cannot grow outside its reservoir but on the other hand viruses have a low infectious dose.

3.3.1 Adenoviruses

There is 51 types of human adenoviruses know today. They cause a wide range of infections from gastrointestinal infection to eye infection. The knowledge about the prevalence of enteric adenoviruses in water sources is limited. However large numbers are excreted in human faeces. *Adenoviruses* are not detectable by conventional cultivation. They are resistant to disinfection, including UV light irradiation (SMI 2010a).

3.3.2 Calicivirus

The types of *calicivirus* that infect humans are norovirus and *Sapovirus*. Like Adenovirus they cannot be cultivated, have high resistance to disinfection and are excreted in faeces of infected humans (WHO 2008). The incubation time is 12- 48 hours and the duration of the symptom are usually less than 3 days. The only known reservoir for this virus in humans, and the minimum dose to cause infection has not been determined (WHO 2008; SMI 2010a).

3.4 Protozoa

Protozoa are small organisms that have one cell nuclei. They have the ability to move themselves and a heterotrophic nutrient uptake. Most of the protozoa produce cysts, oocysts or eggs that are extremely resistant to disinfection and also difficult to remove by filtration (WHO 2008).

Giardia lamblia (*Giardia intestinalis*) and *Cryptosporidium parvum* are the protozoa of most interest. These are common enteric pathogens detected in faecal contaminated water. Compared to bacteria they occur in low number and are difficult to detect. They are difficult to cultivate and therefore to detect them fluorescence microscopy

combined with fluorescent labelled immunological strains are used (Toze 1999). These detection methods are time consuming and expensive but tend to be more accurate than for virus and bacteria.

3.4.1 Cryptosporidium

Of all *Cryptosporidium* species, *Cryptosporidium parvum* is the species that causes most infections for humans. Humans and livestock is the main reservoir and the oocyst do not replicate outside its host. *Cryptosporidium* contaminated water can cause a large impact due to the large amount of oocysts that are excreted by infected animals and humans. The infectious dose is as few as 10 oocysts. For properties of the pathogens see Table 3.1. The survival of the oocysts is high, up to months in fresh waters and they are extremely resistant to disinfection. They are resistant to chlorine but UV light radiation inactivates the oocysts (WHO 2008). The incubation time for *Cryptosporidium* is not determined but estimated to be about 7 days (SMI 2010b).

Table 3.1 Pathogens and their properties, (SMI 2010a; SMI 2010b; Svenskt Vatten 2008; WHO 2000; WHO 2005; WHO 2008).

Pathogen	Incubation	Time of symptom	Reservoirs	Infectious dose	Diameter
Campylobacter	1-3 days	3-6 days	Animals	1,000	>0.2 µm
EHEC	3-8 days	10 days	Humans	100	>0.7 µm
Adenovirus	2-7 days		Humans, birds, mammals and amphibians.	N.A.	80 nm
Calicivirus	12-48 h	3 days	Humans	N.A	35-40 nm
Cryptosporidium	7 days	N.A	Humans and livestock	<10	4-6 µm
Giardia	N.A	Up to a year	Humans and animals	<10	8-12 µm

3.4.2 Giardia

Among the *Giardia* species, *Giardia lamblia* is the one of particular interest causing human infection. *Giardia* can multiply in many animals and humans. The infectious dose is fewer than 10 cysts. The cysts can survive up to months in fresh water, like *Cryptosporidium* oocysts. The symptoms can last for a year in some cases. *Giardia* cysts are more resistant than enteric bacteria to chlorine but less resistant than *Cryptosporidium. Giardia* is sensitive for freezing, while most of the other pathogens live longer in cold temperatures.

3.5 Indicators organisms

Indicator organisms are used to reduce the time and cost when analysing the water quality. Indicators are used to determine if the water is contaminated by faecal pathogens. They are also used to analyse the water in the waterworks to evaluate the efficiency of the treatment processes and the disinfection. If an indicator organism is found after the treatment the treatment is not adequate.

As stated above the coliform group includes both faecal and environmental species of bacteria. One criterion for being an indicator is that it should not be able to grow in water, however some groups of total coliform may grow in water especially the heterotrophic coliforms (WHO 2008). When total coliforms are used as indicators it is suggested to be determined if it is faecal coliforms or not, in order to be useful as an indicator.

Due to the protozoa and viruses resistance to disinfection, the total coliforms and *E. coli* are not appropriate indicators for these pathogens but for *Campylobacter* it is an appropriate indicator (WHO 2008). However, the absence of total coliforms may show that there is no faecal contamination of the water.

4 Contamination events in Swedish water supplies

The number of reported waterborne outbreaks in Sweden is limited. It is mainly when many people are affected, i.e. infected, the outbreak is recognised as a waterborne outbreak and is further analysed (Malm et al. 2010). However, larger *Cryptosporidium* outbreaks occurred 2011 in Skellefteå and 2010 in Östersund. Furthermore, in Lilla Edet 2008 *Calicivirus* caused a larger outbreak (Ekvall 2010).

When only a few people are infected the event will in most cases not be possible to identify as a waterborne outbreak since there always are a number of infected people due to various reasons. When analysing waterborne outbreaks it is therefore important to investigate as many cases as possible in order to obtain sufficient information about the causes and their magnitude. In this survey, reported news about disturbances in Swedish drinking water supplies have been reviewed to analyse possible contamination events. The disturbances are reported in the news when different types of unwanted events happen in the drinking water systems. The events include all from a false positive water analysis to a full contamination outbreak. It is only news that is reported in the media and no further investigations of those events are made. It must be noted that there probably also is a large number of relevant events not reported.

The outbreaks reported to the Swedish National Food Agency are 33 cases for the period 1995-2003 (Lindberg and Lindqvist 2005). This can be compared with the 309 reported events with disturbances in the drinking water system in the media for the period 2000-2008 (Malm et al. 2010). In many of the events an indicator organism is found in a sample but no reports of infected people.

The events reported in the media cover a wider range of events, from just one positive sample to a full outbreak of a pathogen, not only the confirmed outbreaks as for the Swedish National Food Agency. The information in the news is most of the time limited but sometimes the cause of the event is reported as well as other relevant circumstances and the identified pathogen. However, based on the large number of reported events a good understanding of the causes can be obtained. With more events reviewed it is easier to get an idea of the causes and the large number of the hidden statistics that are expected to be in the area of drinking water-related risks.

In some cases there are incidents where there is an increased microbial risk but the water is still safe to drink since proper measures have been taken by the water utility.

Among all the 309 events reported in the media during 2000-2008, 155 of them where related to microbial contamination and groundwater supplies. Based on the same news articles reviewed in this thesis, (Malm et al. 2010) conclude that the total reported amount of people infected from groundwater supplies are about 3,500 people of the 35,000 people that in total are connected to the drinking water supplies where people have been infected.

4.1 All reported events

In 104 of the 155 events there were information about the number of days with boiling request and/or if chlorination was added in the distribution system (Figure 4.1). The sum of these days was 1,433, see appendix 1, with a yearly average of 160 days with disturbances. The number of days reported multiplied by the number of connected

people for each event will give about 303,000 customer days of disturbances each year, see appendix 1.

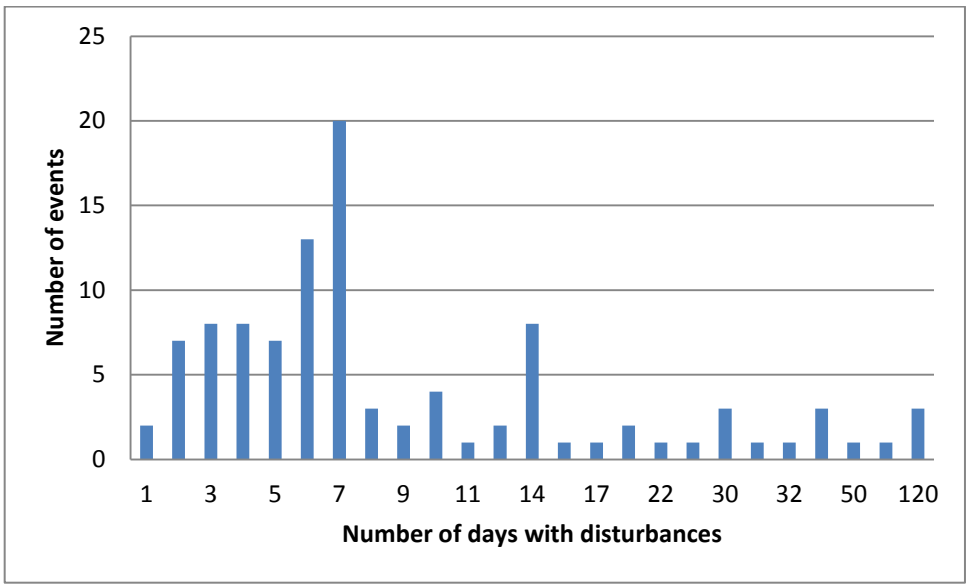

Figure 4.1 The numbers of days with microbial-related disturbances in groundwater supplies and its occurrence during 2000-2008.

4.2 Events with boiling request

When necessary, municipalities can inform the public and request them to boil the water used for drinking and cocking. This is usually done when pathogens in any form is found in a sample analysis of the drinking water. The boiling request can be an indication of a disturbance in the drinking water system. Most of the boiling request occurred between July and October as can be seen in Figure 4.2.

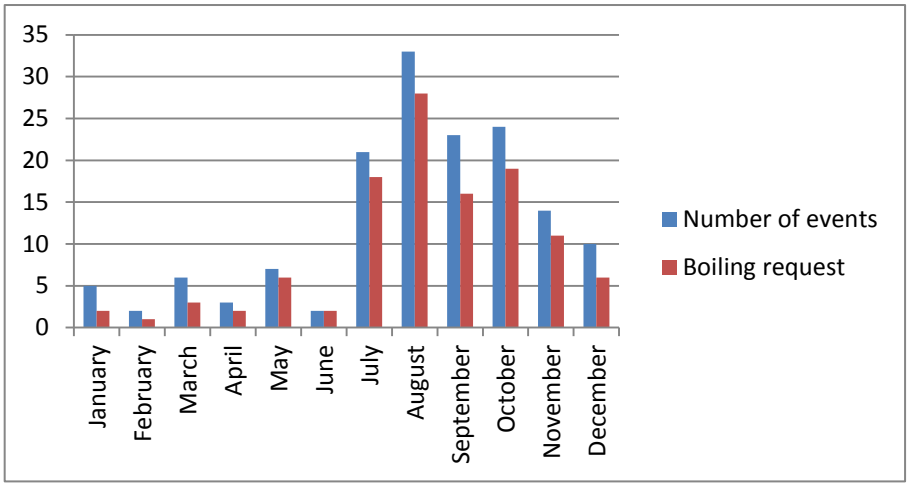

Figure 4.2 Disturbances and boiling request reported in the media during 2000-2008, associated with groundwater and microbial risk.

In total 114 events lead to a request that the people should boil the water used for drinking and cocking. In 42 of the events it occurred or followed after heavy rain,

according to the news. In six of these cases there were people reported infected after drinking the water, with a total of 132 people. In most cases when an indicator organism was found a boiling request was sent out, only in some cases the result of an additional analysis was reported in the media to confirm the presence of a contamination.

Most of the events (66 %) occurred in smaller supply systems. Of the 155 events, 102 occurred in systems with less than 1,000 people connected. These results are similar to the 70 % reported in the study by Lindberg and Lindqvist (2005).

4.3 Flooding or heavy rain

In 50 of the events it was reported in the news that the event was caused by heavy rain or flooding of the water source. The differences between the years are not big but there are great differences between the months. Most of the cases were there also have been reported heavy rain were during July to October, see Figure 4.3. There is a big possibility that this numbers are underestimated since it may not be sure that they report heavy rain in the same news item that they write about the disturbance in the drinking water.

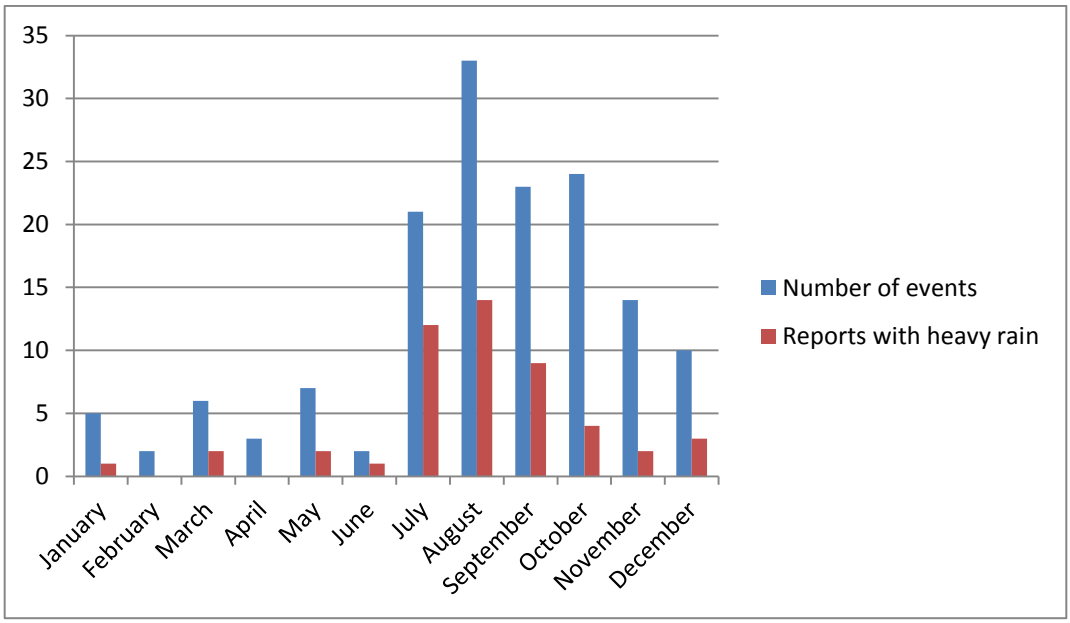

Figure 4.3 Total number of media reports with disturbances during or after heavy rain associated with groundwater and microbial risk during 2000-2008.

4.4 Infected people

In the cases where people were reported infected due to the drinking water, coliform bacteria where commonly found when a large number of people were affected. In one of the cases where coliforms where found and 3,000 people were infected, *Campylobacter* was the causing agent.

Although there was only one event where virus had been isolated and confirmed, *calicivirus* had the largest percentage of people infected (Figure 4.4). In 11 of the

cases it was stated that people got infected by the drinking water, with a total of 4,286 people infected during the nine years.

It is difficult to detect the agent causing an infection. In 43 of the cases no agent where found. The hidden statistics are huge and hard to measure and quantify. For the events studied here, the causing agent was only detected in two cases. It is the two cases where most people were infected. *Campylobacter* caused the largest event in Söderhamn 2003 were 3,000 of 22,000 people were infected. In the second largest case *calicivirus* infected 400 of 1,500 people in Malung 2002.

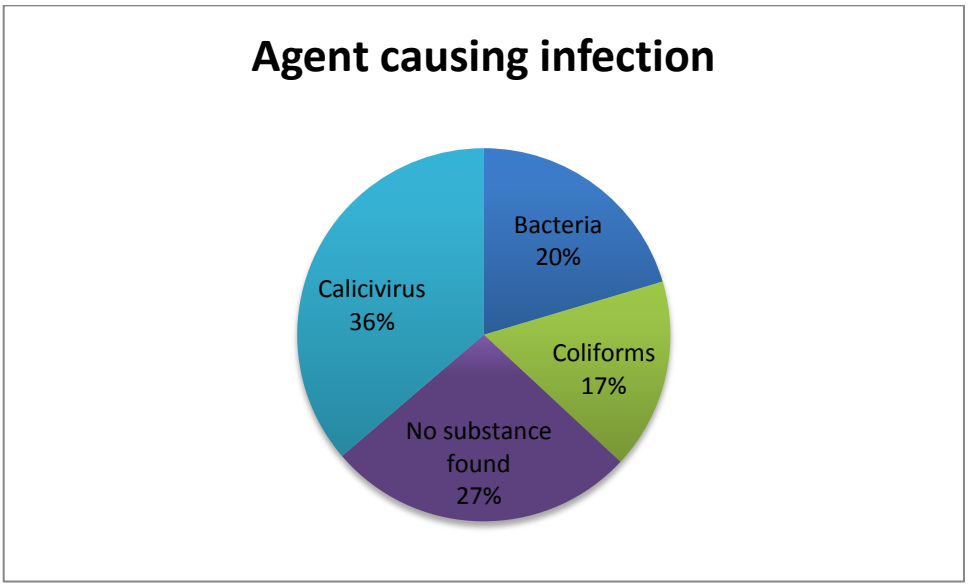

Figure 4.4 The percentage of total number of infected people associated with outbreaks in groundwater supplies and the causing agents during 2000-2009.

The number of people reported infected by drinking the water from small waterworks (< 1,000 people connected) were 116 out of 3,136 in total, this is about 4 %. The small waterworks are associated with many risk-related events of infection but only a small percentage of people that become infected by drinking the water.

A similar study based on data from 1995 to 2003 (Lindberg and Lindqvist 2005), showed that the amount of people reported infected by the drinking water from small groundwater works were 15 %, 1,582 people of 10,697.

4.5 Uncertainties

In 46 of the 155 events sample failure according to the reports in the media where suspected. The sampling is extremely sensitive. One small finger print in the sampling bottle is enough to ruin the sample.

Some indicator bacteria are commonly used to determine if there is a risk of faecal contamination. In most of the cases where coliforms were found in a sample there was no information if it was faecal coliform or not.

Detection of virus in water is time consuming, expensive and inaccurate (Toze 1999). Bacteria are easier to detect and quantified but still this procedure encounter difficulties in isolation, cultivate them and identify them. Some strains of bacteria can enter a state when they do not cultivate and can thereby not be identified by cultivation (Toze 1999). The presence of pathogens in the water is low. Even during an outbreak, most of the sample can be negative (Lindberg and Lindqvist 2005). Since most of the pathogens are not homogeneous distributed it is even more difficult to get a fair estimation of the risk.

The presence of indicator organism is not necessary correlated with the presence of pathogens. The survival characteristic of the indicator bacteria and the pathogens are different, especially for the enteric viruses and protozoa (Hellard et al. 1997). In a study by Payment and Locas (2011) the positive samples of virus were investigated and in all of them total coliforms were present and *E .coli* was present in 7 of 10 samples.

5 Pathogen transport and removal

The transport of viruses, bacteria and protozoa in water sources is a key aspect that in the end affects the health risk the drinking water consumers are exposed to. Knowledge on transport processes through saturated and unsaturated porous media and the reduction of pathogens are of significant importance in order to protect the water sources. The distance, or retention time, between the pollution source and the abstraction well is one of the most important factors affecting the reduction. Site specific conditions will determine what retention time is necessary in order to achieve the necessary pathogen reduction and thus an acceptable risk level. There are several processes such as physical, chemical and biological processes that govern the transport and the reduction of microbial pathogens. If, for example, the water velocity is high the flow through larger pore will increase resulting in less filtration and adsorption.

5.1 Schmutzdecke

Schmutzdecke is a biological layer that is formed on the surface of the sand in a slow sand filter. This layer is important for the removal of microorganisms (Bauer et al. 2011). It is in the uppermost soil layers where the biological activity is greatest and the reduction of pathogens generally is largest (Jin et al. 2000; Oliver et al. 2006).

In a study by Hijnen et al. (2007) the sand were analysed after a filtration experiment. They found that the highest accumulation of *Cryptosporidium* oocysts were in the top 5-10 cm of the sand (the total depth was 1.5 m). After the dosage of oocyst stopped the breakthrough continued for about 50 day's demonstration the reversible attachment. Also some known oocyst predation planktons were observed in the sand and it was concluded that they were possibly the cause of large reduction of the *Cryptosporidium* in the filter. The reduction rate was observed and from the mass load a log-linear inactivation rate was estimated. The inactivation rates of oocyst in the sand filter were 0.014-0.02 log10 reduction per day. *E. coli* has also shown to be able to accumulate in biofilms and if the conditions are changed, e.g. increased flow, a detachment of *E. coli* from the biofilm can be observed Wang et al. (2011).

5.2 Inactivation

The different pathogens show different inactivation or die-off rate in sand filtration. The enteric viruses persist longer than bacteria. Also the environment is governing the inactivation rate, the same species have different inactivation rate in different environments as well as the different strains within a group. Lower temperature, increased moisture and increased pH tend to reduce the inactivation rate (Oliver et al. 2006). The inactivation of pathogens is more well known than the other reduction mechanisms, and for most of the pathogens the inactivation rate can be described as a first order process (Oliver et al. 2006). For a summary of the inactivation rates see Toze (2003) page 74-75 and Oliver et al. (2006) page 67-68.

To estimate the reduction due to inactivation (Bauer et al. 2011) compared the reduction to an earlier made experiment with similar conditions where the half time inactivation were measured. According to Dizer et al. (2005) the half time inactivation (T50) at a temperature between 11 and 25 °C were 2 days for both somatic and K13

bacteriophages. At 400 cm/d and 50 cm depth the log10 removal due to inactivation will be 0.07 of the total 2.38 log10 reduction. At 90cm depth the inactivation will be 0.55 of total 3.19 log10 reduction (Bauer et al. 2011). According to this comparison the inactivation did not contribute much to the total reduction, however, the residence time were only some hours.

According to Oliver et al. (2006) bacteriophages X174, MS2 and PRD1 have an inactivation rate between 0.01-0.04 per day (temperature 10-12°C). This indicate that the inactivation play a minor role in the reduction of those bacteriophages and most likely for all pathogens.

5.3 Straining

The straining removal is physical filtration, i.e. blocking of pathogens that are larger than the pore space (Sen 2011). Straining is one of the controlling factors when the pore space is smaller than the size of the pathogens, see Figure 5.1. However, the larger pathogens such as *Giardia* have a diameter of 8-12 µm. To prevent *Giardia* by straining the granular medium diameter will be much smaller than sand, clayey-silt have typically small pore space enough (<4 µm) to prevent migration by straining (Oliver et al. 2006). The material in artificial groundwater recharge and slow sand filtration is larger than that and therefore the straining is assumed to play a minor role. Bradford et al. (2002) suggested that straining should be considered when $dp/dm > 0{,}0017$, where (dp) is the colloid diameter and (dm) is the median collector diameter.

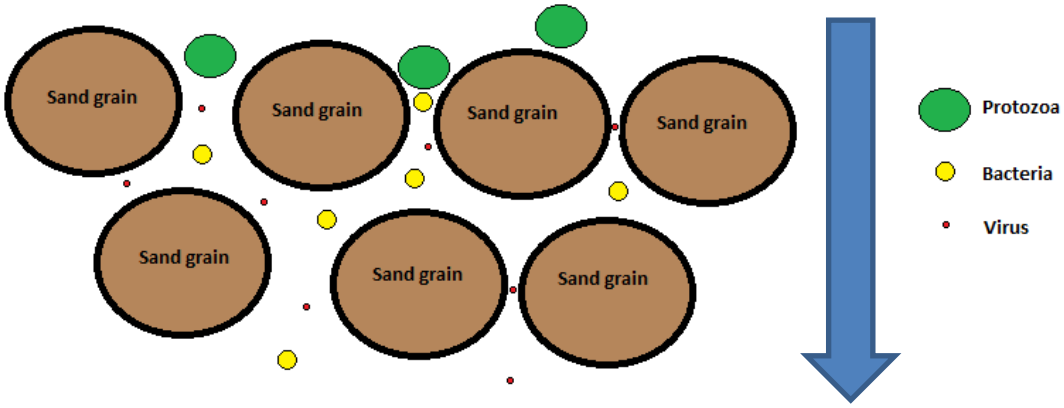

Figure 5.1 conceptual model of straining of the protozoa.

5.4 Saturated conditions

In the saturated flow the transport mechanisms are advection, dispersion and diffusion (Oliver et al. 2006). Factors affecting the pathogen removal in the saturated zone are inactivation (die off), adsorption, filtration and predation.

5.4.1 Adsorption

Three main processes control the movement and fate of microorganisms in porous media. These are transport, attachment/detachment and inactivation. The attachment to grain surfaces is a reversible process where both equilibrium and kinetic mechanisms need to be considered.

The colloidal filtration theory (CTF) can be used to describe the attachment and detachment of microorganisms to sand in slow sand filtration for saturated conditions:

$$\ln(C/C0) = -\frac{3}{2}\frac{(1-\epsilon)L\alpha\eta}{dc} \quad (1.1)$$

C – the influent pathogen concentration

C0 – the effluent pathogen concentration

dc – the diameter of the sand/soil grains

ϵ – the bed porosity

L – the length of the bed

η – the single collector collision efficiency

α- the attachment efficiency

η0 – the theoretical single collector contact efficiency (η0=α*η)

The single collector collision efficiency η is a theoretical parameter based on the physical processes in colloid transport and the interactions between the colloid and the collector (Hijnen et al. 2007). The colloidal filtration theory assumes complete removal by sorption, no removal due to straining. Straining have been shown to influence the removal although it is not considered to be an important mechanism (Tufenkji 2007). Straining of, for example, *E. coli* in course sand is likely to be minimal (Wang et al. 2011). It is also assumed that the saturated medium is homogeneous and that the particles is spherical in shape (Tufenkji 2007).

The experimentally measured attachment efficiencies are much higher than those calculated using the CFT theory (Tufenkji 2007). One possible reason for this might be that the CFT does not consider that the deposition may cause blocking of other particles or detachment of the particles.

The CTF theory can be used preferably to compare the microbial attachment behaviour in soil. The spatial distribution of the retained microorganisms together with the microorganism concentration in the influent will give a characterization of the removal performance in the filter media.

5.4.2 Charges

For the saturated transport the organism's isoelectric point is of interest. The isoelectric point is the pH at which the organisms have a net surface charge of zero. The higher the pH of the solution is compared to the organism, the more negative surface charge the organism gets. Under most conditions the microbial pathogens carry a negative charge. Most of the mineral surfaces also carry a negative charge. Under these conditions most pathogens will be repelled by the grain surfaces and a highly negative charge between the collector and the organism cause a stronger repelling force (Oliver et al. 2006). Positively charged minerals (e.g. iron hydroxides) will promote the adsorption of negatively charged viruses (Bhattacharjee et al. 2002). A decrease in the ionic strength, such as heavy rain, may cause detachment.

5.4.3 Attachment efficiency in sand and soil

In a study by Schinner et al. (2010) the transport behaviour of five bacterial pathogens through high purity quarts sand in columns were analysed. The sand mean diameter (d50) was 0.763 mm and the coefficient uniformity (d60/d10) was 1.2. The ionic solutions (IS) were 10 and 30 mM KCl with a pH at 5.9. The porosity of the sand was 0.35.

The bacteria studied were *E. coli* O157:H7, *Y. enterocolitica*, *E. faecalis*, *M. aeruginosa* and *Anabaena flosaquae*. The two latest are cyanobacteria.

The study by Schinner et al. (2010) also included an analysis of filtration through natural agricultural soil, where *E. coli* O157:H7 and *E. Faecalis* were considered. The soil consisted of 85 % sand, 9.7 % silt and 5.3 % clay. The porosity of the agricultural soil was 0.42. The soil main mean diameter (d50) was 0.240 mm and coefficient uniformity (d60/d10) of 2.4. The grain size distribution of the agricultural soil was bimodal distributed. The heterogeneity in the soil may impact the straining potential.

The cell size and shape are important for colloidal transport (Salerno et al. 2006). There was a significant variability in cell size within the populations, mean diameter can be seen in Table 5.1 This may result in an over or underestimation of the transport behaviour if the size distribution is wide (Schinner et al. 2010).

Table 5.1 The diameter of pathogens analysed in the literature (Schinner et al. 2010).

Bacteria	**Mean equivalent spherical diameter** (ESD_{50})
E. coli O157:H7	0.74 mm
Y. enterocolitica	0.82 mm
E. faecalis	0.84 mm
M. aeruginosa	2.6 mm
Anabaena flos-aquae	160 mm

For *E. coli* the attachment efficiency (α) is approximately 1 in both IS solutions, so the maximum deposition rate for *E. coli* is reached at very low IS, increased salt concentration will not increase the deposition rate. The mean diameter of the *E. coli* organism (0.74 mm) is slightly smaller than the sand collector diameter (0.76 mm).

For the agricultural soil experiment the attachment efficiency depends on the mean collector diameter. The mean diameter for the soil may not be representative due to the heterogeneity, the calculated attachment efficiency (α) may be overestimated. For the *E. coli* $\alpha_{average}$ is 0.082 which is much lower than for sand. For the *E. faecalis* α is 0.39 at 10 mM IS and 0.68 for 30 mM IS.

The mean collector diameter for the soil (0.24 mm) is smaller than the mean diameter of all the microorganisms. The big difference in the α value between the quartz sand and agricultural soil show that the α value is different for each media and may cause an overestimation of the removal efficiency if not sufficient evaluated. This shows how the collector size properties influence the removal of the pathogens in saturated flow.

5.4.4 Flow velocity and depth depending reduction in small scale field experiment

Wastewater influenced surface water were filtered through a sand filtration facility and the removal of bacterial faecal indicators, coliphages and enteric *adenoviruses* were analysed in a study by Bauer et al. (2011). The pathogens studied were somatic phage, K13 bacteriophages, enteric *adenoviruses* and faecal bacteria. The dose of wastewater was adjusted to get a concentration of 104 somatic bacteriophages per 100 ml.

The sand in this study was free from clay material and had a size d10 of 0.269 mm. The ratio between d10 and d60 was 2.42, and the porosity was 0.32.

Samples were taken at five different depths: 30, 60 90, 120 and 150 cm. The facility was built to utilize river bank filtration (RBF), with both vertical and horizontal flow and a tangential current to the filter surface. The water level was 30 cm above the surface. The tangential flow might influence the formation of schmutzdecke du to shear stress.

The temperature during the 67 days of the experiment was between 4 and 9 °C. The pore water velocities (PWV) were adjusted between 50, 100, 200 and 400 cm/d. The highest removal of the bacteriophages was at 50 cm/d. However, the removal did not differ substantially for the PWV rates at 100, 200 and 400 cm/d.

As can be seen in Table 5.2 the log10 removal increased with depth. The study by Bauer et al. (2011) also a showed a higher reduction rate in the uppermost layer of the sand filter. The temperature also showed to influence the removal, a second experiment with identical design was conducted by Bauer et al. (2011) with water temperature between 11 and 25 °C, where the removal rate where generally higher.

Table 5.2 Log10 removal at PWV 50 cm/d (Bauer et al. 2011).

Pathogen	30 cm depth	60 cm depth	90 cm depth
Somatic phages $\log_{10}$ removal	1.55	2.03	3.19
K13 phages $\log_{10}$ removal	1.69	2.17	3.25

A second filtration experiment was conducted by Bauer et al. (2011) in a conical formed filter. In this filter the water were infiltrated vertically. The water level was 20

cm above the filter surface and the sampling occurred at 90 cm depth. The PWV was 900 cm/d. The examined pathogens were somatic and K13bacteriophages, enteric adenoviruses, intestinal enterococci and *E. coli*. The log10 removals of K13 bacteriophages at 900 cm/d were larger than at 50 cm/d, this indicates that the velocity does not play the most important role at those flow rates. The measured concentration and the log10 removal can be seen in Table 5.3.

Table 5.3 The log10 reduction at 90 cm depth and PWV at 900 cm/d (Bauer et al. 2011).

Pathogen	**Inflow concentration** (Organisms/100 ml)	**Measured concentration at 90cmdepth** (Organisms/100 ml)	**Log_{10} removal**
Somatic bacteriophages	19,735	36	2.74
K13 bacteriophages	7,675	2	3.58
E. coli	195,273	15	4.11
Intestinal enterococci	35,920	8	3.65
Adenovirus	7,600 +- 2,180	<10	>2.88

The influent phage concentration varied from day to day depending on the quality of the wastewater. This variations increase the uncertainty in the removal rate calculations. In addition to the temporal variability also measurement errors contribute to the uncertainty.

5.4.5 Delayed breakthrough for *Cryptosporidium* and *C. perfringens*

Cryptosporidium parvum oocyst and spores of the bacteria *C. perfringens* (used as a surrogate) were added in the influent through a slow sand filter in a study by Hijnen et al. (2007). The sand in the filter bed had a diameter d50 of 0.28 mm and a porosity of 0.41. The depth of the sand was 1.5 m and the flow through the sand was 720 cm/d. The filter had been running prior to the experiment and a schmutzdecke was already present.

The temperature was between 8.2 and 18.8 °C and the pH was 8.0. The effluent was sampled during the dosage and 22 weeks after the dosage had stopped in order to determine the delayed breakthrough behaviour.

The colloid filtration model was used and the elimination in the filter bed was assumed to be only due to attachment and detachment.

The average concentration of *Cryptosporidium* oocysts in the influent was 314.6 per litre and in the effluent 0.0016 oocysts per litre. Based on the average concentration the decimal elimination capacity (DEC) reduction was 5.3 log10. A mass balance was also used in order to calculate the elimination and accumulation in the filter bed. The mass balance decimal elimination capacity (DECm) for the oocysts was calculated based on the number dosed to the influent and after passing the filter. The DECm for oocyst was 4.7 log10. The DECm is estimated to be a more realistic value due to zero counts will underestimate the average counts. The DEC value for the *C perfringens* spores was 3.9.

The accumulated numbers of oocysts in the filter were 1.8 % after 184 days and 0.2 % after 253 days. The reduction of retained oocysts in the sand was calculated to be 0.014-0.02 log10 reduction per day. This indicates a low risk of delayed breakthrough for infectious oocysts due to the rapid decline most likely caused by predation, since eight different species of oocyst predator were observed (Hijnen et al. 2007). The straining removal mechanism is consider to have a minor role of removal (Hijnen et al. 2007).

5.4.6 Summary of the logarithmic reductions during saturated conditions.

For the saturated conditions the reduction of all the pathogens from the experiments described above follows a similar pattern, see Figure 5.2. The quantities of the reduction results are, however, too few to make a generic model. If the predictions of pathogen removal are based on column or field studies with conditions different from the specific system, there is a considerable risk of overestimations (Oliver et al. 2006).

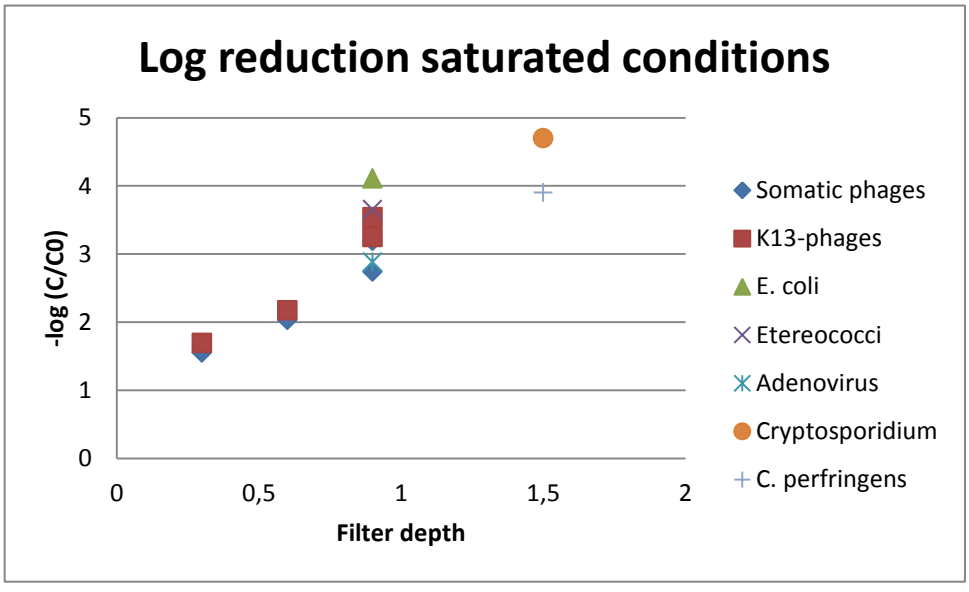

Figure 5.2 Depth and log10 reductions of pathogens in saturated conditions (from Bauer et al. (2010) and Hijnen et al. (2007)).

5.5 Unsaturated conditions

The unsaturated zone comprises the zone between the surface and the groundwater. In the unsaturated zone the hydrogeological process are more complex than in the saturated zone. This is because the presence of the gaseous phase and the flow discontinuities. The behaviour of microorganisms are also more difficult to predict in the unsaturated zone (Oliver et al. 2006; Sen 2011). The general characteristics of the unsaturated zone are presence of oxygen, gas water interface and higher content of organic matter. The reduction in the unsaturated zone plays an important role in reducing the pathogens and needs to be considered (Sen 2011). It is important for future assessments to make a model for the reduction in the unsaturated zone since most transport studies with, for example, viruses have been in saturated porous media. Not many studies have been conducted in unsaturated porous media.

In the unsaturated zone the pathogens can be captured in the liquid gas interface, the solid liquid gas interface and by straining, see Figure 5.3 (Sen 2011). The air water interface (AWI) is an important barrier for the pathogens because it can serve as a collector of colloids as microbial pathogens. The sorption of viruses at the AWI is caused by the hydrophobic interactions (Sen 2011).

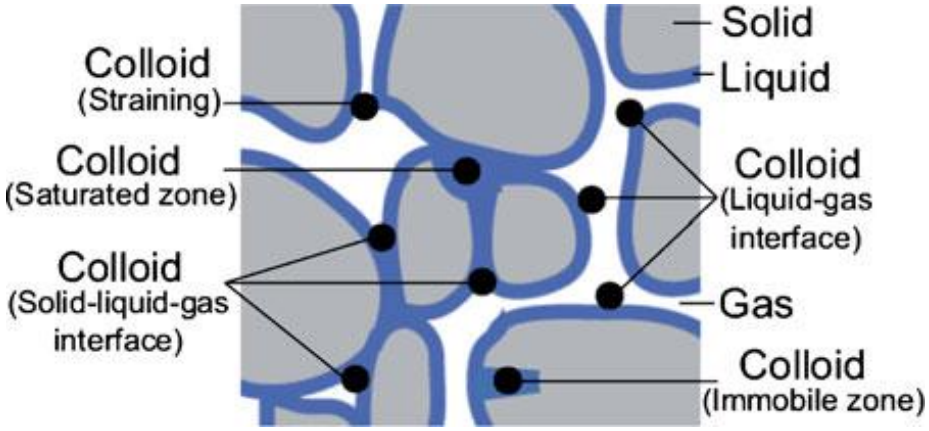

Figure 5.3 A conceptual model of the attachment of colloids in the three phases (Sen 2011).

A larger scale column study were performed by Zagerholm et al. (2007). The columns were filled with 5.4 m of soil, with the sieving characteristics d10, d60 and d90 of 0.11, 0.7 and 3 mm respectively. The total porosity was calculated to be 0.37 and the effective porosity was 0.24. The experiments were conducted under oxic conditions since there was oxygen still present in the outflow.

The bacteriophages MS2 and X174 were studied during saturated and unsaturated conditions for different flow depths, the reduction of the bacteriophages can be seen in Table 5.4.

Cl. perfringens and coliforms were also added to the influent and the effluent were analysed to estimate the reduction of the bacteria. There were no detected bacteria in the effluent so the minimum reduction of *Cl. perfringens* where 5.4 log10 and coliforms 2.7 log10.

Table 5.4 The inflow concentration (C0) and the measured concentration (C) from the Zagerholm et al. (2007) experiment.

Travelled through	**X174** (C)	**X174** Log10 (C/C0)	**MS2** (C)	**MS2** Log10 (C/C0)	**Travel time** (hours)
Inflow	7.00E+05	1	3.60E+06	1	0
2m Saturated	1.15E+04	1.78	1.3E+06	0.44	47
1m Unsaturated	1.00E+02	3.85	1.30E+05	1.44	6
2m Unsaturated	1.50E+00	5.64	2.20E+04	2.21	18
2m Unsaturated	6.00E-01	6.07	8.20E+04	1.64	24
4m Unsaturated	1.00E-01	6.85	2.00E+01	5.26	43

According to Zagerholm et al. (2007) the reduction of pathogens are larger in the unsaturated zone compared to the saturated zone. One reason can be that in the saturated zone the water flow mainly through larger pores and the contact time between the pathogen and the media decreased.

A similar study was made by Lundh et al. (2006). The columns were filled with 5.4 m of soil, with the sieving characteristics d10 and d60 of 0.04-0.07 mm and 0.21-0.50 mm respectively. The porosity was 0.41 and the conductivity 11.6-12.0. The pH was around 6.9. The infiltration velocity was 0,5 m/d. In two experiments (Experiments 1 and 2) the pathogens were added in pulses to the influent and in a third experiment (Experiment 3) the pathogens were added continuously. The studied pathogens were the bacteriophage MS2 and X174 as in the study referred to above. In experiment 1 and 2 the bacteriophages were added during 40 minutes in pulses and in experiment 3 during 7 days. The log10 reduction for the maximum and minimum measured concentrations, together with the initial concentration for experiment 3 can be seen in Tables 5.5 and 5.6. The inflow concentration is measured in plague forming units, i.e. the number of infective viruses.

Table 5.5 The inflow concentrations of MS2 in the experiment by Lundh et al. (2006) and the log10 reduction after measured meters of unsaturated zone.

MS2				
Travelled through	**Experiment 1**	**Experiment 2**	**Experiment 3** (maximum detected)	**Experiment 3** (minimum detected)
Inflow C (pfu/ml)	$1.00 \cdot 10^{10}$	$2.00 \cdot 10^{8}$	$3.60 \cdot 10^{6}$	$3.60 \cdot 10^{6}$
1m Unsaturated	2.06	1.12	0.97	1.67
2m Unsaturated	2.26	1.30	2.21	2.82
2m Unsaturated	2.66	1.52	1.64	2.19
4m Unsaturated	5.08	2.52	4.23	5.38

As can be seen in Figure 5.4 Experiment 1 have a higher log10 reduction of MS2 than Experiment 2, it may be due to the higher concentration in the inflow. The inflow concentration in Experiment 1 is 50 times higher than in Experiment 2. In experiment 3 the minimum and maximum concentration is represented. The difference between the maximum and minimum concentration is smaller than the difference between Experiment 1 and 2. The inflow concentration for all three experiments is high and the log10 reduction may be much lower with lower inflow concentrations.

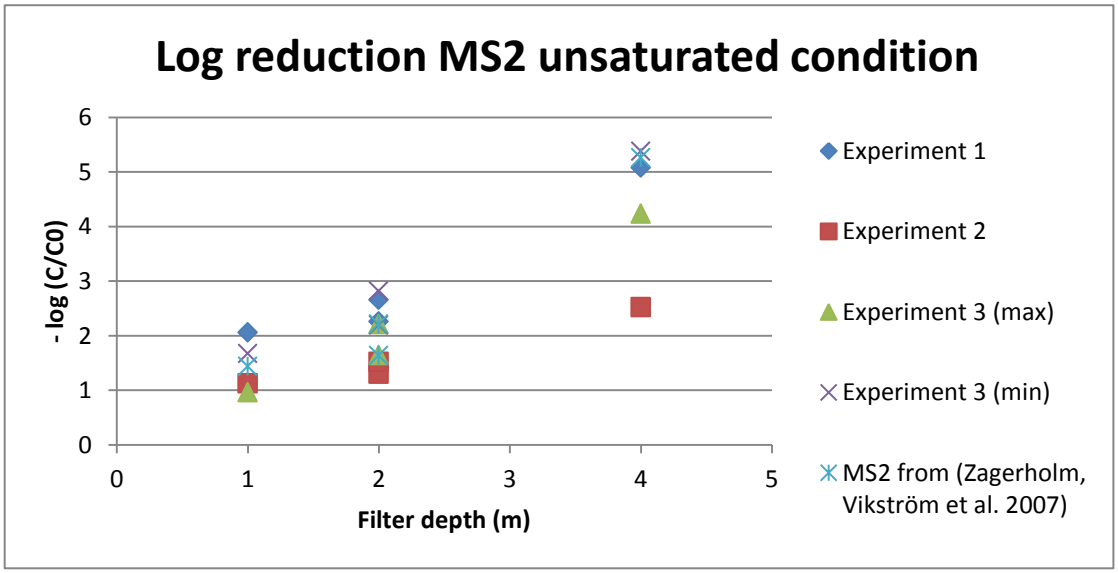

Figure 5.4 Depth and log10 reduction of MS2 in unsaturated conditions based on Lundh et al. (2006).

Table 5.6 The inflow concentrations of X174 in the experiment by Lundh et al. (2006) and the log10 reduction after measured meters of unsaturated zone.

X174				
Travelled through	**Experiment 1**	**Experiment 2**	**Experiment 3** (maximum detected)	**Experiment 3** (minimum detected)
Inflow C (pfu/ml)	$1.00 \cdot 10^{8}$	$1.40 \cdot 10^{8}$	$7.00 \cdot 10^{5}$	$7.00 \cdot 10^{5}$
1m Unsaturated	1.47	1.94	3.80	4.37
2m Unsaturated	1.89	2.18	4.43	6.02
2m Unsaturated	2.68	2.41	6.02	7.15
4m Unsaturated	4.85	3.75	5.97	7.15

For the bacteriophage X174 the difference in log10 reduction between Experiments 1 and 2 is only 1.1 log10 at 4 meters depth compared to 2.56 for the MS2 bacteriophage. For X174 the difference between the minimum and maximum detected pathogens for Experiment 3 is similar to MS2 but the log10 reduction rate seems to slow down after 2 meters, see Figure 5.5.

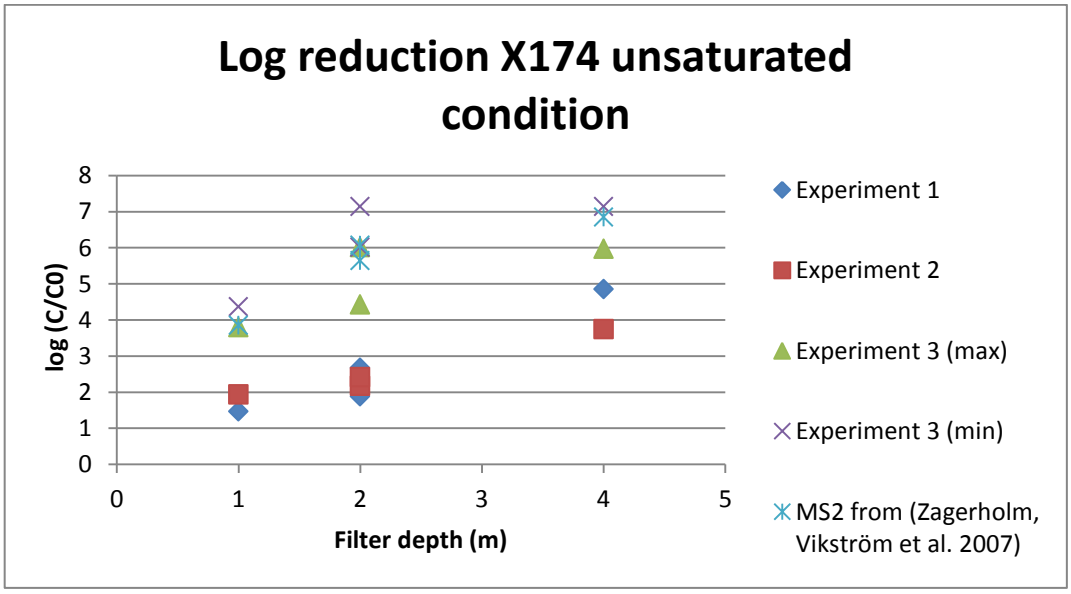

Figure 5.5 *Depth and log10 reduction of X174 in unsaturated conditions based on Lundh et al. (2006).*

The MS2 and X174 reductions for the Zagerholm et al. (2007) experiment which was deducted with the similar inflow concentrations are within the interval for Experiment 3.

5.5.1 Inflow concentration relation to reduction per filter depth

In all the experiments and reports studied in this thesis the inflow concentrations are all far above the QMRA model recommended value by Lundberg et al. (2009) recommended value. In Figure 5.6 and 5.7 the inflow concentration (particle per millilitre) can be seen together with the log10 reduction and the depth for the measurement. The two highest inflow concentrations have the lowest reduction for X174. The reduction deviation seems to be more plant specific rather than concentration specific but this need to be further investigated due to the limited amount of studies.

The MS2 reduction in Figure 5.7 show less deviation compared to the X174 reduction. The deviation in the reduction for MS2 is not only 1 log10 after 1 meter of unsaturated filtration. The inflow concentration seems to not be the most important factor in the reduction rate.

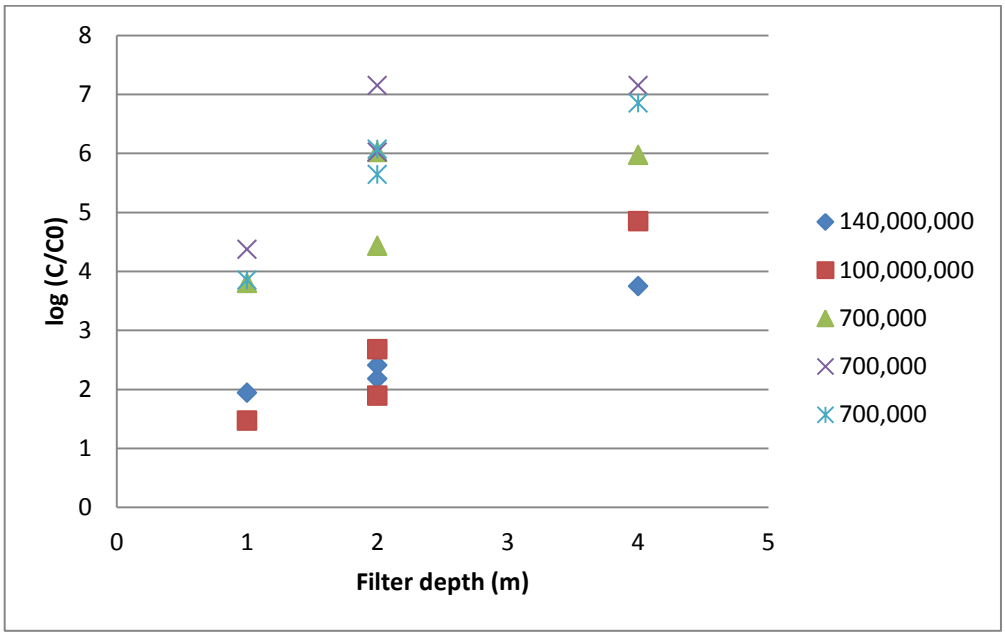

Figure 5.6 *Depth and log10 reduction for different inflow concentrations of X174 in saturated conditions (the inflow concentrations (particle per millilitre) can be seen to the right) (Lundh et al. 2006; Zagerholm et al. 2007).*

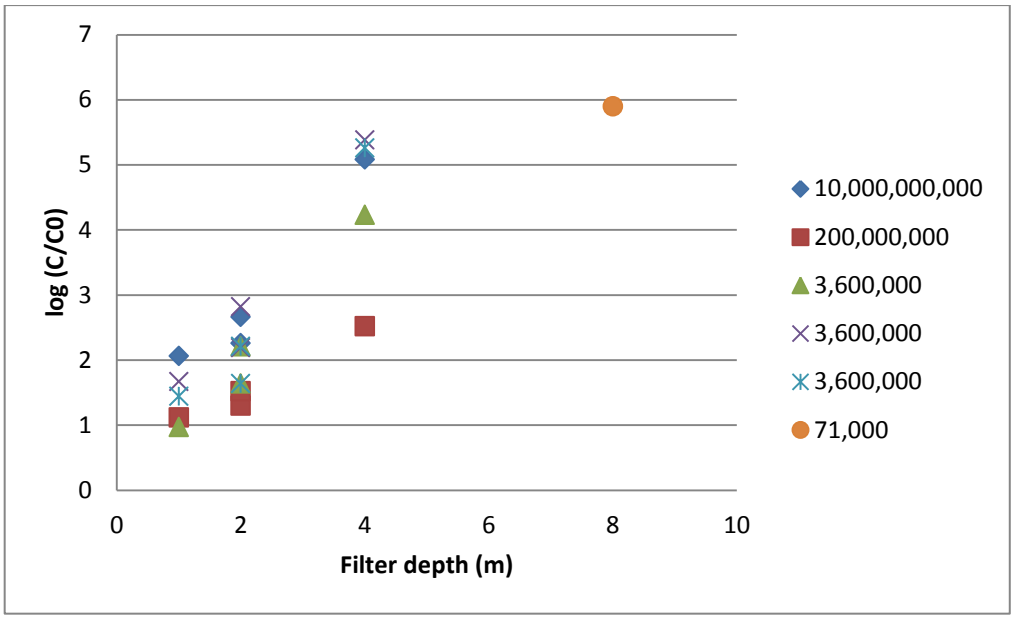

Figure 5.7 *Filter Depth (meter) and log10 reduction for different inflow concentrations of MS2 in saturated conditions, (the inflow concentrations (particles per millilitre) can be seen to the right),(Lundh et al. 2006; Schijven et al. 2000; Zagerholm et al. 2007).*

5.5.2 Retention in unsaturated conditions

To determine the role that the unsaturated flow conditions play in virus reduction in sand columns Jin et al. (2000) conducted several column flow experiments in both saturated and unsaturated columns (21.7 % saturation). The bacteriophages studied were MS2 and X174 filtered through a 10 cm long column packed with well sorted quarts with a mean diameter of 0.60-0.85 mm. The removal for both bacteriophages increased under unsaturated flow conditions, for X174 the removal was more than 40 times more during unsaturated conditions.

For the saturated system 70 % of the retained MS2 could be eluted. In the saturated and the unsaturated studies the same percentages of X174 were recovered indicating increased removal due to increased sorption, not inactivation.

The retained MS2 bacteriophages in the unsaturated column could not be recovered and were therefore suspected to be removed due to inactivation, whereas the retained X174 could be recovered and stayed viable.

The reduction of microorganisms in unsaturated flow is associated with the air water interface (AWI). This interface does not exist in saturated system. The AWI seemed to affect MS2 more than X174.

In the saturated experiment the X174 reduction was larger than for MS2. The different saturated transport behaviour might be explained by the different isoelectric points for the different bacteriophages. The isoelectric point is the pH where the molecule has a zero net charge at the surface. For X174 the isoelectric point is at pH 6.6 and for MS2 it is at pH 3.9. The electrostatic repulsion for MS2 is greater due to the more negative charge at pH solution of 7.5.

The forces that governing the sorption are electrostatic attraction, repulsion, van der Waals forces, covalent ionic interactions and hydrophobic effect (Jin et al. 2000). However, there are many processes occurring during the transport and inactivation, many of them are yet to be known and poorly understood.

5.5.3 Deep well injection

A field study of a pilot deep well infiltration and its removal of microorganisms were performed by Schijven et al. (2000). The influent water were highly concentrated with bacteriophages (MS2 and PRD1) and *E. coli* (WR1). Instead of *Cryptosporidium*, spores of *Clostridium bifermentants* (R5) were used as surrogates. Monitoring wells were placed at the distances 8, 12, 22 and 38 meters from the injection well. The temperature of the water was 12°C. The porosity was 0.32. The groundwater in the study was anoxic and the calcium content lower than 0.3 %. The average grain size diameter was 0.27 mm.

The inactivation was assumed to be much smaller than the detachment. The collision efficiency α for MS2 and R5 were $1.4*10^3$ and $8*10^3$ at 8 meters from the injection well. Those low values show unfavourable conditions, sandy soil with high pH that increases the electrostatic repulsion.

Within the first 8 meters from the well ferric oxyhydrates precipitated, providing positive charged patches, where the negative charged microorganisms may be attached. From the injection well and 10 meters there were oxygen present, after 10 meters the condition was anoxic. In the oxic area there were aerobic bacteria present that may enhance the inactivation of the bacteriophages and viruses (Schijven et al. 2000).

After 8 meters from the well MS2 had been reduced by 6 log10, when comparing this value with Figure 5.4 it seems to follow the pattern for unsaturated condition.

The following 30 meters it was reduced only about 2 log10 more. R5 was reduced 5 log10 the first 8 meters and the following reduction was negligible, as is can be seen in Table 5.7. WR1 was reduced by 7.5 log10 the first 8 meters. The logarithmic reduction was estimated to decrease nonlinear with the distance. The reduction in the oxic area is much higher than in the anoxic area, this support the increased reduction due to AWI.

The inactivation rate was analysed at the injection well and at 8 meters from the well. The inactivation rate at the well was 0.081 per day for MS2, 0.060 per day for PRD1 and 0.063 per day for WR1. Eight meters from the well the inactivation of MS2 was 0.039 per day. To verify the measured concentration the numbers of microorganism passed were also measured, there were no difference between the log10 removals.

Table 5.7 The measured concentrations for the selected pathogens and the depths studied by Schijven et al. (2000).

Place	**MS2**		**PRD1**		**R5**		**WR1**	
	Organisms per litre	Log_{10} (C/C0)	Organisms per litre	Log_{10} (C/C0)	Organisms per litre	Log_{10} (C/C0)	Organisms per litre	Log_{10} (C/C0)
Well	$7.1 \cdot 10^{7}$		$4.3 \cdot 10^{5}$		$1.6 \cdot 10^{4}$		$1.1 \cdot 10^{6}$	
8m	95	5.9	0.89	5.7	0.29	4.7	0.03	7.5
12m	4.6	7.2	0.038	7.1	0.70	4.4	-	-
22m	0.73	8.0	-	-	0.097	5.2	-	-
38m	0.26	8.4	-	-	0.16	5.0	-	-

6 QMRA

The Swedish Water & Wastewater Association financed a project in which a model for quantitative microbial risk assessment (QMRA) for drinking water production was developed. The model was developed as a result of an increasing interest for proactive and risk-based approaches, which is emphasised within the drinking water sector. A risk-based approach is preferred compared to the end-product testing in order to find and reduce the risks. A proactive approach aims to reduce the risks, i.e. prevent unwanted events before they occur. The purpose of the QMRA model was to provide a general model that can be used by the drinking water producers, i.e. mainly municipalities. A drinking water producer may provide water to many people and microbial contaminants in the drinking water may cause serious consequences. A model as the QMRA can provide important information to the work of managing risks. The model is available for downloading at www.svensktvatten.se, see also (Lundberg et al. 2009).

The user of the model can define the different treatment processes used within a treatment plant and study how it reduces pathogens. The user can change treatment steps and try different scenarios such as changes in raw water quality and treatment failure. The model can assist in making a risk assessment of the waterworks and make a stress test of it.

The model can also be used to quantify the pathogen reduction necessary to get an acceptable risk for a specific water source. In order to get relevant and accurate results from the model it is important that the input data reflect the reality as much as possible.

6.1 The QMRA model

The model can be used to study a water treatment plant and find the weaknesses, highest acceptable pathogen concentration in the raw water, run failure simulations etc. Data and probability distributions for presence of pathogens are available in the model together with the reduction ability for the common treatment processes in Sweden. A dose response relationship is also included in the model for calculating the probability of infection.

In order to ensure as accurate results as possible from the QMRA model the pathogens in the source water needs to be assessed together with the performance of the treatment barrier. The strengths of the model are scenario analysis and comparisons of process combinations. The model is mainly developed for surface water as the water source. For the artificial groundwater recharge the transport length or residence time should be considered when estimating the infiltration reduction of pathogens. In Figure 6.1 a flowchart of the QMRA model is showed, the ovals are the input data, the rounded rectangle are the model processes and the rectangle are the results.

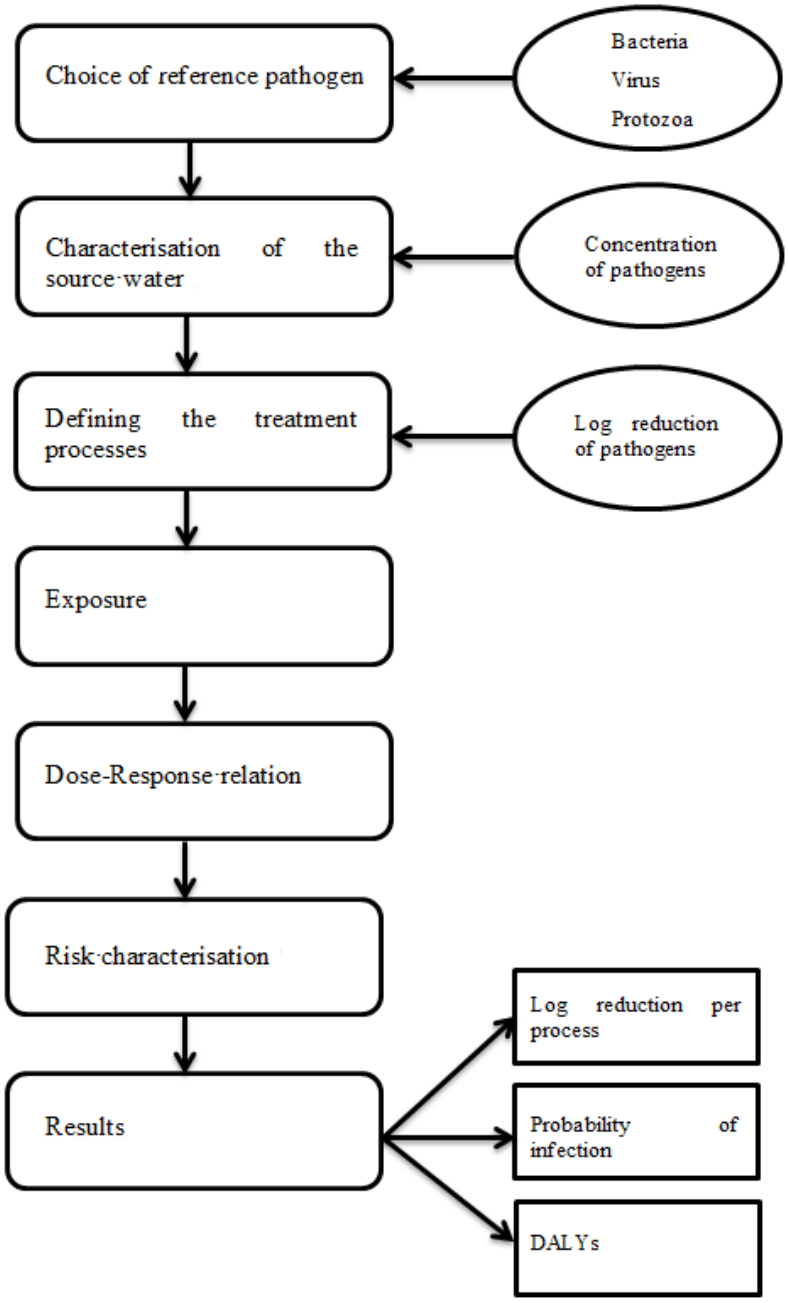

Figure 6.1 Flowchart of the QMRA model.

6.1.1 Reference pathogen

The first step when applying the QMRA model is to define the pathogens to be analysed. The available pathogens in the model are *Campylobacter*, *E. coli* 0157 (EHEC) and *Salmonella* for representing bacteria. Adenovirus, rotavirus and *norovirus* are representing viruses, and *Cryptosporidium* and *Giardia* are representing protozoa.

The model is made primary for the above mentioned pathogens and other representative pathogens should not be used.

6.1.2 Characterisation of the source water

To characterize the source water there is two ways, one way is to describe the amount of the selected pathogens in the source water, a log-normal distribution is often used for the presence of pathogens concentration in the raw water. The other way is to describe specific point sources of wastewater which will reach the water inlet. In the latter case the model considers the inactivation and dilution for the selected pathogens. There are also default values for typical surface water sources that can be used if no data are available. The default values can be seen in Table 6.1, two of the pathogens are represented as distributions and the rest are given as point values.

Table 6.1 The default values for pathogen presence in surface water used in the QMRA model (Lundberg et al. 2009)

Pathogen	**Default concentration** (Organisms/l)
Campylobacter	1
Salmonella	1
EHEC	0.1
Norovirus	1
Rotavirus	Log-normal: mean=1; stdev=3
Adenovirus	1
Giardia	0.5
Cryptosporidium	Log-normal: Mean=0.4; stdev=2.1

6.1.3 Defining the treatment process

The third step in the model is to define the treatment processes. The model contains the common treatment processes used in Sweden. It is also possible to define and add additional processes. For every process the number of parallel lines should be added. All the parallel lines in the treatment process are assumed to be the same, this is simplifies for failure simulations. However, when wells for artificial groundwater are used the depths and retention time are different and this is not considered in the QMRA model. It is not possible to define wells for the artificial groundwater, therefore all the values regarding the infiltration must be entered manually.

When defining the reduction of pathogens in the barriers, the reduction can be entered either as a distribution or as a point value. For a point value the in and out concentration of the pathogens are the input data and the reduction is calculated. For the distribution the choice of distribution and the values can be added. If the treatment performance is unknown, default values can be used. The reliability of the data and sub-optimal condition are also possible to define.

6.1.4 Exposure and Dose-Response

For the exposure, default values or local exposure data can be defined. The dose-response relations are predefined and no choice can be made here, the model will calculate the dose-response relationship from defined default data.

6.1.5 Results

If the input data is defined using distributions, which often is the case, the results of the QMRA model are presented as distributions of the log reduction per treatment process, the probability of infection and in DALYs (Disability Adjusted life Years).

The DALY is here a measure of the loss of years of healthy life due to a disease spreading through the drinking water system. One DALY is one year loss of healthy life. The DALYs can preferable be used to compare different treatment processes (Lundberg et al. 2009).

6.2 QMRA and groundwater infiltration

The QMRA model does not consider artificial groundwater infiltration or sand filtration deeper than 1.4 meters. In order to make the model work for artificial groundwater infiltration it is necessary to enter data on the reduction manually. It can be done as an additional treatment step or enter the chosen reduction values in the slow sand filtration treatment. There is only limited data available for reduction through sand filtration in the model.

To make a relevant QMRA the specific pathogen reduction should be measured. The specific infiltration site characteristics may differ from other sites and a general value for the infiltration reduction may give misleading results far from the reality of the specific site. This can be considered by entering a distribution for the reduction to consider the uncertainty if a site investigation is not possible.

The QMRA model cannot handle clogging or other physical problem related to infiltration of water. Clogging and freezing may block parts of the infiltration system causing higher flow in other parts of the infiltration. In order to prevent undesirable high flow velocity in parts of the filtration it is important to maintain the infiltration construction.

6.2.1 Default values

There are default values for slow sand filtration in the QMRA model. The default values comes from the Microrisk report *Efficacy of water treatment* (Hijnen et al. 2006), and are triangular distributed (Table 6.2). The default values are for normal slow sand filtration with a depth of 0.7-1.4 meters and a filtration rate < 1 m/h (Lundberg et al. 2009; Hijnen et al. 2006)

Table 6.2 The default values for the triangular distributed reduction of the pathogens (Lundberg et al. 2009).

	Minimum log_{10} reduction	**Most common log_{10} reduction**	**Maximum log_{10} reduction**
Bacteria	1.2	2.7	4.8
Virus	0.6	2.2	4.0
Protozoa	0.3	3.8	6.5

Since the water characteristics and the reduction processes in the artificial recharge plant can change, it was suggested in the Microrisk project that the reduction values in Table 6.1 are used for artificial recharge and river bank filtration (Hijnen et al. 2006).

6.2.2 ODP values

Norsk Vann has, partly in collaboration with the Swedish Water and Wastewater Association, developed a procedure for evaluating treatment barriers for waterworks, called Optimal desinfeksjonspraksis (ODP) (Norsk Vann 2009). For artificial recharge, the ODP procedure recommends a maximum pathogen reduction value upon the retention time between the infiltration basin and the production well (see Table 6.3).

Table 6.3 The recommended maximum log10 reduction for artificial infiltration treatment barrier according to the ODP procedure (Norsk Vann 2009).

Retention time	**Maximum log_{10} reduction**		
	Bacteria	Virus	Protozoa
>60 days	3.0	3.0	3.0
30-60 days	2.5	2.0	1.5
10-30 days	2.0	1.0	1.0
3-10 days	1.5	0.5	0.75

6.2.3 Literature survey data

The reduction data presented in Chapter 5 is summarized in Table 6.4. The summarized sample reduction from Chapter 5 was collected from different depths (0.9-38 metres). The correlation with this data and an artificial groundwater recharge should be analysed before an assessment of the treatment barrier is done.

Phages were added to viruses and for *Cryptosporidium* the spores of *C. perfringens* was used as surrogates and added to the *Cryptosporidium* data.

Table 6.4 The log10 reduction capacity for micro-organisms.

	Studies	**Data**	**Mean**	**50th percentile**	**Min/Max**
Cryptosporidium	2	6	4.65	4.7	3.9 - 5.2
Bacteria	4	5	4.67	4.11	2.7 - 7.5
Virus	4	49	3.87	3.58	0.97 - 8.4

7 Case study Dösebacka water supply

The Dösebacka water supply in Kungälv is a groundwater system with artificial groundwater recharge. It produces 2.2 million m^3 per year (Zagerholm et al. 2007). The water source is the nearby river Göta älv. The water from the water intake is pumped to a sedimentation basin see Figure 7.1. After the sedimentation basin the water flows to the infiltration basins. There are nine infiltration basins where the infiltration rate is between 190 and 1040 mm/day. The distance from the infiltration basins to the wells are 100 to 150 meters.

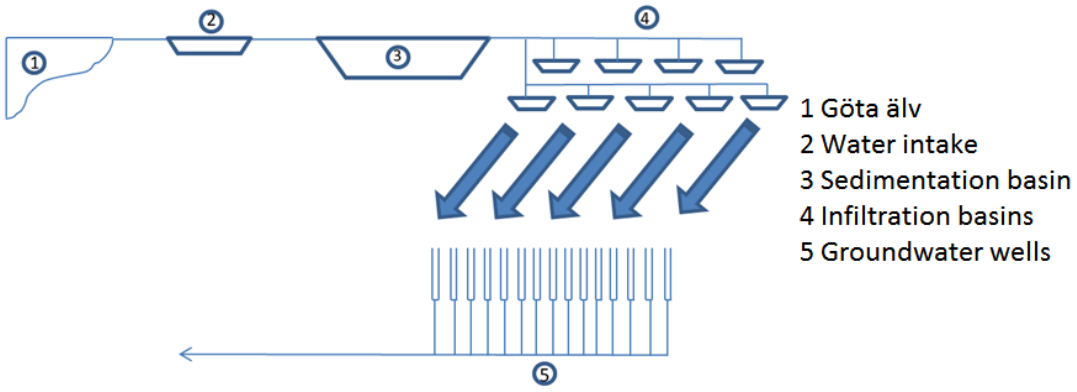

Figure 7.1 A conceptual model of the Dösebacka artificial groundwater infiltration.

There are fifteen groundwater wells. The retention time, i.e. the time from the infiltration basins to reach the production well, is at shortest 21 days and the longest more than 500 days. Two of the 15 wells have shorter retention time than 60 days (Zagerholm et al. 2007).

The pH in the ground water is between 6.8 and 7.2 and the porosity in the sand is 20-33 %. Some of the production wells give water with high turbidity, this water is filtrated with sand filtration.

The area around the water treatment plant is relatively flat. Under the river there is a layer of clay. Beneath the clay layer there is a layer of gravel where the groundwater can move. However, the clay layer is not completely covering the river bottom. In parts of the river edges there is a layer of gravel which enables induced infiltration, i.e. river water may reaching the well through the gravel layer. The wells are drilled through the clay layer reaching the gravel layer.

7.1 Reference pathogens and characterisation of the water source

The choice of reference pathogens are Campylobacter, norovirus and *Cryptosporidium.* The chosen pathogens and the concentrations are based on measurements in the river Göta Älv downstream Dösebacka (Åström 2011). The data on the pathogen presence in the river is limited, only few measurements were found and the deviations between them are high.

The selected pathogen concentrations in the river were measured in 2004 during a sewer discharge period and are given as most likelihood estimate in a Poisson distribution. For the QMRA model, the pathogens are assumed to be homogenous distributed in the river, see Table 7.1 (Åström 2011).

Table 7.1 The measured concentration (organisms per litre) of pathogens in Göta älv (Åström 2011).

	Campylobacter	**Norovirus**	**Cryptosporidium**
Units per Litre	10	148	0.13

7.2 Infiltration depth

The depths from the infiltration basins to the production wells were estimated by Zagerholm et al. (2007) and from the report the estimated unsaturated flow was calculated, see Table 7.2.

Table 7.2 The wells, the well depth and the estimated unsaturated zone

Well	**Well depth*** (m)	**Unsaturated Flow** (m)
GRP 1	31	N.A.
GRP 2	37	2
GRP 3	46	2
GRP 4	41	N.A.
GRP 5	45	N.A.
GRP 7	47	N.A.
GRP 8	20	N.A.
GRP 9	17	N.A.
GRP 10	46	N.A.
GRP 11	N.A.	0-1

*estimated well depth (Zagerholm et al. 2007)

The unsaturated zone was calculated by Zagerholm et al. (2007) from elevation map of the wells and the groundwater level in observation wells nearby. This could result in some error and should be changed whenever new information is available. The unsaturated zone may be from zero and higher, supposedly between 1-2 meters. The saturated zone is at least 17 meters if no unsaturated zone is present in well GRP 9. The reduction for the infiltration plant have been assessed before and no pathogens were found in the effluent and based on this it was concluded that the reduction at least is of 3 log10 for *C. Perfringens* and minimum 4.7 log10 for coliform bacteria (Zagerholm et al. 2007).

7.3 Treatment processes

The treatment processes in Dösebacka are artificial groundwater recharge and sand filtration. The sedimentation is not considered to be a part of the barrier and no reduction of pathogens are assumed there.

The treatment barrier was simulated three times with different input data in the QMRA model. The first simulation is made using the default values for slow sand filtration Lundberg et al. (2009). The second simulation is the ODP values and the input data for the third simulation is from the literature survey data.

The rapid sand filtration is currently used for some of the wells which have higher turbidity, about 40 % of the water is going to the sand filtration. The sand filtration is a rapid sand filtration but is not considered as an additional treatment step in this study because it is only used for the wells with higher turbidity.

In the ODP procedure the recommended values are 0,5 log10 reductions for bacteria and 0.25 and 0.5 log10 reduction for virus and protozoa respectively for rapid sand filtration (Norsk Vann 2009). The report by Hijnen et al. (2006), suggests a mean elimination capacity distribution, see Table 7.3.

Table 7.3 The performance of rapid sand filtration (Hijnen et al. 2006)

	Bacteria	**Virus**	***Cryptosporidium***
Min	0.1	0.1	0.0
Highest probability	0.6	0.8	2.0
Max	1.5	3.8	3.1

The performance values suggested in the Microrisk project (Lundberg et al. 2009) was selected and used to represent rapid sand filtration in the QMRA model. These values are presented as distributions and the ODP recommended values lies within the distribution interval. It should further be noted that the Microrisk reduction is higher than the ODP values.

7.3.1 Default values

The default log10 reduction values in the QMRA model for slow sand filtration were entered as a triangular distribution. The default slow sand filtration is here representing the artificial infiltration. The values can be seen in Table 7.4.

Table 7.4 The default performance of slow sand filtration.

	Bacteria	**Virus**	**Protozoa**
Min	1.2	0.6	0.3
Highest probability	2.7	2.2	3.8
Max	4.8	4.0	6.5

7.3.2 ODP values

According to the ODP procedure the reduction for two of the wells would be maximum 3.0 log10 reduction for all three pathogens types. This is within the reduction interval given in the default vale for slow sand filtration in the QMRA model, but lower than the maximum values. The log10 reduction values according to ODP can be seen in Table 7.5 and are assumed to be triangular distributed.

Table 7.5 The filter performance according to ODP.

	Bacteria	**Virus**	**Protozoa**
Min	1.5	0.5	0.8
Highest probability	2.5	2.0	1.5
Max	3	3.0	3.0

7.3.3 Literature data

The log10 reduction from the literature data can be seen in Table 7.6. The reduction is assumed to be triangular distributed.

Table 7.6 The filter performance according to literature review.

	Bacteria	**Virus**	**Protozoa**
Min	2.7	1.0	3.9
Highest probability	4.7	3.9	4.7
Max	7.5	8.4	5.2

The default values for the QMRA model and the recommended values in the ODP procedure regarding the pathogen reduction for artificial groundwater recharge as a treatment barrier are lower than all the studied experiments. One possible explanation to this may be that the literature data is based on longer filtration.

7.4 Results

7.4.1 Pathogen concentrations after filtration.

The pathogen concentration post filtration varies between the organisms, see Table 7.7. The *Cryptosporidium* (protozoa) concentration is close to zero in all three cases, this means that based on all three data sets almost all the *Cryptosporidium* oocysts are removed. The literature data has close to 100 times less concentration of both Campylobacter and norovirus than the other two data sets.

Table 7.7 The microorganism concentration (organisms per litre) after filtration.

	Bacteria	**Viruses**	**Protozoa**
Default	$1{,}39 \cdot 10^{-2}$	$8{,}31 \cdot 10^{-1}$	0,00
ODP	$4{,}31 \cdot 10^{-2}$	2,00	0,00
Literature data	$1{,}24 \cdot 10^{-4}$	$7{,}09 \cdot 10^{-3}$	0,00

7.4.2 The log reduction after treatment.

Since almost no *Cryptosporidium* passed the filtration the QMRA model did not provide any information for the *Cryptosporidium* reduction. The ODP recommended reduction values for artificial recharge were the lowest. The Default values are higher, however, the highest default reduction for bacteria are less than the 50 % percentile for the literature data reduction, which can be seen in Figure 7.2.

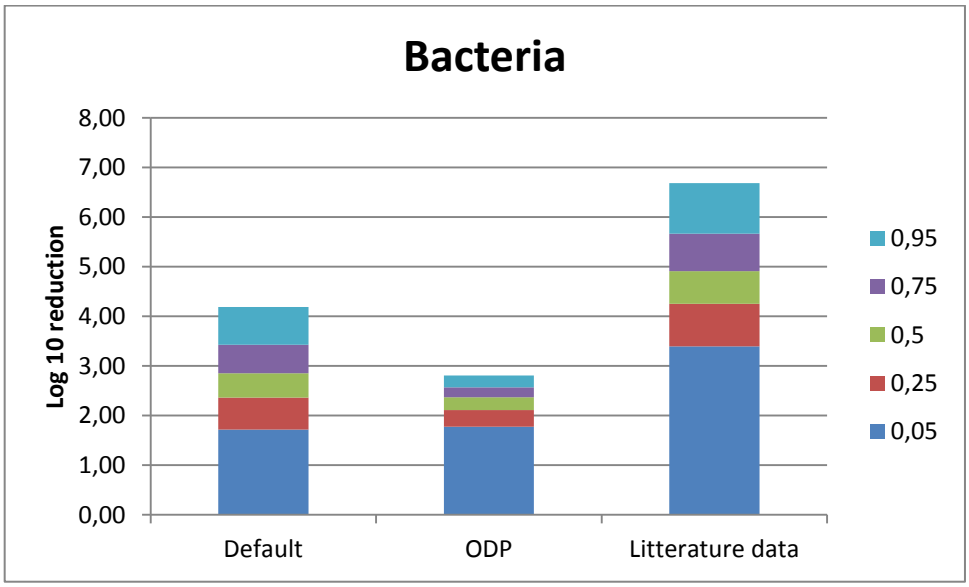

Figure 7.2 The probability bands of bacteria log10 reduction by filtration.

For the virus reduction the differences between the default and the ODP reduction are less than for bacteria. For the literature data the 95th percentile reduction are higher for the virus than for the bacteria, but for the 5th percentile the reduction is the opposite. The reduction is thus higher for virus in the literature data compared to bacteria but the deviation is higher. This shows a greater uncertainty which can be seen in Figure

7.3, however the uncertainty are probably larger for the bacteria due to the limited amount of data.

It is difficult to know how well the results reflect reality when the reductions from the literature data reduce the pathogens about 1000 times more than the default reduction. However, even in case of the 50th percentile reduction the literature data reduction is higher than the maximum for the default reduction. The 25th percentile is close to the maximum.

However, the literature data is based on measurements and the default values might have a lower reduction in order to not overestimate the reduction potential.

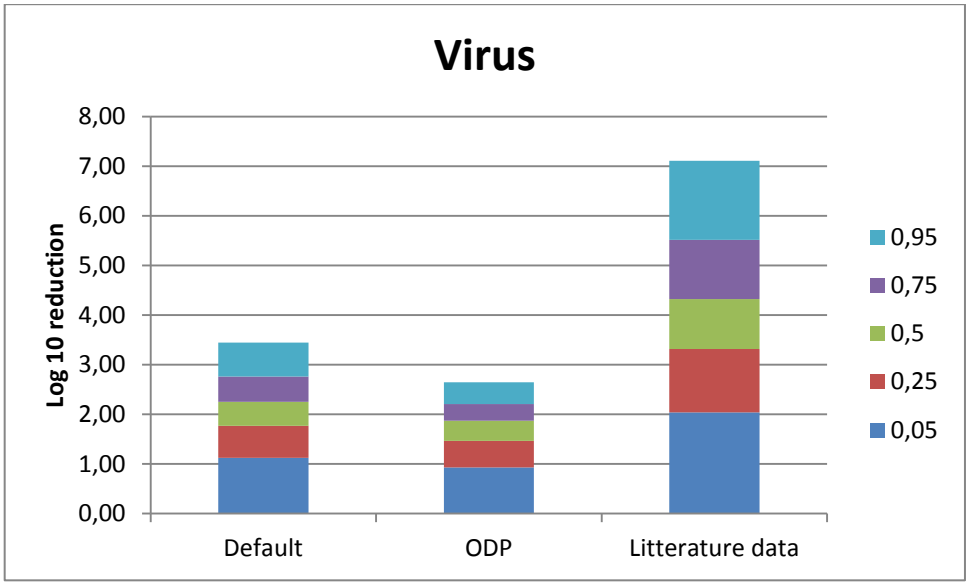

Figure 7.3 The probability bands of virus log10 reduction by filtration.

7.4.3 The probability of infection.

The daily probability of infection calculated in the QMRA model is similar for the default reduction and the ODP reduction values, both for bacteria and virus, see Figures 7.4 and 7.5. The inflow concentrations of microorganisms are measured during a sewer breakage. During normal circumstances the inflow concentrations are far lower. The high values were chosen in order to represent harsh conditions. Thus, the simulations represent a kind of worst case.

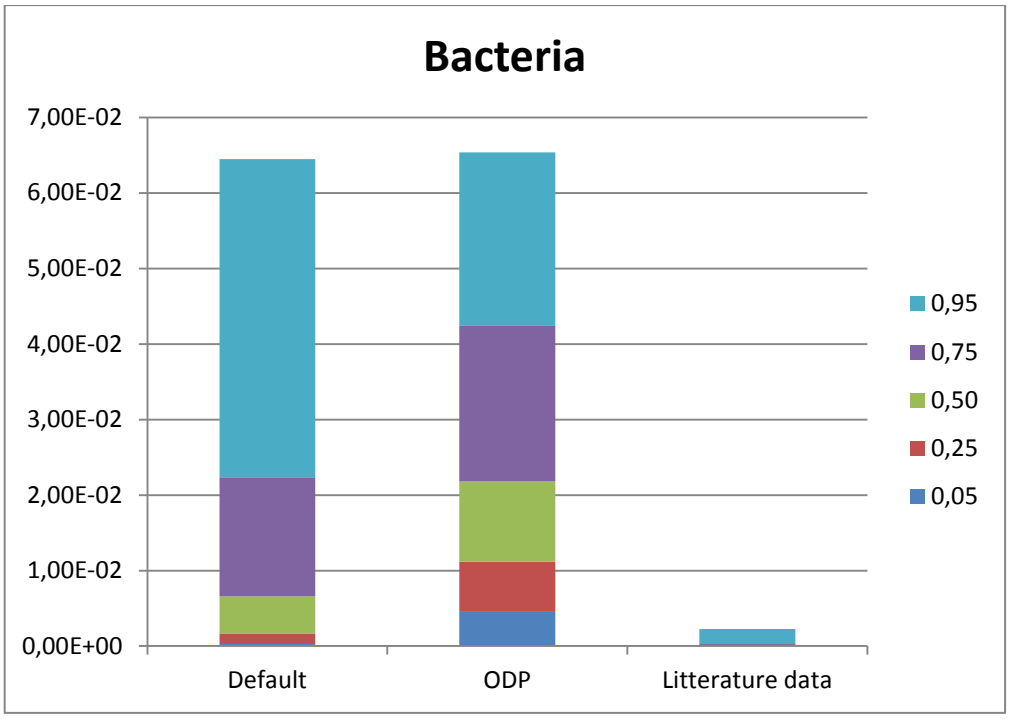

Figure 7.4 The probability bands for daily infection by bacteria.

The literature data gives a much smaller probability of infection compared to the other reductions for the bacteria. For the virus the difference is not that high as for bacteria but still there is a big difference.

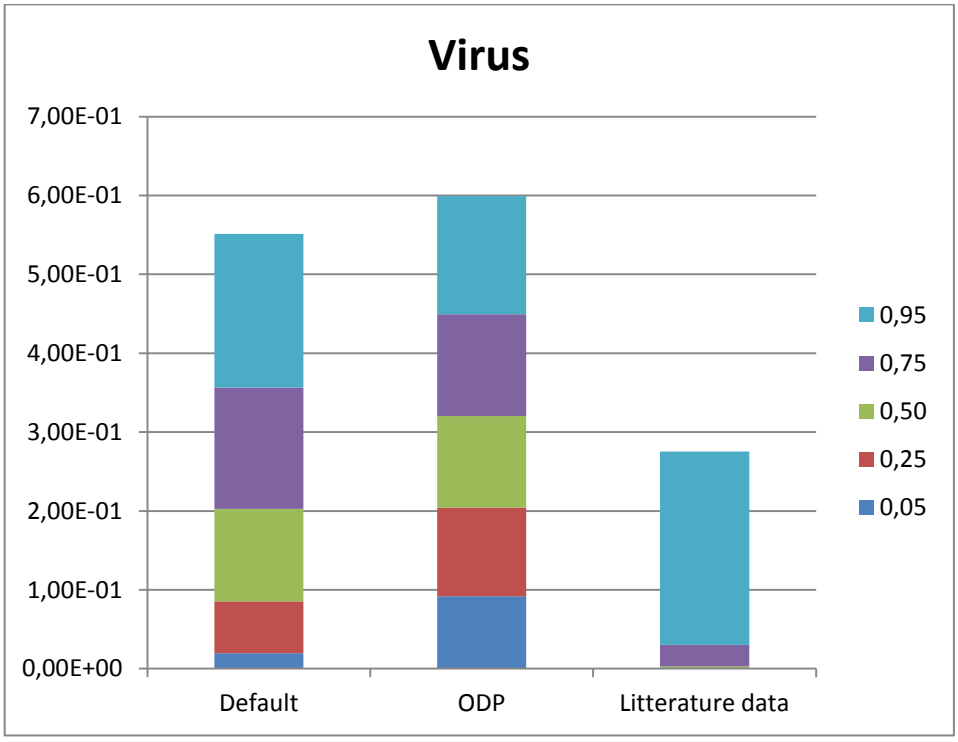

Figure 7.5 The probability bands for daily infection by virus.

For the protozoa, the ODP data result in the highest probability of infection, almost 20 times higher than the default values, see Figure 7.6. It is important to note that the interest of this thesis is the difference in reduction and the probability of infection is only to show the difference between the reduction data.

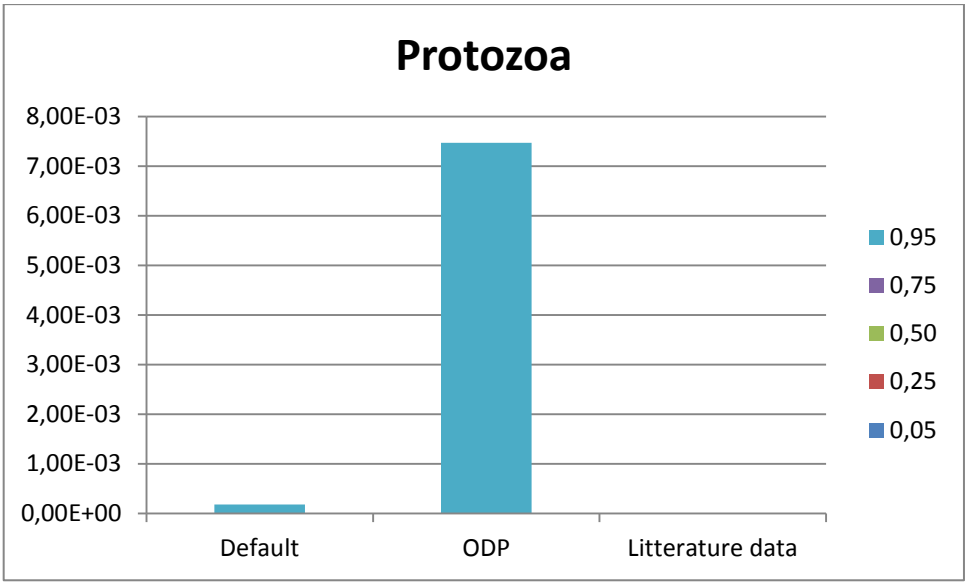

Figure 7.6 The probability bands for daily infection by protozoa.

8 Conclusion and Recommendation

In order to achieve acceptable risk levels the water sources needs to be well protected and sufficient treatment barriers must be installed. It is important to know the sources of pathogens to each water source, and reduce these as far as possible.

According to the survey presented in this thesis, flooding is in many cases (50 cases of total 141) the event causing a contamination of the raw water source. It seems that many of these places are easily affected when heavy rain occurs and they are vulnerable to run-off. To prevent the surface flow reaching the groundwater source during an excessive rain fall is an effective measure to prevent direct contamination of the well. Since it is likely that the surface water is contaminated by sewer discharge, protective measures may also be needed to prevent this.

A good understanding of the processes that influence the pathogen reduction in artificial recharge systems is important for making reliable risk models, especially for unsaturated flow. The movement and survival of pathogens are specific for every site and simplified models should be regarded with caution. However, the reality is always very complex and also simple models can help to better understand and predict the risks.

In order to model the treatment barriers today, the site-specific reduction needs to be measured to get reliable results. The studies of microbial pathogen reduction are few. A general guideline for the pathogen reduction regarding artificial groundwater infiltration and granular filtration would make it easier to predict the risk. This is for the future to be done; today the knowledge about many of the parameters is yet to be known. By modelling input data by means of probability distribution it is possible to consider existing uncertainties. Regarding the artificial groundwater recharge the uncertainties can be reduced when more data becomes available. In order to reduce the uncertainty when available data are scarce, comparison of different datasets, as done in this thesis, is a way to see the variation in the results. There is data for different types of infiltration characteristics so probably data for a similar infiltration facility is available.

The possibility to accurately predict and estimate the transport and reduction in soil filtration is at present limited since there is still missing knowledge about the subsurface processes, mainly during unsaturated conditions. In studies where oxygen is present in some parts of the filtration the reduction of pathogens have increased. This shows the importance of the unsaturated zone.

Due to the many parameters affecting the reduction of pathogens in the soil the uncertainties may be high but with sufficient flow time artificial recharge may constitute an efficient barrier reducing the risk to an acceptable level.

The reduction depends on the retention time and the infiltration depth and this is not considered in the QMRA model. The default values are valid only for slow sand filtration up to 1.5 meters dept. The most difficult part is to find and verify the reduction during artificial groundwater recharge with larger filtration depths and to know the uncertainties about them in order to make them relevant and applicable. The reductions are site specific and might change drastically under other circumstances and are thus not applicable to all infiltration facilities.

It is difficult to find relevant data about the pathogen concentrations in the source water. There is a limited number of pathogen types included in the model, and no

information about the relation between the pathogens or any common reference pathogens. A general relation guideline between the presence of the pathogens in the model and common reference pathogens would help to predict the pathogen concentration in the source water when local data is absent.

Additional knowledge and information would make it possible to further improve the applicability of QMRA on artificial groundwater recharge. However, if uncertainties as well as some other aspects are considered it is possible to obtain useful and important information on microbial risks using the QMRA model studies in this thesis.

9 References

Bauer, R., H. Dizer, et al. (2011). "Removal of bacterial fecal indicators, coliphages and enteric adenoviruses from waters with high fecal pollution by slow sand filtration." Water Research **45**(2): 439-452.

Bhattacharjee, S., J. N. Ryan, et al. (2002). "Virus transport in physically and geochemically heterogenous subsurface porous media." Journal of Contaminant Hydrology **57**: 161-187.

Bradford, S. A., S. R. Yates, et al. (2002). "Physical factors affecting the transport and fate of colloids in saturated porous media." Water Resources Research **38**: 1327.

Cabral, J. (2010). " Water Microbiology. Bacterial Pathogens and Water." International Journal of Environmental Research and Public Health **7**(10): 3657-3703.

Dizer, H., H. S. Wolf, et al. (2005). "Die Novelle der EU-Badegewässerrichtlinie Aspekte der Risikobewertung bei der Grenzwertsetzung ()." Bundesgesundheitsblatt **48**: 607-614.

Ekvall, A (2010). Utbrott av calicivirus i Lilla Edet. Svenskt Vatten Utveckling. **2010-13**

EPA, U. S. (1992). Guidelines for Water Reuse. U. S. E. P. Agency. Washington.

Goss, M. and C. Richards (2008). "Development of a risk-based index for source water protection planning, which supports the reduction of pathogens from agricultural activity entering water resources." Journal of Environmental Management **87**(4): 623-632.

Hansson, G. (2000). 100-årig teknik inom svensk dricksvattenförsörjning. VA-FORSK, VAV AB. **2000-5**.

Hellard, M., M. I. Sinclair, et al. (1997). "Drinking water and microbial pathogens issues and challenges for the year of 2000." J. Public Health Med.(19): 129-131.

Hijnen, W., G. Medema, et al. (2006). Efficacy of water treatment processes, MICRORISK.

Hijnen, W. A. M., E. F. Beerendonk, et al. (2006). "Inactivation credit of UV radiation for viruses, bacteria and protozoan (oo)cysts in water: A review." Water Research **40**(1): 3-22.

Hijnen, W. A. M., Y. J. Dullemont, et al. (2007). "Removal and fate of Cryptosporidium parvum, Clostridium perfringens and small-sized centric diatoms (Stephanodiscus hantzschii) in slow sand filters." Water Research **41**(10): 2151-2162.

Jin, Y., Y. Chu, et al. (2000). "Virus removal and transport in saturated and unsaturated sand columns." Journal of Contaminant Hydrology **43**(2): 111-128.

Lindberg, T. and R. Lindqvist (2005). Dricksvatten och mikrobiologiska risker. S. National Food Agency. **Rapport 28**.

Lundberg, J., J. Ansker, et al. (2009). MRA - Ett modellverktyg för svenska vattenverk. S. V. Utveckling. **2009-05**.

Lundh, M., E. Holmström, et al. (2006). "Reduktion av naturligt organiskt material och mikroorganismer i konstgjord grundvattenbildning." VA-Forsk rapport(2006-19).

Malm, A., T. Pettersson, et al. (2010). "Health effects of quality disturbances in Swedish water distribution networks (In Swedish)." Nordic Drinking Water Conference, Copenhagen **7th**.

Norsk-Vann (2009). "Veiledning til bestemmelse av god desinfeksjonspraksis." **170**: 47.

Oliver, S., H. Guy, et al., Eds. (2006). Protecting Groundwater for Health. London, World Health Organization.

Payment, P. and A. Locas (2011). "Pathogens in Water: Value and Limits of Correlation with Indicators." Ground Water **49**(1): 4-11.

Rollins, D. M. and R. R. Colwell (1986). "Viable but nonculturable stage of Campylobacter jejuni and its role in survival in the natural aquatic environment." Applied and environmental microbiology **52**: 531-538.

Schijven, J. F., G. Medema, et al. (2000). "Removal of microorganisms by deep well injection." Journal of Contaminant Hydrology **44**(3-4): 301-327.

Schinner, T., A. Letzner, et al. (2010). "Transport of selected bacterial pathogens in agricultural soil and quartz sand." Water Research **44**(4): 1182-1192.

Schoenen, D. (2002). "Role of disinfection in suppressing the spread of pathogens with drinking water: possibilities and limitations." Water Research **36**(15): 3874-3888.

Sen, T. (2011). "Processes in Pathogenic Biocolloidal Contaminants Transport in Saturated and Unsaturated Porous Media: A Review." Water, Air, & Soil Pollution **216**(1): 239-256.

SMI (2010a) Sjukdomsinformation om adenovirus.

SMI (2010b). "Sjukdomsinformation om cryptosporidiuminfektion."

SMI (2010c). "Sjukdomsinformation om campylobacterinfektion."

Svenskt-Vatten (2008). Råvattenkontroll - Krav på råvattenkvalitet. S. W. W. Assosiation.

Thomas, C., D. J. Hill, et al. (1999). "Evaluation of the effect of temperature and nutrients on the survival of Campylobacter spp. in water microcosms." Journal of Applied Microbiology **86**(6): 1024-1032.

Toze, S. (1999). "PCR and the detection of microbial pathogens in water and wastewater." Water Research **33**(17): 3545-3556.

Toze, S. (2003). "Pathogen survival in groundwater during artificial recharge." International Association of Hydrological Sciences(IAHS Publ. 285.2004).

Tufenkji, N. (2007). "Modeling microbial transport in porous media: Traditional approaches and recent developments." Advances in Water Resources **30**(6-7): 1455-1469.

Tufenkji, N. and M. Elimelech (2003). "Correlation Equation for Predicting Single-Collector Efficiency in Physicochemical Filtration in Saturated Porous Media." Environmental Science & Technology **38**(2): 529-536.

Wang, A., B. Lin, et al. (2011). "The Impact of Biofilm Growth on Transport of Escherichia coli O157:H7 in Sand." Ground Water **49**(1): 20-31.

WHO (2000) Campylobacter. Fact sheet N°255

WHO (2005) Enterohaemorrhagic Escherichia coli (EHEC), . Fact sheet N°125

WHO (2008). Drinking water quality, Chapter 11 -Microbial fact sheets. W. H. Organization. Geneva.

Zagerholm, B., M. Vikström, et al. (2007). Modellberäkning av konstgjord grundvattenbildning, Svenskt Vatten Utveckling.

Åström, J. (2011). Microbial Risks in Surface Water Sources. Department of Civil and Environmental Engineering

Water Environment Technology. Gothenburg, Chalmers University of Technology. Doctoral Thesis**:** 66.

Appendix 1

The number of events and the number of days with disturbances.

Number of days with disturbances	Number of events
1	2
2	7
3	8
4	8
5	7
6	13
7	20
8	3
9	2
10	4
11	1
13	2
14	8
15	1
17	1
20	2
22	1
25	1
30	3
31	1
32	1
35	3
50	1
90	1
120	3

Days of disturbances, connected people and possible sample failure

Number of days with disturbances	Connected people	Connected people * Days	Possible sample failure
1	1500	1500	yes
1	12000	12000	yes
2	1985	3970	yes
2	83	166	yes
2	50	100	yes
2	302	604	yes
2	50	100	yes
2	N.A.	N.A.	yes
2	N.A.	N.A.	yes
3	1600	4800	
3	180	540	
3	600	1800	yes
3	198	594	yes
3	282	846	yes
3	10000	30000	
3	600	1800	
3	2224	6672	
4	N.A.	N.A.	
4	137	548	
4	475	1900	yes
4	372	1488	no
4	17000	68000	yes
4	3000	12000	no
4	301	1204	yes
4	194	776	yes
5	1500	7500	yes
5	96	480	
5	200	1000	yes
5	133	665	no
5	200	1000	yes
5	600	3000	

5	155	775	no
6	62	372	
6	10	60	
6	338	2028	
6	500	3000	
6	3988	23928	
6	372	2232	
6	1486	8916	yes
6	70	420	no
6	158	948	
6	720	4320	yes
6	3808	22848	no
6	2224	13344	no
6	640	3840	
7	2000	14000	
7	100	700	
7	859	6013	
7	278	1946	
7	218	1526	yes
7	250	1750	yes
7	450	3150	
7	114	798	yes
7	1682	11774	yes
7	260	1820	yes
7	264	1848	
7	271	1897	no
7	858	6006	yes
7	600	4200	yes
7	1176	8232	no
7	449	3143	yes
7	40	280	
7	60	420	yes
7	105	735	no
7	800	5600	no

8	230	1840	
8	60	480	yes
8	140	1120	
9	431	3879	
9	400	3600	
10	160	1600	
10	72	720	
10	154	1540	no
10	241	2410	no
11	465	5115	
13	280	3640	yes
13	150	1950	yes
14	132	1848	
14	5400	75600	
14	569	7966	no
14	90	1260	
14	161	2254	no
14	1144	16016	yes
14	608	8512	
14	N.A.	N.A.	no
15	301	4515	yes
17	60	1020	
20	300	6000	
20	N.A.	N.A.	no
22	700	15400	no
25	500	12500	
30	60	1800	
30	30	900	
30	N.A.	N.A.	
31	100	3100	
32	1853	59296	
35	189	6615	
35	2275	79625	
35	60	2100	no

	50	84	4200	
	90	22000	1980000	
	120	372	44640	
	120	100	12000	
	120	200	24000	no
Sum	1433	121298	2726983	174334
Average	14	1225	26221	4981
Median	7	282	1948	1750

Appendix 2

Case study Dösebacka

The municipality is considering rebuilding the Dösebacka water supply and in this appendix the effects on the risk level by possible changes are analysed. The studied barriers are artificial groundwater recharge, upflow media filter (Dynasand) and UV disinfection. Rapid sand filtration is included although it is not always considered to be a microbial barrier.

The pathogens chosen for this test was *Campylobacter*, Norovirus and *Cryptosporidium*. The input concentrations of these pathogens were defined as a single value of 10 organisms per litre to simplify the comparisons of the treatment barriers.

The first barrier is artificial groundwater filtration, the input data were both default values for slow sand filtration from the QMRA model (see Table 1) and the literature data for comparison (see Table 2). All barriers were set to consist of 15 parallel units.

Table 1 *The default values for the triangular distributed reduction of the pathogens in artificial groundwater recharge (Lundberg et al. 2009).*

	Minimum log_{10} reduction	**Most common log_{10} reduction**	**Maximum log_{10} reduction**
Bacteria	1.2	2.7	4.8
Virus	0.6	2.2	4.0
Protozoa	0.3	3.8	6.5

Table 2 *Performance of slow sand filter according to literature review.*

	Minimum log_{10} reduction	**Most common log_{10} reduction**	**Maximum log_{10} reduction**
Bacteria	2.7	4.7	7.5
Virus	1.0	3.9	8.4
Protozoa	3.9	4.7	5.2

After the artificial infiltration there will be Dynasand filters with a log10 reduction represented by a triangular distribution, see Table 3.

Table 3 The log10 reductions with Dynasand filtration (Lundberg et al. 2009).

	Minimum reduction	**Most likely reduction**	**Maximum reduction**
Bacteria	0.8	1.4	3.3
Virus	0.1	0.9	3.9
Protozoa	0.8	3.0	5.4

The last treatment barrier is the UV disinfection. The fluency dose was set to be 30 mJ per cm^2. Using the QMRA model the reduction for the UV disinfection barrier was calculated as a point values: 5.3 log10 for bacteria, 3.2 log10 for viruses and 3 log10 for protozoa.

The QMRA barriers were modelled in three steps, the different sand filtrations were modelled individually with UV as the last treatment step. The result of the QMRA model for comparing the Dösebacka water supply barriers can be seen in Figure 1. The result is not considered to represent the actual barriers but can be used for a relative comparison of the different barriers.

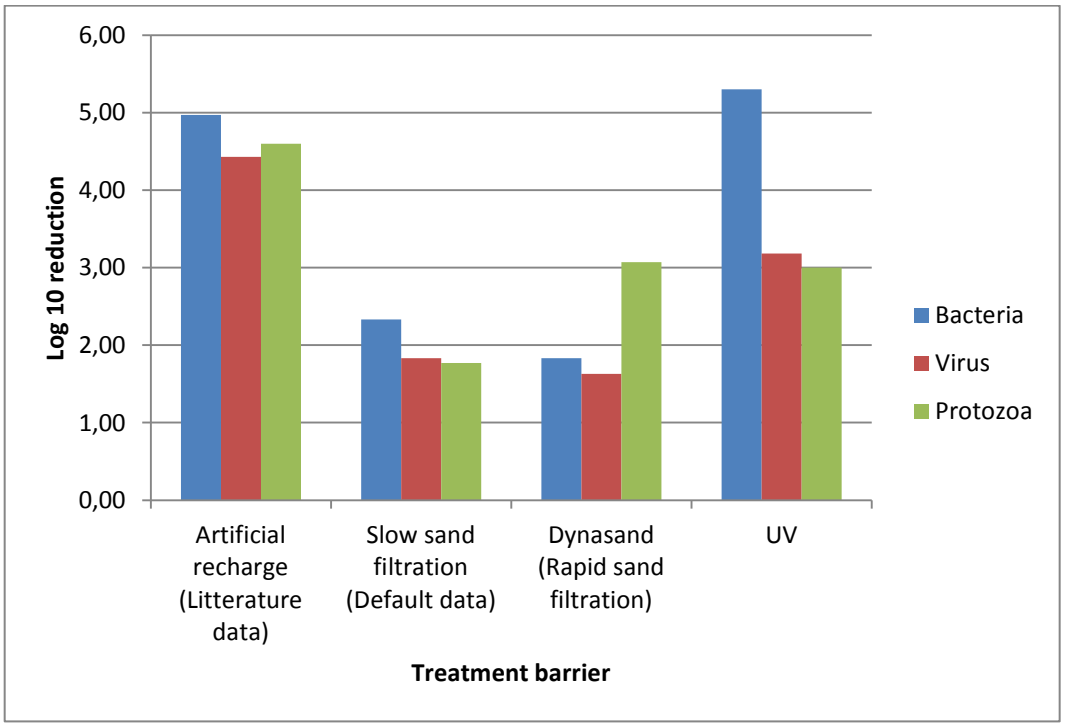

Figure 1 The mean value of the treatment barriers from the QMRA model.

According to the results in Figure 1 and Table 4, and the underlying data, the UV barrier reduces the bacteria and viruses more than 2 log10 compared to the rapid sand filter, see Table 4. If an additional treatment barrier is to be constructed the UV barrier seems to be a preferable option regarding the virus and bacteria reduction in this

simulations. If the protozoa are expected to be the major risk for this supply system, rapid sand filtration is slightly more effective in this modelling than UV regarding the protozoa. If the literature data are to trust, then the artificial groundwater recharge is effective and an important part of the water treatment system.

Table 4. The mean log10 reduction of the different barriers.

	Artificial Groundwater recharge			
	Literature data	**Default data**	**Rapid sand filtration**	**UV**
Bacteria	4,97	2,33	1,83	5,30
Virus	4,43	1,83	1,63	3,18
Protozoa	4,60	1,77	3,07	3,00