Adapted operation of drinking water systems to cope with climate change

SEVENTH FRAMEWORK PROGRAMME

Adapted operation of drinking water systems to cope with climate change

COLOPHON

Title
Adapted operation of drinking water systems to cope with climate change

Report number
PREPARED 2013.047

Deliverable number
D5.5.2

Author(s)
Contributions from:
Elsa Mesquita, Maria João Rosa, José Menaia (LNEC)
Edwin Kardinaal (KWR)
Bjørnar Eikebrokk (SINTEF)
Edited by Patrick Smeets (KWR)

Quality Assurance
By Patrick Smeets

This report is:
PU = Public

Summary

Introduction
Most water supply systems have evolved over the past decades into systems that can supply sufficient and safe water under the current conditions. Climate change will affect these conditions, requiring a more rapid adaptation to new conditions. Rapid response is needed during events that cause an immediate threat to water quality. But also more gradual changes such as temperature rise can be regarded a rapid change with respect to the rate at which a distribution system can be adapted. This section discusses the risks from these rapid changes that can affect water supply and how they can be addressed.

Water supply systems can be affected in various ways and through different causes. The effect of changes in raw water quality on drinking water production and distribution were studied. Direct risks to drinking water infrastructure, for example by flooding, are not the focus of this study.
The changes can be categorised as:
- New or increasing levels of contaminants in raw water
- Changes in raw water quality that affect treatment performance
- Changing environmental conditions causing (growth) processes to occur in raw or treated water

These changes can have different causes and are often a result of various combinations. These causes were identified in the study based on expected climate change in Europe and more specifically the effects in four climatic regions. Selected risks that were considered most relevant by the participating partners were studied in more detail, including adaptation of treatment systems to deal with these rapid changes.

Risks to the water cycle
Raw water quality can be affected by changes in rainfall pattern in all climatic regions. Both heavy rainfall events and prolonged periods of drought can cause a rise of contaminants in the raw water. Heavy rainfall can lead to run-off carrying contaminants from agriculture, industry or the urban environment into the water. In addition combined sewer overflows (CSO's) and by-passing or even wash-out of wastewater treatment plants can occur. Contaminants that thus get into the water are particles (turbidity), pathogenic micro-organisms, nutrients, pesticides, oil and various other contaminants. Prolonged periods of drought lead to less dilution of contaminants, especially in rivers that receive (treated) waste water. Thus the concentration of contaminants can rise significantly. All climatic regions are expected to experience this. In the Mediterranean region the increase in draught will force water suppliers to use sources with poorer water quality, such as wastewater impacted surface water. Higher temperatures and changing frost periods are causing an increase of natural organic matter (NOM) in the raw water, leading to colour problems, but also problems in treatment. The combination of nutrients and higher temperatures can favour blooms of cyanobacteria

(blue-green algae) that can release toxic substances in the water. The change of raw water quality and treatment performance can lead to a change in treated water quality. Even when the water complies to drinking water standards, it can still change the processes that occur in the distribution system. Together with other changes such as temperature and patterns of use, this can lead to regrowth problems in the network or the household.

Responding to Cyanotoxins

Cyanobacteria are among the oldest organisms on earth, and are widely distributed in all kind of ecosystems. The ability to float gives several cyanobacteria genera an advantage over other organisms in the water competing for sunlight. Under specific conditions the cyanobacteria can form dense blooms which can be harmless but in some cases are toxic. Although dense blooms can easily be detected visually, it is hard to estimate if there is a risk of toxins, since certain strains do and others don't produce the toxins. A range of toxins can be produced, for which analysis is time consuming and expensive and conditions can change rapidly. Alternatively molecular techniques have been developed to detect specific toxin-producing species of bacteria to improve the risk estimate. Three levels of prevention of blooms and removal of toxins can be distinguished: reduction of bloom formation (catchment level), bloom control within the actual water system (basin level) and during the drinking water production by removing cells and toxins. (production level). Measures in the catchment can focus on limitation of phosphorus either by diverting phosphorus streams and preventing runoff, or by removing phosphorus by adding a coagulant. At the catchment level, vertical mixing by dosing air or oxygen with a bubble aerator or dosing of hydrogen peroxide can be effective. In drinking water treatment the first steps should focus on removing cyanobacteria cells without disrupting them. Remaining dissolved toxins can then be removed by activated carbon or oxidised. Systems that apply pre-oxidation of raw water to improve their coagulation-filtration run the risk of disrupting cells resulting in release of toxins.

Responding to increase in NOM

An increase in NOM, measured as colour, UV-absorbance and total organic carbon (TOC) has been observed in surface waters in recent years. It is likely that the increase in NOM is due to a number of drivers, e.g. increased precipitation and increased runoff from drainage areas, increased temperatures and increased primary production, increased biological activity, less acid rain, etc. NOM has significant impacts on water treatment as well as distribution processes, impacts that may significantly increase the overall risk levels in drinking water supply:

- Affect colour, taste and odour levels in water
- Control most treatment processes, and affects overall treatment performance, incl. barrier efficiency
  - Challenge process control systems (increased seasonal variability in raw water quality, incl. NOM content and NOM nature)
- Increase coagulant demand and sludge production rates
- Affect filter run lengths, filter backwash and energy use
- Affect disinfectant demand and/or disinfection efficiency
- Form DBPs during chlorination, ozonation, etc
- Affect stability and removal of inorganic particles and pathogens and increase mobility of micro pollutants
- Adsorbs to metal precipitates, affect corrosion processes and biological stability in distribution systems
- Increase soft deposits, biofilm formation and regrowth in DS
- Foul membranes, block AC pores and/or outcompete T&O, micro pollutants, etc for AC adsorption sites
- Increase the organic loads and affects design and operation of ozonation-biofiltration systems

In order to cope with increasing NOM, treatment performance optimization is required. Additional treatment steps and combinations of treatment technologies may be required, when single-step NOM removal processes are no longer be capable of reducing NOM to acceptable levels. A step-wise strategy is proposed to cope with the increasing NOM levels:

1. Identify the seasonal variability in NOM concentration and characteristics in the present situation, and predict future NOM concentrations and characteristics in raw and treated waters in 10, 20 and 40 years time
2. Perform an internal benchmarking of the current water treatment operation performance status versus an optimized situation
3. Based on the above steps 1 and 2, and from comprehensive evaluations of NOM content, NOM nature, NOM treatability, and NOM biodegradability in raw and treated waters, evaluate the applicability and possible benefits of additional treatment steps like activated carbon adsorption (AC), ion exchange (IEX), nano filtration (NF), advanced oxidation processes (AOP), UV-disinfection, etc
4. Implement selected changes to the treatment processes, i.e. shift to the best available treatment technology (BAT) and the best operation performance (BOP) in terms of water quality/safety and sustainability/use of resources and emissions/environmental impacts

Several Norwegian utilities have performed full-scale treatment optimization trials during the last few years. Rapid NOM fractionation and BDOC measurements are among the more advanced tools used in process performance diagnosis and optimization efforts.

Responding to microbial growth in distribution systems
Micro-organisms may enter the distribution system through treatment, leaks or during construction works. Although many microbial species that ingress

into drinking water distribution system (DWDS) are unable to survive or multiply, many bacteria colonize DWDS inner surfaces by forming biofilms. Biofilms are multicellular communities of microorganisms embedded in a fibrilar matrix in the sediment or attached to the pipe wall. Regardless the presence of a disinfectant residual biofilm sessile communities persist as the predominant form of microbial regrowth and the origin of most DWDS water planktonic microorganisms. The DWDS biofilm may host pathogens (e.g. *Legionella pneumophila*, *Campylobacter jejuni*, *Mycobacterium avium*) and opportunistic species of several genera (e.g. *Aeromonas*, *Pseudomonas*, *Klebsiella*). Moreover, the biofilm fibrilar matrix has the ability to sequester and hold/release viral (e.g. adenovirus, rotavirus, norovirus virions) and protozoan (e.g. *Cryptosporidium parvum* oocysts) pathogenic forms including amoebae carrying high numbers of *L. pneumophila*. Hence, DWDS biofilms have an important potential to control drinking water safety. Therefore, although no direct evidence of negative effects of DWDS biofilm on the general population health has been produced, biofilm hazards should be considered, especially with respect to immune-compromised people.

Climate change effects can lead to an increase of biofilm formation through an increase of temperature, NOM concentration and chlorine demand. Higher temperatures lead to higher biological growth rate. An increase of NOM in the raw water followed by oxidation processes may lead to an increase of biodegradable matter, allowing more growth. Residual chlorine in the DWDS does not prevent the formation of biofilm, but can control it to some extent. Rising water temperatures increases chlorine disinfection effectiveness, but also to higher chlorine consumption. By increasing the waters disinfectant demand and consumption, rises in DWDS-NOM and temperature will consequently lead to increased formation of DBPs and, possibly, biodegradable NOM derivatives, which in turn may contribute to lower DWDS waters' biological stability. Most rapid change is expected from NOM in the raw water and the ability of water treatment to deal with these changes.

Biological stability of the water and biofilm growth cannot be measured directly. Methods for total organic carbon (TOC), dissolved organic carbon (DOC), assimilable organic carbon (AOC), biodegradable DOC (BDOC), biofilm formation rate (BFR), ATP analysis, quantitative PCR, heterotrophic plate counts (HPC), flow cytometry and DOC analysis by on-line spectroscopy all have their strengths and weaknesses to assess biological stability or biofilm formation.
Biofilm formation problems are most effectively prevented by sufficient reduction of biodegradable matter by treatment before the water enters the DWDS. Biologically stable water can be characterised as AOC < 10 μg C/L or BFR < 10 pg ATP cm^{-2} day^{-1}. If this cannot be achieved, measures in the DWDS are needed. Given that in practice it is not possible to control DWDS water temperature, the use of disinfectant residual appears as the only possible measure to mitigate enhanced regrowth in the DWDS itself. Superchlorination may be advised to protect consumers upon detection of significant increments in bacterial numbers or pathogens. Booster dosing may be required to control chlorine concentrations in low flow extremities of

DWDS networks. Hydraulic or chlorine models are useful tools to support siting of rechlorination locations. Nevertheless, it should be taken into account that chlorine can reduce the numbers of active planktonic bacteria, but have little to no effect on their sessile counterparts. In addition, while being effective against many bacteria that can occur in biofilms, chlorine has low efficacy against some pathogens, including those of the *Mycobacterium* group.

In addition to the selection of provisional sources wherever they are available, the proactive implementation of multibarriers upstream of DWDS and the use of disinfectant residual as a supplementary barrier, the design, operation and condition of network pipelines are also important DWDS characteristics in minimizing regrowth.

As it is discussed in Chapter 6, the selection/control/protection of source water quality, the ability to foresee or rapidly detect changes in the quality of the abstracted water and the capacity of WTPs to respond to the changes and produce water with the required biological stability are the most effective factors to prevent regrowth. Nevertheless, some DWDS characteristics that may potentiate regrowth of sessile and planktonic microorganisms need to be considered, particularly while raises in the water temperature occur. These characteristics mainly concern:

- The condition of network pipes
- DWDS design and operation
- The effectiveness of maintenance activities.
- The type of network materials in contact with the water

In most developed countries often the maintenance of effective concentrations of disinfectant residual is erroneously viewed as an effective barrier against DWDS microbial contaminants. Accordingly, the biological stability of the water is not a generalized concept and TOC, colony counts and faecal indicators are the related parameters that are commonly monitored. Research in the Lisbon network showed that sessile colonization was relatively weak in intensity and that no meaningful risks were associated to the DWDS biofilm. In the Netherlands no disinfectant residual is applied, and focus is on producing biologically stable water.

Adapting drinking water treatment

Utilities face new challenges arising from faster and more severe raw water quality variations promoted by climate change that are expected to lead to an overall deterioration of the water quality used for drinking water production. Many drinking water treatment plants were designed to remove (or inactivate) particles, colloidal matter and microorganisms through a treatment train that generally includes chemical pre-oxidation, coagulation/flocculation/ sedimentation, filtration and disinfection (AWWA 1999). The occurrence and detection of emerging (micro)contaminants in water sources and advances in knowledge of toxicological and epidemiological data of those contaminants are driving increasingly stringent water quality standards in parameters resistant to conventional treatment. To handle these challenges the utilities generally rely on conventional surface water treatment assisted by ozonation and PAC (or, less frequently, GAC) adsorption. Increasing oxidation can enhance disinfection and breakdown of microcontaminants, however it can also lead to increased formation of disinfection by-product with adverse health effects. Special care must therefore be taken to ensure a safe disinfection and (at least partial) removal of microcontaminants while minimizing DBP formation.

The ability to detect rapid changes that lead to risk is a key-issue for deciding and implementing the adequate preventive and corrective actions. This will require:

1. Implementing pro-active measures to identify changes in quantity and quality of water resources;
   a. Anticipating the water source pollution – modelling intense rainfall events (frequency and intensity), runoffs and droughts;
   b. Characterizing the water source pollution – monitoring (volume and water quality parameters) of intense rainfall events and wastewater discharges in the watershed;
   c. Characterizing the source water availability and quality – modelling the water quality in different scenarios;
   d. Regular monitoring/inspection of the source water quality – visual inspection, e.g. of water scums, turbidity and colour, including as much as possible parameters of rapid determination for early warning of quality changes requiring treatment adaptation (e.g. cyanobacterial blooms, muddy and clay waters);
2. Implementing pro-active measures to identify the impact of raw water quality changes in the produced water quality;
   a. Monitoring the critical treatment steps' effectiveness and efficiency, using as much as possible reliable online measurements;
   b. Modelling WTP response to raw water quality changes.

Current operational practices of conventional water treatment are often insufficient to deal with expectable risks due to climate change. Measures to mitigate these risks include operation practices of conventional treatment and WTP upgrade with advanced treatments. Climate change may affect the target contaminants for the processes, potentially requiring a different optimisation. Interfering 'species' can also affect this optimisation by affecting the treatment efficiency or by causing adverse effects on water quality. This report addresses the integrated control of the contaminants challenging surface water conventional treatment in climate change scenarios.

When optimization of existing processes provides insufficient improvement of treatment, utilities need to consider advanced or alternative treatment processes for WTP upgrade. Examples of such processes are dissolved air flotation, alternative oxidants and advanced oxidation processes (AOP), activated carbon (bio)filtration (GAC/BAC), membrane pressure-driven processes (microfiltration – MF, ultrafiltration – UF, nanofiltration – NF and reverse osmosis – RO) and hybrid processes of adsorption and low-pressure membranes (e.g. PAC/UF or PAC/MF). The advantages and disadvantages of these processes are discussed in this report.

In Portugal a case study of preparing a water treatment system for climate change was resulted in the following 3-level strategy to ensure a safe water supply:

1. Optimisation of WTP unit operations for removal of toxins and/or microorganisms - identification of the limiting steps and operating conditions These procedures were implemented in the WTP operation manual and a performance assessment system was developed to promote the benchmarking of the water treatment in terms of effectiveness and efficiency.
2. Development of studies for WTP upgrade in case the monitoring program indicates limited results of step 1 strategy. Advanced technologies that would become very attractive would be dissolved air flotation for cyanobacteria removal; ultrafiltration for particles removal, including bacteria and protozoan cysts, virus and cyanobacterial cells, and PAC/ultrafiltration or nanofiltration for further removal of low MW organics, including toxins and THM precursors.
3. A contingency plan was developed with management procedures' application whenever the monitoring program indicates the treated water is not safe for human supply.

Conclusions

Several challenges of rapidly changing raw water concentrations due to climate change were discussed. Rapid can be interpreted at different time scales of hours (turbidity from runoff), days (cyanobacterial bloom), weeks (drought) or years (increasing NOM). Utilities need to prepare by planning responses on an adequate time scale. Several examples were presented here.

Contents

2 Introduction

2.1 Scope of the study

Climate change will pose several challenges to drinking water supply systems. Various effects can result in deterioration of the raw water quality, requiring improved treatment. Changing water quality and environmental conditions can also negatively affect the conditions during treatment and distribution. Goal of the study was to provide examples of water supply system adaptation to deal with the most relevant risks in three different European regions.

2.2 Approach

First the climate driven risks for water supply were selected from report D2.1.1 *Overview of climate change effects which may impact raw water quality and water treatment* (Ugarelli et al, 2011). Three research institutes from three different regions made inventories of relevant risks in their regions and how they have been dealt with. Each describes:

- The relevant risks
- Current operational practices
- Detecting rapid changes in risks
- Corrective response and curative measures
- Experiences

2.3 Use of study results

Water supply companies can use the examples to assess which risks may be relevant for their systems. Appropriate responses can be identified and the feasibility in each situation can be evaluated. Water supply companies can then prepare their systems by implementing timely detection, and adequate response strategies or protocols, depending on the time scale at which a risk becomes relevant. These steps can be performed in the context of the water cycle safety plan (WCSP) framework developed in work area 2.

2.4 Abbreviations and acronyms

AOC: assimilable organic carbon
ATP: adenosine-triphosphate
BAC: biological activated carbon
BDOC: biodegradable dissolved organic compounds
C/F/S: coagulation/flocculation/sedimentation
DAF: dissolved air flotation
DBPs: disinfection by-products
DOC: dissolved organic carbon
DW: decanted water
DWDS: drinking water distribution systems
EBCT: empty bed contact time
EC: electrical conductivity
EDCs: endocrine disrupting compounds

FW: filtered water
GAC: granular activated carbon
GHGs: greenhouse gases
MIB: 2-methylisoborneol
NOM: natural organic matter
OBPs: oxidation by-products
OBPFP: oxidation by-products formation potential
OW: ozonated water
PAC: powdered activated carbon
PACl: polyaluminium chloride
RW: raw water
SUVA: specific UV_{254nm} absorbance, defined as UV_{254nm}/DOC
T: temperature
T&O: taste and odour
THMs: trihalomethanes
THMFP: THM formation potential
TOC: total organic carbon
TW: treated water
WTP: water treatment plant
VOCs volatile organic compounds

3 Overview of climate change effects that may impact water supply systems

3.1 Introduction

Climate changes will impact the water supply systems by altering the quantity and the timing of water availability, changing water quality and reducing the reliability of the infrastructures.

In PREPARED 2011.011 "Overview of climate change effects which may impact raw water quality and water treatment" (D2.2.1 Ugarelli et al, 2011), various impacts of climate change on the urban water cycle were described.

This report presents climate change impacts of specific relevancy for raw water sources and watersheds, water treatment and distribution systems.

Effects of climate change that are specifically related to raw water availability, water quantity/scarcity and use of additional/supplementary sources will affect optimum treatment process selection and design as well as operation of water treatment and distribution systems. The aspect of water scarcity is, however, covered by other tasks in Prepared and will not be further discussed here.

3.2 Effects of climate change on water supply systems

Table 1 summarizes the impacts of climate change on water quality, treatment and distribution, as well as issues on safety and sustainability/resources use.

Table 1 - Impacts on climate changes on water quality, treatment and distribution as well as safety and sustainability issues

Water supply system element	Impacts on water quality, treatment and distribution	Safety and sustainability
Catchment and raw water sources	Higher temperature and precipitation levels, more intense precipitation, extended growth season and increased primary production, reduced acid rain, etc. can lead to more pathogens, more algae and algae toxins, increasing NOM, colour and UV-absorbance in raw waters. Less ice cover and less snow magazine may negatively affect lake water stratification/stability as well as water quantity and quality in spring and early summer. Quality of water extracted from aquifers may change due to sea level rise and possible saline intrusion	More pathogens and more NOM in drinking water reservoirs. More DBPs and/or reduced disinfection efficiency due to increased disinfectant demand, less UV-transmittance, etc. Less stability/stratification of water sources due to less ice cover and thus risks for more long-term intake of surface water with reduced and more variable quality compared to conventional deep water intakes
Water treatment	Increasing NOM concentrations significantly affect treatment and distribution processes and require optimized treatment performance.	More Natural Organic Matter (NOM), more microorganisms and pathogens, more algae and algae toxins, more seasonal

Water supply system element	Impacts on water quality, treatment and distribution	Safety and sustainability
	Increasing use of resources like coagulation and water treatment chemicals, energy, backwash water, etc. Increased production of sludge/waste and more CO_2 emissions. Need for more sludge processing/management. More variable raw water quality will stress water treatment operations and treatment process control systems. Increased content of algae and planktonic species, substances causing taste and odour as well as low molecular weight, neutral and hydrophilic NOM fractions may require supplementary treatment steps	variability in water quality impose increased risks for treatment barrier failures and disease outbreaks. Increasing use of resources like coagulation and water treatment/regeneration chemicals, energy, backwash water, etc. More sludge/waste/CO_2-emissions. Risk of flooding of water treatment and distribution infrastructure
Distribution	Increased NOM and increased amount of biodegradable natural organic matter (BOM) will facilitate biological regrowth and biofilm formation in distribution networks (also inside buildings). Increased adsorption of NOM/BOM and microorganisms to metal precipitates on pipe walls will increase the amount of soft deposits and thus lead to re-suspension of particles and biofilm, discolouration and customer complaints as well as increased growth/survival of microorganisms including pathogens	More re-suspension and discolouration, more taste and odour problems, and more biofilms and soft deposits due to increased temperature and organic substrate levels. More biological regrowth and growth/survival of micro-organisms and pathogens. Increased risk for intrusion of storm water/waste water as a consequence of increased pipe corrosion (contact with GW table/soil acidity/saline intrusion) and increased pipe breaks due to more landslides, depressurisation, saline water intrusion)

3.3 Region-specific effects of climate change

The major impacts of climate change on water supply systems are extremely region-specific. Some regions will suffer from severe water scarcity/quantity challenges, while other regions will experience increased precipitation and other effects of climate change that may impose severe water quality challenges, e.g. increasing NOM concentrations in surface water sources. Other regions may suffer from sea level increase and intrusion of salt water in ground water reservoirs.

Discussions of major region-specific climate change effects on water supply systems are given below.

3.3.1 Expected effects of climate change on water quality and treatment in the Mediterranean region

Raw water quality problems due to increase of runoff and changes in air temperature

Raw water will be negatively affected by increased runoff in drainage area, increased water temperature, overload of wastewater systems, increased presence of birds and animals and the inclusion of new species in drainage area and reservoirs. The reduced effect of stratification in water supply reservoirs will reduce the protection of water intakes. Deterioration of water quality will probably occur due to shorter times of stable stratification in lakes as a consequence of longer circulation periods.

Treatment plants must be adapted in order to improve the ability to remove microorganisms, Natural Organic Matter (NOM), taste and odour and environmental toxins from cyanobacteria, algae, etc.

Water treatment issues due to sudden variations in raw water quality
Sudden variations in raw water quality create additional difficulties in the operation of water treatment plants; it becomes increasingly difficult to predict the required dose of chemicals, for example. Operation of treatment plants are based on the raw water quality and when it is unstable and vary over time it is difficult to obtain a stable and good drinking water quality. The variation of raw water quality requires continuous monitoring of the raw water in order to be able to detect changes in water quality at any time. Water treatment plants must be adapted and be flexible to take into account the variation of the raw water quality and optimise the treatment processes accordingly. Treatment processes should also be prepared to respond to the changes quickly. The changes in raw water quality may seriously affect water treatability by different technologies, treatment costs, water treatment process selection and treatment process design and operation (Eikebrokk et al., 2004).
Groundwater may need improved treatment due to changes in raw water quality

Like for surface water, in some instances improved and extended treatment of groundwater will be needed due to the changed conditions of groundwater caused by climate changes. In many places, groundwater wells are already contaminated, unprotected or close to becoming dysfunctional due to a lowering of the groundwater table close to or below the bottom of the well.

Increased biological growth due to higher NOM levels
If water treatment plants are not able to cope with the higher NOM levels expected in raw water, NOM will be transferred to the transport system. Natural Organic Matter will act as food source and will facilitate the biological growth in the water bulk flow and in biofilm media on the pipe walls.

Increased water temperature in pipes during the summer months can also contribute to a higher level growth of microorganisms in the network components.

Increase chlorine demand due to temperature increase
Temperature increase can affect bacterial growth rate which is then associated with an increase in chlorine demand to maintain the level of chlorine constant. Chlorine decay in these conditions is then faster and if no action is taken, bacteria regrowth can happen within the water bulk.

3.3.2 *Expected effects of climate change in water systems in the Atlantic region*

Raw water contamination due to increased precipitation
Increased precipitation and consequently runoff may lead to water quality problems such as higher Natural Organic Matter (NOM) water content, higher number of microorganisms in the water and other contamination related problems (e.g. eutrophication).
Longer periods of non-frozen lakes
Warmer climate can bring along longer periods of non-frozen lakes and ponds (ice-free periods) during winter. This makes water easily accessible for animals and birds, allowing new species to live in and around water sources. The presence of a larger number of animals and birds for longer periods of time in the catchment areas raise the risk for microbiological water contamination. Therefore, raw water treatment may need to be improved due to this factor.
Finally, the stratification in wintertime is today secured by the ice coverage. Without this, longer circulation periods may be caused by wind.

Algae growth
Higher water temperature will promote the growth of blue-green algae. Taste and odour compounds and cyanotoxins from these algae are of concern. Warmer water also facilitates microbiological growth in the reservoirs.

Additional water treatment required due to more microorganisms in raw water
More precipitation and more runoff will result in more microbiological activity in the drinking water reservoirs. Water treatment plants will probably have to deal with more microorganisms in water, including parasites because:
More precipitation leads to more runoff from the drainage area of reservoirs. This will bring more microorganisms to the drinking water sources;
Warmer climate will lead to the presence of more animals and birds for a longer period of time (also over winter). This will increase the supply of faeces and microorganisms, including parasites.

Warmer water could also be viewed as another main factor affecting microorganism activity rate and other physic-chemical and biological reactions.

The stratification layers of the surface water reservoirs will function less as hygienic barriers due to changes in water temperature, less freezing of water in winter time and more wind on the surface which will create more circulation in the water.

Due to the presence of more microorganisms in the raw waters, water treatment processes will have to be implemented in order to reduce the risk for microbiological contamination of drinking water. Implementation of early detection tools is beneficial.

Higher amounts of algae and toxins are expected in surface water reservoirs due to warmer weather, more precipitation, more runoff and more strain on the wastewater system. Warmer water and increased access to light during winter months will boost the growth of algae in reservoirs.

Increased risk of infiltration of contaminated water into water supply system due to groundwater level rise
Increased precipitation may raise the groundwater table; if it reaches the level where water supply pipes are installed there is risk of infiltration of groundwater into the pipes.

In some countries water supply system pipes and wastewater system sewers are laid down in the same ditch. The pipes for water and wastewater in such systems go through the same manholes. During extreme weather situations such manholes can be submerged in sewer combined with surface water. Drinking water pipes will then be surrounded by contaminated water. In such situations it is very important that positive pressure occur within the water supply system pipes (Vevatne et al., 2007).

Increased biological growth in biofilms and in suspension in water in pipes due to higher NOM levels in water
The increase in air and water temperature can contribute to the growth of biofilm in the drinking water system components, namely the transmission system components. These conditions can then affect water production for drinking purposes.

Increase chlorine demand due to temperature increase
Chlorine decay increase is also expected to affect the transmission, storage, distribution and plumbing systems due to the increase of microbiological activity associated with the increase in temperature (air and water).

3.3.3 Expected effects of climate change in the Continental region

Increased runoff may lead to raw water contamination
Increased precipitation and consequently runoff may lead to water quality problems such as higher Natural Organic Matter (NOM) water content, higher number of microorganisms in the water and other contamination related problems (e.g. eutrophication).

Longer periods of non-frozen lakes

Warmer climate can bring along longer periods of non-frozen lakes and ponds (ice-free periods) during winter. This makes water easily accessible for animals and birds, allowing new species to live in and around water sources.

The presence of a larger number of animals and birds for longer periods of time in the catchment areas raise the risk for microbiological water contamination. Therefore, raw water treatment may need to be improved due to this factor.

Finally, the stratification in wintertime is today secured by the ice coverage. Without this, longer circulation periods may be caused by wind.

Sea level rise may increase the risk of saline intrusion into groundwater
Sea level rise, combined with changes in rainfall patterns and increased demand for water, will increase the risk to coastal aquifers of saline intrusion and consequent variation of water quality characteristics.

Increase in the number of birds and new species may reside in the reservoir may cause water contamination
The new climate is better suited for many types of animals and among them birds will expand their residence time in many reservoirs. This will increase the risk of contamination from faeces.

Increase of temperature (water and air) reduces surface reservoir capacity
Higher water temperature is better suited for growth of blue-green algae. Warmer water also facilitates microbiological growth in the reservoirs.
Growth of blue-green algae is something that first and foremost affects the bathing water quality since these algae can easily be removed in the water treatment plants.

Decrease of summer precipitation may cause sedimentation problems and water quality problems
In the abstraction system, decrease of summer precipitation may affect water quality. Lower flow rates cause deposition leading to reduces raw water quality. Lower flows lead to greater sedimentation, with blockages causing service failure. Reduced raw water volumes reduce dilution and increase drinking water quality risk.

Increase of temperature (water and air) may impact treatment efficiency and processes
Higher temperature impacts treatment process because of increased planktonic species, of NOM and also changes in water colour, and further:

- Increased algal growth and risk of microscopic organism within the water supply system.
- Discolouration and odour problems caused by the biological consequences of higher temperatures.

- River floods during winter lead to greater sediment level and water quality risk
- Greater sediment levels which increases drinking water quality risk
- Increase of temperature (water and air) may influence micro-biological growth in the distribution system
- Increased rate of micro-biological growth increases risk of residual chlorine depletion and contamination of supplies

- River floods during winter may influence water quality. Flooding causes contaminants to enter pipelines increasing drinking water quality risk
- Increased external corrosion due to higher groundwater table and higher acidity of ground. Due to increased rain and increased runoff and infiltration, the ground water table will be higher. This will cause the drinking water and sewer pipes to more frequent be in contact with the ground water table which will facilitate external corrosion problems.

3.3.4 *Expected effects of climate change in the Nordic, cold-climate region*

Major effects

Major effects of climate change in the Nordic, cold-climate region can be summarized as follows:

- Increasing NOM concentrations
- Warmer, wetter and wilder, i.e. increased temperatures, increased precipitation, increased precipitation intensity, more flooding and more extreme events
- Less snow in winter and less snow magazine
- More vulnerable water sources due to less ice cover and less stratification/stability of deep water layers
- Extended growth season and more primary production in water sheds
- Increased growth rates and volume of deciduous tree species
- More challenges from algae, taste/odour and algae toxins
- Emerging pathogens

These impacts will be further described below.

Increasing NOM

An increase in NOM, measured as colour, UV-absorbance and total organic carbon (TOC) has been observed in surface waters in recent years. It is likely that the increase in NOM is due to a number of drivers, e.g. increased precipitation and increased runoff from drainage areas, increased temperatures and increased primary production, increased biological activity, less acid rain, etc. Possible reasons for the increase in NOM are presented in Figure 1.

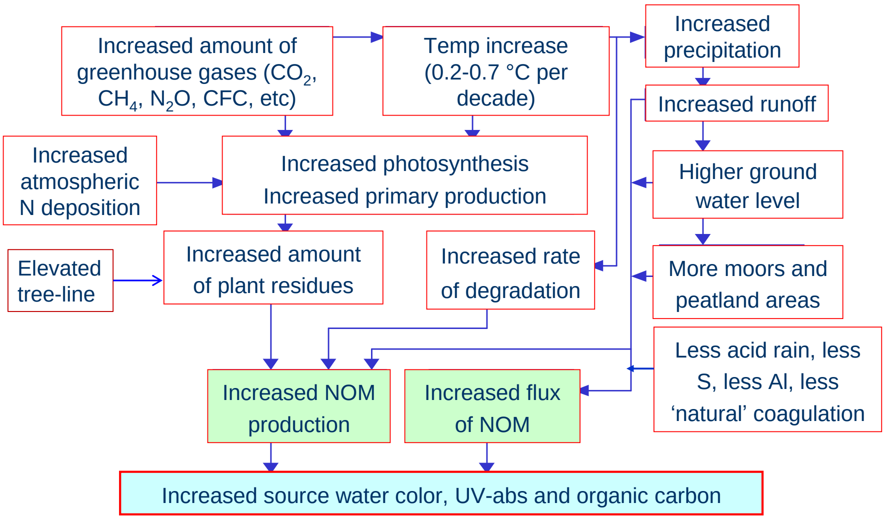


Figure 1. Possible explanations for increasing NOM concentrations (Eikebrokk et al 2004; Modified from Forsberg (1992), Liltved (2002)

It is observed that periods with heavy rain can give a rapid increase in colour and TOC in surface waters. Increasing NOM in drinking water reservoirs can also impose taste and odour problems, in specific if the effects of climate change also include increased algae-related challenges. If not removed properly during water treatment, increasing NOM will increase coagulant and disinfectant demands and sludge production. In addition, increasing NOM may reduce disinfection efficiency and the potential for disinfection by-products formation (BDPs). NOM may also foul membranes and activated carbon filters, thus increasing cleaning/regeneration frequencies and reducing treatment capacities. Increasing NOM may also increase biological regrowth and biofilm formation in distribution networks, not only outside but also inside of buildings thus increasing the risk for health-related problems caused by amoeba and pathogenic microorganisms (e.g. *Legionella*).

Figure 2 shows historic and predicted future NOM concentrations in Oslo's water sources. The predictions are based on historic measurements from the years 1973-2011. The increase in raw water colour levels is evident for all sources, and this will of course significantly affect selection, design and operation of existing as well as future (planned) treatment and distribution processes.

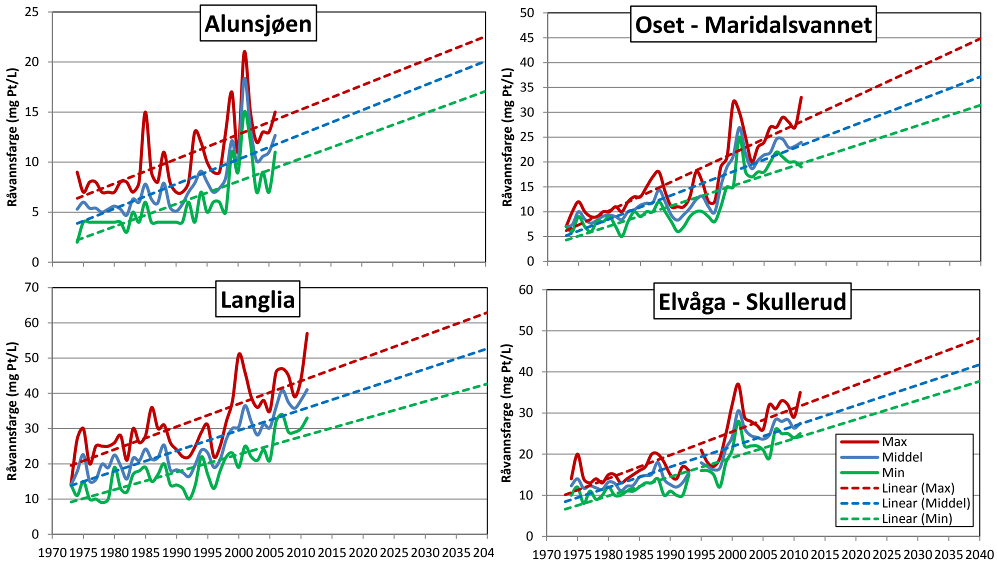


Figure 2. Historic and predicted levels of NOM (colour) in Oslo's raw water sources (lakes).

Similar rates of NOM increase like the ones in Norway are observed also in several other countries, including UK/Scotland, Sweden, Germany, USA, and Canada. In northern and western Scotland, some water sources have reached rather extreme colour levels in the range of 400-600 mg Pt/L.

Contrary to the situation in Norway where the reduction in acid rain is expected to significantly slow down the rates of NOM increase in a few years time, the situation in Scotland shows no sign of declining rates of NOM increase (Weir, 2012).

In some countries (UK, USA, Australia, Norway, etc.), NOM and NOM removal have been priority R&D topics for quite some time. In Sweden, significant R&D activities have been initiated quite recently in order to cope with the increasing NOM levels.

Increasing water temperature levels

Higher water temperatures will trigger the growth of blue-green algae. Increasing NOM (colour) levels may however reduce the risk for growth of blue-green algae due to less light penetration and less availability of visible light.

Increasing temperatures will also facilitate increased biological activity in water reservoirs and pipelines, thus increasing the risk for increased growth and prolonged survival of pathogens. Protozoan parasites like *Cryptosporidium* and *Giardia* will however shorten their survival times when temperature levels increase.

Increasing temperatures may also lead to increased oxygen deficiencies in deep water layers, thus leading to increasing challenges related to iron and manganese.

Increased runoff
In winter, the more frequent changes between cold and mild periods will lead to more situations of rain on snow-covered area thus increasing the intensity and frequency of peak run-offs. This may lead to more flooding in the surrounding areas of the reservoir and to more problems with increasing levels of eutrophication, NOM and microbiological activity in the drinking water reservoirs.

Increasing risk for contamination of surface waters
With increasing precipitation amounts and precipitation intensities, increasing runoff/flooding frequencies and increasing NOM levels there will also be a higher probability of contamination events and more pathogenic microorganisms in surface waters.

Longer periods of non-frozen lakes
Warmer climate can bring along longer periods of non-frozen lakes and ponds (ice-free periods). This makes water bodies more accessible for animals and birds, thus allowing new species to establish in and around water sources.

The presence of a larger number of animals and birds for longer periods of time in the catchment areas may also increase the risk for microbiological water contamination. To cope with this challenge, water treatment may need to be improved in the future.

Finally, the stabile stratification of lakes is secured by the ice cover during wintertime. With a shortening of the time period with a protecting ice-cover, heavy winds during winter time can also disturb the stable stratification and result in mixing of surface water with the deeper water layers of lakes. This will reduce the protective effect of a deep water intake as a hygienic barrier. Additionally, heavier winds on the reservoir surfaces will lead to more water circulation, horizontally as well as vertically.

More microorganisms in groundwater due to increasing runoff
The concentration of microorganisms in groundwater may increase due to increased runoff from the catchment areas and higher presence of animals and birds. There will also be a higher risk for increased pollution and micro biological activity due to increased precipitation. More floods may bring more contamination into catchment areas of groundwater reservoirs.

Salinity problems in coastal areas due to sea level rise
Projected sea level rise and excessive groundwater extraction in coastal areas (in cold climate regions where groundwater is used for water supply) combine to increase the risk of salinity problems in water supplies (Hetzel, 2008).

Groundwater quality changes due to increase in precipitation
Water quality is controlled by land surface characteristics, vegetation cover and soil properties, which may change due to climate changes. For example increased precipitation and wetter climate will facilitate the formation and expansion of marches, which will affect the water quality in such areas.
Groundwater quality may change due to increase of precipitation

Water quality is controlled by land surface characteristics, vegetation cover and soil properties, which may change due to climate changes (Hetzel, 2008). For example increased precipitation and wetter climate will facilitate the formation and expansion of marches, which will affect the water quality in such areas (Zwolsman et al., 2007).

Larger (seasonal) and more rapid variations in raw water quality will pose greater challenges to treatment plant operations
Because of climate change drinking water treatment plants will have to be adaptable to changes in raw water quality. Extreme events, like flooding, will instantly affect the drinking water supply. Water bodies' quality is subjected to weather seasonality which results in large seasonal variations in raw water quality.

Variations in the raw water quality makes it difficult for operators of treatment plants to predict the need for dose of chemicals etc. and it may be difficult to get a stable and good, clean water quality.

A warmer climate may result in more algae and more toxins in raw waters
Higher amounts of algae and algal toxins are expected in surface water reservoirs due to warmer climate, more precipitation, more runoff, more nutrients and more strain on wastewater systems. Higher water temperatures and increased amounts of nutrient will boost the growth of algae in reservoirs. Problems associated with algae blooming are taste and odour and algal toxins.

Need of improved treatment of groundwater
Like for surface water, there will in some instances be a need for improved and extended treatment of ground water due to the changed conditions for groundwater caused by climate change. In many places, groundwater wells are already contaminated , unprotected or close to becoming dysfunctional due to a lowering of the groundwater table close to or below the bottom of the well or due to low and poor maintenance (Hetzel et al., 2008).

Increased use of resources, more waste and more emissions
Increasing NOM levels in raw water will require increased coagulant doses, increased sludge production and reduced length of filter cycles in the water treatment. More frequent membrane washing and more frequent activated carbon regeneration will also increase the use of resources (energy).

Increased risk of contamination from sewers and groundwater

In some countries where water and sewer pipe is in the same ditch, the risk is higher for contamination from sewer due to generally higher flow in sewer pipes. Also, more rain will in some places raise the ground water table and increase the risk for ingress into water pipes. Risk of ingress is especially high during episodes with repair of water pipes or other incidents when pressure might be reduced (like leaks).

Increased risk for biological regrowth

If treatment plants are not able to cope with the high NOM levels in the raw water, this will be transferred into the transport and distribution system. The NOM will work as food and facilitator for biological growth in the bulk flow of water and in biofilm media on the pipe walls. In the pipe network it will be a cause for extended growth problems. Increased water temperature in the pipes in the summer months can also contribute to a higher level growth of microorganisms in the network. High water temperatures will facilitate bacteria growth.

Increased external corrosion due to higher groundwater level

Due to increased precipitation and increased runoff and infiltration, the ground water table will be higher in some places. This will cause the drinking water and sewer pipelines to be in contact with the ground water table more frequently. This may lead to increased external corrosion problems.

3.4 Summary

The waterworks will have to cope with increasing climate-change related challenges in water treatment and distribution:

- More microorganisms
    - Increased concentrations
    - New and emerging species
    - Reduced stratification and less protected deep water in lakes
- More taste/ odour and algae toxins
- More NOM
- Faster and more severe raw water quality changes
- More biofilm formation in the distribution system
- Increased demand for water treatment and disinfection chemicals
- Increased risk of contamination from sewer pipes and groundwater
- Increasing risk for intrusion of saline water into groundwater
- Need for optimized treatment and/or supplementary treatment
    - More challenging treatment
    - More challenging process control
    - More challenging distribution
    - Increasing use of resources (chemicals and energy)
    - Increasing sludge/waste production
    - More emissions of CO_2/GHG

4 Cyanotoxins

Edwin Kardinaal (KWR)

4.1 Challenges to drinking water quality

Cyanobacteria are widely distributed in all kinds of ecosystems. They are commonly found in freshwater systems, in marine systems and furthermore occur in soils, in Antarctic melt water ponds and on rocks. Cyanobacteria inhabit the world's ecosystem for over 3 billion years and are masters in adapting to changing environments. Currently cyanobacteria are part of our natural environment, but in case of mass development cyanobacteria become a nuisance. Especially in surface waters cyanobacteria can form dense blooms resulting in scum or mats frustrating bathing and / or drinking water production.

Worldwide countries rely on the extraction of surface waters for the production of drinking water. Some countries only use some percentages others depend for 100 % on surface waters for the production of drinking water (IWA, 2010). Especially those surface waters enriched with nutrients run the risk of inhabiting the potential of cyanobacterial bloom formation (Paerl & Paul, 2012). Since cyanobacteria can produce a whole range of potent toxins (see Table 1) people consuming waters infected with toxic cyanobacteria may be subject to into serious health risks.

Table 1 Attributes of several cyanotoxins (from: Sivonen & Jones in Chorus & Bartram, 1999; Cox et al., 2005)

Cyanotoxin or irritant	Effects	Cyanobacterial genera
Cyclic peptide		
Microcystin	Liver, skin	*Microcystis, Anabaena, Planktothrix (Oscillatoria), Nostoc, Anabaenopsis*
Nodularin	Liver	*Nodularia*
Alkaloïds		
Anatoxin a	Synaps (nerv)	*Anabaena, Planktothrix (Oscillatoria), Aphanizomenon, Phormidium*
Aplysiatoxin	Skin	*Lyngbya, Schizothrix, Planktothrix (Oscillatoria)*
Cylindrospermopsin	Liver, cell	*Cylindrospermopsis, Aphanizomenon, Umezakia*
Lyngbyatoxin a	Skin, gastro – intestinal track	*Lyngbya*
Saxitoxin	Axon (nerve)	*Anabaena, Aphanizomenon, Lyngbya, Cylindrospermopsis*
Lipopolysaccharide	Skin	All
MBAA (beta-N-methylamino-L-alanine)	Nerve	All*
DAP (alpha-,gamma-diaminobutyric acid)	Cell	All

*the presence and effect of MBBA is under dispute, see e.g. Faassen *et al*, 2012

Cyanobacterial genera like *Anabaena*, *Aphanizomenon*, *Cylindrospermopsis*, *Microcystis* and *Planktothrix* are the main potential producers of a whole range of peptides (Welker and von Döhren, 2006) of which the toxins microcystin, anatoxin and saxitoxin are most relevant since the intake of these cyanotoxins may cause gastrointestinal diseases, liver damage, paralysis or even death in humans and animals (Codd *et al*, 2005). Over the last 20 – 30 years numerous incidents of harmful effects on human beings and the death of animals in combination with massive cyanobacterial blooms have been reported. The first report on animal poisoning due to cyanobacterial blooms dates back to 1833 (Codd *et al*, 2005), but only since the 1980s the study on cyanobacterial toxins has been intensified. This increased interest was merely a result of eutrophication of freshwater bodies leading to an increasing dominance of (toxic) cyanobacteria. Furthermore, improved techniques for the analyses of cyanotoxins helped in further exploration of the cyanotoxins (Sivonen and Jones, 1999).

Besides the production of toxins within aquatic ecosystems, most pelagic cyanobacteria have the ability to float to the water surface and can form dense mats, so called cyanobacterial scum.
The flotation is the result of gas vesicles in the cyanobacterial cells, small hollow cylinders filled with air that provide buoyancy. The volume of gas vesicles is rather constant but the density of cells is changing under influence of light. In the light, dense carbohydrates are formed by photosynthesis that counterbalance the positive buoyancy by gas vesicles (Kromkamp and Mur, 1984; Visser *et al*, 2005). Hence, buoyancy varies in response to irradiance, in the dark or at low irradiance the cells become buoyant and at high irradiance they lose buoyancy and sink. Such buoyancy regulation prevents the colonies from high light intensities at the surface and prevents the cells to sediment in case of stable conditions in the water column (Visser *et al*, 1996). In case of reduced mixing, buoyant colonies can rapidly float up to the water surface of lakes (Huisman *et al*, 2004). This may lead to the formation of dense surface scum (Figure 3).

Figure 3. Scum formation of cyanobacteria in infiltration basin for drinking water production, the Netherlands.

The combination of cyanotoxin production and scum formation may lead to serious toxic surface waters (Figure 4). In the Netherlands toxin concentrations of more than 10.000 µg /l are no exception in inland surface waters. Such concentrations can be harmful to humans or animals consuming such waters.

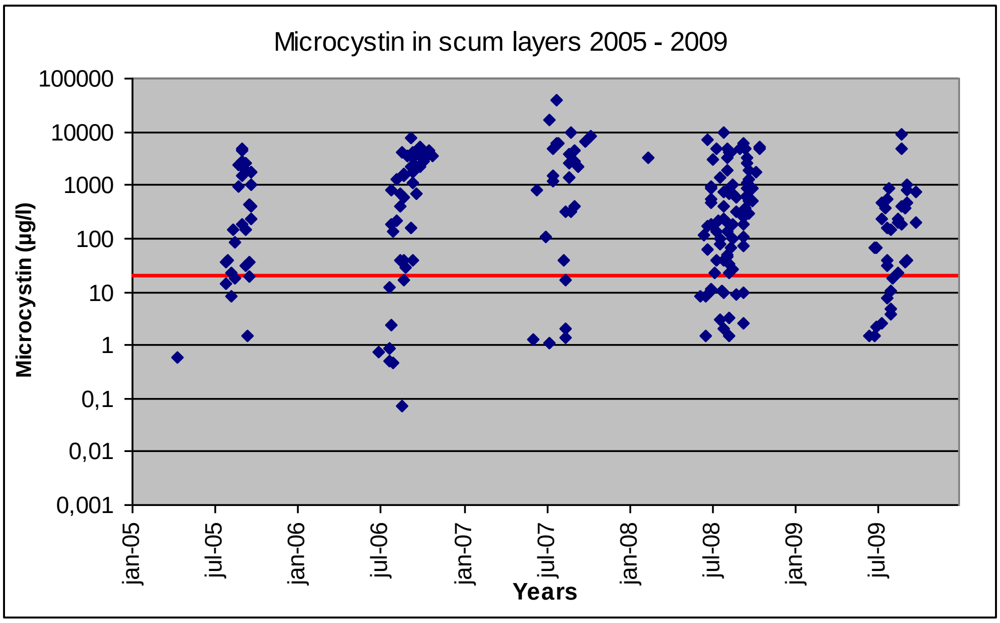


Figure 4. Concentrations of microcystin-LR equivalents in in surface scums of surface waters in the Netherlands, 2005-2009 (Ibelings et al. , 2012). The red line indicates a microcystin level of 20 µg L-1. Note the logarithmic scale.

4.1.1 *Human exposure*

Humans can be exposed to cyanotoxins in three ways: 1. consumption of contaminated drinking water, 2. haemo-dialysis with contaminated water, and 3. recreation in water containing scums or dense cyanobacterial blooms (Codd et al, 2005). Examples of incidents include the outbreak of gastroenteritis in three villages in Sweden that received drinking water contaminated with lake water containing *Planktothrix* filaments in 1994. A total of at least 121 persons fell ill after consumption of the water (Annadotter *et al*, 2001). In Rio de Janeiro, drinking water contaminated with microcystins was distributed to the community, including dialysis centres, in November 2001. Patients were exposed to this water and sub lethal microcystin concentrations were found in blood sera of these patients (Soares *et al*, 2006). In an earlier incident in a dialysis centre in Brazil, water used for dialysis contained microcystin-producing cyanobacteria. Here, 60 patients died as a result of acute liver failure (Carmichael *et al*, 2001).

4.1.2 *Guideline levels*

In general the guideline levels are based upon microcystin. Microcystin is the most abundant cyanotoxin worldwide and toxicology studies are mainly performed with this toxin. Microcystin has numerous structural variants. The microcystin LR structural variant is among the most toxic. The LD_{50} (in mice) of microcystin LR is 50 – 60 $\mu g.kg^{-1}$ bodyweight (BW). When compared to other natural toxins the LD_{50} of microcystin LR is comparable to the LD_{50} of cobra toxin and 10 times lower (i.e., more toxic) than the LD_{50} of curare, a toxic plant extract used by South American Indians for hunting. Based on mouse studies, in which mice were fed with the purified microcystin LR, the No Observed Adverse Effect Level (NOAEL = highest dose associated with the absence of adverse health effects) was 40 $\mu g.kg^{-1}$ BW (Chorus and Bartram, 1999). After a correction factor for the transition of mouse to human exposure the provisional Tolerable Daily Intake (TDI = daily dose causing no effect) for humans was set on 0.04 $\mu g.kg^{-1}$ BW. For the calculation of the safe concentration in drinking water, 1 μg / L, the TDI value was multiplied by the average bodyweight of a human being (60 kg), drinking 2 litres of water per day (0.04 μg /kg × 60 kg / 2 litres of water ≈ 1 μg / L).

4.1.3 *Impact of climate change on cyanobacterial abundance*

Climate change scenarios predict among others the rise of temperature, possibly resulting in the enhanced vertical stratification of aquatic ecosystems, and changes in seasonal weather patterns. The latter may result in increased run-off or droughts. Such alterations are all in favour of the growth of cyanobacteria (see Figure 5). In short:

- Increased water temperature => stronger competitive abilities
- Decrease in water viscosity => allows better vertical transport
- Increase solar irradiation => defence systems allow longer exposure
- Increase in nutrient availability => favour cyanobacterial dominance
- Increase salt intrusion => cyanobacteria are tolerant

- Decrease in run off in dry periods => increase in retention times => favour cyanobacterial dominance

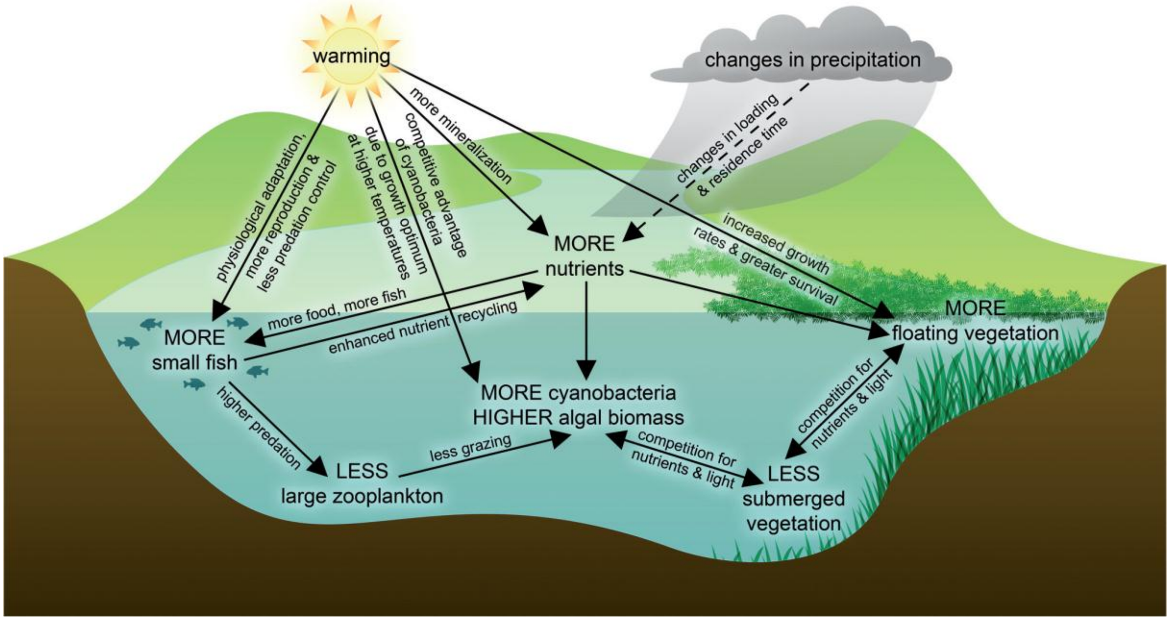


Figure 5. Some relationships now established that link climate change and eutrophication symptoms (Kosten et al, 2011).

The mass development of cyanobacteria has several negative impacts on the ecology of lakes, reservoirs and lagoons. Apart from the toxicity the dense blooms result in a reduced visibility leading to the suppression of the growth of other phytoplankton species and aquatic macrophytes (Scheffer, 1998). Moreover the cyanobacterial blooms may cause oxygen depletion during night time and / or at the time of oxygen demanding biomass degradation. This may lead to major fish kills.
Dense cyanobacterial blooms can also cause taste or odour problems in drinking waters. Substances like geosmin can give drinking water an earthy flavour. (Izaguirre and Taylor, 2004; Uwins *et al*., 2007).

Nutrients
Nutrients involved in massive bloom are mainly nitrogen and phosphorus. Nitrogen can enter water bodies as leachate from soils, as run-off from agricultural land, by way of (un)treated sewage and as atmospheric nitrogen deposition, mainly resulting from fossil fuel combustion.
Phosphorus is biologically available as phosphate. The routes of entry into water bodies are comparable to the routes of nitrogen. However in general in fresh water systems phosphorus is the limiting source for primary production. So the availability of phosphorus frequently places an upper limit upon the biomass and productivity in freshwaters. Some planktonic algae, among which cyanobacteria, have developed mechanisms to store available phosphorus.

Cyanobacterial blooms often develop in eutrophic lakes. This may imply that cyanobacteria require high nutrient concentrations. However, competition experiments have shown that cyanobacteria do not always have a competitive advantage, when compared to green algae. However some cyanobacterial genera do have a greater affinity to phosphorus and nitrogen than other planktonic species. Moreover heterocysts enable other cyanobacterial genera to fix atmospheric nitrogen during aquatic nitrogen starvation and they have a storage capacity for phosphorus. This implies that under conditions of nutrient limitation, cyanobacteria can out-compete other planktonic species. Apart from nutrients, light is an important "source" for cyanobacteria and other algae. Several genera of cyanobacteria (e.g. *Planktothrix*) are good competitors for light. Genera that might have less competitive strength for light may have other capabilities to gain sufficient light for the photosynthesis. Especially the contents of gas vacuoles within certain cyanobacterial cells make the cells float to the surface of the water column, where as other genera may sink to deeper parts of the lake system. Near the surface light conditions are most favourable for growth, so cyanobacterial cells with buoyancy control have a competitive advantage. Moreover the massive growth of buoyant cells may shade other organisms in deeper parts that do rely on light as well.

Temperature effects
Increasing temperature may enhance cyanobacterial growth in several ways. The increase of air temperature may result in higher temperatures of surface waters. In general the assumption is that optimum growth temperatures for cyanobacteria are slightly higher than for diatoms and green algae (Paerl & Huisman, 2009). Although recent studies imply that warmer water will not necessarily result in the competitive advantage of cyanobacteria (Lurling *et al*, 2013). The rise in temperature enhances the formation of vertical stratification. Due to the gas vesicles cyanobacteria can migrate through the water column and prevent sedimentation to deeper layers. Other phytoplankton will sink to the deeper layers. Moreover in warmer water the viscosity will decrease, which enhances the velocity of the vertical transportation. So stratification and viscosity indirectly might increase cyanobacteria growth rates because it improves the competitiveness of buoyant species over sinking species (Huisman *et al*, 2004). The warmer water will also change the bloom-susceptible periods and will affect the composition and the successional patterns of cyanobacterial and eukaryotic algae in aquatic ecosystems. Such long time periods of quiet weather conditions are in favour of the bloom formation of cyanobacteria (Paerl & Paul, 2012)

Carbon dioxide effects
The rising level of carbon dioxide is a key component in the global warming. In surface waters with dense cyanobacterial blooms CO_2 can be depleted to due to high photosynthesis demands. As a result pH may rise up to 10 or even higher. In such circumstances the lack of CO_2 supply can reduce algal growth. Cyanobacteria may once again benefit from their ability to float. The cells can migrate to the water surface and directly intercept CO_2 from the

atmosphere. In contrast, the rising CO_2 levels may lead to acidification of freshwater en marine systems not dominated by high densities of algae.

Rising salinities
As a result of increasing periods of drought the salinity in inland waters and estuaries may rise. Moreover as a result of rising sea water levels the intrusion of salt groundwater may attribute to the further rising salinities in such waters. The increasing demand for fresh water for drinking and irrigation purposes may further enhance the effect of the droughts. Moreover as a result of drought the intrusion of salt wedges in riverine en estuarine systems may also increase the salinities of inland waters.
Rising salinity is (again) favourable for cyanobacteria in comparison to the survival of freshwater algae. Several (potentially) toxic cyanobacterial genera and species are quite salt tolerant. For example the growth rate of *Microcystis* is not negatively affected by salt concentrations up to 10 g L^{-1}, or 30% of seawater salinity (Tonk *et al*., 2007). The same has been observed for some *Anabaena* species that can withstand salt levels up to 15 g L^{-1}. *Anabaenopsis* and toxic *Nodularia* species can even withstand salinities up to 20 g L^{-1} and more. In stratifying systems salt intrusion can result in a stronger stratification with a relative warm freshwater upper layer and a cold saltier deeper part. Stratification is of major advantage for the buoyant genera like *Microcystis* and *Anabaena* (Visser *et al*,. 1995).

Quiet / dry weather
As stated above nutrients may enter water bodies as a result of increasing run off because of increased precipitation. On the other hand climate scenarios predict periods of drought with hardly any rainfall. The decreasing run off in such cases results in relatively quiet waters. Quiet water once again favours the growth of cyanobacteria. The lack of natural mixing (wind induced) makes that cyanobacterial cells migrate through the water column without restrictions. The ability to stay in the upper part of the water make the availability of light ensured. While sinking algal species cannot keep their position within the water column and sink to the bottom where light conditions are less favourable.

Implication for drinking water production
From the paragraphs above it can be concluded that climate change, especially increasing water temperatures and increasing run-off, somehow favour the growth of cyanobacterial biomass. The intake of surface water for the production of drinking water may encounter increased problems. The biomass in itself results in increased organic materials in the production line and possibly in the final end product.
Apart from the increased biomass cyanobacterial cells are well capable in producing a whole set of cyanotoxins (see above). To prevent such toxins ending up in the final tap water at least one step within the production should sufficiently remove such toxins from the water. This is going to be a challenge for future drinking water production utilities that rely on surface water as the major source for drinking water.

4.2 Current operational practices

Despite the potential risk for human public health the incidents with cyanobacteria and cyanotoxins in drinking water have been limited. This is basically the result of the designs of the water treatment plants. Cyanobacterial cells and toxins are mostly removed from the raw waters within conventional drinking water production processes.
However the (massive) blooms of cyanobacteria may have impact on the drinking water production processes. Each treatment step will have its own efficiency in removing cyanobacterial cells and cyanotoxins (see Table 3). Problems that may appear when (massive) blooms of cyanobacteria enter the production process are (Petruševski, 1996; Hitzfeld *et al*, 2000; Hoeger *et al*, 2005):

- If cyanobacteria cells are lysed, toxins enter in the process water, toxins, however, are poorly removed by several treatment procedures like: coagulation, conventional flocculation, filtration, and chlorination.
- Cyanobacteria increase the concentration dissolved organic carbon (DOC). High DOC concentrations can disrupt the coagulation process, resulting in a higher coagulant dose required, besides such concentrations can frustrate ozonation processes (higher O_3 demand and possibly more by products);
- Especially colony or thread-forming algal species can cause the clogging of filters, and back flushing may again lyse cyanobacterial cells;
- Cyanobacteria that penetrate treatment systems that use chlorine for disinfection purposes, increase chlorine demand and act as precursors for the formation of trihalomethanes and other halogenated by-products, such as chloroform;
- The removal of taste and odorous compounds (like found in benthic cyanobacteria) demands an intensive purification process;
- Cyanobacteria that pass filters result in higher AOC-levels (Assimilable Organic Carbon) which promotes after growth of bacteria in the distribution network.

So in the drinking water treatment first focus should be on the removal of cyanobacterial cells, preventing the cells to lyse. Whereas subsequent processes should be able to remove toxins sufficiently.

4.3 Detecting rapid changes that lead to risk

The intake of cyanobacterial biomass in drinking water utilities can change rapidly. Weather and wind circumstances can make cyanobacterial biomass change within hours. A continuous monitoring is preferable but not always possible. The analyses of the actual danger, the toxins, take some time. At the moment results are available the actual situation in the reservoir may be changed. So instead of measuring cyanotoxins the observation of cyanobacterial biomass as indicator of actual risk is performed.

4.3.1 *Abundance and biomass of cyanobacteria*

In many countries risk assessment in relation to cyanobacteria focuses on cyanobacterial abundance and biomass rather than on microcystin (or other toxin) concentrations. As a result, the question raises how to determine these parameters with reliable approaches. The disadvantage of this approach is that no clear relation between a biomass signal and toxins exists because all cyanobacteria contribute biomass or contain pigments, while not all cyanobacteria produce toxins. Thus, measures of cyanobacterial abundance or biomass always provide only an indication of the upper limit of cyanotoxin concentrations to expect. However, this information is very useful for risk assessment and management especially in (daily) monitoring. Different methods to estimate the abundance or biomass of cyanobacteria can be applied:

Chlorophyll a

Method based on spectrophotometric analysis after extraction of the pigments with e.g. ethanol. Advantage of this method is the ease of use. The parameter can be routinely measured and is therefore relatively fast and cheap. Disadvantage is that the method is not specific for cyanobacteria; all phytoplankton contains chlorophyll a. Extraction and analysis of specific cyanobacterial pigments like phycocyanin is less straightforward. However, the analysis of Chlorophyll a can be supplemented by qualitative microscopy, i.e. estimating (without cell counting, usually within 10 minutes or less) whether the phytoplankton seen in the microscopic image largely consists of cyanobacteria or not.

Fluorescence

Method based on different light absorption spectra (excitation wavelengths) of pigments in cyanobacteria, green algae and diatoms. Sensors can be applied in field situations. Advantage of the method is the instant results of the measurements, which can be obtained in-situ. The parameter can be routinely measured and is therefore relatively fast and cheap. Some sensors measure not only Chlorophyll-a, but also a pigment specific to cyanobacteria, i.e. phycocyanin and thus allow their distinction from other phytoplankton.

Microscopic counts / estimates

Method based on the analyses of concentrated cells in a sedimentation chamber, counted under an inverted microscope. Advantage of this method is the direct insight in the composition of the phytoplankton population in general and the possible abundance of potential toxin producing genera / species of cyanobacteria. Disadvantage is that the method is time consuming (and therefore relatively expensive) and a specialist job. The quantification is complicated by growth forms of cyanobacteria, like colonies, filaments or coiled or twisted growth forms. Cell counts between laboratories have been known to show a high level of variation.

Biovolume

Like above (microscopic counts), but in addition the cell numbers are multiplied by the cell volume of each genus / species to provide an estimate

of biomass. Advantage of using this method is that the relative biomass of cyanobacteria in relation to other phytoplankton genera can be estimated. Cell sizes are incorporated in the measurements, dominance of phytoplankton genera and species can be easily evaluated.
Disadvantage of the method is comparable to the microscopic counts. Furthermore, this method introduces an additional parameter, with a certain distribution and variation and scope for potential errors, the volume of each cell.

DNA-copy detection
Method based on the extraction of DNA and the multiplication of certain gene targets. Targets are either on the phycocyanin genes or on the genes encoding for the toxin production. Advantage of the method is the objective way of quantification. The targets are genus specific. The method is relatively fast (result within hours), and sensitive, so that low concentrations of cells can be detected. Disadvantage is the need for a well-equipped laboratory. Besides, the relation between cell numbers and DNA copies can be a source of variation. Overestimation of the cell densities may occur. Detection of toxic genes may not relate to actual toxin production and concentrations.

4.3.2 *Toxin analyses*

The diversity of cyanotoxins as they may be produced by numerous genera of cyanobacteria is illustrated in Table 2. The characters of the set of cyanotoxins differ in for example hydrophobicity and the mode of action in the receiving organism. The diversity of the cyanotoxins makes it hard to scan the whole spectrum of cyanotoxins with one single analysis method. In Table 2 the analysing methods are illustrated and along with the toxins that can be detected.

Table 2. Range of analysis methods for the detection of cyanobacterial toxins (Loftin et al., 2010)

Bioloigcal Assays:	Anatoxins	Cylindrospermopsins	Microcystins	Nodularins	Saxitoxins
Mouse	Y	Y	Y	Y	Y
PPIA	N	N	Y	N	N
Neurochemical	Y	N	N	N	Y
ELISA	?	Y	Y	Y	Y
Chromatographic Methods:					
Gas Chromatography:					
GC/FID	Y	N	N	N	N
GC/MS	Y	N	N	N	N
Liquid Chromatography:					
LC/UV (or HPLC)	Y	Y	Y	Y	Y
LC/FL	Y	N	N	N	Y
Liquid chromatography combined with mass spectrometry:					
LC/IT MS	Y	Y	Y	Y	Y
LC/TOF MS	Y	Y	Y	Y	Y
LC/MS	Y	Y	Y	Y	Y
LC/MS/MS	Y	Y	Y	Y	Y

What method fits best for the actual toxin analysis depends on the research / monitoring questions. In case of monitoring for regular monitoring programs a fast and cheap method would prevail over an analytical tool that is very specific and sensitive. In case of scientific questions this may be the other way around.
For most methods some comments can be summarized:

- Bio assays: not specific may react on other substances than cyanotoxins, can be used for first screening.
- Phosphatase inhibition: rapid accurate method, so far just used for the detection of microcystins.
- ELISA: kits are available for (almost) all cyanotoxins. Good method for general screening, no information on specific (more or less) toxic variants (especially microcystins).
- HPLC UV / LF: a selective and quantitative method. Possibly not all toxins can be detected with this method.
- LC/MS: very selective can be used for a wide spectrum of cyanotoxins. Investments are pretty high and well trained employees are essential for operating the system. Extracting the cyanotoxins from cells and surface water is labour intensive.

4.4 Corrective responses and curative measures

Worldwide, drinking water production utilities and managers of bathing waters locations may experience severe problems with dense blooms of cyanobacteria. As a consequence bathing and /or drinking water production need to be banned or stopped to prevent risks of (severe) health problems.
The first step to in prevent people being exposed to cyanobacteria and cyanotoxins is the reduction of the actual cyanobacterial biomass in the water systems, like streams, rivers, ponds and storing basins. Measures in reducing the growth of cyanobacteria are for example the reduction of concentrations of nutrients in the catchment area or in the water system itself.
Whenever a bloom of cyanobacteria does occur, measures within the basis itself can lead to the control of the overwhelming growth of the cyanobacterial community. Such measures can reduce concentrations of cyanobacteria to acceptable levels.
Also in case of drinking water production, measures within the production process itself will prevent the abundance of cyanotoxins in drinking water. Condition in that case is that the focus in first production steps should be on the removal of intact cyanobacterial cells and later in the process the removal of cyanotoxins.
So, basically three levels of prevention / removal can be distinguished: reduction of bloom formation (catchment level), bloom control within the actual water system (basin level) and during the drinking water production by removing cells and toxins. (production level). Below several samples of measures that proved to be successful on the three different levels are presented.

4.4.1 *Catchment measures*

Measures in the catchment should in the first place focus on the reduction of nutrients, in fresh water systems basically on phosphorus.
For an effective strategy in reducing the nutrient load in a water body an inventory of all nutrient sources needs to be performed. Together with the water quantities that do enter the system, an impression of the most important nutrient sources may lead to selective measures with a high impact on the actual nutrient load.
Examples of measures to prevent the entrance of nutrient rich waters within the water system are the diversions of streams, improving the phosphorous removal of waste water treatment systems, or removal of phosphorous enriched sediments. Sediment can also be capped with substances that bind to the phosphorous like Al^{3+} or Phoslock® and in addition the sediment can be covered with phosphorous free sand (Lurling & Oosterhout, 2013).

4.4.2 *Basin measures*

Measures on the basin level that proved to be effective are the addition of hydrogen peroxide (low concentrations) or the artificial mixing of (deep) basins.
First tests with the addition of low concentration of hydrogen peroxide on a full lake scale were performed in 2009 in the Netherlands (Mathijssen *et al*, 2012). A standing population of *Planktothrix* resulted in a ban for swimming in lake Koetshuisplas in Veendam in the north of the Netherlands. The addition of hydrogen peroxide in low concentrations (end concentrations of 2 mg / l) resulted in a complete collapse of cyanobacterial population. Also toxin levels dropped dramatically to levels below detection. Effects on other (non cyanobacterial) organisms could hardly be detected. Those that did occur, were restored within weeks the population of e.g. zooplankton returned to initial levels (before addition of the hydrogen peroxide).

In the Netherlands a long history exists on the artificial mixing of lakes and basins. Since the early 90th's of the last century vertical mixing of deep systems is a successful method in controlling the cyanobacterial growth (Visser *et al*, 1996, Huisman *et al*, 2004). Both in lakes with several functions (bathing, living boats, and fishing) and storing basins (raw water for drinking water production) the method has been applied for years. The basic principle of the method is pumping air into tubes that are located close to the bottom of the lakes. The air escapes from the tubes like from an upside down showerhead. The air bubbles result in an upstream flow and the surrounding water is "lifted" to the water surface. In the other direction, surface water will flow to the deeper parts of the lake to "replace" the water that flows to the surface. This circulation needs to be faster than floating cyanobacteria can migrate to the surface naturally. Cyanobacteria are thus forced to migrate to deeper parts of the lake / basin where light penetration is low and growth conditions for cyanobacteria are less favourable than at the surface. This way the competitive advantage of the floating cyanobacteria is reduced and so is the growth (see Figure 6).
Recently water managers slightly changed the concept: instead of pumping air, oxygen is being pumped into the tubes close to the bottom of the lakes.

Apart from the "air lift" the oxygen levels near the bottom increases. As a results phosphorous, stored within the bottom, will be fixed to iron. This method prevents anaerobic conditions that could lead to internal phosphorous enrichment.

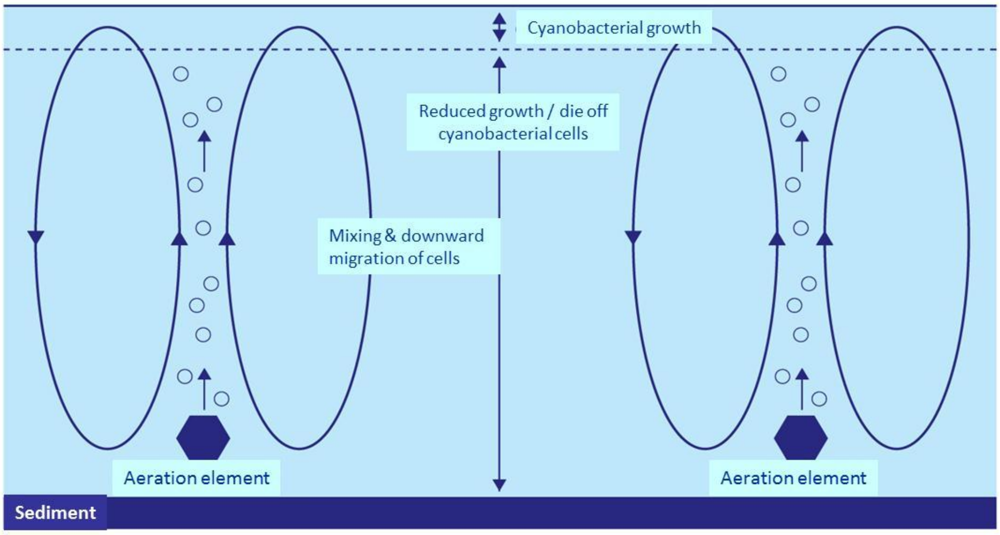


Figure 6. Circulation in a lake system due to aeration of the water column. Circulation will result in a reduced growth of cyanobacterial cells.

Apart from vertical mixing water managers of bathing water locations experimented with horizontal mixing. Especially on sites where scums tend to accumulate on bathing water locations, the application of this method seems successful. In combination with dams to lead the water that contains high densities of cyanobacteria away from the bathing location, this method is helpful.

4.4.3 *Production measures*

The efficiency of removing cyanobacteria and the cyanotoxins during drinking water production strongly depends on the production process itself and the number and types of barriers included. Production processes should in the first place focus on the removal of intact cells of cyanobacteria, since most toxins are cell bound. So by removing the cells the toxins will be removed as well.

As discussed above the first step is reducing the growth of cyanobacteria in the source water. The best way to do so is by creating storing basins that, in case of cyanobacterial dominance, can be treated more easily than the whole water systems.

When pumping water into such basins, the first treatment step can be applied: by adding flocculants (e.g. Al^{3+}) floating substances, like cyanobacteria, may sink to the bottom of the basin and be stored in the slick. Besides such flocculants may reduce the nutrient load into the basin, so the growth of cyanobacteria within the basin may be slowed down.

The actual efficiency of applied methods in drinking water production is summarized in Table 3 below.

Table 3 Ranges of removal efficiency of treatment processes.

Production process	**Removal efficiency *Microcystis* cells**	**Removal efficiency extracellular microcystins**	**References (see 4.7 for details)**
Micro sieves	60%	0%	12
Softening ($Ca(OH)_2$)	0%	0%	n.a.
Coagulation/flocculation/sedimentation	70 – 100%	0 - 12%	3, 11, 15, 17, 20, 21, 16, 23
(rapid) sand filtration	>60%	0 %	8, 11, 20
Slow sand filtration	99%	63 - 99%	5, 6, 11, 20
Coagulation, Flotation	80%	0%	20
Bank filtration	99,9%	75 - 99%	9
Dune infiltration	>90%-8log	NB	10
Granular Activated Carbon	>60%	40-90%	7, 11, 20, 22
Biological Activated Carbon	NB	70->90%	11, 22
Powder Activated Carbon	0%	40 – 98%	7, 11, 13, 20, 22
Ozone	39 - 72%	50 – 100%	7, 9, 11, 14, 15, 18, 19, 20
Chlorine dioxide (0,04-0,2 mg/l)	30-75%	<10%	1
UV disinfection*	<20%	<10%	1, 2, 16
UV/ H_2O_2**	NB[2]	50 – 90%	16, 20
Micro filtration/Ultra filtration	99%	0%	4, 16, 23
Nanofiltration/ Reversed Osmosis	99%	95%	4, 16,20, 22

* Possibly contra productive in case cyanotoxin cells break up as a result of the UV radiation (Ding, 2010). Cyanotoxins will not be disintegrated with doses of 40-70 mJ/cm². Higher doses may have a positive effect.
**No data on inactivation of cyanobacteria in a UV/H_2O_2 process could be found. Cheng *et al*. (2009) state that possible effects can be contra productive when cyanobacterial cells are disrupted and cyanotoxins end up in the end product.

The efficiency of cyanotoxin removal depends on a range of variables like the initial water quality (DOC-concentration, pH), biomass and composition of biofilms, retention time of the water within the treatment and the temperature of the water.
In general the water treatment processes using coagulation, sedimentation and filtration are more efficient in removing cyanobacterial cells than in cyanotoxins (Doomen *et al*, 2008). For removal of cyanotoxins, e.g. biological activated carbon, back-flush and regeneration of the treatment systems seem important factors. Using chemicals (H2O2) or radiation (UV) a serious risk for cell lysis is present, resulting in free toxins in the end product.

4.5 Experiences and cases in the Netherlands

In 2012 the results of a study focussing on the efficiency of cyanotoxin and cyanobacterial removal were published. Below the summary of the study is presented.

In the Netherlands 40% of the drinking water originate from surface waters. Due to the widespread distribution of cyanobacteria, one can suppose that source water for the production of the drinking water will contain cyanobacteria and cyanotoxins. The question arises: do we have a problem, can all cyanobacteria and cyanotoxins be removed by the treatment barriers or can cyanotoxins end up in the drinking water that will be distributed to the customers. The study focussed on two questions:

1. Can cyanobacterial *cells,* originating from surface water, end up in drinking waters?
2. Can cyano*toxins,* originating from surface water, end up in drinking water

For answering these questions an inventory was made of the cell densities in the source waters, what the actual toxin concentration theoretically could be and what the removal efficiency of the organisms and / or the toxins was.

The study was performed for six drinking water companies all using surface water as a source for the drinking water production. The maximum concentrations of cyanotoxins in the drinking water were calculated. In addition, such concentrations were compared to the provisional guideline levels of the WHO (for microcystin LR <1 µg/l).
In this study a worst case scenario was used, meaning the maximum density of cyanobacteria found in the source waters was used as a starting point. Based upon the density, a toxin concentration for the initially used raw water was calculated. Moreover the lowest numbers of removal efficiency (table 4-3) were used for the indication of the removal. Based upon this approach a maximum residual concentration in the drinking water was calculated.
Two hypothetical situations were considered (see Figure 7):

<u>Situation 1</u> All cyanotoxins are cell bound. By removing the cells, the cyanotoxins will be removed as well (suggesting the cells don't break up during the treatment process).
<u>Situation 2</u> The theoretical calculated start concentrations of the toxins consist totally of free cyanotoxins.

In literature the main focus in removing cyanotoxins is on the cyanotoxin microcystin. This toxin is most common in surface waters worldwide and also in Dutch waters. So, other toxins were not considered in this research.

Data of cyanobacterial densities were collected from the drinking water companies. The cyanobacteria concentration and the composition of the cyanobacterial community in the surface waters varied enormously, ranging from hundreds to millions of cells per millilitre. As a result the (theoretical) cyanotoxin concentration could range up to 250 µg/l. Such concentration should not be a problem for the drinking water process, as long as the removal efficiency is sufficient.
The actual removal efficiency depends on factors like the water quality (DOC concentration and pH), operational circumstances within the treatment system and the temperature of the water. In this research for each step the worst case situation was considered, meaning the lowest numbers of a range of efficiency removal (e.g. Table 3) were used in the calculations.
For other micro–organisms than cyanobacteria, the use of legislative quantitative microbiological risk assessments (QMRA) in the Netherlands is required. The approach used in this study for the removal of cyanobacteria and the cyanotoxins was based on the QMRA method.

Treatment	Cell removal %	Toxin removal %
Reservoir softening NaOH	0%	0%
pH-correction CO_2	0%	0%
Micro sieve 35µm	50%	0%
Fe-flocculation	75%	0%
Rapid filtration	60%	0%
UV/H_2O_2	90%	50%
1st Activated carbon filter	60%	70%
2nd Activated carbon filter	60%	70%
Micro sieve 30µm	0%	0%
pH-correction	0%	0%
Disinfection 0,1 mg/l ClO_2	0%	0%

Figure 7. Example of treatment for drinking water production and the removal efficiencies of cells and toxins. For each treatment step the lowest known removal efficiency was applied in the calculations.

Assuming situation 1 when all cyanotoxins are cell bound (see above): the combination of Fe-flocculation, rapid (sand) filtration and UV/H_2O_2 is effective in removing cyanobacterial cells. Moreover, GAC is an effective barrier as well.

Assuming situation 2 where all the toxins are available as free cyanotoxins: the toxins are removed successfully by UV/H_2O_2 and (biological) activated carbon.
In other situations then the one shown in Figure 7 techniques like membrane filtration (nanofiltration and reverse osmosis) and ozone techniques seem successful in removing / degrading the toxic molecules. The use of UV / H_2O_2 is potentially a good method for removing cyanotoxins, however this method is (so far) poorly described in scientific literature (note Table 3).
So far research is conducted mainly on lab scale or pilot scale, examples of research in a full scale operating system hardly exist and should be the focus of future research. In future the use of techniques like membrane filtration and advanced oxidation will increase and so will the effectiveness of the removal of organic micro pollutants, including the cyanotoxins.
Based upon this research it can be stated that for the Dutch drinking water production cyanobacteria and cyanotoxin are no risk for the drinking water quality. However, considering a worst case scenario (all cells that enter the purification plant break up and the toxins end up as free toxins in the process), it cannot be excluded that elevated concentrations of cyanotoxins do exceed the provisional WHO guideline level (1 µg / L microcystin MC-LR) sporadically. Next question to be answered is: does this result in any risk for the public health? The WHO guideline is based upon a lifetime exposure. A lifetime exposure can be excluded, especially when seasonal fluctuations of cyanobacterial densities in source water are taken into account. An acute risk arises when concentrations of the toxin microcystin exceed the toxic level of around 1800 µg / L. Such concentrations will never end up in drinking water in the Netherlands.
Footnote for this research is the fact that only microcystins are considered. Other toxins may react differently in the drinking water production process. However, although potential producers of other toxins than microcystin do appear in reservoirs, the density of such organisms (e.g. *Anabaena*) is relatively low when compared to the density of the potential microcystin producer *Microcystis*.

4.6 Recommendations for research

The actual biomass of cyanobacteria may increase as a result of the effects of climate change. Recommendations based upon this study are to standardize the quantification of cell densities and use new (molecular) tools for rapid and accurate measurements. Moreover the side effect is the possible increase of the clogging of the drinking water treatment processes. So measurements to control the actual biomass are necessary. What measurement works best is dependent of the location.
As a result of the climate changes the composition of the cyanobacterial population might change. Other cyanobacterial genera may dominate, resulting in the dominance of other cyanotoxins. The monitoring of the algal population in drinking water basins remains essential.
The removal efficiency for other toxins than microcystins ,should be part of future research as well as the quantification on full scale production locations. Such studies should result in an insight of the removal efficiencies of the different techniques.

In order to gain a better understanding of the interactive roles that human activities and climatic changes play in controlling CyanoHAB dynamics on the ecosystem scale, the following set of research and assessment priorities emerge:

- Dedicated monitoring of cyanobacteria-dominated ecosystems during the next decades, using traditional sampling coupled to online real-time techniques (e.g. smart buoys), and remote sensing.
- Studies of the annual life cycle of cyanobacteria (Verspagen et al., 2004; Hense and Beckmann, 2006), with special emphasis on the impacts of climatic variables on different phases of the life cycle (Wiedner et al., 2007).
- Laboratory and field assays testing temperature and dissolved inorganic carbon responses of cyanobacterial species and strains under varying nutrient loading scenarios.
- Studies of the development of cyanobacterial surface blooms, and their dependence on mixing processes in aquatic ecosystems (Walsby et al., 1997; Huisman et al., 2004; Jöhnk et al., 2008).
- Studies of the selection of toxic versus non-toxic strains within cyanobacterial species, and their seasonal succession (Via-Ordorika et al., 2004; Kardinaal et al., 2007a;Welker et al., 2007). For example, it has recently been shown that competition among toxic versus nontoxic strains is a major determinant of the overall toxicity of Microcystis blooms, and is strongly affected by the length of the spring–summer period and light availability (Kardinaal et al., 2007b).
- Studies of viral infections, which may lead to mass lysis of cyanobacteria within a few days (Van Hannen et al., 1999; Hewson et al., 2001; Tucker and Pollard, 2005). This begs the question of whether the viral– cyanobacterial interactions depend on temperature, and perhaps could respond strongly to climate change. Clearly, more research is needed in this area.
- Studies of the potential predators of cyanobacteria (e.g. large zooplankton species, zebra mussels), and their responses to climate change.
- Assessments of the impacts of human and natural hydrological modifications (reservoir construction, dams, diversions, droughts, storms and floods) on cyanobacterial bloom potentials and persistence.

4.7 References

Annadotter, H., Cronberg, G., Lawton, L., Hansson, H.B., Göthe, U. & Skulberg, O. (2001) *An extensive outbreak of gastroenteritis associated with the toxic cyanobacterium Planktothrix agardhii (Oscillatoriales, Cyanophyceae) in Scania, south Sweden.* In Cyanotoxins– occurrence, causes, consequences. Edited by: Chorus I. Berlin: Springer-Verlag, 200-208.

Carmichael, W. W., Azevedo, S.M., An, J.S., Molica, R.J., Jochimsen, E.M., Lau, S., Rinehart, K.L., Shaw, G.R. & Eaglesham, G.K. (2001) *Human* *fatalities from cyanobacteria: chemical and biological evidence for cyanotoxins.* Environmental Health Perspective. 109(7): 663–668.

Chorus, I. & Bartram, J. (1999) *Toxic cyanobacteria in water: A guide to their public health consequences, monitoring and management.* E & F.N. Spon, Londen. ISBN 0-419-23930-8.

Codd, G.A., Morrison, L.F. & Metcalf, J.S. (2005) *Cyanobacterial toxins: risk management for health protection.* Toxicology & Applied. Pharmacology. 203 (3): 264 - 272.

Cox, P.A., Banack, S.A., Murch, S.J., Rasmussen, U., Tien, G., Bidigare, R. R., Metcalf, J. S., Morrison, L. F., Codd, G.A. & Bergman, B. (2005) *Diverse taxa of cyanobacteria produce β-N-methylamino-L-alanine, a neurotoxic amino acid.* The National Academy of Sciences of the USA. Proceedings of the National Academy of Sciences 102, (14): 5074 – 5078.

Doomen, A., de Hoogh, C. & Abrahamse, A. (2008) *Cyanobacteria, climate change and drinking water. Does current knowledge sufficiently support impact assessments? A literature review.* Kiwa Water Research BTO 2008.002.

Faassen, E.J., Gillissen, F. & Lürling, M. (2012) *A comparative study on three analytical methods for the determination of the neurotoxin BMAA in cyanobacteria.* PLoS ONE. 7 (5).

Hitzfeld, B.C., Hoeger, S.J. & Dietrich, D.R. (2000) *Cyanobacterial toxins: Removal during drinking water treatment, and human risk assessment.* Environmental Health Perspectives 108 (1): 113-122.

Hoeger, S.J., Hitzfeld, B.C. & Dietrich, D.R. (2005) *Occurrence and elimination of cyanobacterial toxins in drinking water treatment plants.* Toxicology and Applied Pharmacology 203: 231-242.

Huisman, J., Sharples, J., Stroom, J. M., Visser, P. M., Kardinaal, W. E. A., Verspagen, J. M. H. & Sommeijer, B. (2004) *Changes in turbulent mixing shift competition for light between phytoplankton species.* Ecology 85: 2960-2970.

Ibelings, B.W., Stroom J.M., Lürling, M.F.L.L.W. & Kardinaal, W.E.A., (2012) NETHERLANDS: Risks of toxic cyanobacterial blooms in recreational waters and guidelines. In: I. Chorus (ed.); Current approaches to Cyanotoxin risk assessment, risk management and regulations in different countries.

Izaguirre, G. & Taylor, W.D. (2004) *A guide to geosmin- and MIB-producing cyanobacteria in the United States.* Water Science & Technology. 49(9): 19–24.

IWA, 2010. *International statistics for water services.* Specialist group statistics and economics.

Jöhnk, K. D., J. Huisman, J. Sharples, B. Sommeijer, P. M. Visser & J. M. Stroom, (2008) *Summer heatwaves promote blooms of harmful cyanobacteria.* Global Change Biology. 14 (3): 495–512.

Kromkamp, J.C. & L.R. Mur, L.R. (1984) *Buoyant density changes in the cyanobacteria Microcystis aeruginosa due to changes in the cellular carbohydrate content*, FEMS Microbiol. Lett. 25:105-109.

Kosten, S., Huszar, V.L.M. , Bécares, E., Costa, L.S., van Donk, E., Hansson, L.-A., Jeppesen, E., Kruk, C., Lacerot, G., Mazzeo, N., De Meester, L., Moss, B., Lürling, M., Nõges, T., Romo, S., Scheffer, M. (2012) *Warmer climates* *boost cyanobacterial dominance in shallow lakes,* Global Change Biology 18 (1):118-126

Loftin, K., Graham, J. & Rosen, B. (2010), *Analysis and Interpretation for Cyanobacterial Toxin Studies* Workshop Guidelines for Design, Sampling National Water Quality Monitoring Conference Denver, CO, USA April 26, 2010.

Lürling, M. , Oosterhout, F.V. (2013) *Controlling eutrophication by combined bloom precipitation and sediment phosphorus inactivation* Water Research, 47(17):6527-6537

Lürling, M., Eshetu, F., Faassen, E.J., Kosten, S & Huszar, V.L.M. (2013) *Comparison of cyanobacterial and green algal growth rates at different temperatures.* Freshwater Biology. 58(3): 552-559.

Matthijs, H. C. P., Visser, P. M., Reeze, B., Meeuse, J., Slot, P. C., Wijn, G., Talens, R., Huisman, J. (2012) *Selective suppression of harmful cyanobacteria in an entire lake with hydrogen peroxide.* Water research, 46: 1460-1472.

Moss, B., Kosten, S., Meerhoff, M., Battarbee, R.W., Jeppesen, E., Mazzeo, N., Havens, K., Lacerot, G., Liu, Z., De Meester, L., Paerl, H. & Scheffer, M. (2011) *Allied attack: climate change and eutrophication.* Inland Waters. 1:101-105.

Paerl, H.W. & Otten, T.G. (2013) *Harmful Cyanobacterial Blooms: Causes, Consequences, and Controls.* Microbial Ecology. 65(4):995-1010.

Paerl, H.W. & Paul, V.J. (2012) *Climate change: Links to global expansion of harmful cyanobacteria.* Water Research 46(5): 1349-1363.

Paerl, H.W. & Huisman, J. (2009) *Climate change: a catalyst for global expansion of harmful cyanobacterial blooms.* Environmental Microbiology Reports 1 (1): 27 – 37.

Petruŝevski, B. (1996) *Algae and particle removal in direct filtration of Biesbosch water.* Thesis Technische Universiteit, Delft.

Scheffer, M. (1998) *Ecology of shallow lakes.* Kluwer Academic Publishers, Dordrecht. ISBN 1.4020.2306.5.

Soares, R.M., Yuan, M., Servaites, J.C., Delgado, A., Maglhaes, V.F., Hilborn, E.D., Carmichael, W.W. &Azevedol, S. (2006) *Sublethal exposure from microcystins to renal insufficiency patients in Rio de Janeiro, Brazil.* Environ. Toxicol. 21:95-103.

Tonk, L., Bosch, K., Visser, P.M. & Huisman, J. (2007) *Salt tolerance of the harmful cyanobacterium Microcystis aeruginosa.* Aquatic Microbial Ecology, dare.uva.nl, ISBN: 978-90-76894-79-9.

Uwins, H.K., Teasdale, P. & Stratton, H. (2007) *A case study investigating the occurrence of geosmin and 2-methylisoborneol (MIB) in the surface waters of the Hinze Dam, Gold Coast, Australia (Conference Paper).* Water Science and Technology. 55(5):231-238.

Visser, P. M., Ibelings, B.W., Mur,L.R. & Walsby, A.E. (2005) *The ecophysiology of the harmful cyanobacterium Microcystis: features explaining its success and measures for its control.* In J. Huisman, H.C.P. Matthijs, and P.M. Visser (eds), Harmful Cyanobacteria, p. 109-142. Springer, Dordrecht, the Netherlands.

Visser, P. M., Ibelings, B. W., Vanderveer, B., Koedood, J. & Mur, L. R. (1996) *Artificial mixing prevents nuisance blooms of the cyanobacterium Microcystis in Lake Nieuwe Meer, the Netherlands.* Freshwater Biology 36**:** 435-450.

Visser, P.M., Ibelings, B.W., Mur, L.R. (1995) *Autumnal sedimentation of Microcystis spp. as result of an increase in carbohydrate ballast at reduced temperature.* Journal of Plankton Research. 17 (5):919-933.

Welker, M., & von Döhren, H. (2006) *Cyanobacterial peptides: nature's own combinatorial biosynthesis.* FEMS Microbiol. Rev. 30:530-563.

References of Table 3

No.	Reference
1	Cheng, X., Shi,. H., Adams, C., Timmons, T., Ma, Y., (2009) Effects of oxidative and physical treatments on inactivation of *Cylindrospermopsis raciborskii* and removal of cylindrospermopsin; Wat. Sci. Technol., 60 (3), 689-697
2	Ding, J., Shi, H., Timmons, T., Adams, C., (2010) Release and removal of microcystins from *Microcystis* during oxidative-, physical-, and UV-based disinfection; J. Environ. Eng., 136 (1), 2-11
3	Drikas, M., Chow, C.W.K., House, J. & Burch, M.D. (2001) Using coagulation, flocculation and settling to remove toxic cyanobacteria. Journal AWWA 93 (2): 100-111
4	Gijsbertsen-Abrahamse, A.J., Schmidt, W., Chorus, I. & Heijman, S.G.J. (2006) Removal of cyanotoxins by ultrafiltration and nanofiltration. J. Membrane Science 276, 252-259.
5	Grützmacher, G., Böttcher, G., Chorus, I. &Bartel, H. (2002) Removal of microcystins by slow sand filtration. Environmental Toxicology 17: 386-394
6	Grützmacher, G., Wessel, G., Chorus, I. &Bartel, H. (2006) Removal of cyanobacterial toxins (microcystins) during slow sand and bank filtration. IWA publishing. Recent progress in slow sand and alternative biofiltration processes edited by R. Gimbel, N.J.D. Graham, M.R. Collins.
7	Hitzfeld, B.C., Hoeger S.J., & Dietrich, D.R. (2000) Cyanobacterial toxins: Removal during drinking water treatment, and human risk assessment. Environmental Health Perspectives 108 (1): 113-122
8	Ho, L., Meyn, T., Keegan, A., Hoefel, D., Brookes, J., Saint, C.P. & Newcombe, G. (2006) Bacterial degradation of microcystin toxins within a biologically active sand filter. Water Research 40: 768-774
9	Hoeger, S.J., Hitzfeld, B.C. & Dietrich, D.R. (2005) Occurrence and elimination of cyanobacterial toxins in drinking water treatment plants,. Toxicology and Applied Pharmacology 203: 231-242
10	Hoogenboezem,W., Schijven,J., Nobel, P. & Bergsma,J. (1999) De verwijdering van bacteriofagen tijdens duinpassage. H_2O 22 .
11	Knappe, D.R.U., Belk, R.C., Briley, D.S., Gandy, S.R., Rastogi, N., Rike, A.H., Glasgow, H., Hannon,E. Frazier, W.D., Kohl, P. & Pugsley, S. (2004) Algae detection and removal strategies for drinking water treatment plants. Awwa Research Foundation 90971.
12	Knol, T. (2009) Cyanotoxins in drinking water treatment, MSci report Delft University of Technology,
13	Maatouk, I., Bouaïcha, N., Fontan, D. & Levi, Y. (2002) Seasonal variation of microcystin concentrations in the Saint-Caprais reservoir (France) and their removal in a small full-scale treatment plant. Water Research 36: 2891-2897
14	Miao, H. & Tao, W. (2009) The mechanism of ozonation on cyanobacteria and its toxins removal. Separation and Purification Technology 66: 187-193
15	Pietsch, J., Bornmann, K. & Schmidt, W. (2002) Relevance of intra- and extracellular cyanotoxins for drinking water treatment. Acta Hydrochimica et Hydrobiologica 30 (1): 7-15.
16	Qiao, R.P., Li, N., Qi, X.H., Wang, Q.S. & Zhuang, Y.Y. (2005) Degradation of microcystin-RR by UV radiation in the presence of hydrogen peroxide. Toxicon 45: 745-752
17	Rapala, J., Niemelä, M., Berg, K.A., Lepistö, L. & Lahti, K. (2006) Removal of cyanobacteria, cyanotoxins, heterotrophic bacteria and endotoxins at an operating surface water treatment plant. Water Science & Technology 54 (3): 23-28

No.	Reference
18	Rodriguez, E., Onstad, G.D., Kull, T.P.J., Metcalf, J.S., Acero, J.L. & von Gunten, U. (2007) Oxidative elimination of cyanotoxins: Comparison of ozone, chlorine, chlorine dioxide and permanganate. Water Research 21: 3381-3393
19	Rositano, J., Newcombe, G., Nicholson, B. & Sztajnbok, P. (2001) Ozonation of NOM and algal toxins in four treated waters. Water Research 35 (1): 23-32
20	Svrcek, C. & Smith, D.W. (2004) Cyanobacteria toxins and the current state of knowledge on water treatment options: a review. Journal of Environmental Engineering and Science 3: 155-185
21	Teixeira, M.R. & Rosa, M.J. (2007) Comparing dissolved air flotation and conventional sedimentation to remove cyanobacterial cells of *Microcystis aeruginosa* Part II: The effect of water background organics. Separation and Purification Technology 53: 126-134.
22	Warren, A., Drogui, P. & L'Aurion, J. (2010) Revue sur l'état actuel des connaissances des procédés utilisés pour l'élimination des cyanobactéries et cyanotoxines lors de la potabilisation des eaux. Revue des Sciences de l'eau, 23(4), 391-412
23	Westrick, J.A. & Szlag, D.C. (2010) A review of cyanobacteria and cyanotoxins removal/inactivation in drinking water treatment. Anal. Bioanal. Chem, 397, 1705-1714

5 NOM increase

Bjørnar Eikebrokk (SINTEF)

5.1 Challenges to drinking water quality

Significant increases in the concentration of natural organic matter (NOM) have been observed in surface waters in large regions of the world (UK/Scotland, Sweden, Norway, Finland, Russia, The Baltic states, North America, Canada, Australia, etc.). Possible reasons for the increase in NOM are discussed by Eikebrokk *et al*. 2004. Haaland *et al*. (2010) were able to explain more than 80 % of the observed colour increase in Oslo's water sources since the early 1970s by two variables (ref. Figure 1): Increasing precipitation and less acid rain.

NOM has significant impacts on water treatment as well as distribution processes (Eikebrokk et al, 2007), impacts that may significantly increase the overall risk levels in drinking water supply:

- Affect colour, taste and odour levels in water
- Control most treatment processes, and affects overall treatment performance, incl. barrier efficiency
    - Challenge process control systems (increased seasonal variability in raw water quality, incl. NOM content and NOM nature)
- Increase coagulant demand and sludge production rates
- Affect filter run lengths, filter backwash and energy use
- Affect disinfectant demand and/or disinfection efficiency
- Form DBPs during chlorination, ozonation, etc
- Affect stability and removal of inorganic particles and pathogens and increase mobility of micro pollutants
- Adsorbs to metal precipitates, affect corrosion processes and biological stability in distribution systems
- Increase soft deposits, biofilm formation and regrowth in DS
- Foul membranes, block AC pores and/or outcompete T&O, micro pollutants, etc for AC adsorption sites
- Increase the organic loads and affects design and operation of ozonation-biofiltration systems

5.2 Current operational practices and challenges

It is evident from the above list of impacts of NOM on water treatment and distribution that the existing treatment technologies and current operation practices may not be capable of coping with the strong increase in NOM that are observed in many countries and regions.

The situation at Görvälnverket in Sweden (a treatment facility supplying the northern part of Stockholm) may be used as an illustrating example (Ericsson 2012): Granular activated carbon (GAC) filters are applied at this plant as a barrier against micro pollutants and petroleum spills from the heavy boat traffic on the water source, Lake Mälaren. Due to a significant trend towards increasing NOM concentration levels in the source water, there is a corresponding increase of DOC levels in coagulated and filtered water entering the GAC filters. Even though the adsorption capacity for algae metabolites (e.g. *Geosmin*) seems to be well maintained, the adsorption capacity and barrier efficiency of the downstream GAC filters against chemicals/petroleum products is diminished very quickly. The costs and environmental aspects of high regeneration frequencies are quite significant. The observed effects of increasing NOM are likely due to the pore blockage ability and high affinity for the GAC adsorption sites from the large and reactive NOM molecules. Therefore, an additional nanofiltration (NF) treatment step prior to the GAC filters is likely to be installed in the near future in order to decrease the NOM (DOC) concentration levels and maintain GAC as an effective barrier against chemicals.

Another example of the challenging NOM increase is taken from treatment facilities in Norway using ozonation and biofiltration. The applied ozone doses are mainly controlled by the residual colour and/or UV-transmittance in treated water. Thus a high/increasing NOM concentration requires a high/increasing ozone dosage. As a result of increasing ozone dosages, more hydrophilic and more biodegradable organic matter are transformed from the hydrophobic and less biodegradable and dominating humic NOM fractions. The subsequent biofilters are however in many cases not designed to cope with high loads of biodegradable NOM. Thus finished water with high residual BOM/BDOC concentrations is distributed. This may result in significant regrowth and biofilm formation in the chlorine-free distribution systems. In addition, residual organic matter and microorganisms may adsorb to metal precipitates on the pipe walls thus contributing to more "soft" deposits that may sustain growth and survival of pathogens and create discolouration and customer complaints due to periodic resuspension events.

From the above, it is rather evident that the NOM concentration increase may increase the risk levels and number of failures in treatment and disinfection barrier efficiencies. Increases in NOM concentration and NOM concentration variability can be monitored through parameters like DOC, UV-absorption and colour. However, treatment and distribution-relevant features of NOM nature (surface charge/charge density, hydrophobicity, biodegradability, etc.) cannot be measured on-line in the same simple way.

NOM-related impacts on water treatment and distribution that may lead to increased failure risk levels are specific to the applied treatment technology. For major treatment technologies, some of these impacts are described below.

5.2.1 Enhanced coagulation

Normally, some 50-80 % of NOM (TOC) can be removed by coagulation. The hydrophilic, neutral NOM-fractions (NEU) however, are not amendable to removal by coagulation. Thus if the composition of NOM is not significantly altered by climate change, increasing NOM concentrations in the source water will lead to an increase in all residual NOM fractions, including the

NEU fraction. Thus the residual DOC concentration will increase in waters treated by coagulation processes when the NOM concentration increases.

Based on a large number of pilot-scale experiments, Figure 8 shows the qualitative as well as the quantitative effects of increasing raw water NOM (colour) concentration on major aspects of enhanced coagulation-contact filtration process operations (Eikebrokk et al., 2004). From the results in Figure 8, an increase in raw water colour from 20 to 40 mg Pt/L resulted in:

- An increase in the number of daily filter runs/backwashes, and in the backwash water consumption and ripening water production by a factor of 2.2
- An increase in the coagulant dose demand and in the production of sludge by a factor of 1.9
- An increase the residual TOC concentration by a factor of 1.35
- A decrease in treatment capacity to 87 % of the level at 20 mg Pt/L
- A decrease in filter run times to 44 % of the levels at 20 mg Pt/L

At some specific NOM concentration level, enhanced coagulation processes can no longer cope with the deteriorating raw water quality as a stand-alone NOM removal process, mainly because of reduced treated water quality, high chemical demand, excessive sludge production, short filter runs, high backwash water consumption, short filter runs due to rapid head loss development and/or early filter break-through, reduced treatment production capacity, etc.

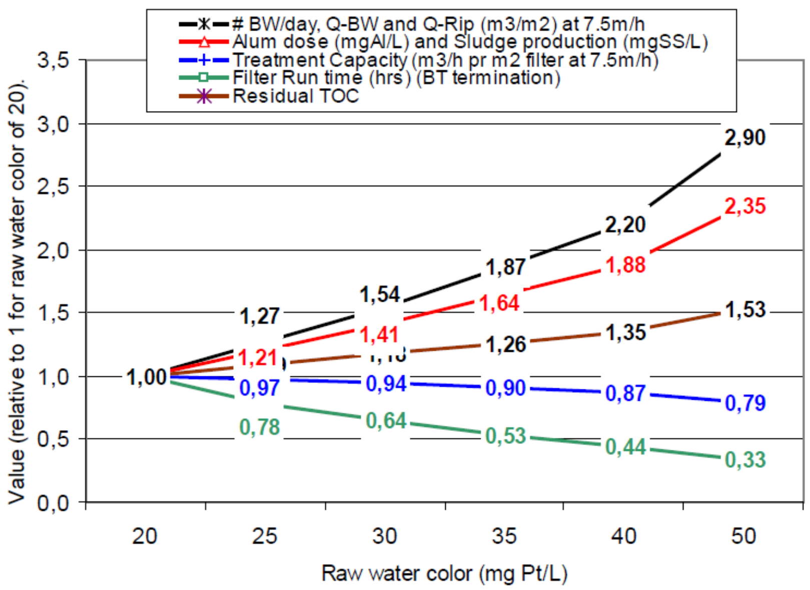


Figure 8. Impacts of increasing raw water NOM concentrations on enhanced coagulation operation performance (Eikebrokk 2002)

A situation with increasing NOM concentrations along with increasing occurrence of algae/algae will represent specific treatment challenges that require combinations of treatment technologies and overall optimization efforts.

5.2.2 *Activated carbon adsorption*

Activated carbon adsorption is normally used for the removal of taste and odour-forming substances, algal toxins, pesticides and other types of microcontaminants in water.

NOM can be a severe competitor to the above-mentioned substances and can significantly impact not only the efficiency of the adsorption process, but also the dose requirements for powder activated carbon (PAC), and the lifetime for activated carbon (GAC) filters, mainly for two reasons:

1. NOM is normally present in concentrations of a few milligrams per litre, while the microcontaminants are present in far lower concentrations (nanograms or micrograms per litre). Thus NOM can out-compete the above-mentioned substances for adsorption sites, thereby reducing the adsorption efficiency and the adsorption capacity for these substances
2. The large NOM molecules can block activated carbon pores, thereby decreasing adsorption capacity

As an illustrating example, Figure 9 shows the PAC dose required to reduce the taste and odour compound, methyl iso-borneol (MIB), by 50 % in waters with different NOM (DOC) concentrations. Similarly, increasing NOM (DOC) will significantly decrease adsorption efficiency for algae toxins like microcystin (Cook and Newcombe, 2003). Besides, NOM will win the competition with microcontaminants in most cases (Newcombe et al. 2002).

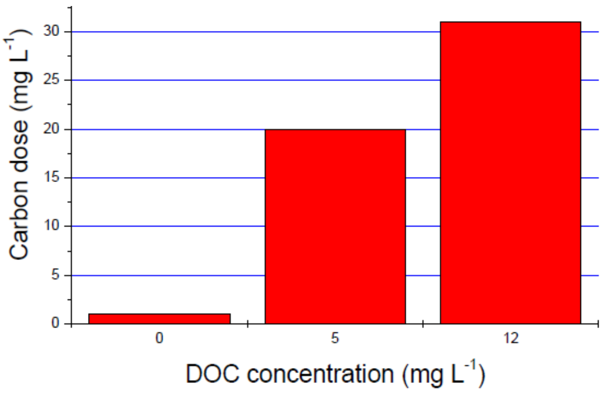


Figure 9. PAC dose requirements to reduce MIB-concentration by 50 % in waters of different DOC concentration, 60 min contact time (CRC 2005)

5.2.3 *Ozonation-biofiltration*

Ozonation-biofiltration processes may be a good NOM removal technology for raw waters with low NOM concentration levels, e.g. colour of about 30 or less. This process is normally effective in removing colour and UV-absorbing NOM fractions (80-90 %) while the bulk DOC removal efficiency is normally rather poor (20-30 %).

Figure 10 illustrates the ozonation-biofiltration process applied in Skien, Norway. Here metals like iron, manganese and aluminium from the raw water precipitate in the alkaline prefilter thus providing excellent adsorbents for NOM. Especially the iron hydroxide is known as a very effective NOM adsorbent. Thus, similar amounts of NOM are removed in the alkaline prefilter and in the biofilter at this facility. The chlorine dosing system is mainly a back-up to the installed UV-disinfection system. It is also used to help controlling the heterotrophic plate counts (22 °C) at levels below 100/mL. The applied chlorine doses are however very low, typically 0.1-0.2 mg/L, thus leaving no free chlorine residuals in the distribution network.

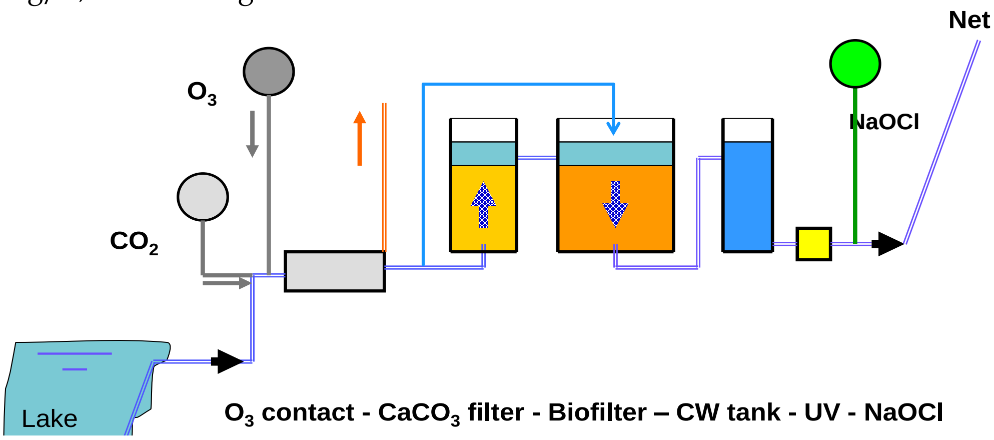


Figure 10. Ozonation-biofiltration, alkaline prefiltration and UV-disinfection treatment used in Skien, Norway.

This ozonation-biofiltration treatment process is sensitive to high levels of NOM in the raw water, mainly due to the following factors:

- There is stoichiometry between the NOM concentration in the raw water and the ozone demand for controlling residual colour and UV-absorption. Thus higher NOM requires higher ozone doses
- Increasing NOM and increasing ozone doses lead to increased transformation of hydrophobic NOM to hydrophilic and more biodegradable NOM thus increasing the load of biodegradable NOM to the biofilter
- Application of ozonation-biofiltration processes for treatment of raw waters with (too) high NOM content, may lead to biofilter overload and poor removal of NOM and BOM, thus yielding problems with biological regrowth and biofilm formation in the distribution system. Similar results can arise from inadequate ozone dose control and/or poor biofilter design

High BOM in finished water may lead to significant regrowth and biofilm formation, unless these effects are controlled by high chlorine residuals. In the Nordic/cold climate region however, chlorine-free distribution systems are very common. Thus increasing NOM and BOM concentrations is a significant challenge also with respect to biological stability during in water distribution systems.

5.2.4 *Membrane filtration*

NOM can foul membranes and thereby decrease treatment capacity and increase the need for membrane cleaning. Major fouling mechanisms include: i) adsorption of NOM to the membrane surface and plugging of the pores, and ii) formation of NOM-cation complexes and layers/cakes on the membrane surface (Goosen et al., 2004).

Some studies have indicated that allochthonous and hydrophobic NOM from the watershed are the main contributors to membrane fouling (Violleau et al., 2005) while others suggest that the hydrophilic NOM fractions are responsible for membrane fouling (Park et al. 2006) and that high molecular weight, hydrophilic neutral NOM-fractions appear to have a large influence over the fouling rate in membrane filtration of surface waters. Thus hydrophilic membranes have lower fouling rates than more hydrophobic membranes. Coagulation will almost always reduce the rate of membrane fouling. Adsorbents (PAC, MIEX, etc.) generally require extended contact times or high doses to be effective in reducing membrane fouling. Addition of particles can improve membrane performance by increasing the porosity of the filter cake. There is a need for tailored membrane materials, coagulants and adsorbents that minimize membrane fouling (CRC 2005).

NOM adsorbed onto membranes can be difficult to remove by physical or chemical cleaning (Cherkasov et al., 1995).

When using membrane filtration (UF) with precoagulation, increasing NOM will increase the coagulant dose requirements, the amount of sludge produced and will thus also increase the need/frequency for membrane cleaning.

In the Nordic/cold climate region membrane processes are dominated by nanofiltration (NF) systems. Although no coagulation is applied in these systems, the need for good fouling control for maintaining treatment capacity will in general be more challenging as the NOM concentration increases and thereby also the potentials for membrane fouling and complex formation.

5.3 Corrective responses and curative measures

In order to cope with increasing NOM, treatment performance optimization is required. In addition, single-step NOM removal processes may no longer be capable of reducing NOM to acceptable levels. Thus additional treatment steps and combinations of treatment technologies may be required.

Thus, when facing a situation with increasing NOM concentration levels in source waters, treatment process optimization and implementation of best operation practice with respect to coagulant doses, polymer use, filter design, filter material, etc. is a natural first step.

Treatment performance optimization will normally help a lot, but only up to a certain level of raw water NOM concentration increase. When this level is exceeded, supplementary treatment steps are required, e.g. magnetic ion exchange (MIEX), membranes (NF, UF with pre-coagulation), ozonation-biofiltration, activated carbon filtration, advanced oxidation processes (AOP), etc.

Increasing NOM will result in increasing dose requirements for PAC or increasing contact times for GAC filters. For maximum efficiency, NOM should be removed as far as possible prior to activated carbon adsorption processes. Possible pre-treatment technologies includes enhanced coagulation, membrane processes (NF or UF with pre-coagulation), ion exchange (MIEX), AOPs, etc.

It is not only the increasing NOM concentration levels that impacts on treatment, NOM nature, NOM treatability and NOM biodegradability may also change. Therefore, NOM characterization and treatability assessments are required in order to identify best available technologies (BATs) and best available operations (BOPs).

Also the selection of supplementary treatment steps should be based on comprehensive assessments and characteristics of NOM nature and concentrations, future trends and predictions, etc.

Future process control options and adaptations should be based on NOM nature characteristics as well, not only on NOM concentration measurements based on in-line DOC or UV-absorbance monitoring. In fact, UV-absorbance or even colour may be more adequate parameters than DOC for NOM coagulation treatability assessments and coagulation process control systems. This is true primarily for raw waters that contain significant amounts of hydrophilic, neutral NOM that is not amendable to removal by coagulation. This NOM fraction that contributes to DOC but not to colour and UV-absorbance, may increase in the future due to a warmer climate and more algae-laden raw water sources.

A step-wise strategy is proposed to cope with the increasing NOM levels:

5. Identify the seasonal variability in NOM concentration and characteristics in the present situation, and predict future NOM concentrations and characteristics in raw and treated waters in 10, 20 and 40 years time
6. Perform an internal benchmarking of the current water treatment operation performance status versus an optimized situation, e.g. by applying the TECHNEAU optimization procedures that are based on extended analysis of raw water/NOM quality and variability (Eikebrokk et al. 2007)
7. Based on the above steps 1 and 2, and from comprehensive evaluations of NOM content, NOM nature, NOM treatability, and NOM biodegradability in raw and treated waters, evaluate the applicability and possible benefits of additional treatment steps like activated carbon adsorption (AC), ion exchange (IEX), nanofiltration (NF), advanced oxidation processes (AOP), UV-disinfection, etc.

8. Implement selected changes to the treatment processes, i.e. shift to the best available treatment technology (BAT) and the best operation performance (BOP) in terms of water quality/safety and sustainability/use of resources and emissions/environmental impacts

In more detail, the corrective responses may include:

- Optimization of existing treatment processes as described for the specific treatments.
- Adaptations of treatment steps to cope with operational challenges:
    - High/rapid head loss increase in direct or contact filtration: Optimization of coagulant doses/minimization of sludge production; Use of filter material with less head loss (e.g. Filtralite®); Optimization of pre-treatment steps (e.g. settling, flotation)
    - Short filter runs/low capacity at coagulation facilities: Use of filter aid polymers to make flocks stronger and/or increase attachment forces and thereby extend filter runs and/or allow for higher filter load rates
    - Membrane fouling/capacity decline: More efficient washing solutions and/or washing routines; Improved pre-treatment steps, e.g. optimized enhanced coagulation
    - Rapid fouling of GAC filters by NOM with corresponding need for frequent regeneration: NF-treatment prior to GAC to reduce NOM load
    - Inadequate NOM removal in ozonation-biofiltration processes: Reduce the production and load of biodegradable NOM to the biofilter by: i) Alkaline pre-filtration to provide pre-additional NOM removal through adsorption to metal hydroxide precipitates, ii) Optimization of ozonation conditions and applied ozone doses to minimize ozone-induced production of biodegradable NOM fractions
    - Reduced coagulant demand: Use of alkaline filter layers or separate alkaline post-filtration at enhanced coagulation facilities to allow for reductions in applied coagulant doses, reduced sludge production, increased filter run lengths and decreased demand for filter backwash water
- Improved process control
    - Better adaptation of coagulation and ozonation-biofiltration treatment processes/operations to raw water/NOM concentration, NOM nature and seasonal variability
- Additional treatment steps for improved NOM removal
    - MIEX, NF, Ozonation-biofiltration, AOPs, etc.
    - Alkaline filtration prior to biofiltration in ozonation-biofiltration processes in order to reduce the load of biodegradable NOM to the biofilter and thereby provide less biodegradable matter and less regrowth during distribution.

5.4 Preventive measures

In the Nordic region, NOM is mainly allochthonous, i.e. originating from degradation of organic material (e.g. leaves, litter, etc.) in the watershed. There is significant variation, however in the relative contribution to TOC, colour, etc. from different types of forest and tree species.

Thus, a selective take-out of the most NOM-producing tree species may contribute to a reduction in the NOM concentration levels in source waters. However, although applicable in theory, this preventive measure may have some limitations in practice. Changes in land use may be another option.

5.5 Experiences and cases in Norway

Due to the severe increase in NOM concentrations in the southern part of Norway (ref. Fig. 3-2), several utilities have performed full-scale treatment optimization trials during the last few years. Rapid NOM fractionation and BDOC measurements are among the more advanced tools used in process performance diagnosis and optimization efforts.

Coagulation facilities

At enhanced coagulation facilities the main objective has been to find the balance between safe and sustainable operation, i.e. how to minimize the use of resources like chemicals and energy while still maintaining excellent water quality and safety levels.

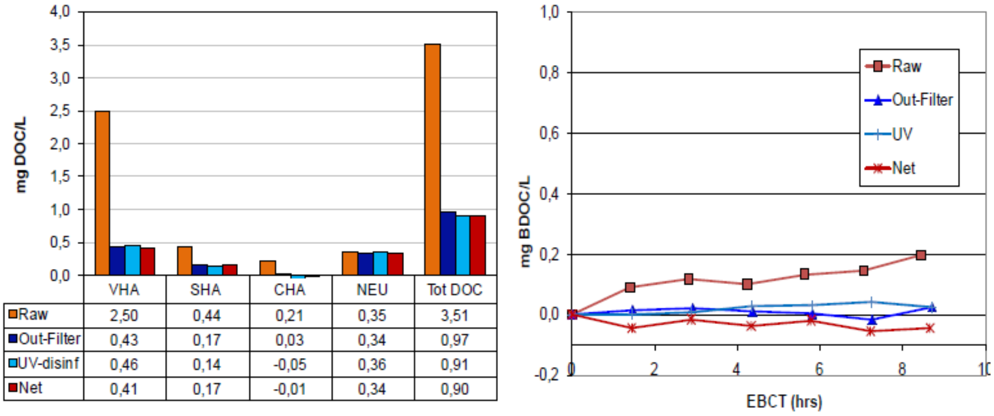
The optimization efforts follow the TECHNEAU enhanced coagulation procedures (Eikebrokk et al., 2007), based on characterization of NOM by:

a. Rapid fractionation, where DOC is split into four fractions: VHA (Very Hydrophobic Acids), SHA (Slightly Hydrophobic Acids), CHA (CHArged hydrophilic substances), and NEU (NEUtral hydrophilic substances) according to the procedures suggested by Chow et al, 2004)
b. Biodegradability (BDOC) analyses based on DOC monitoring in influent and effluent water from a BDOC set-up with six biofilters-in-series filled with glass beads (Eikebrokk et al, 2007)

Based on routine water quality monitoring and the above NOM characterization tools, optimization of treatment performance was performed, i.e. a best possible adaptation of treatment conditions to the variability in raw water quality and NOM concentrations.

Figure 11 shows a typical example of DOC and NOM-fraction concentrations in raw and treated waters in a coagulation facility in Bergen, Norway. The results show that high removal efficiencies are obtained for DOC, and for all of the NOM fractions except the neutral hydrophilic fraction (NEU). The biodegradable organic carbon (BDOC) concentration is low in the raw water, as it is in most Norwegian lake waters. However, the BDOC concentration is reduced even more by coagulation, to levels close to zero. This is likely the major reason why biological regrowth is a minor challenge in in distribution network at most utilities using enhanced coagulation treatment in Norway.

From the optimization efforts described above, utilities are able to better adapt their coagulation conditions to the long-term trends and seasonal variability in raw water quality thereby maintaining good water quality and safety levels. In addition, a more sustainable operation performance has been achieved due to significant reductions in the use of resources like treatment chemicals and energy.



	VHA	SHA	CHA	NEU	Tot DOC
Raw	2,50	0,44	0,21	0,35	3,51
Out-Filter	0,43	0,17	0,03	0,34	0,97
UV-disinf	0,46	0,14	-0,05	0,36	0,91
Net	0,41	0,17	-0,01	0,34	0,90

Figure 11.NOM fraction concentrations and BDOC profiles in samples from raw water, coagulated and filtered water, UV-disinfected water and in distributed water at SVD WTP, Bergen, Norway in Aug 2009 (Eikebrokk et al. 2010)

Ozonation-biofiltration facilities

At ozonation-biofiltration (OBF) facilities, increasing NOM concentrations can severely influence treatment performance and treated water quality including biological stability. Examples from two utilities at the west coast of Norway may serve as illustrating examples of the importance of the NOM concentration levels for this treatment technology.

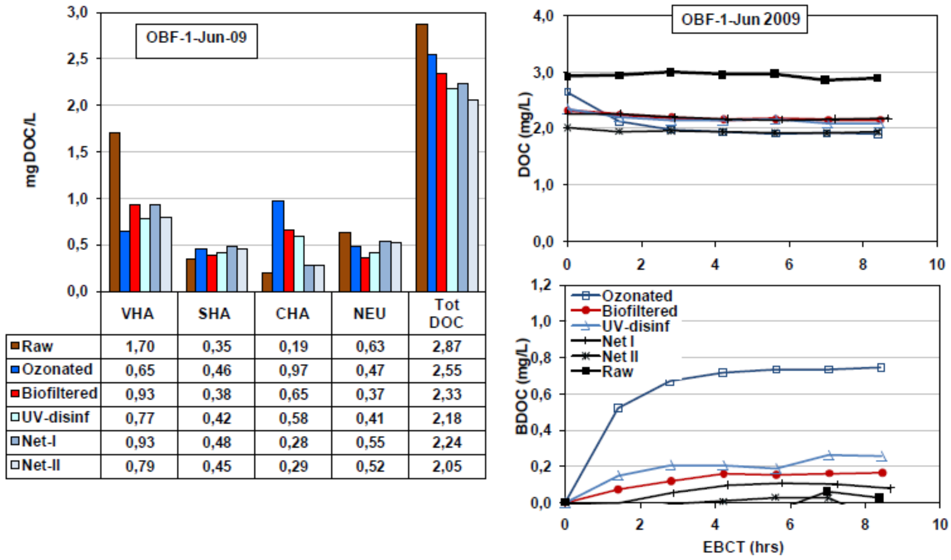
Results from an OBF utility treating raw water with a low to moderate NOM content (TOC and DOC of 2-3 mg/L) is shown in Figure 12. NOM fraction and BDOC concentrations in samples taken from raw water, during different treatment steps and from distributed water are presented.

Similarly, Figure 13 shows the data from an OBF utility treating raw water with a relatively high NOM content (TOC and DOC of 5-6 mg/L).

It is evident from both Figure 12 and 13 that the ozonation process transforms NOM, mainly the biologically stable (humic) VHA fraction, into more biodegradable fractions, mainly CHA. Thus the BDOC levels are significantly increased by ozonation. The organic load to the biofilter is increased by the pre-ozonation process, in specific at the utility treating the more concentrated raw water where the NOM concentration and ozone dose levels are higher.

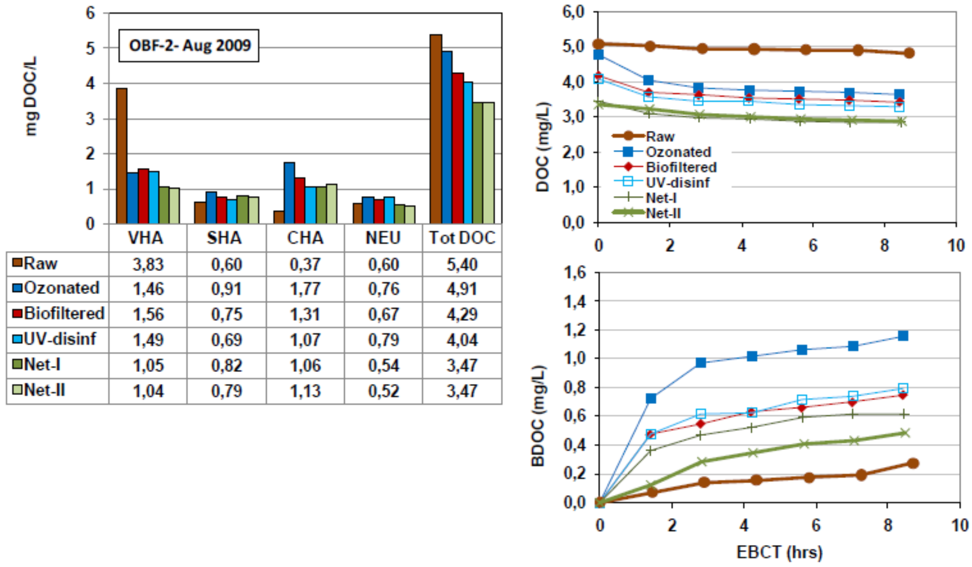
The biofilter (designed for 20-30 min EBCT) is capable of removing some of the ozone-produced BDOC, but not all. The utility treating the low concentrated raw water is capable of maintaining BDOC levels in finished (UV-disinfected) water (0.2 mg/L) that is considered low enough to control regrowth and biofilm formation in the distribution network. This is however

not the case at the utility treating the more concentrated raw water. Here significant amounts of residual BDOC are present in distributed water (0.8 mg/L). As a result, more severe biological regrowth challenges are present at this facility.
High raw water NOM concentrations requires high ozone doses in order to control treated water colour and UV-transmittance at required levels for efficient UV-disinfection performance. At these high ozone dose levels however, significant amounts of hydrophilic and more biodegradable NOM are formed thus reducing the biological stability of treated water and causing regrowth and biofilm formation in the distribution network.



	VHA	SHA	CHA	NEU	Tot DOC
Raw	1,70	0,35	0,19	0,63	2,87
Ozonated	0,65	0,46	0,97	0,47	2,55
Biofiltered	0,93	0,38	0,65	0,37	2,33
UV-disinf	0,77	0,42	0,58	0,41	2,18
Net-I	0,93	0,48	0,28	0,55	2,24
Net-II	0,79	0,45	0,29	0,52	2,05

Figure 12. NOM fraction concentrations and BDOC profiles in samples from raw water, effluent water from different treatment steps (ozonated, biofiltered, UV-disinfected at 40 mJ/cm²), and in distributed water samples (net samples) at a OBF facility treating raw water with low to moderate NOM content (Eikebrokk et al. 2010)



	VHA	SHA	CHA	NEU	Tot DOC
Raw	3,83	0,60	0,37	0,60	5,40
Ozonated	1,46	0,91	1,77	0,76	4,91
Biofiltered	1,56	0,75	1,31	0,67	4,29
UV-disinf	1,49	0,69	1,07	0,79	4,04
Net-I	1,05	0,82	1,06	0,54	3,47
Net-II	1,04	0,79	1,13	0,52	3,47

Figure 13. NOM fraction concentrations and BDOC profiles in samples from raw water, effluent water from different treatment steps (ozonated, biofiltered, UV-disinfected at 40 mJ/cm²), and in distributed water samples (net samples) at a OBF facility treating raw water with high NOM content (Eikebrokk et al. 2010)

NOM fractions and biological stability
The results presented in Figures 11 to 13 above indicate that BDOC is correlated to the hydrophilic NOM fractions. This is verified in Figure 14 that shows data from sampling of raw water and effluent water from different treatment steps at 10 Norwegian water works using enhanced coagulation or ozonation-biofiltration treatment. While no correlation ($R^2 < 0.1$) was found between BDOC and the hydrophobic NOM fractions (VHA, SHA), excellent correlations (R^2 0.86) was found between BDOC and the hydrophilic CHA fraction, and between BDOC and the sum of the hydrophilic NOM fractions (CHA+NEU).

Thus, in order to control biostability and regrowth, i.e. maintain BDOC levels at 0.2-0.3 mg/L or lower, these results indicate that the hydrophilic NOM fraction concentrations should be controlled at levels of 0.5-0.7 mg CHA/L and 1-1.3 mg CHA+NEU/L, respectively in cold water distribution systems with no chlorine residuals.

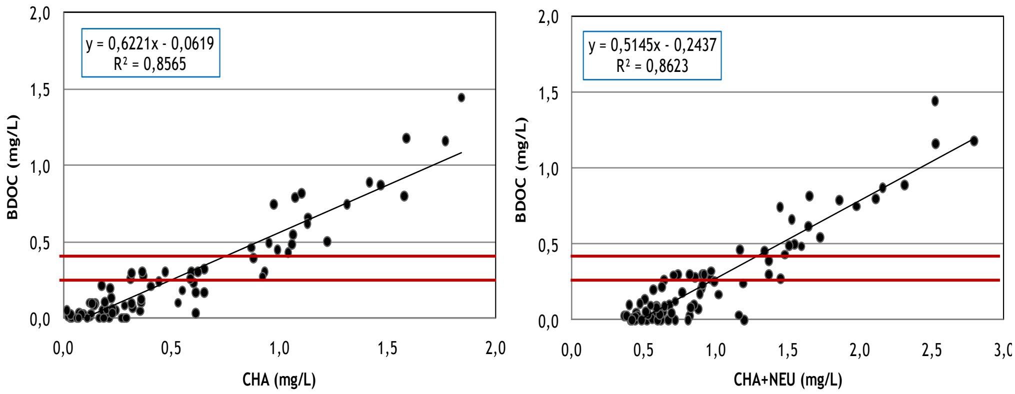


Figure 14. Correlation of BDOC with CHA-fraction concentrations (left) and with CHA+NEU fraction concentrations (right). Data from raw water and effluent water from different treatment steps at 10 Norwegian utilities applying coagulation or ozonation-biofiltration treatment (Eikebrokk 2012)

5.6 Tools, manuals, guidelines, documentation and references

More description of the rapid NOM fractionation, column-in-series BDOC analysis and suggested coagulation optimization procedures can be found in the TECHNEAU web-sites (www.techneau.eu) (Eikebrokk et al. 2007).

5.7 References

Cherkasov, A.N., Tsareva, S.V. and Polotsky, A.E. (1995). Selective properties of ultrafiltration membranes from the standpoint of concentration polarization and adsorption phenomena. *Journal of Membrane Science*, 104:157

Chow, C., Fabris, R., and Drikas, M. (2004): A rapid fractionation technique to characterise natural organic matter for the optimisation of water treatment processes. *J. Water Supply: Research and Technology – AQUA*, 53, No.2, 85-92

Cook, D. and Newcombe, G. (2003). Effect of NOM concentration and character on the adsorption of microcystin analogues onto PAC. Proc. Australian Water Association Fed. Convention, Ozwater, Apr 2003, Perth, Australia

CRC for Water Quality and Treatment (2005). Natural Organic Matter: Understanding and Controlling the Impact on Water Quality and Water Treatment Processes. ISBN 187661644x, Australia

Eikebrokk, B. (2012). Brown waters in Norway: Characterization and treatment. Presented at the IWA World Water Congress & Exhibition, Busan, Korea, Sep 16-21, 2012

Eikebrokk, B. (2010). TECHNEAU Case study Bergen: Optimization of water treatment. TECHNEAU Report D. 7.11.3 B, www.techneau.eu

Eikebrokk, B., Juhna, T. and Melin, E (2007). Water treatment by enhanced coagulation and ozonation-biofiltration: Operation optimization procedures and trials. TECHNEAU Report D5.3.2A, www.techneau.eu

Eikebrokk, B., Vogt, R:D. and Liltved, H. (2004). NOM increase in Northern European source waters: discussion of possible causes and impacts on coagulation/contact filtration processes. *Water Science and Technology: Water Supply*, Vol 4, No 4, pp 47-54, IWA Publishing

Eikebrokk, B. (2002). Increasing color in Norwegian raw waters: Potentials for the optimization of coagulation processes. Proc.: The 3rd Nordic Water Supply Conf., Gothenburg, Sweden, Sep 2002 (In Norwegian).

Ericsson, P. 2012) Pers. comm

Goosen, M.F.A., Sablani, S.S., Al-Hinai, H., Al-Obeidani, S. Al-Belushi, R. and Jackson, D. (2004). Fouling of reverse osmosis and ultrafiltration membranes: A critical review. *Separation Sci. & Technol.*, 29:10:2261

Haaland, S., Hongve, D., Laudon, H., Riise, G.and Vogt, R. D. (2010). Quantifying the drivers of the increasing colored organic matter in Boreal surface waters. Environ. Sci. Techn.

Newcombe, G., Morrison, J., Hepplewhite, C. and Knappe, D.R.U (2002). In the adsorption competition between NOM and MIB, who is the winner, and why? *Water Science and Technology: Water Supply* Vol 2, No. 2, pp 59-67 IWA Publishing

Park, N., Kwon, B., Kim, S.D. and Cho, J. (2006). Characterizations of colloidal and microbial organic matters with respect to membrane foulants. *Journ. Membrane Sci.*, 275:29

Violleau, D., Essis-Tome, H., Habarou, H., Croué, J.P. and Pontie, M. (2005). Fouling studies of a polyamide nanofiltration membrane by selected natural organic matter: An analytical approach. *Desalination*, 173:223

Weir, P. (2012). Pers. Comm.

6 Effect of climate change on microbial regrowth in drinking water distribution systems

José Menaia, Elsa Mesquita (LNEC)

6.1 Challenges to drinking water quality

Drinking water treatment plants (WTP) must produce microbiologically safe water (WHO 2011). However, that does not mean that treated water must be sterile, which would be economically and technologically impracticable. Accordingly, water leaving WTP invariably transports low levels of microorganisms, mainly non-pathogenic heterotrophic and aerobic spore-forming bacteria (Payment & Robertson 2004, WHO 2011).
However, pathogens may accidentally escape treatment upon WTP failure or incapacity to timely adjust to sudden degradation of source water quality, which, among other possible causes (Beuken et al. 2008), may result from intense rainfall events.
In addition to those eluding treatment processes, microorganisms may enter into DWDS downstream of treatment plants (WTP), provided that ingress of contaminated materials occurs during improperly made repairs, backflow or back-siphonage or through breaches in non-pressurized systems (e.g. fissures and cracks in pipes or storage tanks, leaking joints) (Beuken et al. 2008). Although many microbial species that ingress into DWDS are unable to survive or multiply (Reasoner et al. 1989), many bacteria colonize DWDS inner surfaces by forming biofilms (Flemming et al. 2002). Biofilms are multicellular communities of microorganisms embedded in a fibrilar matrix. This extracellular polymeric structure (EPS) is made of compounds produced by microorganisms, mainly polysaccharides and proteins (Flemming et al. 2007). Regardless the presence of a disinfectant residual biofilm sessile communities persist (Menaia and Mesquita 2004) as the predominant form of microbial regrowth and the origin of most DWDS water planktonic microorganisms (Geldreich 1996, Flemming et al. 2002). To same extent, the latter may be inactivated in the bulk water whenever an effective concentration of disinfectant residual is maintained (Srinivasan et al. 2008). However, DWDS biofilm may host pathogens (e.g. *Legionella pneumophila*, *Campylobacter jejuni*, *Mycobacterium avium*) and opportunistic species of several genera (e.g. *Aeromonas*, *Pseudomonas*, *Klebsiella*) (Lehtola et al. 2006, Torvinen et al. 2007, Wingender and Flemming 2011). In addition to having EPS matrix protection against residual disinfectants, biofilm bacterial phenotypes are remarkably more resistant than their planktonic counterparts (LeChevallier et al. 1988; Lewis et al. 2001, Gagnon et al. 2005). Moreover, the biofilm fibrilar matrix has the ability to sequester and hold/release viral (e.g. adenovirus, rotavirus, norovirus virions) and protozoan (e.g. *Cryptosporidium parvum* oocysts) pathogenic forms (Wingender and Flemming 2011, Helmi et al. 2008), including amoebae carrying high numbers of *L. pneumophila* (Kuiper et al. 2004). Hence, DWDS biofilms have an important potential to control

drinking water safety. Therefore, although no direct evidence of negative effects of DWDS biofilm on the general population health has been produced, biofilm hazards should be considered, especially with respect to immune-compromised people (WHO 2011).

On the other hand, biofilm-born coliforms and thermotolerant-coliforms (e.g. *E. coli*) may be released to the water (Fass et al. 1996), then giving raise to the so-called coliform events and thus lead to false positive results of faecal contamination (LeChevallier et al. 1987). Conversely, the non-detection of thermotolerant coliforms (e.g. *E. coli*) may give a false sense of water safety, as those indicator microorganisms are more sensitive to disinfectants' residual than many waterborne pathogens (Larsson et al. 2008, Payment and Robertson 2004, WHO 2011).

Accordingly, the mitigation of DWDS biofilm development must be viewed as a measure to minimize drinking water risks. Such approach is particularly important in the current climate change context, owing to the foreseeable direct and indirect effects of temperature and rainfall pattern changes, including increments in source water natural organic matter (NOM) contents, DWDS biofilm species composition and growth. Owing to their capacity to adsorb and concentrate organic compounds, sediments that accumulate in DWDS may also promote microbial regrowth. Organic substrates for biofilm regrowth may also be leached from plastic pipes (Chambers et al. 2004).

Apart from the loads and types of microorganisms entering DWDS, the overall microbial regrowth is mainly determined by (Norton and LeChevallier 1997, Levy 2004):

- The availability of biodegradable organic compounds (i.e. suitable for utilization as substrate and carbon source for microbial growth);
- The water temperature;
- The capability of disinfectant residual (i.e. chlorine, monochloramine or chlorine dioxide,) to inhibit microbial growth.

Hence, in the developing climate change context microbial regrowth in DWDS will be mainly governed by the interplay between these three factors. Indirectly, their influence will depend on the behaviour of the disinfectant residual, which concentration, along with temperature, governs its biofilm growth inhibition capability. In turn, the disinfectant residual concentration throughout DWDS pipeworks will depend on its decay rates that increase with temperature and NOM increments. Thus, on balance, foreseeable climate change impacts on DWDS biofilm and planktonic microbial regrowth mainly arise from the water temperature, NOM and disinfectant residual interrelated effects, as it is discussed below.

Temperature

Forecasted temperature increases and heath waves will impact microbial regrowth in DWDS. It is well established that biological systems develop faster within a specific range of increasing temperatures. Likewise, the fostering effect of temperature increases on DWDS microbiota (re)growth is known (Percival et al. 2000). At least in absence or at low (<0.1 mg/L) chlorine concentrations (Hallam et al. 2001, Ndiongue et al. 2005), the water temperature influences DWDS biofilms development significantly. Under these conditions, which occur in many systems including the non-chlorinated

ones, biofilm may grow twice faster at 20ºC than at 10ºC (Hallam et al. 2001, Ndiongue et al. 2005).
Conversely, when disinfectants are present at minimally effective concentrations, microorganisms' inactivation rates generally increase exponentially with temperature, typically following an Arrhenius type relationship (WHO 2004). However, disinfectant residual concentrations decay faster at higher temperatures, so that chlorine inhibitory effects on biofilm development are weakened. Hence, on balance rises in DWDS water temperature may promote biofilm growth (Percival et al. 2000, Hallam et al. 2001, Ndiongue et al. 2005). Such effect may be more important in old DWDS with cast iron pipes, since corrosion rates increase with temperature and corroded pipes provide larger and protective surfaces for biofilm development, and have enhanced disinfectant residual demands (USEPA, 2006).
Several studies have shown that planktonic and biofilm bacteria regrowth rates raise more intensely as temperature increases above ca. 15ºC (Batté et al. 2006). Likewise, water temperatures higher than 15ºC may promote the persistence of pathogens (e.g. *Mycobacterium avium*) in DWDS biofilms (Torvinen et al. 2007).

NOM

Raises in source waters NOM, particularly in those of surface origin, are expected consequences of on-going climate changes (UNECE 2009).
In WTP having oxidation/disinfection as treatment steps, refractory NOM (e.g. humic materials) will be broken-down by disinfectants (e.g. ozone, chlorine, chlorine dioxide) into simpler biodegradable derivatives (Ramseier et al. 2011). The same happens in DWDS water, whenever chlorine dioxide or chlorine is used as disinfectant residual. Hence, particularly if no effective biological filtration of raw (e.g., biofiltration, soil passage) or treated water (e.g., sand biofilters, BAC) is used, WTP may face difficulties in removing NOM and producing biologically stable water. Concomitantly decreases in DWDS water biological stability may occur. Such circumstances are likely to arise whenever WTP are challenged by peaks in source water NOM, like those caused by extreme rainfall and algal bloom events (Paerl & Paul 2012). Then, degradation of DWDS microbial water quality may occur, including the occurrence of coliform and *Legionella* spp. events (Wullings et al. 2011).
On the other hand, average or peak increases in DWDS water NOM will lead to higher consumption of disinfectant residual and, thus, to the increased formation of undesirable disinfection by-products (DBPs), as it is discussed below.

Disinfectant residual

In most countries, a concentration of disinfectant (chlorine, chloramine or chlorine dioxide) is kept in the water that leaves WTPs and is maintained throughout DWDS. The so called disinfectant residual is used to counteract microbial regrowth in DWDS, even if generally limited in efficacy against biofilm microorganisms (LeChevallier et al. 1988, Gagnon et al. 2005).
Owing to their relatively slow decay, chlorine (i.e. HOCl and OCl^-) and, less frequently, monochloramine are mostly used as disinfectant residual. While

more unstable, chlorine dioxide is used in some small DWDS, particularly in recent years because of concerns and regulation on chlorine and chloramines DBPs, which include a large variety of toxic compounds (e.g. trihalomethanes, haloacetic acids) (WHO 2011). However, due to its relatively fast decay, chlorine dioxide is only suitable for DWDS where the water has short travelling times.
Due to its lower reactivity against EPS, monochloramine may penetrate biofilms deeper and, thus, be more effective than chlorine against biofilm microorganisms (Van der Wende & Characklis 1990, Norton & LeChevallier 1997). Likewise, by decaying slower monochloramine may be an advantageous alternative to free chlorine in DWDS with long residence times and elevated temperatures. However, their potential for the formation of nitrite and DBPs (e.g. N-Nitrosodimethylamine and Haloacetonitriles) needs to be considered (WHO 2011).Disinfectant residual must be kept above a prescribed minimum concentration needed for effective disinfection, but within a limit allowing for minimisation of the formation of DBPs, in addition to odour and taste. However, disinfectants are consumed by reacting with the water´s NOM while it travels through DWDS, then leading to the formation of DBPs. These may include biodegradable NOM derivatives.
On the other hand, disinfectant residual consumption will also be increased by global warming and heat waves, as the rates of the reactions between the disinfectants and NOM increase significantly as temperature raises. Water temperature rises of 5 to 10ºC may double of chlorine decay rate coefficients (Kastl et al 1999, Powell et al. 2000). Monochloramine may decay twice faster when the water temperature is raised by 16ºC (Sathasivan at al. 2009). Therefore, by increasing the waters disinfectant demand and consumption, rises in DWDS-NOM and temperature will consequently lead to increased formation of DBPs and, possibly, biodegradable NOM derivatives, which in turn may contribute to lower DWDS waters’ biological stability.

6.2 Current operational practices

As pointed out by van der Kooij (2003), increased concerns on drinking water safety, the perception of DWDS biofilm significance for the water’s microbiological quality and the understanding of the importance of biodegradable carbon availability as key factor for regrowth in DWDS, led to the development of the assimilable organic carbon (AOC) and BDOC concepts and methods, as measures of the biological stability of DWDS water. However, owing to the complexity and diversity of organic compounds that control the water’s biological stability – i.e. which may support microbial regrowth- these cannot be determined by chemical analysis. Hence, AOC (van der Kooij et al. 1982) and BDOC (Joret & Levi 1986) were developed as microbiological methods to assess the biological stability of DWDS water samples.
The AOC method (van der Kooij et al. 1982) measures the growth of *Pseudomonas fluorescens* strain P17 and *Spirillum* sp. strain NOX in pasteurized water samples. Growth is generally measured by plate counts or ATP determinations (van der Kooij et al. 1982, LeChevallier et al. 1993). The AOC concentration is calculated with reference to the corresponding yield values for acetate, hence it is expressed as acetate-carbon equivalents/L. The method

has been used in several countries, usually with adaptations or improvements. The latter include the use of natural microbial communities as inocula and the use of turbidity (Hambsch & Werner 1990) or flow cytometry to measure growth (Hammes & Egli 2005).

The BDOC method (Joret & Levi 1986), measures the consumption (mg/L) of dissolved organic carbon (DOC) in water samples incubated with sand from active biofilters' sand bed. Faster evaluations of BDOC are produced by determining in the laboratory DOC reductions during biofiltration of water samples (Lucena et al. 1990).

Contrarily to NOM – often measured as total organic carbon (TOC) – AOC or BDOC control is not widespread among WTPs worldwide. With some exceptions (e.g. Lisbon DWDS), apparently the control of treated water biological stability is mostly practiced in DWDS with no disinfectant residual, like most DWDS in the Netherlands and Denmark, and in parts of other Nordic countries, Germany, Luxembourg and Switzerland.

In the Netherlands biologically stable drinking waters are defined as those having AOC levels lower than 10 µg C L^{-1} and biofilm formation rate (BFR) values below 10 pg ATP cm^{-2} day^{-1} (van der Kooij 2003). The latter, is a flow-through test which measures the biomass developed on the surface of glass cylinders placed on top of each other inside a vertically positioned glass column. Water flows downward and cylinders are collected at regular intervals for biomass quantification, which is done with ATP analysis (Van der Kooij et al. 1995).

Conversely, BDOC values lower than 0.2 mg/L are generally accepted as those of waters with biological stability elsewhere (Dukan et al. 1996).

Nevertheless, factors other than the biological stability of the water may also contribute to increase the levels of microorganisms in DWDS water. In addition to the possibility of being due to ingresses of microbial and/or biodegradable organics contamination into pipes, such increase may be promoted by the release of AOC like compounds from pipes' material and sediments (van der Kooij 2003). By providing larger and protective surfaces for biofilm development and enhancing disinfectant residual decay pipe corrosion tend to boost regrowth. Accordingly, maintenance of DWDS pipes' condition – i.e. proper cleaning, repair and replacement – is also important to lessen microbial regrowth in DWDS.

As the present World Health Organization Guidelines for Drinking-water Quality (WHO 2011) may reflect, the water biological stability is not yet generalized concept. The Guidelines mention AOC as a principal determinant of planktonic and biofilm HPC regrowth, in addition to temperature, lack of disinfectant residual and stagnation. However, do not explicitly describe or prescribe AOC – or BDOC – as a DWDS water monitoring parameters. Nonetheless, the Guidelines consider the need to remove biodegradable organics formed during ozonation as important to avoid undesirable regrowth downstream of WTPs. However, even if not always taken a measure for adaptation to climate changes or to mitigate regrowth in DWDS, the removal of TOC/DOC in WTPs is generally targeted. Thus, while more often directed to the reduction of DBPs formation, indirectly such aim contributes for the improvement water biological stability in DWDS.

6.3 Detecting rapid changes that lead to risk

Advanced methodologies (Lautenschlager et al. 2013) and methods like ATP analysis (van der Wielen & van der Kooij 2010, Vital et al. 2012), quantitative PCR (van der Wielen & van der Kooij 2013) and flow cytometry (Vital et al. 2012) are available to rapidly assess increases in DWDS water microbial contents. However, while very useful to assess the biological stability of treated waters on a routine basis, these approaches only allow detecting regrowth after it has happened. In addition, no established methods are currently available for real time measurement of AOC/BDOC. Alternatively, DOC analysis by on-line spectroscopy (van den Broeke et al. 2006) may be used as a coarse indicator of treated water biological stability.

Nevertheless, as it is detailed in Chapter 7, rapid changes in treated water biological stability most often arise from limitations of WTP to respond to sudden variations in raw water quality. Thus, the timely detection of rapid changes that may lead to excessive microbial regrowth in DWDS mostly depends on the capacity to understand and preventively gauge pertinent alterations in source water quality. Of these, are particularly important those leading to increases in the water's NOM contents, like algal blooms and runoff. The latter is often accompanied by increases in raw water microbiological loads that concurrently challenge WTP capacity. Ingress of organic contaminants into DWDS, including those that may arrive with intrusions, often leads to abnormal consumption of disinfectant residual. Thus, alike for other parameters like pressure and turbidity, frequent or on-line monitoring of chlorine may be a useful indicator of sudden changes in DWDS quality and safety.

Nevertheless, although not amenable for rapid detection, increases in heterotrophic bacteria levels remain the most suitable signal of regrowth occurrence in practice (van der Kooij 2003, WHO 2011). Accordingly, in addition to faecal indicator bacteria, heterotrophic plate counts (HPC) should be monitored in appropriate locations at adequate regular basis, and recorded to provide background knowledge of DWDS microbiology.

Still, while useful, HPC represents only a very small fraction of the microorganisms present in the water and do not allow for the timely detection of increments in DWDS water microbial levels, as HPC usually requires more than 3 days to produce results. Thus, the alternative or complementary use of the above mentioned molecular biology methods are highly advised. Of these, the determination of the waters ATP contents is particularly suited owing to its practicability, rapidity, low detection limit and relationship with the water biological stability (Deininger & Lee 2001, van der Wielen & van der Kooij 2010).

6.4 Corrective responses and curative measures

Regrowth essentially depends on DWDS water biological stability, which is in turn primarily governed by WTP capacity to remove NOM and produce/release of AOC/BDOC. Hence, counteractive measures to effectively respond to episodes of enhanced regrowth can only be taken upstream of DWDS networks. As is discussed in Chapter 7, adaptation of WTP operation and processes or its interim complementation with additional

treatment steps (e.g. GAC filtration, BAC filtration) may be required when alternative sources are not available for provisional use.
Given that in practice it is not possible to control DWDS water temperature, the use of disinfectant residual appears as the only possible measure to mitigate enhanced regrowth. Superchlorination may be advised to protect consumers upon detection of significant increments in bacterial numbers or pathogens, particularly in DWDS with a history of coliform (e.g. *E. coli*) or pathogen events (WHO 2011). Booster dosing may be required to control chlorine concentrations in low flow extremities of DWDS networks. Hydraulic or chlorine models are useful tools to support siting of rechlorination locations.
Nevertheless, it should be taken into account that chlorine can reduce the numbers of active planktonic bacteria, but have little to no effect on their sessile counterparts (Chauret et al. 2005, Srinivasan et al. 2008). In addition, while being effective against many bacteria that can occur in biofilms (e.g. *E. coli*), including pathogens (e.g. *Campylobacter* spp., *Legionella* spp.), chlorine has low efficacy against some pathogens, including those of the *Mycobacterium* group.

6.5 Preventive measures

In addition to the selection of provisional sources wherever they are available, the proactive implementation of multibarriers upstream of DWDS and the use of disinfectant residual as a supplementary barrier, the design, operation and condition of network pipelines are also important DWDS characteristics in minimizing regrowth.

As it is discussed in Chapter 7, the selection/control/protection of source water quality, the ability to foresee or rapidly detect changes in the quality of the abstracted water and the capacity of WTPs to respond to the changes and produce water with the required biological stability are the most effective factors to prevent regrowth. Nevertheless, some DWDS characteristics that may potentiate regrowth of sessile and planktonic microorganisms need to be considered, particularly while raises in the water temperature occur. These characteristics mainly concern:

- The condition of network pipes
- DWDS design and operation
- The effectiveness of maintenance activities.
- The type of network materials in contact with the water;

In addition to reducing pipes' hydraulic capacity and structural strength, thus promoting fissuring and the likelihood of intrusions, pipe corrosion leads to increased disinfectant residual demand and enhanced development of biofilm. Internal corrosion of unprotected metallic pipes (e.g. cast iron, steel) fosters intense development of biofilm by augmenting roughness, thus providing larger and protective surfaces for sessile colonization, which may also be promoted by the absorption of biodegradable organic compounds on the corroded surfaces and corrosion debris (van der Kooij 2003, Levi 2004).

The design and operation of DWDS encompass important aspects of distribution network hydraulics of great importance for drinking water microbiological quality and safety. These, mainly concern the water's travelling times, flow and composition.

Oversizing and/or looping of DWDS result in long travel times and, thus, favour the conditions for microbial regrowth by leading to zones with decreased concentrations of disinfectant residual and low-flow/non-flow. The latter, in addition to chlorine depletion, are prone to the build-up of sediment beds, which accumulate biodegradable organic compounds and nutrients, thus being privileged sites for microbial (re)growth ((van der Kooij 2003, Batté et al. 2006). With this respect, DWDS peripheral parts and dead-ends are particularly critical. In addition, in DWDS loops sudden changes in flow velocity and/or direction may occur that disturb sediments and slough biofilms, then leading to discolouration and the associated degradation of the microbial quality of the water.

Changes in the water composition are other possible causes of rises in DWDS network water microbial levels. Biofilm growth enhancement may arrive with decreases in the water's biological stability whenever that results from mixing of waters from different sources. On the other hand, abrupt changes in the type of disinfectant residual may lead to dislodging of biofilm microorganisms.

Microbial regrowth is likely to be intensified in DWDS with improper maintenance. In addition to the above described consequences from fissured and/or corroded/corroding pipes, allowing for sediment accumulation promotes regrowth and, upon re-suspension, leads to discolouration and increments in the water' s microbial contents (Batté et al. 2006).

In addition to corrosion, regrowth enhancement by DWDS materials in contact with the water may arrive from the leaching out of organic compounds, which expectably increases with temperature. In addition to some kind of plastic pipes and valves constituents, materials with such potential may include those of sealants (e.g. polyamide, silicone, rubber), liners (e.g. resins), coatings and lubricants (LeChevallier 2003).

In what concerns actions to prevent biofilm and planktonic microorganisms regrowth in DWDS, it can be concluded that these primarily relate to measures that can be implemented to assure the biological stability of the water entering into DWDS. In addition, certain features of DWDS design, materials, operation and maintenance may also impact microbial regrowth, as it is summarized below:

- In order to minimize travelling times, avoid low and no-flow conditions, as well as to mitigate the occurrence of changes in flow velocity and direction, desirably DWDS must be properly dimensioned and devoid of loops;

- With respect to DWDS materials that contact water, in addition to being non-corrodible and with surfaces of low rugosity, they must not have a potential to release biodegradable organic compounds;

- In addition to minimizing travelling times, DWDS operation needs to be done as to avert mixing leading to fast changes in the waters composition, particularly those causing decreases in bulk water biological stability, in addition to avoid sudden changes in flow velocity and/or direction;

- Maintenance should be assured and carried out methodically, so that to improve DWDS pipelines condition, including the replacement of corroding pipes and the cleaning operations needed to minimize critical accumulation of sediments

Finally, the use of effective concentrations of disinfectant residual as a secondary barrier, particularly upon rises in DWDS water microbial contents, should not be disregarded (WHO, 2011).

6.6 Experiences and cases

The Netherlands

The Netherlands, where paradigmatically chlorination is not used for primary disinfection or as disinfectant residual, the control of drinking water biological stability is a common practice to limit regrowth in DWDS (Smeets et al. 2009). There, biologically stable water (AOC < 10 μg C/L; BFR < 10 pg ATP cm^{-2} day^{-1}) is generally produced by aeration and sand filtration of underground water (van der Kooij 2003, Smeets et al. 2009).
For the same purpose, waters of surface origin are treated more extensively by physical (e.g. coagulation-sedimentation) and chemical (e.g. ozonation) processes. Biofiltration is used to remove AOC, principally from waters treated with ozone (van der Kooij 2003 Smeets et al. 2009). Concurrently, tested biostable materials are strategically used in DWDS, including in the replacement of old pipes. Likewise, sediment accumulation is minimized by methodological flushing (Smeets et al. 2009).
Through these practices, sound management and selection of water sources, appropriate treatment, adequate DWDS network design and operation (e.g. keeping high flow velocities and pressure throughout DWDS) and rigorous maintenance utilities in the Netherlands supply microbiologically safe water without disinfectant residual DBPs. Such condition is corroborated by the results of well-designed monitoring programs, which allow for timely detection of changes in the waters quality and safety (Smeets et al. 2009).
Therefore, by advancing the undertaking of the above discussed measures to detect rapid changes that lead to risk, as well as the corresponding corrective, remedial and preventive ones, the drinking water utilities in the Netherlands are likely to be prepared to respond to the on-going climate changes. Such preparedness is being proactively strengthened by research on the possible competitive advantage of some opportunistic pathogens (e.g. *L. pneumophila*, *P. aeruginosa*, *Stenotrophomonas maltophilia*, *Aspergillus fumigatus*) at higher temperatures. The research is being funded by the water companies, which thus will be aware of possible climate change effects on DWDS water safety.

Portugal

The Portuguese utilities supply good quality and safe drinking water. This is corroborated by the close to 100% compliance with the pertinent Portuguese legislation (ERSAR 2013), which mirrors and is more stringent than its background document, the European Drinking Water Directive (EC 1998). Nonetheless, as in most developed countries, while the occurrence of biofilm is commonly acknowledged and seen as a potential cause of degradation of DWDS water microbiological quality, often the maintenance of effective concentrations of disinfectant residual is erroneously viewed as an effective barrier against DWDS microbial contaminants. Accordingly, the biological stability of the water is not a generalized concept and TOC, colony counts and faecal indicators are the related parameters that are commonly monitored.

To the best of our knowledge, in Portugal only EPAL-Lisbon, which supplies water to ca. 3 million inhabitants in Lisbon and in other 35 municipalities, analyses for AOC at a regular basis. Such practice is a follow up of studies done for the characterization of the Lisbon network colonization by biofilm and associated risks (Menaia et al. 2008).

The studies comprised the analysis of biofilm samples (62) scrapped from pipe lengths (52) collected throughout the Lisbon DWDS during pipe renewal or burst repairs. In addition to total protein as an indicator of the colonization intensity, samples were analysed by cultural methods for total counts, coliforms, *E. coli*, intestinal enterococci, *Clostridium perfringens*, *Pseudomonas aeruginosa*, *Salmonella* spp. and *Legionella* spp. Likewise PCR was used for *Aeromonas hydrophila*, *Campylobacter jejuni*, *Legionella* spp., *Legionella pneumophila* and *Mycobacterium* spp. detection.

Results showed that the Lisbon network sessile colonization was relatively weak in intensity and that no meaningful risks were associated to the DWDS biofilm.

6.7 Tools, manuals, guidelines, documentation and references

As is repeatedly stressed above, preparedness to respond to climate changes that challenge DWDS water quality and safety, including those leading to increases in bacterial (re)growth, can only be effectively achieved if proactive strategies comprehending measures upstream of DWDS are taken, like those presented in Chapter 7 (e.g. Table 4).

Nevertheless, part of the impacts of such challenges (e.g. temperature and NOM increases) consists on the aggravation of processes that are already motives of concern at DWDS level. Hence, understanding of such processes, as well as of the means available for their control or mitigation is of critical importance. With this regard, examples of useful publications and manuals are:

- Heterotrophic bacteria in drinking water distribution system: a review (Chowdhury, S. 2012)
- Safe piped water: managing microbial water quality in piped distribution systems (WHO 2004)
- Heterotrophic plate counts and drinking-water safety (WHO 2003)
- Health risks from microbial growth and biofilms in drinking water distribution systems (USEPA 2002)

- Microbial regrowth in drinking-water supplies. Problems, causes, control and research Needs (van der Kooij & van der Wielen, 2013).

6.8 References

Batté, M., Féliers, C., Servais, P., Gauthier, V., Joret, J.-C., Block, J.-C. 2006. Coliforms and other microbiological indicators occurrence in water and biofilm in full-scale distribution systems. *Water Sci. Technol*. 54, 41–48.

Beuken, R., Reinoso, M., Sturm, S., Kiefer, J., Bondelind, M., Åström , J., Lindhe, A., Lars Losén, Petterson, T., Machenbach, I., Melin, E., Thorsen, T., Eikebrokk, B., Hokstad, P., Røstum, J., Niewersch, C., Kirchner, D., Kozisek, F., Weyessa-Gari, D., Swartz, C., Menaia, J. (2008) *Identification and description of hazards for water supply systems. A catalogue of today's hazards and possible future hazards*. Techneau report D4.1.4

Chambers, K., Creasey, J. Forbes, L. (2004) Design and operation of distribution networks. *In Safe Piped Water: Managing Microbial Water Quality in Piped Distribution Systems*. R. Ainsworth (ed.), IWA Publishing, London, UK

Chowdhury, S. (2012) Heterotrophic bacteria in drinking water distribution system: a review. *Environ. Monit. Assess*. 184,6087-6137

Deininger, R.A., Lee, J.Y., (2001) Rapid determination of bacteria in drinking water using an ATP assay. *Field Anal. Chem. Technol*. 5, 185-189

Dukan, S., Levy, Y., Piriou, P., Guyon, F., Villon, P. (1996) Dynamic modelling of bacterial growth in drinking water networks. I. 30, 1991–2002

EC (2008) *Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption*. European Commisssion

ERSAR (2013) Controlo da qualidade da água para consume humano. *Relatório Anual dos Serviços de Águas e Resíduos em Portugal*. Entidade Reguladora dos Serviços de Água e Resíduos (2012), Lisboa, Portugal

Fass S., Dincher M. L., Reasoner D. J., Gatel D. and Block J. C. (1996) Fate of *Escherichia coli* experimentally injected in a drinking water distribution pilot system. *Water Res*. 30, 2215–2221

Flemming, H.-C., Percival, S.L. & Walker, J.T. (2002) Contamination potential of biofilms in water distribution systems. *Water Sci. Technol.* 2 (1), 271–280

Flemming H.-C., Neu T.R. and Wozniak D. (2007) The EPS-matrix: the "house of biofilm cells". *J. Bacteriology* 189, *7945–7947*

Gagnon, G.A., Rand, J.L., O`Leary, K.C., Rygel, A.C., Chauret, C., Andrews, R.C. (2005). Disinfectant efficacy of chlorite and chlorine dioxide in drinking water biofilms. *Water Res*. 39, 1809-1817

Geldreich, E.E. (1996) *Microbial quality of water supply in distribution systems*. CRC Press, Boca Raton, Fla.

Hambsch, B., Werner, P. (1990) Automated Measurement of Bacterial Growth Curves for the Characterization of Organic Substances in Water. *Proc. AWWA Water Quality Technology Conf.*, San Diego, California

Hallam, N.B., West, J.R., Forster, C.F., Simms, J. (2001) The potential for biofilm growth in water distribution systems. *Water Res*. 35,4063-4071

Hammes, F.A., Egli, T. (2005) New method for assimilable organic carbon determination using flow-cytometric enumeration and a natural microbial consortium as inoculum. *Environ. Sci. Technol*. 39, 3289-3294

Helmi, K., Skraber, S., Gantzer, C., Willame, R., Hoffmann, L., Cauchie, H.-M. (2008) Interactions of *Cryptosporidium parvum*, *Giardia lamblia*, Vaccinal Poliovirus Type 1, and Bacteriophages ϕ X174 and MS2 with a drinking water biofilm and a wastewater biofilm. Appl. Environ. Microbiol. 74, 2079–2088

Joret, J.C., Levi, Y. (1986) Méthode rapide d'évaluation du carbone éliminable des eaux par voie biologique. Trib. Cebedeau 510(39), 3–9

Kastl, G. J., Fisher, I.J., Jegatheesan, V. (1999) Evaluation of chlorine decay kinetics expressions for drinking water distributions systems modelling. J. Water Supply Res. Technol.–Aqua 48(6), 219-226

Kuiper, M.W., Wullings, B.A., Akkermans, A.D., Beumer, R.R., van der Kooij. D. (2004) Intracellular proliferation of *Legionella pneumophila* in *Hartmannella vermiformis* in aquatic biofilms grown on plasticized polyvinyl chloride. Appl. Environ. Microbiol. 70,6826–6833

Larsson, S., Mezule, L., Juhna, T. (2008) Survival of *E. coli* in drinking water biofilm: the application of FISH technique. *Techneau report* D.3.6.8.1

Lautenschlager, K., Hwang, C., Liu, W-T., Boon, N., Köste, O., Vrouwenvelde, H., Egli ,T., Hammes, F. (2013) A microbiology-based multi-parametric approach towards assessing biological stability in drinking water distribution networks. *Water Res*. 47, 3015-3025

LeChevallier, M.W. (2003) Conditions favouring coliform and HPC bacterial growth in drinkingwater and on water contact surfaces. In J. Bartram, J. Cotruvo, M. Exner, C. Fricker, A. Glasmacher (Eds.), *Heterotrophic Plate Counts and Drinking-water Safety*. IWA Publishing, London, UK

LeChevallier, M.W., Babcock, R.M. and Lee, R.G. (1987) Examination and characterization of distribution system biofilms. *Appl. Environ. Microbiol*. 54, 2714–2724

LeChevallier, M.W., Cawthon, C.D., Lee, R.G. (1988) Factors promoting survival of bacteria in chlorinated water supplies. Appl. Environ. Microbiol. 54, 649-654.

LeChevallier, M.W., Shaw, N.E., Kaplan, L.E.,Bott, T. (1993) Development of a rapid assimilable organic carbon method for water. *Appl. Environ. Microbiol*. 59, 1526-1531

Lehtola, M.J., Pitkänen, T., Miebach, L., Miettinen, I.T. (2006) Survival of *Campylobacter jejuni* in potable water biofilms: a comparative study with different detection methods. *Water Sci. Technol*. 54(3), 57–61

Levi, Y. (2004) Minimizing potential for changes in microbial quality of treated water. In *WHO Safe Piped Water: Managing Microbial Water Quality in Piped Distribution Systems*. R. Ainsworth (ed.), IWA Publishing, London, UK

Lewis, K. (2001) Riddle of biofilm resistance. *Antimicrob. Agents Chemother*. 45, 999-1007

Lucena, F., Frais, J., Ribas, F. (1990) A new dynamic approach to the determination of biodegradable dissolved organic carbon in water. *Environ. Technol*. 12, 343–347

Menaia, J., Mesquita, E. (2004) Drinking water pipe biofilms: present knowledge concepts and significance. *Water Sci. Technol.: Water Supply* 4 (2), 115–124

Ndiongue S., Huck P.M., Slawson R.M. (2005) Effects of temperature and biodegradable organic matter on control of biofilms by free chlorine in a model drinking water distribution system. *Water Res*. 39, 953–964

Norton, C.D., LeChevallier, M.W. (1997). Chloramination: its effect on distribution system water quality. J. AWWA 89,66–77

Payment, P, Robertson, W. (2004) Water microbiology. In *Safe Piped Water: Managing Microbial Water Quality in Piped Distribution Systems*. R. Ainsworth (ed.), IWA Publishing, London, UK

Paerl, H.W., Paul, V.J. (2012). Climate change: Links to global expansion of harmful cyanobacteria. *Water Res*. 46, 1349–1363.

Percival, S.L., Walker, J.T., Hunter P.R. 2000. *Microbiological aspects of biofilms and drinking water*. CRC Press, Boca Raton, FL

Powell J.C., Hallam N.B., WestJ. R., Forster C.F., Simms J. (2000) Factors which control bulk chlorine decay rates. *Water Res*. 34, 117-126

Ramseier, M.K., Peter, A., Traber, J., von Gunten, U. (2011) Formation of assimilable organic carbon during oxidation of natural waters with ozone, chlorine dioxide, chlorine, permanganate, and ferrate. *Water Res*. 45, 2002-2010

Reasoner, D.J., Blannon, J.C., Geldreich, E.E. and Barnick, J. (1989) Nonphotosynthetic pigmented bacteria in a potable water treatment and distribution system. *Appl.Environ. Microbiol.* 55, 912-921

Sathasivan, A., Chiang, J., Nolan, P. (2009) Temperature dependence of chemical and microbiological chloramine decay in bulk waters of distribution system. *Water Sci. Technol.: Water Supply* 9(5)493-499

Smeets, P. W. M. H., Medema, G. J., van Dijk, J. C. (2009) The Dutch secret: how to provide safe drinking water without chlorine in the Netherlands. *Drink. Water Eng. Sci.* 2, 1–14

Srinivasan, S., Harrington, G.W., Xagoraraki, I., Goel, R. (2008) Factors affecting bulk to total bacteria ratio in drinking water distribution systems. Water Res. 42,3393–3404

Torvinen, E., Lehtola, M.J., Martikainen, P.J., Miettinen, I.T. (2007) Survival of *Mycobacterium avium* in drinking water biofilms as affected by water flow velocity, availability of phosphorus, and temperature. *Appl. Environ. Microbiol.* 73,6201- 6207.

UNECE (2009) *Guidance on Water and Adaptation to Climate Change*. United Nations Economic Commission for Europe

USEPA (2002) Health risks from microbial growth and biofilms in drinking water distribution systems. US Environ. Protection Agency. Office of Water

USEPA (2006) *The Effectiveness of disinfectant residuals in the distribution system*. US Environ. Protection Agency. Office of Water

van den Broeke, J., Langergraber, G., Weingartner, A. (2006) Online and insitu UV/vis spectroscopy for multi-parameter measurements: a brief review. *Spectroscopy Europe* 18(4), 15-18

van der Kooij, D. (2003) Managing regrowth in drinking water distribution systems. In J. Bartram, J. Cotruvo, M. Exner, C.Fricker, & A. Glasmacher (Eds.), *Heterotrophic Plate Counts and Drinking-water Safety*. IWA Publishing, London, UK van der Kooij, D., Visser, A., Hijnen, W.A.M. (1982) Determining the concentration of easily assimilable organic carbon in drinking water. *J. Am. Water Works Assoc.* 74, 540–545

van der Kooij, D., Veenendaal, H.R., Baars-Lorist, C., van der Klift, H.W. and Drost, Y.C. (1995b) Biofilm formation on surfaces of glass and Teflon exposed to treated water. *Water Res.* 29, 1655–1662

van der Kooij, D., van der Wielen P.W.J.J. (2013) Microbial growth in drinking-water supplies. Problems, causes, control and research needs. IWA publishing, London, UK.

van der Wende, E., Characklis W.G. (1990) Biofilms in potable water distribution systems. In *Drinking Water Microbiology*. G.A. McFeters (Ed.) Springer-Verlag: New York, NY.

van der Wielen P.W.J.J., van der Kooij, D. (2010). Effect of water composition, distance and season on the adenosine triphosphate concentration in unchlorinated drinking water in the Netherlands. *Water Research* 44:4860-4867

van der Wielen P.W.J.J., van der Kooij, D. (2013). Non-tuberculous mycobacteria, fungi and opportunistic pathogens in unchlorinated drinking water in the Netherlands. *Applied and Environmental Microbiology* 79:825-834.

Vital, M., Dignum, M., Magic-Knezev, A., Ross, P., Rietveld, L., Hammes, F. (2012) Flow cytometry and adenosine tri-phosphate analysis: Alternative possibilities to evaluate major bacteriological changes in drinking water treatment and distribution systems. *Water Res.* 46, 4665-76

WHO(2003) *Heterotrophic Plate Counts and Drinking-water Safety*. World Health Organization. J. Bartram, J. Cotruvo, M. Exner, C. Fricker, A. Glasmacher (Eds). IWA Publishing, London, UK

WHO (2004) *Safe piped water: managing microbial water quality in piped distribution systems. World Health Organization*. R. Ainsworth (ed.), IWA Publishing, London, UK

WHO (2004) *Water Treatment and Pathogen Control*. M. W. LeChevallier and K.K. Au (Eds.). World Health Organization. Geneva

WHO (2011) *Guidelines for drinking water quality*. 4th ed. World Health Organization. Geneva

Wingender, J., Flemming, H.-C. (2011) Biofilms in drinking water and their role as reservoir for pathogens. *Int. J. of Hygiene and Environ.* Health, 214, 417– 423

Wullings, B.A., Bakker, G., van der Kooij, D. (2011) Concentration and diversity of uncultured *Legionella* spp. in two unchlorinated drinking water supplies with different concentrations of natural organic matter. *App. Environ. Microbiol.* 77, 634-641

7 Adaptation of treatment processes

Maria João Rosa, Elsa Mesquita (LNEC)

7.1 Challenges to drinking water quality

Climate change, mainly the temperature change, the increase of extreme hydrological events, soil drying-rewetting cycles and solar radiation will affect the surface water quality and treatment plants must be adapted to face the expected modifications (Bartram et al. 2009, Delpla et al. 2009).

Temperature affects almost all physicochemical and biological processes in water bodies. Temperature rising favours dissolution, solubilisation, complexation, degradation, evaporation and biological processes, among others, leading to an increase of dissolved and colloidal substances in water and the decrease of dissolved gases (e.g. oxygen) (Delpla et al. 2009).

Intense rainfall events and droughts will modify raw water quality by direct effects of dilution or concentration of dissolved substances (Delpla et al. 2009) and will promote runoff and transport of organic matter and microorganisms, including pathogens into lake and rivers (Roig et al. 2011). In fact, during the rainy seasons the presence of enteric pathogens resistant to chlorination (e.g. *Cryptosporidium*) is higher compared to the other seasons (Roig et al. 2011). The contamination of freshwater with microbial pathogens may also occur in heavy rainfall events if the rainwater is discharged in combined sewer systems with limited capacity. In these cases the sewers can overflow directly into surface water body (Charron et al. 2004), contributing to the loading with coliform bacteria (Pednekar et al. 2005).

Impacts of soil drought–rewetting cycles on water quality include those related with the enhanced decomposition of organic matter and its leachate into streams promoted by those cycles. After intense rainfall periods, the increase of surface water turbidity often occurs in parallel with a rise in dissolved organic carbon (Ribau Teixeira et al. 2002, Ribau Teixeira and Rosa 2003, Evans et al. 2005).

Solar irradiation increase (warming temperatures and UV radiation) may affect NOM contents (type and concentration) in fresh water systems due to photolysis and cyanobacterial or algal bloom events (Soh et al. 2008). UV transformations of photosensitive organic micropollutants (e.g. pharmaceuticals and other emerging microcontaminants) into unknown forms should be seriously taken into account (Delpla et al. 2009). Cyanobacterial bloom events are also associated to the increase of T&O compounds (MIB and geosmin) and to the eventual presence of cyanobacterial-toxins in the water. Climate changes (warmer temperatures, increased P concentration and rainfall events) will lead to favourable conditions for cyanobacterial growth so an increase of the frequency and intensity of bloom events is expected in surface waters (Reichwaldt and Ghadouani 2012, Paerl and Paul 2012).

The survey performed within PREPARED WP 5.2.4 allowed to conclude that the utilities from the Mediterranean zone cities/regions – Barcelona, Istanbul, Lisbon and Algarve – have in common the majority of the water quality challenges, namely faster and more severe raw water quality changes and an overall increase in water temperature, NOM concentration, turbidity, microbial load, nutrients, (micro)algae and/or cyanobacteria, T&O compounds and cyanotoxins, most often hepatotoxic microcystins. Anthropogenic organic micropollutants, including endocrine disrupting compounds (EDCs, e.g. pharmaceuticals and pesticides) are also an emerging issue.

7.2 Current operational practices

Many drinking water treatment plants were designed to remove (or inactivate) particles, colloidal matter and microorganisms through a treatment train that generally includes chemical pre-oxidation, coagulation/flocculation/ sedimentation, filtration and disinfection (AWWA 1999).

The occurrence and detection of emerging (micro)contaminants in water sources and advances in knowledge of toxicological and epidemiological data of those contaminants are driving increasingly stringent water quality standards in parameters resistant to conventional treatment.

In addition, the utilities are now facing new challenges arising from faster and more severe raw water quality variations promoted by climate change that are expected to lead to an overall deterioration of the water quality used for drinking water production.

To handle these challenges the utilities generally rely on conventional surface water treatment assisted by ozonation and PAC (or, less frequently, GAC) adsorption. Adjustment of chemicals' (mainly ozone, chlorine and coagulant) and PAC dosing, as well as of sludge wasting and filtration cycles (filtration rates, filtration time, backwashing), were identified as key-measures for adapting the treatment.

However, the increase of suspended particles and dissolved organic matter in raw water may compromise the water treatment efficiency, therefore increasing the risk of drinking water contamination with pathogens and micropollutants, e.g. cyanotoxins, pesticides, pharmaceuticals and cosmetics.

Moreover, an increase in NOM contents (e.g. humic and fulvic acids and algal/cyanobacterial organic matter) may promote the undesired DBP formation. DBPs such as trihalomethanes, haloacetic acids and haloacetonitriles are potentially carcinogenic and or endocrine disruptors among other health effects. Non-halogenated biodegradable DBPs may compromise the microbiological stability of distributed water. As discussed in Chapter 7, easily biodegradable organic compounds (BDOC) produced during NOM oxidation may promote bacterial regrowth in the distribution systems and sustain biofilm development on pipe walls, where pathogenic (bacteria, protozoa, viruses, (oo)cysts, endospores) organisms may grow and/or be entrapped.

Special care must therefore be taken to ensure a safe disinfection and (at least partial) removal of microcontaminants while minimizing DBP formation, e.g. those regulated in EU drinking water quality Directive, THMs and bromate.

The type and concentration of DBPs depend on the oxidant type and dose (concentration x contact time) as well as on the raw water quality in terms of temperature, pH, alkalinity and DBP precursors, i.e. NOM (type and concentration) and, when ozonation is used, bromide contents. Climate change will therefore impact the DBP formation due to temperature increments and changes in the water NOM and inorganic (alkalinity, conductivity, pH) matrices.

Most microcontaminants are not effectively removed by conventional water treatment (Rosa et al. 2009, WHO 2011) and may constitute a health-issue in drinking water supply in climate-change sensitive-areas, when no additional physical (e.g. nanofiltration or reverse osmosis membranes), biological (e.g. slow sand filtration, GAC biofiltration) or hybrid (fine-PAC adsorption/membrane (bio)reactor) barriers exist in the WTPs.
The control of micropollutants might be particularly challenging in drought scenarios and intense rainfall events whenever the raw water becomes highly turbid, NOM and microbiologically loaded, characteristics which strongly hinder the performance of the oxidation, coagulation and adsorption processes due to the increased demand of oxidant(s), coagulant(s) and adsorbent.

Strong variations in water background inorganic matrices driven by water scarcity and rain events may also severely impact the treatment effectiveness and efficiency for removing particles, microorganisms, colloid matter, NOM and other organics. Bromate formation may also become an issue in water scarcity scenarios, when the inlet bromide concentration increases due to saline intrusion in groundwater and surface waters used for drinking water production (USEPA 2005, Rosa et al. 2009, WHO 2011).
Measures to mitigate these risks are detailed in 7.4, including operation practices of conventional treatment and upgrade with advanced treatments.

7.3 Detecting rapid changes that lead to risk

An effective and efficient treatment adaptation relies on a deep and comprehensive understanding of the raw water source, and of the treatment plant capacity and limitations.

The ability to detect rapid changes that lead to risk is a key-issue for deciding and implementing the adequate preventive and corrective actions. This will require:

3. Implementing pro-active measures to identify changes in quantity and quality of water resources;
    a. Anticipating the water source pollution – modelling intense rainfall events (frequency and intensity), runoffs and droughts (Beniston et al. 2007);
    b. Characterizing the water source pollution – monitoring (volume and water quality parameters) of intense rainfall events and wastewater discharges in the watershed;
    c. Characterizing the source water availability and quality – modelling the water quality in different scenarios (Kaste et al. 2006, Komatsu et al. 2007), e.g. USEPA recommends WASP7 - Water Quality Analysis Simulation Program (section 7.7);

d. Regular monitoring/inspection of the source water quality – visual inspection, e.g. of water scums, turbidity and colour, including as much as possible parameters of rapid determination for early warning of quality changes requiring treatment adaptation (e.g. cyanobacterial blooms, muddy and clay waters);

4. Implementing pro-active measures to identify the impact of raw water quality changes in the produced water quality;
   a. Monitoring the critical treatment steps' effectiveness and efficiency, using as much as possible reliable online measurements;
   b. Mondeling WTP response to raw water quality changes (e.g. van Leeuwen et al. 2005, Rietveld et al. 2010, Vieira et al. 2010, Silva et al. 2012).

The water safety plans (WHO 2009) and ISO 22000 (the family of International Standards addressing food safety management) (section 7.7) are robust methodologies for mapping the critical water quality parameters and the critical treatment steps for each WTP. WTP modelling is a very useful tool for identifying and managing critical changes that lead to risk.

The effectiveness and cost-efficiency of the monitoring programs can be improved by a combination of raw water quality management and treatment control. Examples of relevant parameters are described in Table 4.

Table 4 - Water quality monitoring from source to WTP exit – what, how and where to monitor rapid changes that may lead to risk (examples)

What and how (examples)	Where	References
Temperature (online probes)	RW-S, RW-I, WT-bar	
Turbidity (online probes)	RW-S, RW-I, WT-bar, TW	
Water inorganic matrix (online probes) pH and EC	RW-S, RW-I, WT-bar, TW	
Cyanobacteria (online probes) **Chlorophyl-a and cyanopigments** (online) **Cyanobacteria** (laboratory) Cell counts or biomass volume **Chlorophyl-a and cyanopigments** (laboratory) **Cyanotoxins** (laboratory) Microcystins and other relevant toxins	RW-S, TW positive results require monitoring RW-I and WT-bar for treatment adaptation/optimisation	Chorus & Bartram 1999 Schmidt et al. 2008, 2009
Microbial pathogens (laboratory) *E.coli*, enterococci, *Clostridium* endospores (feacal indicators) Virus and parasitic protozoa	RW-I, TW positive results require moni-toring WT-bar for treatment adaptation/optimisation	Figueras & Borrego 2010 WHO 2011
NOM (DBP precursors) (online probes) TOC, UV/Vis spectra, Fluorescence spectra **NOM (DBP precursors)** (laboratory) UV254nm, DOC, SUVA (filtered samples)	RW-I and/or WT-bar (at least one point prior to chlorination), TW	USEPA 1998, 1999a-b Hurst et al. 2004 Rosario-Ortiz et al. 2007
Bacterial regrowth or biofilm formation potential (laboratory) AOC assay	WT-bar and/or TW	LeChevallier et al. 1993 Hammes & Egli 2005
Suspended matter (online probes) Total suspended solids	RW-I, WT-bar and/or TW	Nechad et al. 2010
Disinfectant residual (online probes) Free chlorine, chloramine	TW	

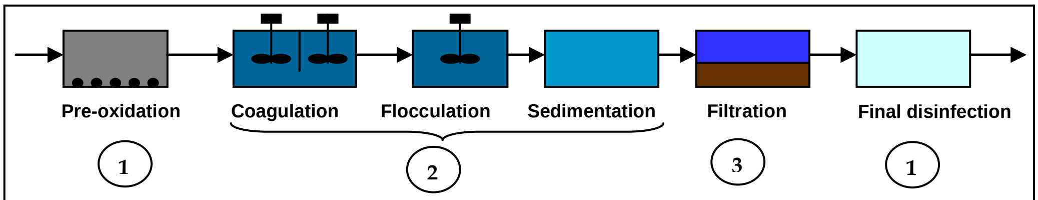
RW-S: raw water source; RW-I: raw water at WTP inlet; WT-bar: water between treatment barriers affecting the given parameter; TW: treated water.

7.4 Corrective responses and curative measures

As introduced in section 7.2, current operational practices of conventional water treatment are often insufficient to deal with expectable risks due to climate change. Measures to mitigate these risks are herein proposed, including operation practices of conventional treatment and WTP upgrade with advanced treatments.

7.4.1 Operation practices of conventional treatment adaptation to climate-change challenges

The so-called conventional surface water treatment generally starts and ends with chemical oxidation (pre-oxidation and final disinfection), and comprises in between passes through clarification by coagulation/flocculation/ sedimentation and rapid filtration (AWWA 1999). Each barrier has specific critical aspects in terms of target contaminant(s) and interfering 'species' (i.e. water quality parameters) (Figure 15) which determine specific adaptive measures.
Measures focussing on specific contaminants are described in earlier chapters. This section addresses the integrated control of the contaminants challenging surface water conventional treatment in climate change scenarios.



Treatment step	Critical aspects	
	Target contaminants	Interfering 'species'
1 Oxidation	Microorganisms, DBP control, T&O compounds, cyanotoxins, iron and manganese, EDCs and other micropollutants	Temperature, NOM, pH, bromide, alkalinity, hardness, ammonium, turbidity, micro(algae) and cyanobacteria, ...
2 Clarification (C/F/S)	Turbidity, micro(algae) & cyanobacteria, NOM (for enhanced coagulation)	Temperature, pH, alkalinity, hardness, NOM (for regular coagulation), ...
3 Filtration	Residual turbidity, micro(algae) & cyanobacteria, other microorganisms	Temperature, ...

Figure 15 - Conventional surface water treatment and identification of critical aspects related with climate change-driven challenges

Pre-oxidation is mainly used to control the biological growth in the WTP, to inactivate biological forms resistant to chlorine (the usual oxidant in the final disinfection step), to oxidise microcontaminants (e.g. T&O compounds, cyanotoxins, EDCs) into less harmful species, to oxidise soluble forms of iron and manganese and produce settleable forms, and to assist the subsequent coagulation. The mainly used oxidants are chlorine, chlorine dioxide, permanganate and ozone.

Depending on the oxidant type and dose (residual concentration x contact time) and on the water quality (e.g. NOM, pH, temperature, alkalinity, ammonium, bromide), different undesirable DBPs arise from the oxidation reactions. If the oxidant is chlorine and the bromide level in raw water is low, the main DBPs are chlorinated forms of trihalomethanes (e.g. chloroform) and haloacetic acids (e.g. di- and trichloroacetic acid). Otherwise, analogue forms of bromide will be also important (e.g. bromodichloromethane, bromodichloroacetic acid). Chlorine dioxide by-products include chlorite and chlorate, undesirable for their adverse health effects. Ozone should not be applied to raw water containing bromide levels higher than 100 μg/L, at which high levels of bromate (at pH > 8.7) or brominated organic compounds (at pH < 8.7) will be probably produced (**Error! Reference source not found.**Figure 16).

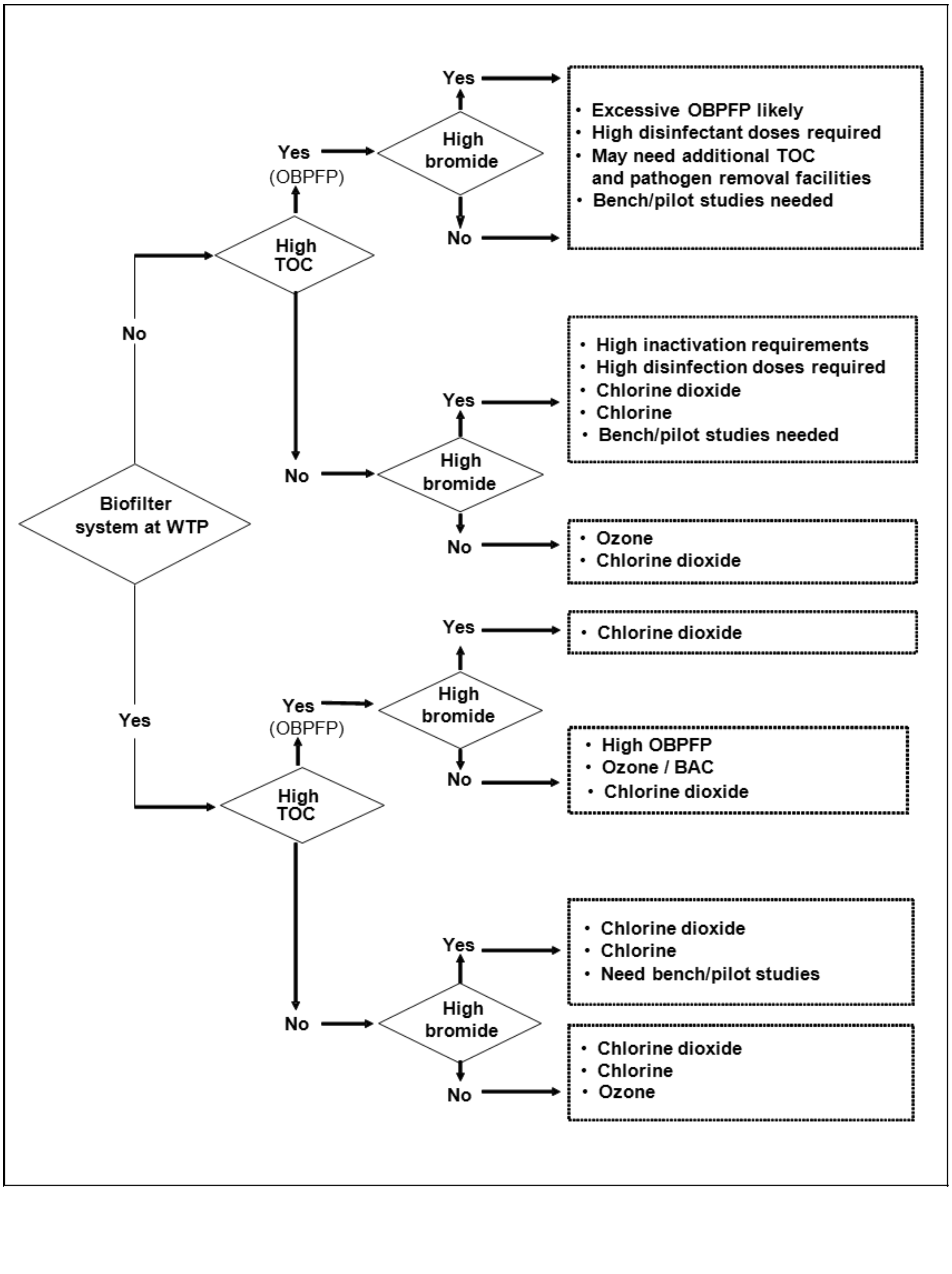


Figure 16 Algorithm for selecting the primary disinfectant (USEPA 1999a)

The DBPs are in general low MW, hydrophilic compounds, therefore difficult to remove in downstream barriers, i.e. C/F/S and filtration.

Increased NOM contents (TOC concentration) in raw waters will certainly increase the DBP formation if no adequate measures are adopted. This will require (USEPA 1999a):

i) adjusting the pre-oxidant dose or even the oxidant used (Figure 15) and
ii) improving the NOM removal in C/F/S prior to the final chlorination (USEPA 1999b) or using a final disinfectant of low NOM-reactivity, i.e. chlorine dioxide or chloramines (Figure 16).

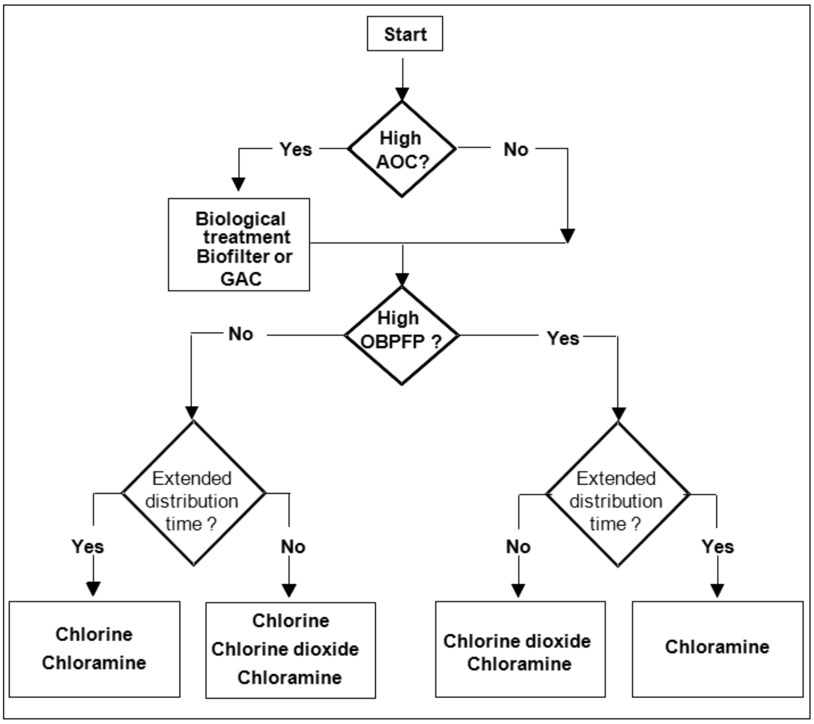


Figure 16 Algorithm for selecting the final disinfectant (USEPA 1999a)

High NOM contents will also increase the final disinfectant demand for controlling the microbial pathogens.

Intense rainfall events may increase ferrous iron and manganese contents in raw water which will increase the oxidant demand. However, if the stoichiometric demand is exceeded, undesired soluble oxidised species are produced, e.g. Mn^{+7} (pink water). In this case, strong oxidant (e.g. ozone) dosing should be reduced and $KMnO_4$ should be used as auxiliary oxidant. Another typical practice for controlling manganese peaks is to allow the sand filters to be covered by a layer of manganese dioxide, which adsorbs and self-catalysis the Mn^{2+} oxidation by the $KMnO_4$ or chlorine applied to the filters – manganese greensand filtration (AWWA 1999).

When toxic cyanobacterial blooms occur in the source water, the pre-oxidation processes should be well controlled since they can promote cyanobacterial cell rupture and subsequent release of cyanotoxin into water. The extracellular cyanotoxins are not easily controlled by conventional water treatment, and cyanobacteria cell lysis should thus be avoided before cell removal. As detailed in Chapter 4, some dissolved cyanotoxins may be destroyed by ozone, however the process efficiency depends on the ozone dose and water quality and impractical high doses would be required for the complete oxidation of the cyanotoxins. Removing the cyanobacteria prior to oxidation process is considered a safer practice.

Coagulation can be effective for NOM removal if performed at low pH (5-6), i.e. enhanced coagulation, which can be achieved by coagulant overdosing and/or pH adjustment. As detailed in Chapter 5, enhanced coagulation efficiency depends on the coagulant type and concentration, on the raw water inorganic matrix (pH, alkalinity, hardness) and on the mixing conditions (time and intensity). For instance, high alkalinity waters will require increased coagulant doses to lower the water pH (AWWA 1999, USEPA 1999b).

Turbidity removal is strongly affected by the water turbidity and alkalinity binomial (AWWA 1999) – high turbidity/low alkalinity waters are easily coagulated by adsorption/neutralization, high turbidity/high alkalinity waters demand higher coagulant doses, low turbidity/high alkalinity waters are sweep coagulated, whereas low turbidity/low alkalinity waters are resistant to coagulation and therefore require the previous addition of turbidity and/or alkalinity (often lime). In climate change scenarios, polyaluminium chlorides are preferable to alum ($Al_2(SO_4)_3.xH_2O$), as these coagulants show higher efficiencies and are less affected by temperature, NOM concentration and alkalinity variations.

Microcontaminants, as well as DBP precursors, may be controlled in conventional WTPs if C/F/S is assisted by PAC adsorption. The PAC doses often required (10-40 mg/L) make this option a good solution for controlling episodes, but limit it's technical-economic and environmental feasibility on a continuous basis.

Tailoring PAC addition is crucial to achieve high removal efficiencies. The physicochemical characteristics of PAC (particle size, porosity and surface chemistry), target pollutants (molecular weight, polarity, charge distribution) and water quality (pH, ionic strength, NOM content) are key variables for they determine the competitive adsorption kinetics and capacity (AWWA 1999, Chorus and Bartram 1999, Campinas and Rosa 2006, Costa 2010). Adsorption modelling is a useful tool to assist PAC selection and optimization (e.g. Viegas and Rosa 2012), but bench or pilot tests are often recommended to characterize the target compound(s) adsorption in the real water matrix.

Increments of temperature and turbidity (suspended solids, including microalgae and cyanobacteria) may promote high filter head losses, thus requiring increased frequency of filter backwash. Filter's shutdown and restart should be gradual to minimise particle breakthrough (Hall et al. 2005).

In some scenarios, e.g. during a cyanobacterial bloom, the recirculation of sludge supernatants must also be thoroughly assessed, and in some situations avoided, as it may constitute a severe input of refractory contaminants (dissolved cyanotoxins from cell lysis) which may compromise the treated water quality (Hall et al 2005).

7.4.2 *WTP upgrade with advanced or alternative treatments*

Advanced or alternative treatment processes for WTP upgrade include dissolved air flotation, alternative oxidants and advanced oxidation processes (AOP), activated carbon (bio)filtration (GAC/BAC), membrane pressure-driven processes (microfiltration – MF, ultrafiltration – UF, nanofiltration – NF and reverse osmosis – RO) and hybrid processes of adsorption and low-pressure membranes (e.g. PAC/UF or PAC/MF).

Dissolved air flotation (DAF) is very efficient for removing low-density particles, e.g. microalgae and cyanobacteria, protozoan (oo)cysts, and for treating NOM-rich waters. For these types of waters, it may thus be advantageous to replace C/F/S clarification by C/F/DAF.
For instance, dissolved air flotation (DAF) of flocculated waters (i.e. C/F/DAF) has proven to be very effective for treating cyanobacterial-rich waters, more than the conventional C/F/S (Ribau Teixeira and Rosa 2007, Ribau Teixeira et al. 2010) or the flock blanket clarification (98% vs. 77%; AWWA 1999).
Lab studies with *Microcystis aeruginosa* single cells (lab cultured, and ideal surrogate for assessing the removal efficiency of the particles of problematic size range (3-10 µm) of protozoa (Vlaski et al. 1996)) showed C/F/S and C/F/DAF were able to remove the cells with no microcystin increase in the treated water, and both were affected by the water NOM. However, C/F/DAF was less affected by the NOM concentration and type and showed higher removal efficiencies of *M. aeruginosa* cells (above 92% in terms of chlorophyll a) using more cost-effective operating conditions (mixing intensity and time, recirculation ratio) including lower coagulant dose (Ribau Teixeira and Rosa 2007).
A more recent study with single cells of *M. aeruginosa* and filaments of *Planktothrix rubescens* and four synthetic waters with different NOM contents (Ribau Teixeira et al. 2010) showed C/F/DAF high removal efficiencies of cyanobacterial cells and filaments (90-100% for intracellular microcystins and 92-99% for chlorophyll a) from waters with both hydrophobic and hydrophilic NOM provided the specific coagulant dose was ensured. No apparent cell damage and release of microcystins to water were observed with the operating conditions tested and the specific coagulant demand was severely affected by the DOC nature, hydrophobic DOC requiring ca. the triple of hydrophilic DOC, i.e. 0.7 vs. 0.2-0.3 mg Al_2O_3/mg DOC.

AOPs comprise of the formation of the hydroxyl radical, which is a strong oxidant (von Sonntag 2007). O_3-based AOPs include ozone at pH 8-10, ozone with UV radiation and/or H_2O_2, ozone with TiO_2 and/or H_2O_2. The addition of H_2O_2 to the existing ozone oxidation was, a few years ago, the more frequent and lower cost option for upgrading a WTP with an AOP (von Gunten 2003). Examples of other AOPs include UV with H_2O_2, an option

which has gained in popularity (Meunier et al. 2006, Mamane et al. 2007, Hofman-Caris and Beerendonk 2011), the Fenton process and catalytic oxidation. According to Von Gunten (2003).
Despite its high oxidation potential for microbial pathogens and micropollutants, several AOP side-aspects should be considered. In addition to increased chemicals and energy consumption, water pH and alkalinity play a key role in AOP performance, since bicarbonate and or carbonate ions compete for the hydroxyl radical, respectively in high alkalinity waters and at high pH values (USEPA 1999a). Also, AOPs are less effective than ozone for oxidizing ferrous iron and manganese (USEPA 1999a), do not ensure a disinfectant residual and may form DBPs, including bromate. Finally, NOM oxidation is not complete and yields low molecular weight biodegradable organic compounds (AOC) which support biological regrowth in distribution networks, besides being DBP precursors although less problematic than the hydrophobic fraction.
As for ozone oxidation, AOPs may be used for primary or secondary oxidation, i.e. before or after water clarification, the latter with lower THM potential formation due to previous partial removal of NOM.

AOPs increase water BDOC and AOC, and should therefore be complemented with downstream **biofiltration** (e.g. biologically active carbon filters – BAC) (Figure 15, 16). Accordingly, ozonation or AOP promotes the biological activity in **GAC filters**. While requiring proper management of biofilm development and activity, the adsorption-biodegradation synergy established in **BAC filters** improves process effectiveness, efficiency and reliability.

A recent lab study with NOM and microcystins (Mesquita 2012, Mesquita et al. 2012) allowed for concluding that the biological activity inevitably develops in GAC filters fed with biodegradable organics. The biological activity has a remarkable effect on NOM breakthrough due to a continuous bio-regeneration of the filter medium – BAC filters with 3-4 months operation presented similar efficiency to GAC filters with no biological activity and 7-8 days of operation. The biological activity contributes also to microcystin-LR removal (and most probably other biodegradable cyanotoxins and microcontaminants), provided that BDOC is feeding the filters. In addition, in BDOC absence episodes, the biological activity and microcystin biodegradation may be supported by the adsorbed BDOC. The oxygen and BDOC consumption rates increase with their supply rates and therefore relatively short empty bed contact times (EBCTs) (10 min.) favour the biological activity compared to longer EBCTs (15 to 20 min.). The biofilm decreases the carbon adsorption of microcystin-LR and NOM but it minimizes/prevents desorption and subsequent breakthrough of the pollutants accumulated in the filter, which otherwise would probably occur when their feed concentration decreases. This phenomenon and the biodegradation/bioregeneration ability make BAC a double barrier against microcystins and other microcontaminants in the treated water.

In a climate change scenario, when ozone-resistant, non-biodegradable and non-adsorbable pollutants are an issue, secondary ozonation or AOP) combined with downstream BAC filtration may constitute a permanent barrier against refractory pollutants, in addition to increased BDOC and AOC, cyanotoxins and other organic micropollutants (e.g. EDCs, pharmaceuticals). BAC/GAC filtration may also control inorganic pollutants by microbial/chemical reduction, e.g. bromate (Marhaba 2000, Rosa et al. 2009). Excessive biofilm development is BAC's major disadvantage since it would require improved washing cycles and may lead to microorganisms' release in the treated water.

UV radiation is also gaining interest in water treatment since it can be effective for destroying/inactivating biological forms resistant to chemical oxidation, including *Cryptosporidium* oocysts and *Giardia* cysts (USEPA 2006). However, the UV dose required for water disinfection depends on the nature of the microorganisms and on the water turbidity/transmittance. The most UV-resistant organisms are viruses, specifically Adenoviruses, and bacterial spores. The protozoon *Acanthamoeba* is also highly UV-resistant. Bacteria and (oo)cysts of *Cryptosporidium* and *Giardia* are more susceptible with a fluence requirement of 20 mJ/cm^2 for an inactivation credit of 3 log (Hijnen et al. 2006). To enable accurate assessment of the effective fluence in continuous flow UV systems in water treatment practice, biodosimetry is still essential, although the use of computational fluid dynamics improves the description of reactor hydraulics and fluence distribution. For UV systems that are primarily dedicated to inactivate the more sensitive pathogens (*Cryptosporidium*, *Giardia*, viruses), additional model organisms are needed to serve as biodosimeter. For turbid waters (> 10 NTU) UV disinfection is limited and for certain microorganims the required doses are very high.

Membrane technology includes a broad range of solutions for adapting the treatment systems to deal with the climate change driven risks.
While the low-pressure (0.1-1 bar) microfiltration membranes (0.1-2 μm) safely ensure turbidity and pathogens removal, including *Cryptosporidium* oocysts and *Giardia* cysts, UF (2-200 nm or 50-100 kDa MW cut-off; 0.5-5 bar trans membrane pressure) can also remove viruses and coagulated NOM (Ribau Teixeira et al. 2002).NF membranes are more selective (0.5-5 nm or 150-300 Da MW cut-off; 5-20 bar) and, in addition to effective removal of turbidity and pathogens, they present high removals of dissolved organics – NOM, micropollutants and DBPs above 150-300 Da, e.g. cyanotoxins (Ribau Teixeira and Rosa 2006, 2012), EDCs, pesticides, pharmaceuticals – and partial to high removal of inorganics (e.g. bromate, bromide, hardness ions, heavy metals, nitrate, salinity).Reverse osmosis is adequate for desalination and is widely used in water scarce regions, where brine disposal in the receiving waters (ocean) is not an issue.

Hybrid processes of **PAC/MF** or **PAC/UF** assure pathogen (protozoa, bacteria and viruses) and turbidity removal, and the control of NOM (coagulated and dissolved), DBPs and microcontaminants (Campinas and Rosa 2006). They integrate PAC ability for adsorbing coagulated and dissolved organics, including viruses and membrane fouling material (Campinas and Rosa 2010a), with the high particles' retention of the low-pressure UF and MF membranes (Campinas and Rosa 2010b, 2011). Major advantages of PAC/UF(MF) processes in a climate change context are related to: i) low unit energy consumption (due to the low trans membrane pressures applied and PAC ability for assisting the membrane fouling control); ii) small PAC particle size, which improves adsorption kinetics and, compared to conventional PAC addition to C/F/S, allows superior water quality with lower PAC consumption and sludge production (Campinas and Rosa 2010c); iii) process flexibility, by easily adjusting PAC type and dose to the target contaminant(s) and feed water quality.

The successful application of membrane technology relies on an effective membrane fouling control, which otherwise may prohibitively decrease the membrane flux and deteriorate its selectivity. It may be ensured through adequate pre-treatment (Ribau Teixeira and Rosa 2003), membrane cleaning, and operating conditions, which should allow maximum water recovery rates with minimal unit energy and chemicals' consumption.

Both conventional and advanced technologies, except those involving contaminant destruction, like chemical or biological oxidation, produce solid and liquid wastes with high contaminant concentrations. Adequate treatment and disposal of those wastes should be guaranteed.

Table 2 summarizes the ability of the alternative and advanced treatment options above discussed for adapting conventional WTPs to deal with the climate change driven challenges.

Table 2 - Effectiveness of alternative and advanced processes for macro and microcontaminant control in a climate change scenario (adapted from Rosa et al. 2009)

<table>
<tr><th>Contaminants</th><th>C/F+DAF</th><th>UV</th><th>GAC</th><th>BAC</th><th>MF</th><th>UF</th><th>PAC/UF</th><th>NF [a]</th><th>RO</th></tr>
<tr><td>Protozoa (cysts, oocysts)</td><td>+/-</td><td>+</td><td colspan="2">+/-</td><td>+</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Bacteria (vegetative forms)</td><td>-/+</td><td>+</td><td colspan="2">-</td><td>+</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Bacteria (endospores)</td><td>-/+</td><td>-</td><td colspan="2">-</td><td>+</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Helminth eggs</td><td>+/-</td><td>-</td><td colspan="2">+/-</td><td>+</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Cyanobacteria</td><td>+</td><td>- [b]</td><td colspan="2">+/-</td><td>+</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Enteroviruses</td><td>-</td><td>-</td><td colspan="2">+/-</td><td>+/-</td><td>+</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>NOM_SUVA < 3 L/(mgC·m)</td><td>-/+</td><td rowspan="2">+/-</td><td>-/+</td><td>+</td><td>-</td><td>-</td><td>+/-</td><td>+</td><td>+</td></tr>
<tr><td>NOM_SUVA > 4-5 L/(mgC·m)</td><td>+/-</td><td>+/-</td><td>+/-</td><td>-/+</td><td>+/-</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>AOC</td><td>-</td><td>+/-</td><td>-/+</td><td>+</td><td>-</td><td>-</td><td>-/+</td><td>+/-</td><td>+</td></tr>
<tr><td>THM</td><td>-</td><td></td><td colspan="2">+</td><td>-</td><td>-</td><td>+</td><td>+/-</td><td>+</td></tr>
<tr><td>HAA</td><td>-</td><td></td><td colspan="2">+</td><td>-</td><td>-</td><td>+</td><td>+/-</td><td>+</td></tr>
<tr><td>Bromate</td><td>-</td><td></td><td>+/-</td><td>+/-</td><td>-</td><td>-</td><td>-/+</td><td>+</td><td>+</td></tr>
<tr><td>Bromide</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+/-</td><td>+</td></tr>
<tr><td>Chlorate</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+</td><td>+</td></tr>
<tr><td>Chloride</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+/-</td><td>+</td></tr>
<tr><td>Nitrate</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+</td><td>+</td></tr>
<tr><td>Sodium</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+/-</td><td>+</td></tr>
<tr><td>Sulphate</td><td>-</td><td></td><td colspan="2"></td><td>-</td><td>-</td><td>-</td><td>+</td><td>+</td></tr>
<tr><td>Microcystins</td><td>+ or
-/+ [c]</td><td></td><td colspan="2">+ or
/- [d]</td><td>-</td><td>-</td><td>+ or
+/- [d]</td><td>+</td><td>+</td></tr>
<tr><td>T&O (MIB, geosmin)</td><td>-/+</td><td></td><td colspan="2">+</td><td>-</td><td>-</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>VOCs</td><td>+ [e]</td><td></td><td colspan="2">+</td><td>-</td><td>-</td><td>+/-</td><td>-/+</td><td>+</td></tr>
<tr><td>EDCs and pharmaceuticals
(hydrophobic and chemically resistant)</td><td>-/+</td><td></td><td colspan="2">+</td><td>-</td><td>-</td><td>+</td><td>+</td><td>+</td></tr>
<tr><td>Pesticides
(including chemically resistant)</td><td>-/+</td><td></td><td colspan="2">+ or +/- [d]</td><td>-</td><td>-</td><td>+</td><td>+</td><td>+</td></tr>
</table>

Sources: AWWA (1999), USEPA (1999a-b, 2005, 2006), Marhaba (2000), Huang et al. (2004), WHO (2004, 2011), Hall et al. (2005), Ribau Teixeira and Rosa (2003, 2006, 2007, 2010, 2012), Campinas and Rosa (2006, 2010a-c, 2011), Mesquita et al. (2006, 2012), Mesquita (2012)

-	Not adequate
-/+	Limited effectiveness
+/-	Partial control if adequate operation conditions are guaranteed
+	Effective provided adequate operation conditions are guaranteed
	No information available

[a] Considering 200 Da MW cut-off

[b] UV should not be used to control cyanobacteria, since it leads to cell rupture and cyanotoxin release

[c] Effective removal of intracellular toxins; no significant removal of dissolved toxins

[d] Depends on chemical characteristics of the target compound

[e] There are volatilization conditions in C/F/DAF

7.5 Preventive measures

Better than acting correctively is to prevent risks from occurring. This can be achieved by preventing hazard(s) and or minimizing their consequence(s) and frequency.

As far as climate change driven risks to drinking water treatment is concerned, preventive measures encompass the implementation of:

- global policies for minimizing the climate change drivers (i.e. for reducing the GHG emissions and increasing their sequestration), which will reduce the challenges to drinking water quality (detailed in section 7.1);
- management planning including:
  - pro-active measures for improving the reliability of the source water quantity and quality, e.g.:
    - efficient water use;
    - watershed pollution control (agricultural, industrial, domestic, stormwater);
    - proper water management, including the variation of the water abstraction depth to prevent/minimize scums and algal and cyanobacterial biomass from entering the WTP;
    - alternative water sources, e.g. water reuse, water transfer between supply systems;
  - pro-active measures for improving the detection of trigger factors of rapid changes that lead to risk (detailed in section 7.3);
  - pro-active measures for improving WTP ability for timely adaptation of the treatment process to changes in raw water quality, e.g.:
    - performance assessment and benchmarking treatment plants praxis;
    - water safety plans;
- contingency plans for extreme events (e.g. drought, intense rainfall, and pollution peaks).

7.6 Experiences and cases in Portugal

Portugal and the Algarve region in particular is a case study from which many lessons can be learned with regard to the adapted operation of drinking water systems to cope with climate change pressures, namely with severe droughts and intense rainfall events as illustrated herein.

Algarve, in south Portugal, with its sun and white sandy beaches, is a worldwide known summer destination (the biggest in Portugal) and one of the best golf destinations in the world, with 39 golf courses in operation, approved or under construction, and 24 planned (Freire 2010). Tourism is the most important economic activity in the region and drives a strong seasonal demand for water supply during the summer. The water consumption of the golf industry will soon be equivalent to approximately 200 thousand inhabitants.

Since the year 2000, the Algarve Multi-municipal System (currently covering 15 municipalities) has supplied water to approximately one million people in high season. Surface water sources include five dams, namely Odeleite -

Beliche, Funcho, Bravura, and very recently (since the year 2012) the main dam Odelouca. The system includes four WTPs – Alcantarilha (design capacity of 259 000 m³/day) and Fontaínhas (29 000 m³/day) in western Algarve; Tavira (190 000 m³/day) and Beliche (13 000 m³/day) in eastern Algarve – which were designed to treat surface water using conventional treatment of pre-oxidation (with ozone or chlorine dioxide in Fontaínhas WTP), remineralization (in eastern Algarve WTPs, whose raw waters are very soft, aggressive), coagulation with polyaluminum chloride and PAC addition (whenever necessary), flocculation, lamellar sedimentation, rapid sand filtration and final chlorination (Rosa et al. 2004a, Campinas and Rosa 2010c, 2011, Silva et al. 2012). To face the strong seasonal demand, the big plants include three or four treatment lines in parallel.

For instance, Alcantarilha WTP faced from its start-up a strong seasonal variation in raw water quality together with a seasonal water demand (ca. 180 000 people during winter and 650 000 people in summer, year 2002) (Rosa et al. 2004a). Continued monitoring showed that seasonal variations correspond to two major types of raw water quality: clear waters (1-6 NTU) and turbid waters (25-40 NTU, sometimes up to 1000 NTU) (Ribau Teixeira et al. 2002). Increases in turbidity usually occur after intense rainfall periods (e.g. during winter 2002/03 and autumn 2003, Figure 17) or are related with particle re-suspension in the WTP effluent main, due to the increase in flow rates to fulfil the water demand during the high season (e.g. in June-July 2003), and give rise to higher organic carbon contents (Ribau Teixeira et al. 2002, Ribau Teixeira and Rosa 2003, Rosa et al. 2004a).

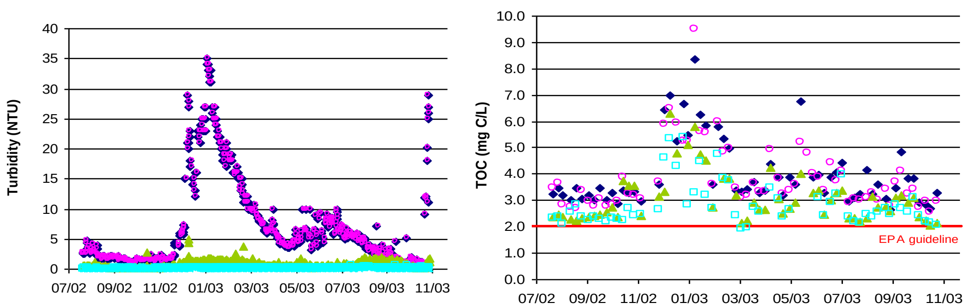


Figure 17- Turbidity (left) and TOC (right) values in Alcantarilha WTP RW (♦), OW (○), DW (▲) and TW (□), Aug. 2002-Oct. 2003 (Rosa et al. 2004a)

In 2002 a reinforced monitoring program was therefore implemented in Alcantarilha WTP (Figure 18) inspired on the HACCP (Hazard analysis and critical control points) principles of Codex Alimentarius. The plan aimed at assessing the levels of different contaminants and to establish trends; to identify and track the occurrence of new hazardous chemicals; to assess and optimize the WTP treatment performance and also to provide data to help future developments in drinking water quality standards. It provided the basis for the pioneer development of the water safety plan for the Algarve Multi-municipal System and for the ISO 22000:2005 (Food safety management systems) and ERP 5001:2007 (drinking water quality) certifications.

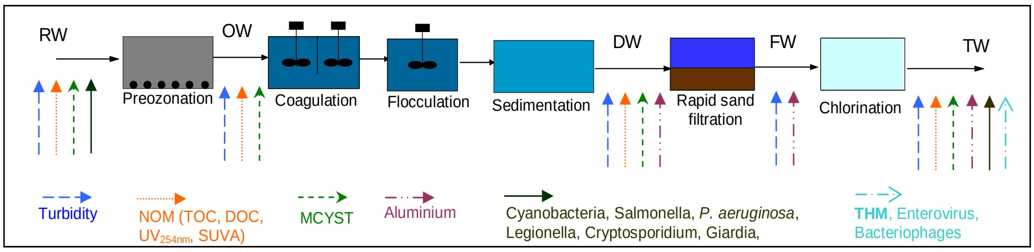


Figure 18 - Reinforced monitoring program at Alcantarilha WTP (Rosa et al. 2004a)

The monitoring program included legally established parameters and emerging contaminants of health and environmental concern yet not regulated at that time by the national or the European legislation, such as the waterborne disease organisms *Cryptosporidium*, *Giardia*, *Salmonella* spp, enterovirus, *Legionella* spp., and the indicator bacteriophages. Cyanobacteria and cyanotoxins (particularly the hepatotoxic microcystins, MCYST) were also included due to the historical record of cyanobacterial blooms in the Algarve surface waters. The DBPs THMs, which became regulated in December 2003, were also included (WHO 1993, 1996, 1998, 2002), as well as their precursor NOM (expressed as TOC, DOC, UV_{254nm} and SUVA).

Despite large fluctuations and high influent values in turbidity in wet months, the treated water presented very low and fairly constant turbidity values (an average of 0.12±0.05 NTU; 99% of the samples below 0.4 NTU), thus far below the national and EU standards for drinking water. This was possible due to an increased C/F/S and filtration removal efficiency of inlet turbidity (Figure 19). The 99% plateau was reached for inlet turbidity values above 9 NTU. Considering that these waters present a low to moderate alkalinity (50-75 mg/L as $CaCO_3$), the inlet turbidity increase improved colloidal matter removal by adsorption and charge neutralization mechanisms at low coagulant doses. High removals were also achieved with low turbidity waters since a polyaluminum chloride coagulant was used, whenever necessary with a flocculant aid.

TOC and turbidity seasonal patterns in raw water were quite similar, which emphasizes turbidity as a good and easy-to-assess indicator of water quality for operational adjustments of water treatment. However, since Alcantarilha conventional treatment was operated for clarification purposes rather than enhanced NOM removal, TOC removal efficiencies did not significantly increase with the inlet concentration (Rosa et al. 2004a), and thus treated water with lower quality was observed during the peak season (Figure 15).
Besides the hydrophilic nature and low MW of the ozonated water NOM (influent to C/F/S stages) (SUVA < 3 L/(mgC.m), Figure 18), which make colloidal suspensions hard to destabilize (Edzwald and Van Benschoten 1990), the high basicity of the PACl coagulant did not allow the water pH decrease required for enhanced coagulation to occur.

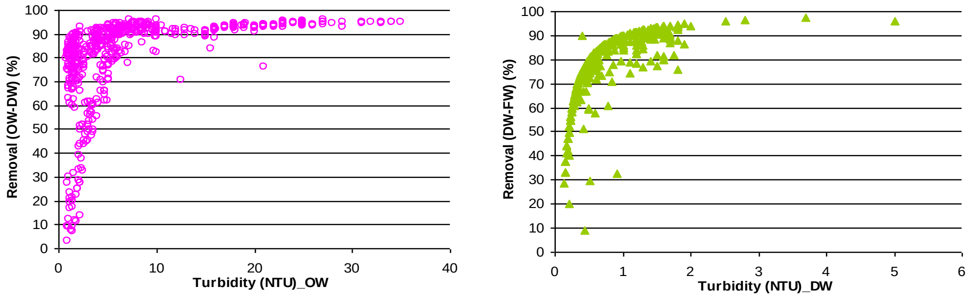


Figure 17- Turbidity removal in Alcantarilha WTP C/F/S (○) (left) and filtration (▲) (right) vs. inlet turbidity, Aug. 2002-Oct. 2003 (Rosa et al. 2004a)

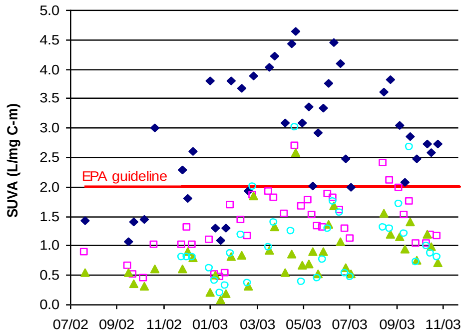


Figure 18 - SUVA values in Alcantarilha WTP RW (♦), OW (○), DW (▲) and TW (□), Aug. 2002-Oct. 2003 (Rosa et al. 2004a)

As expected, UV_{254nm} absorbing substances were easier to remove by C/F/S than overall TOC, and their removal by this process increased with the inlet concentration, whereas no such correlation was observed for ozonation (RW-OW) (Figure 19).

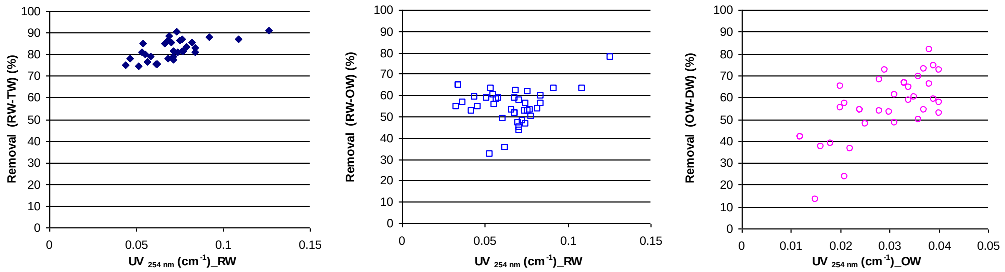


Figure 19 - Removal of UV_{254nm} absorbing substances in Alcantarilha WTP vs. inlet turbidity (Aug. 2002-Oct. 2003): overall removal (♦) (left), ozonation removal (□) (centre) and C/F/S removal (○) (right) (Rosa et al. 2004a)

According to USEPA (1998), the low SUVA values in the water prior to chlorination (< 2 L/(mgC.m) in decanted water, Figure 18) indicate a low THM formation potential (THMFP), since hydrophobic DOC has higher potential to form THM than hydrophilic DOC (Galapate et al. 2001). Indeed, total THM values in the treated water (14.2±2.5 µg/L) were always far below the national, European and USEPA standards (Figure 20). Despite the low THMFP, the total THMs in the treated water correlated with DOC concentration in decanted water (Figure).

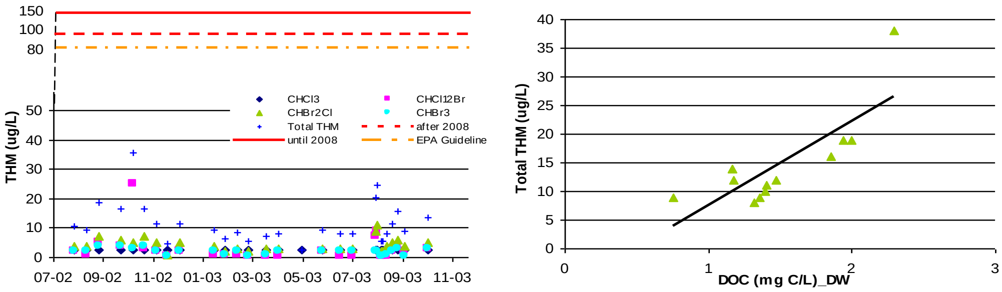


Figure 20 - Total THMs in Alcantarilha WTP RW (♦), OW (○), DW (▲) and TW (□) (left) and total THMs in treated water vs. DOC in decanted water (▲) (right), Aug. 2002-Oct. 2003 (Rosa et al. 2004a)

Cyanobacteria were detected in Funcho dam reservoir (source water) throughout most of the year (Rosa et al. 2004b), therefore reaching Alcantarilha WTP. In March 2003, during a surface bloom, cyanobacteria were not detected in Alcantarilha WTP raw water, indicating an adequate management of the water depth abstraction at Funcho Dam reservoir (Rosa et al. 2004a). Dissolved MCYST was never detected in the raw water, and intra-cellular MCYST was quantified only once and at very low concentration (October 2003). Cyanobacteria were always absent in the treated water and MCYST were systematically below the quantification limit of 0.014 ug/L, far below the WHO provisional guideline value of 1 ug MC-LR/L in drinking water.

Occasionally, the monitoring program implemented at Funcho Dam reservoir identified the presence of other cyanobacterial genera, e.g. *Aphanizomenon* and *Phlanktotrix*, which are potential producers of a potent neurotoxin, anatoxin-a. This toxin was included in the monitoring programs implemented for Funcho Dam reservoir and Alcantarilha WTP since April 2004.

During the same study (Rosa et al. 2004a), the pathogens *Salmonella* spp. and *Pseudomonas aeruginosa* were always fully removed in Alcantarilha WTP.

In the beginning of the high season (June 2003), when the temperature increased together with the drinking water demand, *Cryptosporidium*, *Giardia*, and *Legionella* were analysed in raw and in treated water, and enterovirus and bacteriophages in the treated water. None of them were detected in both waters. It was therefore not possible to check Alcantarilha WTP ability to remove these microbial contaminants, as well as the microcystins.

However, in addition to the referred assessment through this monitoring program, a WTP management protocol was developed including a 3-level strategy in terms of these hazards' removal to ensure a safe water supply:

1. Optimisation of WTP unit operations for removal of toxins and/or microorganisms - identification of the limiting steps (e.g. the recirculation of sludge treatment streams) and operating conditions (particularly, the type and dose of oxidants, coagulant, flocculant and PAC) (e.g. Rosa et al. 2009). These procedures were implemented in the WTP operation manual to guarantee their application whenever a bloom episode occurs. A performance assessment system was developed to promote the benchmarking of the water treatment, i.e. the continuous improvement of

its performance in terms of effectiveness and efficiency (Vieira et al. 2010, Silva et al. 2012).

2. Development of studies for WTP upgrade in case the monitoring program indicates limited results of step 1 strategy. In fact, one may expect limited performance of the conventional treatment with preozonation and PAC adsorption if the occurrence of the above referred hazardous substances becomes frequent. In this case, complementary and advanced technologies would become very attractive, such as: dissolved air flotation for cyanobacteria removal (Ribau Teixeira and Rosa 2007, Ribau Teixeira et al. 2010); ultrafiltration for particles removal, including bacteria and protozoan cysts, virus and cyanobacterial cells (Ribau Teixeira et al. 2004, Campinas and Rosa 2010b); and, for further removal of low MW organics, including toxins and THM precursors, PAC/ultrafiltration (Campinas and Rosa 2006, 2010a, 2010c, 2011), nanofiltration (Ribau Teixeira and Rosa 2006, 2012), GAC adsorption (Costa 2010) and GAC filters with controlled biological activity (BAC) (Mesquita et al. 2006, Mesquita 2012).
3. A contingency plan was developed with management procedures' application whenever the monitoring program indicates the treated water is not safe for human supply. This plan includes instructions to interrupt Alcantarilha's WTP production and to manage an alternative supply water system using water produced in other water treatment plants, which use different surface water and/or groundwater sources (Rosa et al. 2004a, Ribeiro et al. 2008).

In addition to strong seasonal variation of the raw water quality, episodes of severe water scarcity occurred in the region before the Odelouca dam became available in 2012 as the main raw water source of the Algarve Multi-municipal System.

During the hydrological year 2004/2005, a serious situation of drought occurred in Portugal (Figure 21) which led to the preparation of a contingency plan for the Algarve Multi-municipal System, in addition to the development of an action plan establishing measures and investments aimed at the reinforcement of Algarve's water supply capacity (Ribeiro et al. 2008). One of the most important measures consisted in establishing the interconnection between the two subsystems (eastern and western Algarve subsystems) so that the water may be imported from one subsystem to another. This allowed improving the overall resilience of the Algarve Multi-municipal System against source water scarcity and quality deterioration.

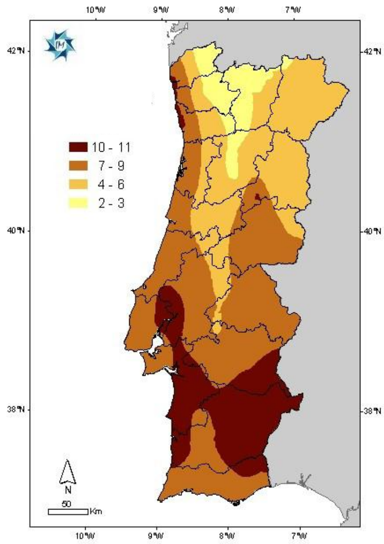


Figure 21 - Spatial distribution of consecutive months of severe to extreme drought in Portugal during the hydrological year 2004/2005 (MAOTDR et al. 2005)

As far as drought-related issues are concerned, many lessons can be learned from the years 2004/2005 experience in Algarve. The drought severely damaged the regularization of the flow rate in the region's reservoirs and increased the risk of seawater intrusion in the main aquifer system, due to the very low water levels observed and total absence of natural recharge (Ribeiro et al. 2008). Following the meetings of the national Drought Commission, Águas do Algarve, SA and the holding Águas de Portugal, SGPS prepared an action plan including immediate and short-term measures to mitigate the drought effects.
Immediate measures included, amongst others, water savings and reactivation of groundwater abstraction from municipal boreholes. Short-term measures included the reinforcement of the connection between the eastern and the western Algarve, and technical-economic studies on alternative water sources, i.e. treated wastewater use and water desalination.
As such options resulted as not economically attractive, it was decided to reinforce the connection between the two subsystems and go ahead with the Odelouca dam construction, the main water source originally planned (in the 90's) for the Algarve Multimunicipal System.

The immediate solution in western Algarve was to reactivate abstraction from the groundwater municipal boreholes (with one tenth the cost of water desalination). This required Alcantarilha WTP to adjust to different types of raw water – mixtures of surface and groundwater and only groundwater. Fortunately, unlike in many other aquifer systems located in agricultural areas, nitrate and pesticides were not an issue. On the other hand, while lower microbial loads and NOM concentration entered the plant and made the water treatment easier, the bromide concentration and the water turbidity/alkalinity challenged the pre-ozonation and the C/F/S steps, respectively, and required chemicals dosing fine-tuning for successful control of bromate formation and of residual turbidity and aluminium (Rosa et al. 2009, Vieira et al. 2010).

The assessment of Alcantarilha WTP performance, with respect to treated water turbidity during the years 2001-2007 (Vieira et al. 2010), showed few situations of unacceptable performance, mostly related with changes in raw water quantity and quality observed in the low to high season transitions and during the 2004-2005 drought. In this period, a lower performance was coincident with higher volumes of raw groundwater.

7.7 Tools, manuals, guidelines, documentation and references

Examples of tools available to assist the assessment of needs and for managing the adaptation of WTPs to cope with climate change are presented in Table 5.

Table 5 - Examples of relevant guidelines/manuals

Guidelines/Manuals	Description
ISO 22000 *Food Safety Management*. International Standardization Organization. Geneva	Standard for food safety management systems, incorporating HACCP and hygiene prerequisite principles.
Bartram J., Corrales L., Davison A., Deere D., Drury D., Gordon B., Howard G., Rinehold A., Stevens M. (2009) *Water safety plan manual: step-by-step risk management for drinking water suppliers*. World Health Organization. Geneva	Practical guidance to facilitate water safety plans development, focusing particularly on organized water supplies managed by a water utility or similar entity.
WHO (2009). World *Vision 2030: The resilience of water supply and sanitation in the face of climate change* Health Organization. Geneva	This publication refers to how anticipated climate change may affect drinking water services.
WASP7 - Water Quality Analysis Simulation Program	This model helps users interpret and predict water quality responses to natural phenomena and manmade pollution for various pollution management decisions. WASP is a dynamic compartment-modelling program for aquatic systems, including both the water column and the underlying benthos.
Sinisi L., Aertgeerts R. (2011) *Guidance on water supply and sanitation in extreme weather events*. World Health Organization. Geneva	This publication describes how adaptation policies should consider the new risks from extreme weather events, how vulnerabilities can be identified and which management procedures can be applied to ensure sustained protection of health.

7.8 References

AWWA (1999) *Water quality and treatment – A handbook of community water supplies*. 5th ed. American Water Works Association. McGraw-Hill. New York.

Bartram J., Corrales L., Davison A., Deere D., Drury D., Gordon B., Howard G., Rinehold A., Stevens M. (2009) *Water safety plan manual: step-by-step risk management for drinking water suppliers*. World Health Organization. Geneva

Beniston M., Stephenson D.B., Christensen O.B., Ferro C.A.T. et al. (2007) Future extreme events in European climate: an exploration of regional climate model projections. *Climatic Change* 81, 71–95

Campinas M., Rosa M.J. (2006) The ionic strength effect on microcystin and natural organic matter surrogate adsorption onto PAC. *Journal of Colloid and Interface Science* 299(2) 520-529

Campinas M., Rosa M.J. (2010a) Assessing PAC contribution to the NOM fouling control in PAC/UF systems. *Water Research* 44(5) 1636-1644

Campinas M., Rosa M.J. (2010b) Evaluation of cyanobacterial cells removal and lysis by ultrafiltration. *Separation and Purification Technology* 70(3) 345-353

Campinas M., Rosa M.J. (2010c) Comparing PAC/UF and conventional clarification with PAC for removing microcystins from natural waters. *Desalination and Water Treatment* 16, 120-128

Campinas M., Rosa M.J. (2011) PAC/UF for removing cyanobacterial cells and toxins from drinking water. In *Expanding Issues in Desalination*. ed. Robert Y. Ning, InTech. Rijeka, Croatia. 233-252

Charron D.F., Thomas M.K., Waltner-Toews D, Aramini JJ, Edge T, Kent RA, et al. (2004) Vulnerability of waterborne diseases to climate change in Canada: a review. *Journal of Toxicology and Environmental Health Part A* 67, 1667–1677

Chorus, I., Bartram, J. (eds) *Toxic Cyanobacteria in Water: A guide to their public health consequences, monitoring and management*. World Health Organization. Geneva

Costa H. (2010) *Activated carbon adsorption of cyanotoxins from natural waters*. PhD Thesis. Universidade do Algarve, Faro

Delpla I., Jung A.V., Baures E., Clement M., Thomas O. (2009) Impacts of climate change on surface water quality in relation to drinking water production. *Environment International* 35, 1225-1233.

Edzwald J.K., Van Benschoten J.B. (1990) Aluminium coagulation of natural organic matter. In *Chemical Water and Wastewater Treatment*. H.H Hahn and R. Klute (Eds.). Springer-Verlag, Berlin. 341-359

ERP 5001:2007 *Especificação de Requisitos do Produto: Água para Consumo Humano. Variante sistemas de abastecimento público em alta*. APCER - Associação Portuguesa de Certificação (in Portuguese)

Evans C.D., Monteith D.T., Cooper D.M. (2005) Long-term increases in surface water dissolved organic carbon: observations, possible causes and environmental impacts. *Environmental Pollution* 137, 55-71

Fellman J.B., Hood E., Edwards R.T., D'Amore D.V. (2009) Changes in the concentration, biodegradability, and fluorescent properties of dissolved organic matter during stormflows in coastal temperate watersheds. *Journal of Geophysical Research* 114 (G01021) 14 pp.

Figueras M.J., Borrego J.J. (2010) New perspectives in monitoring drinking water microbial quality. *International Journal of Environmental Research and Public Health* 7, 4179-4202

Freire J. (2010) Plans for bulk supply of TWW for golf course and landscape irrigation in the Algarve region. In *Workshop on treated wastewater use in Portugal*. Lisbon, Portugal, Oct.

Galapate R.P., Aloysius U.B., Okada M. (2001) Transformation of dissolved organic carbon matter during ozonation: effects on trihalomethane formation potential. *Water Research* 35(9) 2201-2206

Hall T., Schmidt W., Codd G.A., von Gunten U., Kaas H., Acero J., Heijman B., Meriluoto J., Rosa M.J., Manckiewicz J. et al. (2005) *Best practice guidance for management of cyanotoxins in water supplies*. "TOXIC" EVK1-2002-00107. EU 5FP project, Key Action "Sustainable Management and Quality of Water"

Hammes F., Egli T. (2005) New method for assimilable organic carbon determination using flow-cytometric enumeration and a natural microbial consortium as inoculum. *Environmental Science and Technology* 39, 3289-3294

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

Hofman-Caris C.H.M, Beerendonk E.F. (Eds) (2011) *New concepts of UV/H_2O_2 oxidation*. Project report. BTO 2011.046. KWR – Watercycle Research Institute. WaterRF - Water Research Foundation

Huang W., Chen C., Peng, M.Y. (2004) Adsorption/reduction of bromate from drinking water using GAC: effects on carbon characteristics and long-term pilot study. *Water SA* 30, 369-375

Hurst A.M., Edwards M.J., Chipps M., Jefferson B., Parsons S.A. (2004) The impact of rainstorm events on coagulation and clarifier performance in potable water treatment. *Science of the Total Environment* 321, 219–230

Kaste O., Wright R.F., Barkved L.J., Bjerkeng B., Engen-Skaugen T., Magnusson J., Saelthun, N.R. (2006) Linked models to assess the impacts of climate change on nitrogen in a Norwegian river basin and fjord system. *Science of the Total Environment* 365, 200–222

Komatsu E., Fukushima T., Harasawa H. (2007) A modeling approach to forecast the effect of long-term climate change on lake water quality. *Ecological Modelling* 209, 351–366

LeChevallier M.W., Shaw N.E., Kaplan L.A. Bowt T.L. (1993) Development of a rapid assimilable organic carbon method for water. *Applied and Environmental Microbiology* 59(5) 1526-1531

Mamane H., Shemer H., Linden K. G. (2007) Inactivation of *E. coli*, B. *subtilis* spores, and MS2, T4, and T7 phage using UV/H_2O_2 advanced oxidation. *Journal of Hazardous Materials* 146, 479-486

MAOTDR, MCTES, MADRP AND MAI (2005) *Drought in Continental Portugal. 31st December 2005*. Inter-ministerial technical group final report to the Ministries Council. RCM 83/2005, art.º 8. Ministry of Environment, Regional Policies and Planning; Ministry of Science, Technology and Higher Education; Ministry of Agriculture, Rural Development and Fisheries; Ministry of Interior. Lisbon, Portugal (in Portuguese).

Marhaba, T. (2000) Examining bromate ion removal by GAC through RSSCT and pilot scale columns. *Environmental Engineering and Policy* 2, 59-64

Meunier L., Canonica S., von Gunten U. (2006) Implications of sequencial use of UV and ozone for drinking water quality. *Water Research* 40, 1864-1876

Nechad B., Ruddick K.G., Park Y. (2010) Calibration and validation of a generic multisensor algorithm for mapping of total suspended matter in turbid waters. *Remote Sensing of Environment* 114(4) 854-866

Mesquita E., Menaia J., Rosa M.J., Costa V. (2006) Microcystin-LR removal by bench scale biological-activated-carbon filters. In *Recent Progress in Slow Sand and Alternative Biological Filtration Processes*. IWA Publishing, London. 373-383

Mesquita E. (2012) *Remoção de cianotoxinas da água para consumo humano em filtros de carvão activado com actividade biológica.* Tese de Doutoramento. Universidade do Algarve, Faro (in Portuguese)

Mesquita E., Menaia J, Rosa M.J. (2012) Remoção de microcistina-LR e de matéria orgânica natural por filtros de carvão ativado. In *Proc. 15.º ENaSB*. Évora, Portugal. October (in Portuguese)

Paerl H.W., Paul V.J. (2012) Climate change: Links to global expansion of harmful cyanobacteria. *Water Research* 46(5) 1349-1363

Pednekar AM, Grant SB, Jeong Y, Poon Y, Oancea C. (2005) Influence of climate change, tidal mixing, and watershed urbanization on historical water quality in Newport Bay, a saltwater wetland and tidal embayment in southern California. *Environmental Science & Technology* 39, 9071–9082

Reichwaldt E.S., Ghadouani A. (2012) Effects of rainfall patterns on toxic cyanobacterial blooms in a changing climate: Between simplistic scenarios and complex dynamics. *Water Research* 46, 1372-1393

Rietveld L.C., van der Helm A.W.C., van Schagen K.M., van der Aa L.T.J. (2010) Good modelling practice in drinking water treatment, applied to Weesperkarspel plant of Waternet. *Environmental Modelling & Software* 25, 661-669

Ribau Teixeira M., Lucas H., Rosa M.J. (2002) The role of pH on the ultrafiltration for drinking water production in Algarve (Portugal). *Water Science & Technology: Water Supply* 2(5-6) 367-371

Ribau Teixeira M., Rosa M.J. (2003) pH adjustment for seasonal control of UF fouling by natural waters. *Desalination* 151(2) 165-175

Ribau Teixeira M., Rosa M.J. (2006) Neurotoxic and hepatotoxic cyanotoxins removal by nanofiltration. *Water Research* 40(15) 2837-2846

Ribau Teixeira M., Rosa M.J. (2007) Comparing dissolved air flotation and conventional sedimentation to remove cyanobacterial cells of *Microcystis aeruginosa*. Part II. The effect of water background organics. *Separation and Purification Technology* 53(1) 126-134

Ribau Teixeira M., Sousa V., Rosa M.J. (2010) Investigating dissolved air flotation performance with cyanobacterial cells and filaments. *Water Research* 44(11) 3337-3344

Ribau Teixeira M., Rosa M.J. (2012) How does the adsorption of microcystins and anatoxin-a on nanofiltration membranes depend on their co-existence and on the water background matrix. *Water Science and Technology* 66(5) 976-982

Ribeiro A., Lucas H., Sousa J, Coelho R., Viriato M., Dias S. (2008) Infrastructure strategic management in contingency situations. *Water Asset Management International* 4(3) 17-19

Roig B., Delpla I., Baurès E., Jung A.V., Thomas O. (2011) Analytical issues in monitoring drinking-water contamination related to short-term, heavy rainfall events. *Trends in Analytical Chemistry* 30(8) 1243-1251

Rosa M.J., Cecílio T., Ribau Teixeira M., Viriato M., Coelho R., Lucas H. (2004a) Monitoring of hazardous substances at Alcantarilha's water treatment plant, Portugal. *Water Science & Technology: Water Supply* 4(5) 343-353

Rosa M.J., Cecílio T., Costa H. Baptista R., Lourenço D. (2004b) Monitoring of microcystins at Funcho dam reservoir. In *4th IWA World Water Congress*. Marraquexe, Morocco, Sept.

Rosa M. J., Vieira P., Menaia, J. (2009). *O tratamento de água para consumo humano face à qualidade da água de origem*. Série Guias Técnicos, Vol. 13. Instituto Regulador de Águas e Resíduos and Laboratório Nacional de Engenharia Civil. Lisbon, Portugal. Sept. ISBN 978-989-95392-7-3 (in Portuguese)

Rosario-Ortiz F.L., Snyder S.A., Suffet I.H. (2007) Characterization of dissolved organic matter in drinking water sources impacted by multiple tributaries. *Water Research* 41, 4115–4128

Schmidt W., Bornmann K., Imhof L., Mankiewicz J., Izydorczyk K. (2008) Assessing drinking water treatment systems for safety against cyanotoxin breakthrough using maximum tolerable values. *Environmental Toxicology* 23(3) 337-345

Schmidt W., Petzoldt H., Bornmann K., Imhof L., Moldaenke, C. (2009) Use of cyanopigment determination as an indicator of cyanotoxins in drinking water. *Water Science & Technology*, 59.8, 1531-1540

Silva C., Ramalho P., Quadros S., Alegre H., Rosa M.J. (2012) Results of 'PASt21' – the Portuguese initiative for performance assessment of water and wastewater treatment plants. *Water Science & Technology: Water Supply* 12(3) 372-386

Soh Y.C., Roddick F., van Leeuwen J. (2008) The future of water in Australia: the potential effects of climate change and ozone depletion on Australian water quality, quantity and treatability. *Environmentalist* 28, 158–65

USEPA (1998) *40 CFR Parts 9, 141, and 142 National primary drinking water regulations: disinfectants and disinfection byproducts; final rule*. US Environmental Protection Agency. Federal register/Vol. 63, No 241/Wednesday, Dec. 16/Rules and Regulations. 69390- 69476

USEPA (1999a) *Alternative disinfectants and oxidants guidance manual*. US Environmental Protection Agency. Office of Water. EPA 815-R-99-014

USEPA (1999b) *Enhanced coagulation and enhanced precipitative softening guidance manual*. US Environmental Protection Agency. Office of Water. EPA 815-R-99-012

USEPA (2005) *Technologies and costs document for the final long term 2 enhanced surface water treatment rule and final stage 2 disinfectants and disinfection byproducts rule*. US Environmental Protection Agency. Office of Water. EPA 815-R-05-013

USEPA (2006) *UV Disinfection Guidance Manual for the Final Long Term 2 Enhanced Surface Water Treatment Rule.* US Environmental Protection Agency. Office of Water. EPA 815-R-06-007van Leeuwen J., Daly R., Holmes M. (2005) Modeling the treatment of drinking water to maximize dissolved organic matter removal and minimize disinfection by-product formation. *Desalination* 176 (1–3) 81-89

Viegas R.M.F., Rosa M.J. (2012) Modelação da adsorção de ibuprofeno com carvão ativado no tratamento avançado de água. In *Proc. 15.º ENaSB*. Évora, Portugal. October (in Portuguese)

Vieira P., Rosa M.J., Alegre H., Lucas H. (2010) Assessing the operational performance of water treatment plants – Focus on water quality and removal efficiency. In *IWA World Water Congress*. Montreal, Canada. Sept.

von Gunten U. (2003) Ozonation of drinking water: Part I. Oxidation kinetics and product formation. *Water Research* 37, 1443-1467

von Sonntag C. (2007) The basic oxidants in water treatment. Part A: OH radical reactions. *Water Science and Technology* 55, 19-23

WHO (1993) *Guidelines for Drinking Water Quality. Volume 1 – Recommendations*. 2nd ed. World Health Organization. Geneva

WHO (1996) *Guidelines for Drinking Water Quality. Volume 2 – Health Criteria and other supporting information*. 2nd ed. World Health Organization. Geneva

WHO (1998) *Guidelines for Drinking Water Quality. Addendum to Volume 1 – Recommendations*. 2nd ed. World Health Organization. Geneva

WHO (2002) *Guidelines for Drinking Water Quality. Addendum Microbiological agents in drinking water*. 2nd ed. World Health Organization. Geneva

WHO (2004) *Water Treatment and Pathogen Control*. M. W. LeChevallier and K.K. Au (Eds.). World Health Organization. Geneva

WHO (2011) *Guidelines for drinking water quality*. 4th ed. World Health Organization. Geneva

8 Conclusions and outlook

8.1 Conclusions

The examples show that the expected outcomes of climate change will vary per region, but also per system. Since climate change effects vary and interact differently it is not possible to predict exactly which effects will occur where. By taking the right measures, water supply companies can be prepared to timely detect the risks that occur and to take action. The time scale for occurring risks and required response can vary greatly. For example NOM increase will occur gradually over the years, allowing companies to study the most appropriate response. Cyanobacterial blooms can occur within a few days and although this can be somewhat predicted by weather forecasts and local conditions, these are not very exact. Blooms may occur where they were absent in previous years. Prepared water supply companies can then take action both to reduce the risk and to effectively communicate with the public. Sudden peaks of microbial contamination can occur suddenly, for example after an extreme rainfall event. Technical measures and response protocols need to be readily available within a few hours and these procedures need to be tested to protect public health.

The key characteristic of climate change related risks is that new risk may occur that operators are not familiar with in their situation. This report provides a starting point for the water supply companies to learn from other situations and start preparing for what may happen.

8.2 Outlook

The examples in this report are based on incidents, tests and studies. It is important the water utilities keep good record of the occurrence of risk events and which measures were effective under which conditions. Especially with climate change the different regions can learn from each other. For example, droughts that are already common in some regions may be new to other regions or NOM problems may have already been dealt with in parts of the Atlantic region. By sharing the successes, but also the failures and mistakes, the water supply industry can learn as a whole and implement measures more efficiently and effectively.