Organic matter in water pdf

I'm not robot	reCAPTCHA		
Continue			

```
Covers conventional as well as advanced methods of removal NOM includes methods of characteristic METHOD Explains removal methods, such as: removal methods of characteristics methods of characteristics methods of characteristics methods of characteristics methods.
Explains removal methods, such as: removal by coagulation, electrochemical, advanced oxidation, and integrated methods you currently do not have access to this book However, you can purchase individual chapters directly from the content table or buy the full version. Purchase a book of 1 Separate 50 milliliters (0.21 c) of the sample in a test tube. You
don't need a whole sample of water to determine the demand for chemical oxygen. Set the rest of the sample aside in case you want to use it for additional tests. You can also check multiple 50ml corks (1.8 imp fl oz; 1.7 fl oz) samples simultaneously and average results for greater accuracy. COD is the overall demand for chemical oxygen and it can show
you how much oxygen is used in your body of water. 2 Prepare a test tube with 50 milliliters (0.21 c) of mineral water in most grocery
 stores. You have to take the same steps to the sample and mineral water to have a comparison for the final equation. 3 Add 1 g magnesium sulfate to the sample and tube of mineral water. Don't load up and make sure you don't overload any tube. Leave the
mixture on the counter for 5 minutes to dissolve the solution. You have to go slowly so that magnesium sulfate can be absorbed into the water. 4 Add in 1 g silver sulfate and a few glass beads. Heat 3 to 4 glass beads. Heat 3 to 4 glass beads. Heat 3 to 4 glass beads at 600 degrees Fahrenheit (316 degrees Celsius) for 1 hour. Add 2 beads to each tube as you add 1 g of silver sulfate in each tube. Glass
beads give the liquid in each tube a large area of surface so that they can cool and heat up quickly. 5 Cool the sample in ice water, then add 75 ml of sulphuric acid. Dunk pipes into the icy water quickly, be careful not to get water inside the pipes. Leave them there for 3-5 minutes until they have cooled to room temperature and then add to the sulfuric acid.
Use caution as you put in sulphuric acid, and try not to get it on your skin. If you get sulfuric acid on your skin, rinse it with cool water right away. 6 Add in 25 ml dichromata Use a clean pipette to dump potassium diffromat into each test tube. You don't need to go slow, so you can put in all 25 ml at once. Your sample can change colors while doing so That's
fine. 7 Place the tubes on the capacitor and leave for 2 hours. The capacitor cools the hot gas in the liquid. Put all your test tubes on the capacitor for too long, it can distort your results. 8 Add mineral water to each tube and then cool
them to room temperature. Turn off the capacitor and let them cool enough so you can touch them, or about 2 minutes. Add enough mineral water to each test tubes on the counter for 10-15 minutes until they have cooled. If your test tubes are too small, transfer the liquid to a large
glass container. 9 Titrat sample using the Ferroin indicator. Use a pipette to slowly add a Ferroin 1 drop solution at a time. Keep an eye on how many drops you use in each of them, because you will use that number in your final calculation
10 Stop the titration when the color turns reddish-brown. The goal is to turn your sample from green/blue to red/brown. As soon as the sample changes, stop the caption and write down how many droplets of the indicator you have used. Tip: The color change will happen very quickly, so it won't be hard to detect. 11 Identify the COD by comparing the tube
sample to mineral water. First, subtract the amount in the ml of the solution indicator that you used on the actual sample from the indicator solution you're working on. Multiply this amount by 8,000 and then divide the number by 50 milliliters (0.21
c) to get the amount of chemical demand for oxygen in parts by a million, or mg/liter. For example, if you used 4 ml in the actual sample and 20 ml in mineral water, subtract 20 to 4 to get 16. Then multiply this by 0.1 to get 1.6. Multiply 1.6 x 8000 to get 12,800 and then divide that into 50 ml (your sample size) to get 246 mg/l. 12 Target for COD from 10 to 30
mg/L for an uncontaminated body of water. If the COD is between 10 and 30 mg/L, it is not contaminated. If the COD is between 25 and 50 mg/L, it is sewage. If you are concerned about the amount of COD in the water, contact a wastewater management specialist. Health Canada Ottawa, OntarioJuli 2020
Canada is working with provinces, territories and federal agencies to establish quality guidelines Canada. Over the years, new methodologies and approaches have led Health Canada, in collaboration with the Federal-Provincial Territorial Committee on Drinking Water, to develop a new type of document, the document, provide advice and advice on drinking
water quality on parameters that do not require a formal Guide to Canadian drinking water quality. The guidelines are set in accordance with the Drinking Water Quality Guidelines in Canada specifically for pollutants that meet all the following criteria: If a pollutant of interest does not meet all of these criteria, Health Canada, in collaboration with the Federal
Provincial Territorial Committee on Drinking Water, may decide not to develop a Technical Document. In this case, a guide document may be developed. The guidelines go through a similar process to the Guidelines, including public consultation through the Health Canada website. They are offered as information to drinking water authorities and to assist in
providing guidance in the event of a spill or other emergency. Natural organic matter (NOM) is an extremely complex mixture of organic compounds and is found in all groundwater and surface waters. Although NOM has no direct impact on health, it affects the efficiency of drinking water treatment processes and, consequently, the safety of drinking water.
NOM can also affect consumer satisfaction as it can contribute to unwanted color, tastes and odors in drinking water treatment. This guide document was prepared in collaboration with the Federal Provincial Territorial Committee on Drinking
Water and reviews and evaluates the risks associated with the impact of NOM on drinking water treatment to protect drinking water treatment to protect drinking water treatment to protect drinking water treatment and drinking water treatment to protect drinking water treatment to protect drinking water treatment and drinking water treatment to protect drinking water treatment to protect drinking water treatment and drinking water treatment to protect drinking water treatment to protect drinking water treatment to protect drinking water treatment and drinking water treatment and drinking water treatment and drinking water treatment to protect drinking water treatment to protect drinking water treatment and drinking water treatment an
of disinfection sub-snoughs, and promote the development of biofilms in the distribution system. Its presence can also create conditions that increase the concentration of lead and/or copper in treated water due to its effect on corrosion. NOM's controllability and reactivity vary greatly in Canada, as each source it has unique features. Because NOM consists
of numerous organic compounds, it cannot be measured directly. However, there are a number of other parameters that can be used to indicate the concentration and nature (i.e. chemical, physical and biodegradable properties) of NOM. It is important to understand the differences between the concentrations and nature of the NOM in order to designing and
operating appropriate water treatment processes. Currently, there is no practical health-based value for MLM in drinking water. The development of an effective nom control strategy should be based on good understanding: specific treatment option
(s) to remove NOM, reduce its reactivity to form disinfection by-products, reduce its potential to promote corrosion and produce biologically stable water for distribution. Lack of treatment study specific to specific to specific to specific to reatment or other
unintended consequences. Since water sources or treatment processes may change over time, it is important to monitor the concentration and distribution systems. The purpose of this document is to provide advice to provinces, territories, other government departments and
stakeholders (such as water system owners, consultants, equipment suppliers and laboratories) on the impact of NM on the safety of drinking water. It summarizes the factors that influence the concentration and nature of NOM and
discusses the points to consider in the development of the NOM management strategy. It also provides specific recommendations on water purification, monitoring and quality goals. The guidelines, standards and/or recommendations on water purification, monitoring and quality goals. The guidelines, standards and/or recommendations of other national and international organizations on drinking water may vary depending on the date of the assessments, as
well as on different strategies and approaches. International organizations have not set quantitative limits for NM in drinking water. The United States Environmental Protection Agency's rule on disinfectants and sub-susion disinfectant requires the removal of total organic carbon (TSS) by surface water facilities using conventional or lime-softened water
treatment levels above 2 mg/L in their water source. The World Health Organization proposes to optimize the removal of NOM as a means of minimizing the growth of biofilm in the distribution system. European Union rules include TSEs as an overall indicator of water quality; in some jurisdictions, the demand for chemical oxygen (COD) may be used instead
of TOC. In Australia, a guide has been developed for water utilities to help them understand and impact of NM. Natural Organic Matter (NOM) is an extremely complex mixture of organic compounds that vary greatly in terms of their physical and chemical characteristics. NOM occurs naturally in the environment and can also be the result of human activity.
NOM is in colloidal and dissolved forms in all groundwater and surface waters, as well as in rainwater. Although the environmental impact of NOM is common and does not involve direct health effects, the presence and characteristics of NMD will have significant implications for drinking water treatment processes aimed at protecting public health. NOM plays
a crucial role in the treatment of drinking water for a number of reasons. First of all, NOM can indirectly contribute to health effects in many ways, including: Water canals can also be significantly involved in a number of problems caused by NOM, namely: NOM can also lead to an increase in consumer complaints as it can contribute to undesirable color,
tastes and odors in drinking water. These and other issues are further discussed in subsequent sections of the document. All water safety or water safety plan approach, to ensure water safety. These approaches require a systemic assessment, which includes the characteristics of the water
source, the description of therapeutic barriers that prevent or reduce pollution, the allocation of conditions that can lead to pollution, and the definition of controls. Operational monitoring is then established and operational/management protocols (e.g. standard operational procedures, corrective actions and incident response) are established. Compliance is
monitored and other protocols are implemented to verify the water safety plan (e.g. accounting, consumer satisfaction). Operators are also currently required to ensure the effectiveness of the water safety plan. In developing and implementing a risk management approach, it is important to understand how NOM can indirectly lead to health effects. NOM can
increase until the cloudiness and flow changes and can remain elevated after the turbidity and flow have returned to baseline conditions. Thus, changes in the NOM strategy should be to protect against microbial risks
at any time, while minimizing DBP, lead and copper concentrations and monitoring the formation of biofilm in the distribution system. The water supply may require multiple cleaning processes to effectively balance microbial and
chemical risks throughout the year. Water system owners should contact the relevant drinking water authority in the affected jurisdiction to confirm whether specific applied to their source/system. To determine the most effective treatment option (s) for adequate adequate adequate treatment NOM and to achieve water quality goals related to microbial risk,
DBP, biological stability and corrosion control. A robust, reliable and sustainable clean-up strategy requires a robust, reliable and sustainable clean-up strategy. Monitoring specific to specific to specific sources is needed to assess the seasonal variations of NMMs and to predict extreme conditions due to climate change. The study of curability should include bench
and/or experimental scale testing, as well as potential DBP formation tests that are representative of the distribution system. The concentration and distribution system of the water supply in order to provide a comprehensive monitoring plan and include the characteristics of sources,
operational and compliance monitoring; it must also demonstrate that water quality goals have been consistently achieved to combat microbial risks, DBPs, biological stability and corrosion. Ideally, continuous online monitoring should be used for highly variable sources (i.e. those that fluctuate with precipitation/snow melting events) and critical processes
(e.g. coagulation). Assessment of water sources should be part of normal system estimates. These should include an understanding of the sources of NOM in the watershed/aquifer, conditions that lead to changes in the concentration and/or nature of nom (e.g., precipitation/melting events, algae blooms, drought, fire), and factors that increase the reactivity
of NOM to form DBPs (e.g. reactionary conditions, water age and inorganic compounds such as ammonia, Surface and underground sources should be characterized for NOM and inorganic compounds. variability in treatment and distribution. Constant rapid monitoring and optimization of cleaning will help ensure that water utilities achieve water quality
goals and ensure maximum protection of public health in the full range of water quality conditions. Maintaining modern knowledge about best practices and maintaining knowledge of drinking water advances are important aspects of the approach to water safety planning from source to tap or water safety. organic matter (BOM) stimulates the growth of biofilm
in the distribution system. Distribution can provide a habitat for the survival of pathogens that may have passed through drinking water purification barriers or entered the distribution systems are the maintenance of residual disinfectant, the
limitation of BOM and the control of corrosion. Maintaining the physical/hydraulic integrity of the distribution system and minimizing negative or low-pressure events are other key components of the approach to the safety plan from source to tap or water. Water quality in the distribution system should be regularly monitored, including DBPs and biological
stability indicators (e.g., variability in disinfectant residues, biofilm formation rate, corrosion rate). Operating/maintenance programmes should also be developed (e.g. water age control, water treatment, cross-connection control, asset management) and strict hygiene during the construction of all water facilities (e.g. repairs, maintenance, new installations) to
ensure that drinking water is transported to the consumer with minimal quality. Table 1 outlines the proposed parameters (e.g. disinfectant residues, DBPs) are already monitored in most treatment plants as part of the approach to
the production of safe drinking water. Other parameters are relatively easy to use (e.g. absorption or transmission of UV radiation) and provide quick results. The proposed water quality targets are set out in Table 2. They are offered as a guide only on the basis of a literature review that has been completed to develop the document. Since some water
sources can be extremely reactive (e.g., forming more DBPs), stricter water quality targets may be required. Water channels should use the most appropriate methods and parameters to regularly monitor water quality in the raw water system, process and distribute water, establish baseline conditions and identify changes that require a change in the
process. Low-concentration DBD systems have stable biological water quality (e.g. biostabelability) and baseline data indicating that NOM does not affect corrosion may consider reducing monitoring. NOM is an extremely complex blend of organic compounds that vary in polarity, acidity, charge density and molecular weight; NOM can also range from
biodegradable (i.e. genital or semi-lab) to less biodegradable (i.e. recalcitrant or fireproof). Because NOM contains numerous compounds, it can be classified by its polarity (i.e. hydrophobic or hydro
identified. Complex classes provide the highest level of specificity, specificity, specificity, compounds that may be present (Minor et al., 2014). The size and shape of NOM depends on the pH and low ion strength, it can have a flexible linear
form of thread (Ghosh and Schnitzer, 1980; Brachetta et al, 1997). Some compounds have both hydrophobic, hydrophobic, hydrophobic, hydrophobic, hydrophobic, hydrophobic, le. amfotheric) (Ghosh and Schnitzer, 1980; Brachetta et al., 1997; Her et al., 2007; Amy, 2008). Fractions containing
polysaccharides, proteins and amino acids have the highest molecular weight (10 kDs), while the molecular weight fractions. The
smallest fractions of NOM (0.5 kDa) are usually hydrophilic compounds (Sillanp et al., 2015a). The most biodegradable fractions include lylines, tannins and terpenoids. Concentration and character (i.e. chemical, physical and biodegradable properties) of NOM can be highly variable
due to the numerous hydrological and biogeochemical processes affecting the sources of NOM (Aiken and Cotsaris, 1995). This is summarized in the following sections, along with parameters that have historically been used to quantify organic matter, including: 1) organic color as a measure of humini and fulvic acids; and 2) Organic carbon, a key
component of NOM (Thurman, 1984). Other parameters that can be used to measure and characterize NOM are discussed later in this paper. There are two natural sources of NOM: acochtonous (i.e. derived from the earth's ecosystem) and autochton (i.e. derived from plants and microorganisms growing in the reservoir) (Aiken and Cotsaris, 1995).
Anthropogenic (human) activity can also contribute to NOM. Allochthonous NOM is exported to the aquatic environment as precipitation moves through layers and penetrates organic soil (Aitkenhead-Peterson et al., 2003). Soil, plant debris, microbial biomass and root exudates
contribute to alcochtonic NOM (Kalbitz et al., 2000). Allochthonous NOM tends to be hydrophobic in nature. These and other factors influencing the concentration and nature of the allochthonous NOM are described in table C-3.1 of this paper. Autochton NOM comes from phytoplankton, algae, cyanobacteria and macrophytes (i.e. plants, to the substrate of
lakes or rooted in the and can share 5-100% of the concentration of DOC, depending on certain conditions (Bertilsson and Jones, 2003; Bade et al., 2007; Tomlinson et al., 2016). When all-ton inputs are high, for example in colored water sources or during precipitation/snow events (i.e. storm flow conditions), the proportion of autochton NOMs
is generally low. Conversely, when all-ton inputs are low, for example, in clean water sources or during dry periods when runoff is low, the proportion of autochton NOMs is generally high. Autochthonous NOM includes a wide range of compounds: mono and polysaccharides, amino acids, peptides, proteins, nucleic acids, organic acids, lipids and fatty acids
(Pivokonsky et al., 2006; Henderson et al., 2008). DOC is generated as a result of the production and decomposition of microbial and plant biomass in water sources (Nguyen et al., 2014). Algae tend to dominate large lakes, while macrophytes tend to be the main source in small lakes (Wetzel, 1992; Bertilsson and Jones, 2003). Algae and
cyanobacterial flowers, in particular, are the source of DOC, which can be periodic and intense. Cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential presence of cyanobacterial flowering may be associated with additional water quality problems due to the potential flowering may be associated with a flowering may be associated wit
wastewater treatment and stormwater discharges, agricultural runoff and industrial discharges. Anthropogenic NOM is reportedly hydrophilic in nature (Imai et al., 2001) and rich in nitrogen (Dotson and Westerhoff, 2009; Mitch et al., 2009). Watersheds strongly affected by human sources may observe a decrease in CTO or MLC after the improvement of
wastewater or stormwater treatment (Reckhow et al., 2007). NOM crude water concentrations represent a net effect of hydrological and biogeochemical processes in the watershed or aquifer (Eckhardt and Moore, 1990). The concentration and nature of THE NOM, and therefore its curability (i.e. the potential for removal) and reactivity (i.e. the potential for the
formation of DBPs), vary greatly from one source to another, as each water source has unique characteristics. For example, kerekes et al. (1982) reported that in two lakes in Nova Scotia, only 1 km from each other, the concentration of TOC is 5.6 and 17.2 mg/l, respectively. Longer retention times tend to reduce the concentration of DOC, as noted in table
C-3.1. However Curtis and Adams (1995) it was reported that the evacuation of fire-resistant NOM had led to an increase in MLC concentrations, with increased retention time in Alberta's sub-wet and semi-arid areas. Other researchers r
each other (Aiken and Cotsaris, 1995; Ogren, etc., 2007; Reckhow et al., 2007; Goss and Gorchik, 2013; Kent et al., 2014). NOM concentration of microbes when transported through soil (Thurman, 1985; Aiken and Cotsaris, 1995; Aitkenhead-Peterson
et al., 2003). However, these processes are limited by the amount of biodegradable NOM that is present. Conversely, some groundwater passes through organic-rich aquifers, resulting in high organic carbon concentrations (Thorstenson et al., 1979; Hem, 1985; Aravena et al., 1995; Lemieux et al., 2019). According to published sources, organic carbon
concentrations in some North American groundwater range from 0.1-22 mg/L (see table C-3.2). Seasonal variability may occur on the basis of each of them, so using one sample to represent groundwater tends to be more
hydrophilic and recalcitrant in nature (Diem et al., 2013) and is almost as reactive as surface water NOM-based mg/l DOC (Owen et al., 1995; Reckhow et al., 2017) reported specific DBP harvests of 67 micrograms of trigalomatetan
(THMs)/mg DOC and 29 grams of haloacicic acids (HAAs)/mg DOC for small groundwater reserves using chlorination. Groundwater sources may have higher concentrations of bromide (Chowdhury, 2018) or iodine (Lemieux et al., 2019), which may contribute to their reactivity (i.e. the potential for accident formation). Thus, localized conditions play a very
important role in establishing the concentration and character of NOM (Bourbonniere, 1989; Mulholland, 2003; Reckhow et al., 2007; Sillanpe, 2015). The incident data below highlights the variability that can occur, with or without a related change in THE concentration of DOC. The data also show that while groundwater tends to have lower concentrations of
NMD, some sources may have elevated concentrations. Thus, both surface and underground sources should be described. Concentration and character (i.e. chemical, physical and biogeochemical processes that export, generate or humiliate NOM as described in
table C-3.1.Monitoring data that were available for this review, below (see tables 4 and 5). Data not detected by detection were excluded from the statistical analysis. The results are presented to show the variability of detectable concentrations that can occur spatially. Monitoring data from out and the territories for TSS and the colors in the raw water are
summarized in Table 4. These data indicate that TSS concentrations and organic color vary spatially and tend to be lower in groundwater (although some groundwater sources may have elevated concentrations). Table 5 presents DOC monitoring data collected in 2009 and 2010 from selected drinking water sources in each region of Canada (Health
Canada, 2016). These data also show lower concentrations of MLC in groundwater with minimal variations between the concentration of raw and treated water. For the surface water surveyed, the average treated water of DOC ranges from 3.2-3.4 mg/L in summer and 2.8-3.5 mg/l in winter. Because some jurisdictions are not represented in Table 4, Table C-
3.3 provides long-term DOC surface water monitoring data in Canada (2000-2015) for individual regions or river basins across Canada (Environment Canada, 2017). These data also show a significant variability in NOM concentrations spatially. A number of studies have characterized six factions of NOM (see table 3) found in several Canadian waters
(Montreuil, 2011; Department of Environment and Environment an
concentrations remained low with minimal changes. Goss and Gorchin (2013) studied the river spring in Manitoba, and observed significant temporal variability in six factions of NOM, but with fluctuations in DOC concentrations. Two other studies show that the nature of NOM can vary greatly depending on location. The results of these fractionation studies
demonstrate the variability that may occur in the noM-c nature or without associated changes in DOC concentration. In addition, the results show that hydrophilic neutral fractions may, from time to time, make up a significant portion of NOM. This proportion can be particularly problematic, as discussed in subsequent sections. NOM can also be crushed using
faster methods or depending on its size or fluorescence, as indicated in section B.5.2.4. The biodegradability properties of NOM are discussed in detail in subsequent sections. Environmental factors can alter the concentration of NOM are discussed in detail in subsequent sections.
can affect water sources and water purification processes, as mentioned below. A number of researchers reported an increase in the concentration of NOM and a change in its nature after the snow melted, runoff or torrential rains (Gregory, 1998; Billicka and Gertig, 2000; Tseng et al., 2000; Goslan et al., 2002; Volk et al., 2002; Eikebrokk et al., 2004;
Fearing, etc. etc. 2004a, 2004b; Hurst et al., 2004; Chow et al., 2006; Sharp et al., 2006; Parsons et al., 2016; Lames et al., 2016; Lames et al., 2016; Lames et al., 2016; Carpenter et al., 2016; Lames et al., 2016; Carpenter et al., 2016; Carpenter et al., 2016; Lames et al., 2016; 
NOM in the reservoir (Thurman, 1985; Saraceno et al., 2009). The highest concentrations can occur in summer and autumn, when temperatures are warmer, biological activity is high, and high-intensity/short-lasting process for a
number of reasons. First, changes in water quality occur in storm flow conditions that create challenging coagulation conditions (e.g., pH, alkalinity, ionical force, divalent concentration of ions) (Gregory, 1998; Billicka and Gertig, 2000; Tseng et al,2000; Davis and Edwards, 2014). Second, it was noted that NM increases to changes in turbidity or current and
may remain elevated after the turbidity and flow return to baseline conditions (Soulsby, 1995; Hurst et al., 2004; McVicar et al., 2015; James et al., 2015; James et al., 2016). Thus, if the dose of coagulant is controlled on the basis of flow or turbidity, the coagulant may not be sufficiently dosed, leading to suboptimal coagulation conditions (Hurst et al., 2004; Parsons et al., 2007;
Kundert et al., 2014; McVicar et al., 2015; James et al., 2015; James et al., 2016). It is well known that suboptimal coagulation conditions lead to a significant deterioration in the removal of pathogenic journals (Ongerth and Pecoraro, 1995; Patania, et al., 1995; Edzwald and Kelly, 1998; Coffey et al., 1999; Emelko et al., 1999, 2005; Dugan et al., 2001; Harrington et al., 2001; Harrington et al., 2001;
Huck et al., 2001, 2002; Dai and Hozalski, 2002; Betancourt and Rose, 2004; Hendrix et al., 2005; O'Melia, 2006; Hiinen and Medema, 2007). Rains in winter or spring may be particularly worrying, as low temperatures can reduce the efficiency of the clotting process (Hurst et al., 2004). Over the past few decades, there has been an increase in MLC
concentrations in Canada (Eimers et al., 2008; Keller et al., 2008; Couture et al., 2012; Chowdhury, 2018), North America (Driscoll et al., 2009; Northern Europe (Eikebrokk et al., 2004) and Japan (Imai et al., 2001). In areas where DOC has increased
waters also often become more colored (Ekstr'm et al., 2011; Kritsanne and Extrem, 2012; Weichenmeier et al., 2014). Pagano et al. (2014) conducted a comprehensive review of the growing trends of DOC and noted that researchers suggest a reduction in the deposition of atmospheric acid (i.e. sulphur control) and climate change agents as two key Monteit
et al., (2007) (2007) data for North America, the United Kingdom and Europe have reported that the decrease in acid deposition is due to 85% of the growing trends of the DOC, with the exception of the United Kingdom and Newfoundland. In these regions, the increase in sea salt deposition is due to a decrease in DOC in some areas. The authors found no
trends between DOC and rising temperature or concentration of carbon dioxide in the atmosphere. Extrem et al. (2011) conducted site acidification experiments and confirmed that reducing acid deposition leads to an increase in THE concentration of DOC and color, which means increased mobility of NOM through sulphur emission control. As for organic
color, this trend is related to the combination of iron with DOC (Weyhenmeyer et al., 2014). However, the mechanisms are not fully understood. Black and Christman (1963) also found that iron was always present with an organic color, but no link can be established between iron content and color. Increasing DOC and/or color concentrations can have a
significant impact on water channels through clotting and filtration processes. Anderson et al. (2017) reported a fourfold increase in alum dose (i.e. 12.9 to 49.5 mg/L) and a 1.75-fold increase in lime use at a full-scale facility where the true color increased from about 20 in 1990 to about 50 in 2015. The authors also reported that the hydraulic capacity of the
plant was reduced by 26%. Parsons et al. (2007) reported that the average dose of coagulant in full-scale institutions in the United Kingdom increase in colour. Eikebrokk et al. (2004) conducted experimental studies and reported that an increase in color in waters with low
turbidity by 75% (It; 0.3 units of nephelometric muteness (NTU)) increased the dose of coagulant, sludge production, the number of spin washes and residual tots by 64%, 64%, 87% and 10% respectively. The authors also predict an increase in the
consumption of chemicals to adjust pH and an increase in biological growth in the distribution system due to higher residual organic carbon concentrations. Other researchers noted that higher residual organic carbon concentrations. Other researchers noted that higher residual organic carbon concentrations. Other researchers noted that higher residual organic carbon concentrations.
15 years (2000-2015) for 304 surface water and 137 groundwater sources in Newfoundland/Labrador and observed growing trends for DOC, organic color and trigalometans THMs.Researchers have also found that forest fires can lead to long-term (10 years) water quality degradation, which significantly changes and the nature of NOM, and thus significantly
affects the water treatment processes (Emelko et al., 2011a; Geng et al., 2011; Emelco, 2019). Wildfires are forecast to be a wildfire frequency as a result of climate change and exacerbate the impacts of NMD include: rising water
temperatures; Increased runoff variability and increased nutrient loads due to extreme runoff events (van der Linden et al., 2018). The increase in the frequency and severity of algae growth and cyanobacterium blooms is projected to be associated with these changes (van der Linden et al., 2018). A comprehensive review of the expected impacts of climate
change on the curability of NOM can be found elsewhere (Ritson et al., 2014). Although NOM has no direct health impact, it has a critical impact on drinking water treatment and can contribute to indirect health effects as well as operational and
aesthetic issues. Drinking water purification usually includes physical removal barriers (i.e. primary disinfection) that are prescribed by the journal Inactivation Credits. The NOM has a critical effect on both sides. For chemically auxiliary refinement
processes, NOM has a coagulant requirement that needs to be overcome before the formation of neutrally charged flok particles. Neutrally charged flok particles are necessary to ensure that the filters are appropriate and meet the requirements of murky turbidity for the removal of pathogens (Hall and Packham, 1965; Semmans and Field, 1980; Dempsey et
al., 1984; Edwards and Amirtaraj, 1985; Amy et al., 1989; Edzwald and Van Benshoten, 1990; White et al., 1997; Sheen et al., 2008). NOM concentrations of NOM necessitates an increase without altering turbidity or flow and therefore may go unnoticed. However, elevated concentrations of NOM necessitates an increase without altering turbidity or flow and therefore may go unnoticed. However, elevated concentrations of NOM necessitates an increase without altering turbidity or flow and therefore may go unnoticed.
particles; otherwise there are conditions of suboptimal coagulation and there may be a loss in the removal of the pathogenic journal. James et al (2016) reported a breakthrough of Cryptosporidium and an increase in the number of particles (2-5 microns, 5-10 microns and 10-15 microns) at a full-scale direct filtration station as a result of increased color in the
water source. Other researchers have reported particle breakouts in the 2 microns on an experimental scale during periods of increased TSS (Billica and Gertig, 2000; Carlson and Gregory, 2000). A number of other studies (Ongerth and Pecoraro, 1995; Patania et al., 1995; Dugan et al., 2001; Huck et al., 2001) report that the removal of cryptosporidium by
refinement/filtering can deteriorate significantly under suboptimal coagulant conditions (e.g. decreased by 2.0-3.4 logs compared to optimal conditions). It is well known that NOM is in demand for chemical oxidizers (i.e. chlorine ozone), which must be overcome before the requirements for inactivation of pathogenic journals are met (AWWA,
2011a; MWH, 2012). Grunet et al., (2018) and L'ziart et al., (2018) and L'z
recommended. L'ziart et al. (2019) found that the organic turbidity of 1 NTU, generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic neutage generated by 2 mg/l of gum acids, inhibits the effectiveness of chlorine, while 5 NTU inorganic 
ensure effective inactivation of pathogens and to minimize the formation of DBP (see section B.4.1.2). Additional information on how NOM affects the demand for chemical oxidizers, decay and disinfection is published elsewhere (Health Canada, 2009a, 2018). Several studies have examined the effect of particles on the effectiveness of UV disinfection, and
most of them have concluded that UV-dose-response of microorganisms does not depend on fluctuations in turbidity up to 10 NTU (Christensen and Linden, 2004; Passantino et al., 2004). However, the presence of humic acid particles and coagulants has been shown to significantly affect the
effectiveness of UV disinfection, with lower levels of inactivation being achieved. Templeton et al. (2005, 2007) found that in unfiltered effects samples (range No. 4.4-9.4 NTU) UV disinfection of bacteriophages in the presence of folic acid floxes was reduced by a significant statistical degree (<0.5 logs) compared to water-free. Templeton et al. (2005) also
found that UV-absorbing organic particles (i.e. NOM) protected particles associated with bacteriophage from ultraviolet light, while inorganic particles (e.g., size, structure, chemical composition), number of large particles (e.g. ≥25 microns), the
level of aggregation of pathogens by particulate matter and the desired level of inactivation than on the level of turbidity (Caron et al., 2007; Hargi and Landry, 2007; Templeton et al., 2008; Kollu and Ermepi, 2012). In addition, UV transmission at a wavelength of 254 nm depends on dissolved and particulate matter, which prevents UV radiation from
penetrating through water. In general, each reduction in UV transmission by 10% leads to a 50% reduction in the dose of UV radiation (Hofmann, 2008). Thus, to meet the requirements for inactivation of the water pathogen with UV transmission by 85% more reactors than 95% UV-transmission water. Manufacturers usually specify a minimum UV under
which the system will not function properly. Pre-treatment for NOM removal may be necessary to meet the manufacturer's specification for minimal UV transmission, in order to meet the requirements for pathogenic logging inactivation and to ensure that the equipment works safely. It is important that water channels understand the pathogenic reductions in
logs that can be achieved when working in optimal conditions, and the impact of short- and long-term treatment disorders (Hurst et al., 2004). The application of the reliability index proposed by Huck et al. (2001) provides a simple and practical means of identifying events and periods when clotting and refinement processes become unstable. Information on
how to use a quantitative microbial risk assessment to assess the reliability of physical barriers to removal and/or inactivation is also available (Health Canada, 2019a). Chemical disinfectants react with NOM to the formation of potentially dangerous DBPs (Rook, 1974; Stevens et al., 1976). After years of research, it is generally accepted that all nom factions
contribute to the formation of DBP, although some factions form more accidents than others (Hoehn et al., 1993; Owen et al., 1993; Owen et al., 2014; Hua et al., 2014; Hua et al., 2014). It is also recognized that some unregulated accidents are
more cytotoxic and genotoxic than regulated accidents such as THMs and HAAs (Stalter et al., 2016; Wagner and Pleva, 2017). While THMs and HAAs can be used as indicators of the presence of other DBPs (Health Canada, 2006, 2008a), it is important to recognize that their ways of forming and speed of reaction are different (Reckhow and Singer, 1984;
Liang and Singer, 2003; Hua and Rekhow, 2007; Bond et al., 2012). For example, Plourde-Lescelleur et al. (2015) reported a potential THM formation ratio of 0.66-3.35 for six Canadian surface water sources (test conditions - pH 8.0, residual chlorine 1.0 mg/L at 22 degrees Celsius for 24 hours). Archer and Singer (2006a) reported that as hydrophilic
fractions increased, THMs were predominantly produced relative to HAAs. Thus, THMs and HAAs should be managed accordingly. Other factors influencing the formation of DBP include water temperature, pH, disinfectant conditions (i.e. disinfectant, dose, contact time, residual) and the presence of reactive species such as bromide, iodide, ammonia and
```

```
sulfur (Liang and Singer, 2003; Ates et al., 2007; Hua and Rekhow, 2007b; Reckhow et al., 2007; Fabris et al., 2007; Fabris et al., 2008; Christiana et al., 2009). It has been shown that bromide, in particular, increases the education rates of DBP three to ten times (Symons et al., 1993; Westerhoff, etc., 2004; Heeb et al., 2014). As a result, similar concentrations of DOC can
produce a wide range of DBP, depending on the nature and reactivity of NOM (Fabris et al., 2008). For example, Hua and Rekhou (2007a) reported that raw water reactivity for Winnipeg, Manitoba, (DOC 7.9 mg/L; bromide 46 micrograms/L). In
some cases, NOM is rich in nitrogen and promotes the formation of nitrogen DBPs (N-DBPs) (Leenheer and Croue, 2003; Mitch et al., 2009). Rain phenomena can also transport DOC, which is rich in precursors, leading to increased concentrations of DBP (Goslan et al., 2002; Fearing et al., 2004a, 2004b; Reckhow et al., 2007; Kraus et al., 2010; Delpla and
Rodriguez, 2016; Wright et al., 2016). Bond et al. (2011, 2012) completed a comprehensive literature review and summarized the potential for various components of THE NOM to form THM, HAAs and N-DBPs as indicated in Table 6. Reckhow et al. (2007) reported that tannin and lignin are significant precursors, representing 60% and 25-30% of THM
formation capacity, respectively. The authors also hypothesized that proteins may be important precursors in the early stages of NOM formation, but they will biodegrade (see section B.4.1.3) and terpenoids will take their place as another important source of THM precursors. It is important that water channels understand the source of NOM's specific
reactivity when choosing a disinfectant to mitigate the formation of potentially dangerous DBPs (Hua and Reckhow, 2007a). Determining a specific yield of DBP (i.e. ICP/mg DOC) can help, since the removal of DOC is generally easier to achieve than reduced reactivity, especially with present bromide (Crou' et al., 1993; Goslan et al., 2002; Reckhow et al.,
2007; Fabris et al., 2008; Tubich et al., 2013). Different mitigating measures may be required to minimize one group of DBPs compared to another; technologies that focus on removing specific factions of NOM (Bond et al., 2011) may be required. In addition, Kastl et al. (2016) reported that the requirements for the removal of NOM should be linked to the
terms of the distribution system. For example, a distribution system with a living time of 7 days and a temperature of 15 degrees Celsius (Rodriguez and Serodes, 2001; Kastl et al., 2016). Research, including potential
DBP development tests, should be carried out in assessing various mitigating measures and/or alternative treatment options. In order to ensure the implementation of an effective noM control strategy, a study of the suitability for assessing seasonal variations of NOM: 1) should be developed to assess seasonal variations in NOM; and 2) to be a
representative of the terms of the distribution system. It is important to note that the formation of potential testing methods that use very high doses of chlorine may not be correct differences in DBP DBP bromide (Bond et al., 2014). This is because chlorine can out-of-competition bromine when it is in excess relative to bromine. In typical operating conditions,
bromine is much more efficient than chlorine (Bond et al., 2014). Standard methods are available to assess the potential of DBP formation (APHA et al., 2017). The use of uniform formation conditions (Summers et al., 1996) allows a direct comparison of results to assess the effectiveness of different treatment options (AWWA, 2011a). Practical advice on
assisting water utilities in conducting potential DBP trials (Alexander et al., 2019). The technical guidelines for THM and HAAs state that water utilities should make every effort to maintain concentrations as far as possible. Thus, the removal of NMD is the recommended best practice to minimize the formation of both regulated and unregulated DBPs (Health
Canada, 2006; 2008a). This may require specific monitoring to ensure that precursors are disposed of adequately (see table 1 and 2 in section A.2.3). It is essential that the efforts made to minimize the education of DBP do not compromise the effectiveness of disinfection. More information can be obtained from the relevant technical documents of the guide
on TM, HAA, chlorite/chlorate, N-nitrozodichylamine and bromuata (Health Canada, 2006, 2008a, 2008b, 2011, 2018). The biological stability of water from the point of consumption (Prest et al., 2016). Heterotrophic organisms make up the majority of
bacteria in drinking water and draw their energy for the growth, multiplication and production of biofilm matrix materials as a result of the degradation of organic carbon compounds (Vu et al., 2009; Prest et al., 2016). BOM encourages the growth of bacteria and the development of biofilm in the distribution and plumbing system, which can lead to problems
that are important for public health. Biofilms provide a habitat for the survival of fecal pathogens that may have passed through integrity violations (Leclerc, 2003). It has been shown that enteric viruses and protozoa can be found in biofilms (Howe et al., 2002; Le Chevalier,
2003; Chang and Jun, 2004); although these organisms cannot grow in this environment, they can accumulate and release over a long period of time (Howe et al., 2002; Warneke, 2006; Wingender and Flemming, 2011). In addition, opportunistic pathogens of the plumbing room (OPPPs), such as Legionella pneumophila and non-tubular mycobacteria (e.g.
M. avium, M. intracellulare) have adapted for cultivation and are preserved in and the plumbing biofilm systems (Wilczak, 2006). While biofilm-microorganisms use the components with the shortest half-resolution period of biodegradation first,
they skillfully consume all types of available NOM components, leaving a treatment center to support their growth in the distribution system (Fischer, 2003; Camper, 2014). Table 7 summarizes the half-publishing period of biodegradation for NOM components (Reckhow et al., 2007). Treatment processes also have a significant impact on the composition and
concentration of organic nutrients. For example, oxidizers such as chlorine and ozone produce biodegradable products when reacting with NOM (Alarcon-Herrera et al., 1993; Bersill, 2001; Rek bow, et al., 2007). It is well known that ozone converts NOM into BOM (Owen et al., 1993, 1995); thus, biologically active filtration may be required to stabilize treated
water (GLUMRB, 2012). Chlorine can also respond to organic matter, thereby increasing the amount of organic carbon absorbed (AOC) and biodegradable DOC (BDOC), which can exacerbate the problem of biofilm growth in distribution systems (Drikas et al., 2003; Reckhow et al., 2007; Croft, 2012; Camper, 2014). Limited publications measured this
aspect, and the numerical values reported ranged from site to site (Camper, 2014). LeChevallier et al. (1992) observed an increase in AOC by 20% and 44% when samples of granular activated carbon (GAC)-sand wastewater filter were treated with 1 mg/l free chlorine or 2 mg/l monochloramine for 30 minutes, respectively. The company studied the
microbiological quality of water and pipeline deposits in 16 full-scale distribution systems in Finland. The average rate of bacterial biomass production was higher in the water and pipeline deposits in 16 full-scale distribution systems were affected by
chlorine oxidation (increase in AOC) and bacterial activity (reducing AOC). Different patterns were observed in different distribution systems and by season. In one system, while the following spring there was a 35% decrease in AOC
concentrations in the same areas (Liu et al., 2002). Minimum residues of disinfectants are generally recommended in drinking water supplies in North America to control the growth of biofilm (LeChevallier et al., 1996; LeChevallier and Au, 2004). In some European countries, the approach adopted to achieve biological stability is to reduce the nutrients that
support growth in water (Lautenschlager et al., 2013). In the Netherlands, companies water supply, seek to limit regrowth in distributed water in the absence of residue of disinfectants by way of (10 micrograms/l) (Lautenschlager et al., 2013). BOM levels (e.g. AOC, BDOC) are an important factor in drinking water biostability, but are only one of the
components that influence changes in water quality (Prest et al., 2016). Other compounds that have a role in controlling the growth of microbes in the distribution system have been identified: phosphorus, ammonia, manganese, iron and gum (Camper, 2004; Prest et al., 2016). The biostabelability of distributed water is influenced by numerous factors: the
type and concentration of organic and inorganic nutrients, the type and concentration of residual disinfectant, biofilm and sediment, as well as the conditions, pipe material and diameter, pH, corrosion rate) (LeChevallier et al.; Prest et al.,
2016). Comprehensive reviews of biological stability can be found elsewhere (Pr'vost et al., 2015; van der Kooij and van der Wielen, 2014; LeChevallier et al., 2015b; Prest et al., 2015b; Prest et al., 2016). In general, strategies to improve biostapleability and minimize the development of biofilm in the distribution and plumbing system include optimized NOM removal,
maintenance of effective disinfectant residue, maintenance of low levels of biostaple capacity in treated water (e.g., AOC, BDOC, biofilm formation of storage; There are additional recommendations to monitor the biological stability of drinking water
distribution systems (Health Canada, in preparation). Corrosion is a deterioration of the material that is the result of a reaction with the environment. Corrosion in drinking water distribution systems can be caused by several factors, including the type of materials used in pipes and fittings, the age of pipelines and fittings, the time of water stagnation and the
quality of water in the system (including its pH and alkalinity). Other drinking water quality options that can affect corrosion include temperature, calcium, free residue of chlorine, chloramine, chloride, sulfate and NOM (Health Canada, 2009b). It has been shown that NOM affects the corrosion of lead and copper (Korshin et al., 1996, 2000, 2005; Edwards and
Sprague, 2001; Dry and Korshin, 2007; Liu et al., 2009; Valentina and Lin, 2009; Valentina and Lin, 2009; Shock and Little, 2011; Arnold et al., 2012; Chow et al, 2016). The impact of NOM on metal surfaces can be varied. NOM can provide protective film by reducing corrosion (especially over a long period of time), or it can increase corrosion through a variety of
mechanisms: 1) NOM can be integrated with calcium ion and prevent the formation of a protective scale; or 2) NOM can act as a microorganisms, which in turn can attack the surface of the pipe and (Shock and Little, 2011). Schock et al. (1996) reported various exposures to lead and copper. In some cases, it has been observed that NMS form soluble
organic compounds with lead, which leads to an increase in the concentration of dissolved lead. NOM can also be an adsorb/stick to the inner surface of lead pipes, reducing lead commensurate. Korshin et al. (2005) demonstrated that NOM can influence both morphology (physical structure) and distribution of lead particles and can prevent or worsen the
formation of more stable cerrusite and hydrocerrusite scales. NOM was observed to prevent the formation of cerrusitis formation (i.e. imperfect and scattered crystals were observed). Specifically, in the absence of NOM, it has been found that about 90% of lead particles are zgt; 5 microns. The addition of NOM reduced the size of
the particles as the larger particles broke due to the accumulation of surface charge. The sharpest increases in soluble lead concentrations were between 0 and 3.5 mg/l MLC (Korshin et al., 2005) and 0 to 2 mg/l DOC (Korshin et al., 2000). In the factor experiment, zhou et al. (2015) noted that an increase in NOM (from 1 mg/L to 7 mg/L DOC) resulted in a
significant increase in lead emissions when simulating partial replacement of the lead service line. In The Stand work trueman et al. (2017), the authors noted that the presence of humic acid increased the release of lead from lead coupons as a result of both homogeneous and galvanic corrosion. Although the addition of orthophosphate reduced the release
of lead, the addition of gum substances made orthophosphate less effective. The authors hypothesized that the combination of lead and gum substances suppresses lead precipitation with phosphates. Similarly, there was also an increase in copper emissions even at low concentrations (0.1-0.2 mg/l MLC), although the link between DOC and copper
production was not observed as linear (Korshin et al., 1996). The authors hypothesized that NOM adsorbs inorganic crystalline structures at high concentrations, NOM cannot cover the entire surface, leading to spotted crystalline product coverage and creating conditions for copper
pitting. The apparent contradictory impact of the NOM on copper emissions was further studied by Edwards and Spurs (2001). The authors noted that Prevent the formation of more stable malachite. Copper pipe aging is an important factor in
controlling copper emissions, with older copper pipes associated with more stable tenorite and malachite scales (Lagos et al., 2001; Edwards and McNeil, 2002). NOM can also be copper-free as a food source bacteria that consume dissolved oxygen and trigger re-deposition with suitable catalysts (Edwards and Sprague, 2001). The presence of NOM can
reduce copper emissions, as NOM can sorbate on freshly formed copper pipe surfaces, reducing soluble copper complexity and causing re-deposition (Edwards and Sprague, 2001). In practice, Arnold et al (2012) showed that removing NOM is an effective method for reducing blue water issues in school with new copper plumbing. The authors hypothesized
that the removal of NOM accelerated the natural aging process. Peng et al. (2013) noted that iron release increased in the presence of NOM (DOC 1 mg/L) and that other inorganic (lead, vanadium, chromium, copper and arsenic) can be released from iron at various levels of chloride (0-250 mg/L). The interaction of NOM with metal surfaces is complex, with
several factors influencing interactions, such as exposure time and pH (Korshin et al., 2005; Liu et al., 2009). NOM characteristics have also been noted as important for lead (Dryer and Korshin, 2007). Further research is needed to examine the significant impact and mechanisms of
production of NM And metal both in the distribution system and in the premises of plumbing materials. Currently, researchers recommend removing NM to minimize the concentration on corrosion control, lead and copper is available elsewhere (Health
Canada, 2009b, 2019b, 2019c). The purpose of coagulation is to destabilize (i.e. neutralize the charge) of colloidal particles (including pathogens) so that they are effectively aggregated during flocculation and subsequently removed by refinement and/or filtration. Coagulation also deals with the removal of NOM by phase change, which converts dissolved
organic matter into particles: either directly by precipitation or by advertising for particles created by the coagulant reactions occur with both particles and NOM. Therefore, when a coagulant is added, NOM acts as a ligand that complexes positively charged metal ions,
attaching a coagulant demand that must be overcome before flocculation can occur (Edzwald and Haarhoff, 2012). The nature of the NOM also has a significant impact on the clotting process. For example, for a coagulant concentrations of fulvic acid to a coagulated equal mass compared to gum acids (Edzwald, 1993), it is necessary to double the dose of
coagulant; Rigovello et al., 2011). Coagulation should therefore be seen as an integrated process, which are taken into account their various characteristics of coagulation (Edzwald and Haarhoff, 2012). For The For for the pH conditions of most water sources (pH 6-8), NOM and particles
carry a negative charge, which becomes more negative with an increase in pH. However, a negative charge of NOM is usually between 5-15 zek/mg of carbon, while particles between 0.05-0.5 zek/mg particles, depending on the type of particles (Edzwald, 1993). Pernitsky and Edzwald (2006) evaluated the charge for both NOM and particle components for
different water sources to demonstrate that in most cases caagulant dosage is controlled by NOM rather than mutity. The mutity should increase significantly, in the absence of a related increase in NOM, for the turbidity to control the dose of coagulant. Since NOM concentrations can increase rapidly four to five times during storm events, it is important that
water channels have a good understanding of the impact of NOM on caagulant dosing (Edzwald, 1993; Pernitsky, 2003; Hurst et al., 2015; James et al., 2015; James et al., 2016). Failure to adjust a dose of coagulant in accordance with the change in NOM may contribute to suboptimal coagulation conditions and reduced ability to pathogenic log removal
(Edzwald, 2017). Given the importance of coagulation of chemistry to ensure the removal of pathogenic journals, water channels should consider both NOM and turbidity in determining the optimal pH and coagulant doses (Edzwald, 1993; Edzwald and Haarchoff, 2012). Yar testing is one of the most commonly used methods for
simulating coagulation and determining the potential of water clotting (Black and Willems, 1961; Chow et al., 2004). It should be noted, however, that some factions of the NOM cannot be removed by clotting at any pH or dose (Kavanaugh, 1978; Babcock and Singer, 1979; Owen et al. 1993, 1995; Volk et al., 2002; Chow et al., 2004, 2006; Carpenter et al.,
2013). A more detailed discussion of the principles of coagulation and process optimization is presented elsewhere (Edzwald and Van Benschoten, 1990; Edzwald, 1993; Gregor et al., 1997; Edzwald and Kaminsky, 2009; AWWA, 2011a,
2011b; Edzwald and Haarhoff, 2012; Davis and Edwards, 2014). NOM has been identified in numerous studies as responsible for membrane contamination, which can significantly worsen water purification operations. It is generally accepted that the hydrophilic neutral fraction of NOM, consisting of polysaccharides and proteins in macromolecular and/or
colloidal form (i.e. biopolimiers), is responsible for membrane contamination (Amy and Cho, 1999; Carroll et al., 2000; Fan et al., 2000; Fan et al., 2009; Peldszus et al., 2011; Croft, 2012; Chen et al., 2014; Kimura et al., 2014; Kimura et al., 2014; Rahman et al., 2014; Rahman et al., 2014; Croft, 2015; Cho et al., 2016; Cho et al., 2006; Halle et al., 2007; Amy, 2008; Halle et al., 2008; Halle et al., 2009; Peldszus et al., 2014; Croft, 2012; Chen et al., 2014; Croft, 2012; Chen et al., 2014; Croft, 201
Siembida-Lesh and 2014, 2015; Yamamura et al., 2014; Jeong and Cho, 2016). It is assumed that once pollution is initiated by biopolimers, the reduction of electrostatic forces allows hydrophobic NOM to absorb membranes, resulting in further contamination (Peldszus et al., 2011; Croft, 2012; Chon and Cho, 2016). Rahman et al. (2014) reported that
concentrations of biopolimers up to 0.1 mg/L resulted in reversibility (i.e. removable by washing/air scouring) and irreversible (i.e. removable chemical cleaning) contamination. Her et al., (2007), reported contamination of proteins that were not detected in the forage water due to low concentrations (detection limit is not given). Other factors influencing
membrane pollution include membrane characteristics (e.g., membrane type, pore size distribution, material, surface charge, hydrophobic), operating conditions (e.g. pH, ionian strength, concentration and folant character) (Amy, 2008; Water canals should have a good
understanding of how NOM in their water source will interact with membranes to avoid configurations that carry significant pollution. Preliminary treatment may be necessary to reduce the concentration of biopolymeers (Carroll et al., 2001; Siembida-Lesh et al., 2014; Jeong and Cho, 2016). Pre-treatment should be configured for each
individual source, as effectiveness depends on the source (Fabris et al., 2007; Gao et al., 2011; Siembida-Lesh et al., 2015). Regular flushing and periodic chemical to remove accumulated folates (Alspach et al., 2014). It is well known that NOM is responsible for aesthetic problems such as color,
taste and smell (Hassler, 1947). The color caused by the presence of organic substances can occur in both surface and groundwater (Black and Christman, 1963; Thurman, 1984; Tan and Sudak, 1992). Organic color is usually caused by the presence of gum and fulvical acids, which are substances of black and yellow color (Stevenson, 1982). Black and
Christman (1963) reported that 87% of the compounds responsible for color in 10 American sources were colloidal and 3.5-10 nm in size. In contrast, Ratnaweera et al. (1999) reported that 40% of the compounds responsible for color in seven Finnish sources were zlt; 10 kDa (approximately 1 nm). High-color sources tend to have a higher concentration of
high molecular mass of gum acids, which may explain these differences in the distribution of sizes (Edwards and Amirtharajah, 1985; Aitkenhead-Peterson et al., 2003). Fulvicic acids are a more complex mixture of low molecular mass compounds that more hydrophilic than gum acids, and they have a significant effect on the required dose of coagulant (see
section section In addition, a higher proportion of fulvic acids may not be coagular at any dose of pH or coagulant (Hall and Packham, 1965; Cavanaugh, 1978; Babcock and Singer, 1979). Since humic and fulvic acids are important precursors to DBP, proper color removal can be ensured in time (Chaulk, 2015). For example, Tan and Sudak (1992) reported
a potential formation of 250-262 micrograms/L (7-day potential test for formation at 20 degrees Celsius and about pH 8) for high-color groundwater with natural gum and fulvik acids (TOS 3.93-4.70 mg/l; UV absorption 0.1829-0.1907). Tastes and smells can be caused by volatile compounds produced by microbial biomass (e.g. actinomycetes, cyanobacteria,
fungi) that are washed out of the earth's environment or naturally present in the aquatic system/aquifer (Hrudey et al., 1992; Caitlyn and Watson, 2006; AWWA, 2011a). Watson (2003) has identified about 200 volatile organic compounds that produce unwanted flavors and odors. Terpenoids (e.g. geosmine and 2-methylosobornol), sulfides and
polyunsaturated fatty acids have been identified as the most odorous. Geosmine and 2-methylosobornol are environmentally stable compounds (i.e. not easily biodegradable) that can be transported over considerable distances from the site of the production of compounds (Satchwill et al., 2007). Other researchers identified pyrimidines as problematic
(Chorus et al., 1992; Caitlyn and Watson, 2006; Peter et al., 2009). Sakeus et al (2001) found that actinomycetes and fungi can survive in soft deposits (i.e. accumulated sediments containing organic matter) water distribution systems. As a result, the distribution system can be a source of taste and smell problems. Reactions of chlorine with
NOM can also promote tastes and smells (AWWA, 2011a). In particular, nitrogen-rich NOMs can form odorous aldehydes (Hrudey et al., 2004, 2005) or nitriles (Freuze et 
of disinfectant, disinfectant for the ratio of amino acid molars, pH, temperature and reaction time (Froese et al., 1999; Brosillon et al. 2009). Amino acids have been identified as the main precursor causing odor, and they can be released by licking bacterial or algae cells or when proteins are oxidized (How et al., 2018). Table 8 provides thresholds for odo
concentrations that have been recorded in the literature on these compounds. Since nitriles have much higher concentrations of the odor threshold (see table 8), they are usually not related to the taste and smell of events (Freuze et al., 2005). Connection odor (e.g. terpenoids) and precursors (e.g. amino acids, proteins) are not effectively removed by
conventional treatment. In this way, other processes may to minimize tastes and Smells (Rice and Gomez-Taylor, 1987; Bruchet et al., 1999). After the formation of odorous compounds, they can be stored in the distribution system for up to 500 hours (≈21 days) at 15 degrees Celsius (Freuze et al., 2004, 2005). Concentrations can also
increase in the distribution system by releasing amino acids or peptides from biofilm (Brosillon et al., 2009). Guidelines to assist water utilities in assessing and minimizing unwanted tastes and odors are available elsewhere (AWWA, 2011c). An effective strategy to combat NOM requires a good understanding of the origins, origins and changes that occur at
the water source (Volk et al., 2002). Water canals should have a good understanding: While the numerous organic compounds that promote NOM cannot be measured directly, there are a number of surrogates include TOC, DOC, UV absorption and
chemical oxygen demand (Sillanp et al., 2015b). In addition, UV-absorber and UV transmission are mathematically related; therefore, the latter may also indicate the concentration of NOM. Since NOM is a major factor in organic color, this option can also be relevant (Matilainen et al., 2011). TOC quantifies all organic carbon in the water sample and the
amount of particulate matter and dissolved organic carbon. DOC is promptly defined as organic carbon, which has passed through the 0.45 micrometer filter (APHA et al., 2017). Since the filter can leach some organic carbon into the sample, it is recommended that at least 50 ml of organic water pass through the filter and assembly filter before filtering the
DOC sample (Karanfil et al., 2002, 2005). TOC and DOC are measured indirectly from carbon dioxide, which is produced by the oxidation of chemicals with ultraviolet catalysis or at high-temperature combustion. UV-visible light absorption at 254, 350 and 440 nm can be linearly correlated with THE concentration of DOC in some freshwater systems.
However, linear correlations are less likely to be found in sources with strong autochton or man-made inputs or where DOC has been significantly degraded by natural ultraviolet light (e.g., long-term retention in the lake) (Minor et al., 2014). The UV254 measurement has historically been used in the water industry (Edzwald et al., 1985). Samples must be
filtered to remove particle-related variations in UV absorption (APHA et al., 2017). It is generally accepted that the change in UV absorption absorption provides operators with valuable information about impact on the dose of coagulant, as
the concentrations of NOM may change without any observed in treated water or a wastewater filter. It is important that correlations are developed on the basis of specific sources, as the relationship between NOM and
UV radiation is unique to each source (Pernitsky, 2003). In some cases, it is not possible to establish a correlation between UV254 and DOC (Cho et al., 2010; Sadrnourmohamadi et al., 2013; Small et al., 2014). Monitoring the absorption of UV-visible light over a wider range of wavelengths may be more appropriate in some cases (Wright et al., 2016). In
addition, the lack of correlation may be due to the presence of NOM, which has low absorption of UV (e.g. proteins, sugars) or high nitrate content, which may interfere with this measurement (Leenheer and Crou, 2003). Monitoring the absorption of UV-visible light over a wider range of wavelengths can also provide a more advanced characteristic. UV
transmission is a relative indicator of how much light passes through the water sample (at a wavelength of 254 nm, usually 1 cm) compared to how much light passes through pure deionized water (which has 100 UV transmissions). The demand for chemical oxygen serves to give some insight into the concentration of oxidized organic matter in water sample
(Frisch and Kunin, 1960; Stoddart and Gagnon, 2014). The use of potassium dihromate) has not been sensitive enough for drinking water (Rittman and Huck, 1989). Since then, more sensitive methods have been developed., 1993); another method is the method of photoelectrochemical demand for oxygen (pecoD) using UV-activated titanium dioxide as an
oxidizer (zhao et al., 2004; ASTM, 2017). Information on the average state of organic carbon oxidation during water treatment can also be gleaned using the ratio of COD and TOC/DOC molars (Li et al., 2018). Color is historically measured by colorful methods. The presence of suspended particles (e.g. clay oxides, iron and manganese) can give the waters
a color and should be removed by filtering the sample through a filter of 0.45 microns before measuring the organic color associated with NOM (e.g. color primarily due to the presence of gum and fulvik acids - see section B.4.3); filtered sampling is promptly defined as the true color (APHA et al., 2017). The researchers also used visible light uptake at 420
nm in Measures for Organic Color (Ekstr'm et al., 2011; Weichenmeier et al., 2014). However, the wavelength between 450 nm and is offered as a standard spectrothometric method (APHA et al., 2017). The spectrophotometric method requires the filtering of samples using a filter of 0.45 microns (Hongve and Okesson, 1996; APHA, 2017). Comparing true
and obvious color results can help water channels determine whether color complaints are related to NOM. The visible color is applied to unfiltered samples and is a useful measure for assessing the presence of iron and manganese oxides in the distribution system (Reiber and Dostal, 2000; Imran et al., 2005). Measuring the above parameters is easy and
quick, and some tests can be automated. They may indicate a change in water quality; however, they do not offer information about the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM. Edzwald et al. (1985) found that UV254, divided into mg/l DOC, was a useful indicator of the nature of NOM.
Calculating a particular color (i.e. the true color divided into mg/L DOC) can also provide useful information (Chow et al., 2005). The SUVA concept was developed as an operational indicator of the nature of NOM and the effectiveness of coagulation for the removal of NOM (Edzwald and Van Benschoten, 1990; Edzwald and Tobiason, 1999). Table 9
presents the generally accepted relationship between SUVA, NOM, UV absorption, coagulation and potential TOC removal. SUVA calculation is widely used to assess the nature of NOM because it is easy and inexpensive to identify and is a good indicator of changes in the quality of water sources (Westerhoff et al., 1999; Imai et al., 2001; Weishaar et al.,
2003; Rekhoe et al., 2007). For example, Volk et al. (2002) tracked DOC and UV254 daily for 22 months for the White River (Muncie, Indiana). During this periods of low runoff and high algae activity (i.e. hydrophilic, autochton NOM), while high values were associated
with melting snow and storm runoff. During a typical precipitation event, SUVA increased from 2.6 litres/mg over 12 hours, indicating that hydrophobic, allochthonous NOM is being washed away in a source from the ground watershed. Archer and Singer (2006b) analyzed surface water data for 18 months from the U.S. Environmental
Protection Agency 's Data Collection Regulations (EPA) and found a clear link between SUVA, water sources characteristics, and clotting efficiency to remove organic carbon. In general, the high sources of SUVA, which can be misleading as
to the potential for organic carbon removal through coagulation (Edzwald, 1993). to achieve the DBP's guidelines will depend on the The concentration of NOM and whether you can remove enough reactive NOM. If residual MLC remains reactive to DBP formation after coagulation, other technologies may be needed to remove specific NOM fractions (Bond et
al., 2011). Since humic and fulvic acids are important precursors to DBP, proper color removal can be ensured in time (Chaulk, 2015). Low-source SUA tends to have NOM, which is not subject to coagulation (Pernitsky, 2003). Tracking the use of chemicals (e.g. coagulant dose, demand for chlorine) and calculating a specific dose or demand (i.e. mg/L per
mg/L MLC) can help water channels assess changes in the nature of NOM. For example, Chow et al., (2005) reported that a specific dose of coagulant decreased when NOM all-ton inputs increased. In addition, Hwang et al. (2001) reported that the hydrophilic base share of NOM produced a significant demand for chlorine, as indicated in Table 10. This
fraction consists of compounds that are biodegradable (e.g. amino acids). Thus, the removal of these compounds using biological filtration methods (see section B.6.2.5) may reduce the demand for chlorine and DWP (Pr'vost et al., 1998). Summers et al. (2013) warned that DOC and demand for chlorine do not correlate with THMs and HAA5 (i.e.
monochloroactic acid, monobromoacetic acid, dichloroactic acid, dichloroactic acid, dibromoacceic acid, di
halogenation processes dominate, compared to other reactions that consume chorin (e.g., inorganic oxidation, photolytic and corrosive processes). Caution is recommended when assessing trends in chlorine demand or specific demand for chlorine, as reactions are likely to vary depending on seasonal and weather-related effects, treatment processes and
where chlorine is added. The actual concentrations of DBP measured in the distribution system give a good idea of the reactivity of NOM. It is recommended that the parameters used to characterize NOM be measured in conjunction with DBP samples to assess a specific DBP crop (e.g., ICP/mg MLC). In addition, inorganic compounds that increase the
reactivity of NOM for the formation of DBP (i.e. ammonia, bromide, iodine and sulfur) should be characterized. NOM compounds can be crushed using commercially available solid extraction phase sorbents. However, the measurement of the six factions of the NOM Table C-3.4) is time-consuming and time-consuming (Minor et al., 2014; Goss et al., 2015)
and cannot correlate well with the potential results of the formation of DBP (Wright et al., 2016). 2016). researchers investigated faster assessment methods (Martin-Mousset et al., 2009). Goss et al. (2017) compared three prepackaged cartridges to extract a solid
phase to isolate hydrophobic and hydrophilic fractions from three surface waters in Manitoba. The authors reported that this method could be used in water treatment. In contrast, Wright et al (2016) reported that solid phase extraction
cartridges leaked variable amounts of organic carbon, skewed TOC results and poorly correlated with DBP forming potential results. The authors did not recommend the use of this method as a monitoring tool. NOM compounds can also be physically crushed based on differences in molecular size using membrane fractionation or chromatography of size
exclusion (Koudjonou et al., 2005). NOM is usually split into four sizes ranges: 1, 1-10, 10-30, and 30 kDa. The size exclusion chromatography can be supplemented with organic carbon (LC-OCD) and/or organic nitrogen (LC-OND) (Huber et al., 2011). These methods are usually reported in peer-reviewed literature, but are not yet used in the usual way by
water channels. Fluorescence is another method that shows promise (McKnight et al., 2010; Bridgeman et al., 2011; Marchekhova, et al., 2013; Sanchez et al., 2013). NOM fluorescent fractions show peaks of intensity at certain
wavelengths; this allows them to be classified as terrestrial, microbial or man-made organic matter, as well as humico, fulvik- or protein compounds. Researchers developed mathematical tools or algorithms to evaluate the large data sets that are generated and to compare differences between samples. Field units are in commercial conditions and research
is continuing on how this method can be integrated as a regular monitoring tool (Bridgeman et al., 2011; Murphy et al., 2013; Wright et al., 2016; Peleato et al., 2017; Frank et al., 2018; Li et al., 2014). BDOC and AOC are the two
most widely used parameters for measuring the biological stability of water. BDOC refers to the parts of the DOC available for the use of heterotrophic bacteria (Escobar and Randall, 2001). Testing consists of measuring DOC in water before and after incubation with the inoculate of the natural bacterial population. BDOC is considered a measure of the
hydrolyzed pool available for bacterial regrowth. The AOC is the most easily decomposed part of BDOC BDOC can be taken by bacteria and converted into organic biomass (Escobar and Randall, 2001). The test for the AOC identifies water growth potential by measuring the yield of two pure strains of bacteria (Pseudomonas fluorescens strain P17,
Spirillum strain NOX) over several days and comparing these observations with a calibration curve for growth made using organic carbon standards solutions (e.g. acetate or oxalate) (LeChevallier et al., 2015). AOC concentration is considered to be one of the indicators of water biostaqueness for the growth of heterotrophic bacteria (Escobar and Randall,
2001). AOC is often used as a method for predicting bacterial regrowth because it returns a value that corresponds to bacterial counting (Escobar and Randall, 2001). It has been suggested that the AOC:BDOC ratio could be used as an indication of the relative biological stability of biodegradable organic compounds present in drinking water (Escobar and
Randall, 2001). Both methods take a long time and require a high level of analytical expertise. Camper (2004) and van der Kuy et al (2015) reported that carbon compounds not measured by AOC and BDOC may also influence the growth of biofilm and that these measurements alone may not be sufficient to estimate growth potential. The water industry is
exploring the benefits of faster methods that use adenosine triphosphate (ATP) measurements or flow cytometry technology (Hammes et al., 2015; Van der Kuy et al., 2015; Bezmer and Hammes, 2016; Elhadidy et al., 2016). ATP measurements
are gaining popularity as an indicator of microbiological biomass (Siebel et al., 2015a). However, understanding the value of measurements, as they relate to other water quality measurements such as viable and iconic cell numbers, is necessary when
considering the inclusion of ATP analysis in the monitoring program (Siebel et al., 2008; Hammes et al., 2010). Flow cytometry has been proposed as one of
several methods for assessing biological stability (Lautenschlager et al., 2013; Prest et al., 2013; Prest et al., 2014). The technology is advanced and has significant requirements for hardware, user training and data processing (Hammes and Egli, 2010). Several studies have investigated online biostathability monitoring using flow cytometry, but
standardized methods Not designed for drinking water (Hammes and Egli, 2010; Lautenschlager et al., 2013; Prest, etc. 2013). In a study of the full-scale chlorinated drinking water system, Nescerecka et al. (2014) found that flow cytometry in combination with ATP measurements provides more relevant information than the number of heterotrophic plates to
assess and understand biological stability at different points in the distribution system. Van der Kooij et al. (2015) reports that the use of ATP techniques to assess biofilm formation potential and biofilm accumulation indicators has provided a better understanding of biological instability in distributed water (without disinfectant residues) that would not have
been identified only by an AOC assessment. LeChevallier et al. (2015a) completed a statistical analysis of full-scale data on six water channels and concluded that the most useful measured by ATP accumulated on
soft steel coupons) and changes in corrosion rate (measured by linear polarization). LeChevallier et al. (2015b) provide recommendations to help water utilities produce biological stability of drinking water distribution systems are also available in
preparation for Health Canada. The approach to the source or water safety plan, which includes careful selection of high-quality water (O'Connor, 2002; CCME, 2004; WHO, 2012). Research should be presented, including bench
testing and/or experimental scale, to determine the most appropriate treatment options for the full range of water quality conditions (Valade et al., 2009; Huck and Sozansky, 2011). Temporary fluctuations in the concentration and nature of NOM can have a significant impact on the selection, design and operation of water treatment processes (Sillanp, 2015)
More variable weather conditions associated with climate change will be of increased importance for process selection (Huck and Coffey, 2004) and day-to-day process control (Wright et al., 2016). Water channels should integrate the risks associated with climate change (e.g. algae bloom, drought, fire, flooding) into the process of maximizing the
reliability, reliability and sustainability of their systems (Emelko et al., 2011b; Irias, 2019). In order to properly select, design and operate water-storage facilities, it is necessary to understand the changes in the concentration and nature of THE NOM for the full range of conditions faced during the year, both for surface and surface for groundwater (AWWA,
2011a; Sillanpe, 2015). To determine the most suitable water purification processes should have knowledge of the following (Ivan'ev-Tumbas, 2014): Appropriate type and level of treatment and distribution system (Kastl et al., 2016). A
treatment-specific study should be conducted to evaluate and compare treatment options for the removal of NOM (Goss and Gorczyca, 2013; Plourde-Lescelleur et al., 2015; Kastl et al., 2016). The study of curability should include bench and/or experimental testing and take into account concomitant water quality goals related to microbial risk, DBP,
biological stability and corrosion control. Options to be considered as part of the treatment suitability study include chemical doses and residues, turbidity, organic content (e.g. DOC, UV254, COD, color), organic character (e.g. hydrophobic, size, specific absorption of UV), pH and alkalinity, anions (e.g. bromide, chloride, fluoride, nitrate/nitrite,
orthophosphate, sulfate), the potential for the formation of DBP, which is representative of the distribution system, biostabilitis and corrosion characteristics (Gregor et al.) Sheen et al., 2008; Cho et al., 2010; Brown and Cornwell, 2011). The optimal solution will be specific to the sources, and several treatments may be required to adequately remove NOM at
any time of the year (Collins et al., 1986) Chang et al., 2001; Hua and Rekhow, 2007; Caranfil et al., 2007; Fabris et al., 2008; Christiana et al., 2008; Christiana et al., 2008; Christiana et al., 2015; Sillanpe, 2015). Lack of treatment study specific to specific sources can lead to the choice of inappropriate treatment, increased concentration of by-products for
disinfection after treatment or other unintended consequences. There are a number of treatment options for removing NOM. Optimized coagulation is the most applicability should be carefully analysed on the basis of specific sources, since coagulation could remove only a few
factions of NOM; the remaining fractions (i.e. those that are not removed by clotting) may react with disinfectants, so that the DBP guidelines are not achieved. For example, alcothone NOM is usually hydrophobic in nature and can usually coagulate, while hydrophilic NOM is generally harder to treat (Volk et al., 2002; Chow et al., 2004, 2006). In fact, for
sources high in hydrophilic neutral NOM, coagulation will be ineffective (Chow et al., 2006). As a result, it is very important to test banks and potentially test the formation of DBP to determine the feasibility of optimized to remove NOM. Additional or alternative treatment options include nanofiltration, ion exchange, GAC or powdered activated carbon (PAC),
(PAC), filtration and oxidation processes. The literature warns that the specific output of DBP (i.e. ICD/MG MLC) can sometimes be greater in treated water than in disinfected raw water (Jacangelo et al., 2007, de la Rubia et al., 2008; Department of Environment and Env
2011). This is due to the higher ratio of bromide:DOC after treatment. Since bromide is not removed by most treatment processes, more bromine DBPs can be formed after treatment if the removal of NOM is insufficient. As a result, it is very important to work out a study of curability specific to specific sources to evaluate and compare treatment options; This
study should include bench and/or pilot scale testing to determine the potential of DBP formation. Treatment options and their effectiveness are summarized below. The results are presented to demonstrate that deleting NOM can be very variable. More information about treatment is available from other sources (Parsons et al., 2007; AWWA, 2011a; Bond et
al., 2011; Huck and Sozanski, 2011; Sillanpe, 2015). It is important that water treatment operators understand the mechanisms of removal of NMMs, as changes in cleaning practices can have a significant impact on water quality (Ivan'ev-Tumbas, 2014). Thus, training of operators is also necessary to ensure the effective operation of treatment barriers at any
time (Smeets et al., 2009). Maintaining modern knowledge of best practices and maintaining knowledge of drinking water advances are essential to water safety. Coagulation includes two main mechanisms: one consists
of neutralizing the charge and forming insoluble precipitation; the other includes aluminium or ferric hydroxide floss (i.e. coagulation of the sweep) (Dempsey et al., 1984). Each mechanism is conducive to a specific set of working conditions associated with pH and coagulant dose. As PH increases, NOM becomes more negatively charged, but hydrolysis
coagulant products with lower positive charge dominate. Thus, at 7, a four-fold increase in the dose of a coagulant is necessary to overcome the negative charge of NOM removal is bad (Semmens and Field, 1980; Edzwald and Van Benshoten, 1990; Edzwald and Tobiason, 1999). Physical factors
(such as mixing a coagulant and mixing conditions in a flukulator) can affect the formation of a flok; however, in most cases the coagulant will depend on the characteristics of the water to be treated. Available coagulant variants (e.g.
aluminium and ferric coagulants, inorganic polymer pol
increases as the temperature decreases (Pernitsky, 2003). For alums, optimal performance usually occurs at pH values close to the pH of the minimum tact (i.e. 6.5-6.7 at 4 degrees Celsius and 6.0-6.2 at 20 degrees Celsius) (Edzwald and Kaminski, 2009). Since the pH of the minimum slug is higher at lower temperatures, a higher dose of coagulator may be
required to overcome a more negative charge on NOM with a lower positive charge on hydrolysis coagulant products, as noted above. Strict pH control is necessary for optimal coagulation; PH should be constantly kept from the coagulant products, as noted above. Strict pH control is necessary for optimal coagulation; PH should be constantly kept from the coagulant supplement to after filtering effectively remove the flok particles. Even a small pH change can release a NOM that was
previously included in the floks (Slavik et al., 2012). Yar-testing is recommended to optimize the choice of coagulants. NOM also determines the size, structure and strength of the floxes, controlling both the degree and speed of clarification or filtration processes (Eikebrokk and Saltnes, 2001; Newcombe and Dixon, 2006; Parsons et al, 2007). Several studies
have shown that low-density NOM floxes are more flotation than sediment (Plummer et al., 1995; Edzwald and Kelly, 1998; Edzwald et al., 1999, 2000, 2003; Harrington et al., 2001; Edzwald, 2010; Gregory and Edzwald, 2011). In addition, the formation of sedentary floss (Semmens and Field, 1980) may require the addition of coagulant care (e.g., activated
silica, bentonite, lime, polymer); Edwards and Amirtaraj, 1985; Gregor et al., 2011). Otherwise, operators are not aware of insufficient fading coagulants until there are spikes in sedentary water or cloudiness of filters (Pernitsky, 2003).
Ideally, raw water should be constantly monitored to optimize the dose of coagulant (Pernitsky, 2003; Newcombe and Dixon, 2006; Sharp et al., 2006; Sharp et al., 2008). Online monitoring tools for NOM include TOC, DOC, UV absorber/transmission and COD; to destabilize particles they include the potential of a zeta or streaming current (Conio et al.,
2002; Newcombe and Dixon, 2006; AWWA, 2011b). The maximum removal of NOM was reported when the charge of coagulant particles is almost neutral, measured by the potential of the zeta (Sharp et al., 2006; Sharp, 2015) or streaming (McVicar et al., 2015). Failure to adjust a dose of coagulant in accordance with the NOM change may conditions of
suboptimal coagulation and reduced ability to pathogenic removal of logs (Edzwald, 2017). Table 11 summarizes variability variability variability reported over certain periods. Hargesheimer et al. (1994) reported a 0% deletion at different
times in 1993, namely in early March, May, August and December. It is possible that these time frames represent snow cover (March, December) or base flow conditions with low NOM acoton inputs (May and August). Carpenter et al. (2013) reported a 0% removal in early September 2011 for Plant 1 and August 2011 for Plant 2. PLEASE that at this time
(i.e. at the end of summer) NOM will be generated from autochton sources and is likely to be hydrophilic in nature. Table 12 summarizes TOC compliance monitoring data released by the U.S. Environmental Protection Agency (2016) as part of the third six-year review. The data are the removal of TSEs (percentage) achieved at conventional surface water
treatment plants as a function of the water quality matrix established by the disinfectant/DBP rule. Overall, the U.S. Environmental Protection B.8), although some facilities have failed to meet removal requirements. The report cautions that data analysis cannot
determine which objects can determine alternative performance criteria or which may treat water to less than 2 mg/L (U.S. EPA, 2016). On an experimental scale, Braun et al. (2014) evaluated conventional processing (parallel to membrane filtration, ion and GAC) over a three-year period that included a prolonged drought and two different periods of flooding.
The authors reported variable removal of DOC with conventional treatment (range - 32-61%; alum dose - 20-160 mg/L; pH coagulation - 6.0-6.5). The authors also noted that water quality was the best during this period. During drought periods, NOM is
usually generated by autochton sources and is hydrophilic in nature. This confirms the results of full-scale studies indicating that hydrophilic NOM may be difficult to treat. The results of numerous studies were reviewed and summarized by Bond et al (2011). In these studies, removal percentages were determined for DOC, UV254, predecessors of THM and
precursors of HAA for numerous treatment processes, including coagulation and coagulation in combination with other processes (see table 13). The results published by Plourde-Lescelleur et al. (2015) are also summarized in Table 13; in this study, two coagulants were compared for routine processing and Alum combined with ion exchange, ozonation or
PAC has been evaluated for six Canadian sources of surface water. The results summarized in Table 13 confirm variable performance that can only be achieved by coagulation and that the increase in может быть достигнута путем интеграции коагуляции с другими процессами. В совокупности результаты полномасштабной, экспериментальной и
стендовой шкалы свидетельствуют о том, что коагуляция может быть эффективной, какими бы плохими ни были результаты. Эти выводы подтверждают необходимость проведения применимости коагуляции для адекватного удаления прекурсоров NOM и DBP для полного спектра условий качества
воды. Другие процессы, отмеченные в таблице 13, дополнительно обсуждаются в последующих разделах. В настоящее время в очистке питьевой воды используются четыре типа мембран, управляемых давлением: микрофильтрация (ИF), ультрафильтрация (ИF), и обратный осмос (RO). Мембраны, как правило,
классифицируются по типу веществ, которые они удаляют, операционное давления и используются для удаления частиц/патогенов. Преобладающим механизмом удаления является напряжение или исключение размера.
NF и RO называются мембранами высокого давления и используются для удаления в солуство или диффузивность. The size distribution of NOM varies between sources, but generally over 50% of NOM molecules
have a molecular weight of < 1kDa and 80% have a molecular weight of &lt; 10kDa (Sillanpää et al., 2015a). As a result, a tight NF membrane is required to remove the majority of DBP precursors, as shown in Figure 1. Studies indicate that the optimum MWCO for NOM removal is 0.2-0.3 kDa (Jacangelo et al., 1995, 1997; Bond et al., 2011; Sillanpää et
al., 2015a). Figure 1 illustrates that MF membranes cannot remove any NOM fractions other than biopolymers. UF membranes may remove some NOM, as shown in Figure 1, but DBP formation potential may not decrease adequately. For example, Lamsal et al. (2012) reported an overall DOC removal of 66% for full-scale UF membranes (absolute pore size
= 0.01 μm). THM and HAA formation potentials decreased by 54% and 77%, respectively but remained high at 200 μg/L, respectively (test conditions = chlorine 1.0±0.4 mg/L at room temperature for 24 h). Ødegaard et al. (2000; 2010) reported colour removals of > 85% for 27 full-scale NF membrane plants in Norway (raw water colour = 35-
50 mg Pt/L; MWCO 1-2 кДа). На экспериментальном уровне Braun et al. (2014) оценивали интегрированную систему (номинальный выше для обычного лечения (см. раздел В.6.2.1). Удаление DOC варьировалось от 89-97% (данные
интерпретируются на графике) и представляло собой самые высокие процентные удаления по сравнению с обычным лечением, ионом и GAC, которые были протестированной System. As in conventional treatment (see section B.6.2.1), MF/NF Removing doc during drought. The results published by Bond et
al. (2011) and Plourde-Lescelleur et al. (2015) for NF membranes are summarized in Table 13. The reported removals were high, as were the results of the pilot scale. Taken together, the results of the membrane pore to remove NOM (≈200-
300 Da). However, it is important to consider foul potential as NOM is known to foul membranes. Pollution potential indicators include low SUVA, high hydrophilic fraction, high dissolved nitrogen or high concentration of biopolymer (Lee et al., 2006; Amy, 2008; Croft, 2012; Kimura et al., 2014; Siembida-Lesh et al., 2014). In addition, the UF membranes
cannot remove the AOC (usually It; 1 kDa) unless combined with a rigid NF or RO membrane or biological treatment (Sillanp et al., 1997). Pre-treatment requirements for NOM removal should be considered as part of a study of the
suitability for a source-specific treatment where the optimal MWCO is not used to remove NOM (i.e. 0.2-0.3 kDa). Experimental testing is recommended to assess the potential of contamination and the need for prior treatment (Huehmer and Voutchkov, 2007). Ionic exchange is a process in which ions from raw water exchange ions within the solid phase of
resin. In the early 1960s, it was recognized that ion exchange processes could remove NOM, mainly because NOM was found to contaminate ion resins used to remove involves ion exchange (i.e. electrostatic), with hydrophobic absorption and
hydrogen bonds also plays a role (Fu and Symons, 1990; Bolto et al., 2002). Ionic metabolism can effectively remove charged NOM compounds (Kim
and Symons, 1991; Bolto et al., 2002; 2004; Parsons et al., 2007; Cornelissen et al., 2008; Umbert et al. (1999) reported that the uncollected share of NOM at full-scale ion exchange was more effective for removing DOC (8.1 mg/L decreased to 1.7
mg/l) compared to color removal (75 mg/l pt decreased to 30 mg/l pt). Longer contact time or other treatment may be required if color removal is needed (Hongve et al., Edegaard et al., 1999). The exchange resin, which was specifically designed
to remove NOM NOM Et al., 2000; Drikas et al., 2000; Drikas et al., 2003; Fearing et al. 2004c; Budd et al., 2005; Singer et al., 2007, 2009; Bond et al., 2010; Brown and Cornwell, 2011). Levchuk et al (2018) summed up the results of 22 studies (1 full-scale; 21 benches of scale) conducted in 1997-2016, 50% of which assessed the effectiveness of magnetic ion exchange.
Factors affecting the performance of ion exchange include the concentration and nature of NOM, water quality (in particular, concentration and charged functional groups) and operational variables (resin dose, contact time, frequency of regeneration).
However, the ion exchange processes do not remove the turbidity; so they usually apply with the neuerence removal process (Drikas et al., 2003). Some researchers reported that the exchange of ions was more effective than coagulation for the removal of NOM, either by itself (Drikas et al., 2003; Singer et al., 2007, 2009) or in combination with coagulation
(Drikas et al., 2003; Brown and Cornwell, 2011; Braun et al., 2014). Drikas et al. (2011) reported an average DOC removal of 54% for full-scale ion-exchange coagulation (a dose of resin ≈8-16 ml/L for 10 minutes; a dose of alum ≈6-10 mg/L). The authors explain much of the DOC's removal using resin magnetic ionic exchange resins. Singer et al. (2009)
completed a comprehensive review of numerous full-scale and experimental studies involving 21 sources in Australia and the United States and reported the removal by the presence of hydrophilic NOMs with a base and neutral
charge. Table 14 summarizes the results of a number of other experimental studies. Researchers studying magnetic ion metabolism (one or with coagulation) reported widespread variability in the removal of DOC; removal was reportedly affected by the nature of the NOM (Fearing et al., 2004c; Singer et al., 2007; Braun et al., 2014), with higher DOC
removals observed with an increase in SUVA (Singer et al., 2009). Braun et al. (2014) demonstrated that integrating GAC as an additional process can improve the removal and the highest variability during drought, similar to conventional and membrane
treatments, as discussed above. Fearing et al (2004c) highlighted the variability that occurs due to the dose of resin and the time of contact. Table 15 summarizes the results demonstrate the variability in performance that can occur with resin and system, similar to the
magnetic exchange of resin ion, Above. The use of ion exchanges in conjunction with other processes can have some operational benefits and consequences that should in the study of curability for specific sources. For example, ion exchange before clotting can reduce the dose of coagulant and associated sludge production, reduce the turbidity of
sedentary water, reduce the use of chemicals to adjust pH, reduce the dose of disinfectant and stabilize the residues of the chlorine distribution system (Budd et al., 2005; Brown and Cornwell, 2011). Ion exchange processes may also remove some bromide from sources with low alkalinity and sulphate concentration due to minimal competition for ion
exchanges (Singer et al., 2007). Umbert et al (2008) reported improved activated carbon performance to remove pesticides when ionic exchange treatment (e.g. chloride) and lower doses of sulfate-based coagulants may increase the potential for corrosion
due to changes in the ratio of chloride: sulfate mass. More information on chloride: The ratio of the mass of sulfate and its effects is available in Health Canada (2019b)Collective, complete, experimental and bench-scale results show that ion exchange can be effective when the dose of resin and contact time is optimized for NOM removal, however poor
results have also been reported. These findings confirm the need for a cure study, including bench or experimental scale testing, to determine the optimal configuration and Cornwell, 2011). Consideration should also be given to the processing and
```

disposal of residues from ion exchange processes. Water channels that use ion exchange to remove other anions (e.g. arsenic, chromium, nitrate, uranium) should know that NOM competes for ion exchange sites and can reduce the efficiency of the process (Frisch and Kunin, 1960; Ungar, 1962). To ensure that this process remains cost-effective as

```
intended (Bursill, 2001), pre-processing may be required to remove the NOM. Activated carbon is an absorbent material that provides a surface on which ions or molecules in raw water can concentrate. It can be used in two ways: dung applications using powdered activated carbon (PAC) or fixed-bed reactors with granular activated carbon (GAC) (AWWA,
2011a). Removal mechanisms include the adsorting of dissolved organic matter on a PAC or GAC, as well as the biodegradation of BOM in GAC fixed-bed reactors with active biofilms. Chowdhury et al(2010) found that biofilms can form in GAC macropores even when Chlorine. The main use of PAC and GAC in water purification is to remove micro-subutes
as well as tasteful and odor-causing compounds. The use of PAC offers the advantage of providing virgin carbon when needed (e.g. during the taste and smell season). GAC fixed-bed reactors are operated at a similar pace to the usual fast pace therefore, GAC characteristics (e.g., type, particle size, reactivation method) and working conditions (e.g. filter
speed, empty bed contact time, flushing mode, filter time) affect their performance. A large specific surface area and a well-developed porous GAC structure can provide a high sorbation capacity for organic molecules (Simpson, 2008). However, the GAC has not been widely used as a core NOM control strategy, because GAC's ability to absorb is usually
quickly exhausted (i.e. within a few months) and regeneration can be costly (Pr'vost et al., 1998; Huck and Sozansky, 2011). For example, the U.S. Environmental Protection Agency (2016) reports a 120-day frequency of reactivation of systems with a 6 mg/l using an empty bed contact time of 10 minutes. In addition, once the addictive capacity is exhausted
GAC can continue to remove NOM through a biodegradation mechanism, albeit with lower efficiency (Bond et al., 2011; Gibert et al., 2011a). For THE PAC, dose and contact time are important factors. Results for two full-scale conventional
water-hicwac installations (both using alum coagulation) are reported in the literature. In one case, a PAC dose of 150 mg/L improved the removal of DOC by 20% and THM precursors by 80% (Kristiana et al., 2011). In another case, a 11 mg/l PAC dose improved DOC removal by 7%, but did not improve in the removal of THM precursors (Carri're et al.,
2009). Yar testing is recommended to optimize the type of PAC, dose and contact time. For the GAC, studies show that the volume of pores must be in the size range that corresponds to a specific NOM source for GAC to be effective (Karanfil et al., 2007; Gibert et al., 2013b). As Karanfil et al. (2007) notes, surface area and total pore volume are not
sufficient criteria for choosing GAC to remove NOM, as these parameters do not provide information about the available pore region. The authors suggest that water channels request detailed information on the distribution of pores and pH of the zero fee point for VKS candidates. Rapid tests of small-scale columns should be conducted to compare the
indicators of alternative CEIs, especially for sources with low SUVA content (Ates et al., 2007; Karanfil et al., 2007). GAC particle suasing should also be considered, as abrasions can lead to loss of GAC material and bundle inside the bed, both of which are undesirable (Gibert et al., 2013b). Water canals that use activated carbon to remove pesticides or
other traces of contaminants should know that NOM competes for adsorcies and can reduce the efficiency of the process (Haist-Gulde Happel, 2012). NoM removal may require pre-processing so that the process remains cost-effective (Bursill, 2001). Biological processing involves targeting the removal of the BOM faction, which stimulates the growth of
biofilm in the distribution system (section B.4.1.3) and increases the demand for chlorine (section B.5.2.2) (Pr'vost et al., 2009; Diem et al., 2013). The
recalcitrant or fireproof NOM is unlikely to be removed by biological processes unless it oxidizes to turn it into a BOM. Biological processing generally improves the biological processes of biological treatment of drinking water are the filtration
of the river bank, rapid granular media filtration without maintaining the residue of disinfectant on the bottom and slow filtration of sand. Riverside filtration (RBF) includes the placement of vertical or horizontal water wells near the river for use by the river bank and the adjacent aquifer as a natural filter for the removal of pollutants, including BOM. As water
enters groundwater, concentrations decrease as a result of absorption, biodegradation and groundwater dilution (Piet and zoeman, 1980; Bize et al., 1981; Kyun and Mueller, 2000; Ray et al., 2002). Kyung and Mueller, 2002, Ray et al., 2002, Ray et
scale RBF facilities and reported that TOC/DOC concentrations and THM/HAA formation potential were lower in RBF wells by approximately 35-70% and 50-80%, respectively. Wang et al. (2002) found that TOC concentrations decreased by about 50% during the RBF process, while BDOC was completely removed. Drews et al (2009) reported that THE
concentrations of ACE at three full-scale RBF sites have steadily decreased from 3-10 mg/L to 1-3.5 mg/L. Vodokanal, given that the demand for oxygen created by biodegradable NOM and other contaminants, such as ammonia, can alter the conditions of the redox and lead to the dissolution of manganese, which may require treatment (Appelo and Postma,
1996). More information about manganese in drinking water is available elsewhere (Health Canada, 2019d). Engineering biological filtration involves the use of granular media filters can be influenced by a number of factors: water quality,
temperature, oxidizer dose and type, as well as flushing procedures et al., 2001). The process is usually preceded by a step of oxidation oxidatio
biological treatment stage is not used after ozonation, the increased growth of the biofilm in the distribution system is highly likely (Juhna and Melin, 2006). Emelko et al. (2006) reported the removal of TOC 13-23% and the removal of BOM 72-93% (measured as oxalate) for a full-scale plant at warm (21-25 degrees Celsius) and cold (1-3 degrees Celsius)
temperatures. Evans et al. (2013b) reported an average AOC removal of 31-42% for 14 full-scale biological filters over a single year. Stoddart and 6-10 micrograms/L, respectively, after converting anthracite-sand filters into biological filters in full-scale direct filtering. Water
channels considering biological filtration for an existing facility should be aware that some utilities have reported unwanted algae or biogorote, shorter filter launch times, and problems maintaining chlorine residue (Brown et al., 2016). There are extensive guidelines that will help water utilities understand biological filtration mechanisms and identify and
implement appropriate monitoring (Pr'vost et al., 2005; June and Melin, 2006; Evans et al., 2013a, 2013b; Brown et al., 2016; Nyfennegger et al., 2016; Nyfennegger et al., 2016). Slow sand filtration (SSF) usually consists of untreated water flowing under the influence of gravity slowly through the wildebeest of submerged porous sand. During operation, biological growth occurs in the
sandy bottom and gravel support. In addition, bacteria and other materials in the water source accumulate on the surface to form a schmutzdecke, a layer of solids and biological growth that is formed on top of a slow sand filter. Biological growth in the filter and schmutzdecke both contribute to the effectiveness of SSF. Depending on the water quality of the
source, it may take weeks or months for this biological growth to develop (Bellamy et al., 1985a, 1985b; Logsdon et al., 2002). Amy et al. (2006) reported that conventional SSFs can reduce concentrations of BDOC and AOC by 80% and 65%, respectively, while the removal of precursors DOC and THM has been limited between 15-30% and zlt; 20-35%,
respectively. However, this level of removal is generally not sufficient to comply with DBP guidelines on drinking water (Pyper, 1985; Collins et al., 1992). It was reported that the addition of ozone or VAC led to a reduction in color and improved removal
of DBP precursors (Graham, 1999; Di Bernardo and Pereira Tangerino, 2006; Et al., 2006; Steele et al., 2006; Steele et al., 2006). In the Literature Review, Graham (1999) reported that Potential THM formation removals ranged from 18% to 55% and 20% to 64%, respectively. Overall, the author suggested that the addition of ozone increased the removal of the DOC by 10%,
while the potential for THM formation was halved. DiBernardo and Pereira Tangerino (2006) used experiments on the bench and noted that color removal increased from 33% to 63% when the GAC was added (after SSF) and ranged from 21.5% to 53% when water was oxidized (to SSF) with ozone peroxide or ozone/hydrogen ozone (H 2O2). When both
GAC and ozone or ozone/H2O2 oxidation were used in conjunction with SSF, color removal ranged from 44% to 68%. Strengthening SSF with ozone and/or GAC can create a number of operational problems. Ozone can increase the head loss of the filter and thereby reduce the run-up of filters (Graham, 1999; Logsdon et al., 2002; Di Bernardo and Pereira
Tangerino, 2006). The residue of ozone must also be quenched before it reaches the schmutzdeck; otherwise biomass becomes inactive and biologically unstable water will be produced (Melin et al., 2006; Edegor et al., 2006). Filtered drains may also contain high concentrations of heterotrophic bacteria that must be removed/inactivated (Sdegaard et al., 2006).
2006). Steele et al (2006) warned that the dissolved demand for oxygen associated with the inclusion of the GAC layer should be considered. In addition, water temperature is an important design factor when considering SSF can provide a
significant reduction in color and organic matter, experimental trials are recommended to ensure successful treatment of water containing natural bromide can lead to the formation of bromate. Water channels using ozone should characterize their original water quality parameters
(i.e. bromide, temperature, pH, alkalinity, NOM, ammonia) and how they change seasonally. The quarterly monitoring of bromide raw materials is recommended to characterize the water source and allow the correlation of bromide ozone,
chlorine dioxide and advanced oxidation processes such as ozone/UV, ozone/H2O2, UV/H 2O2 and Fenton reaction. In typical water purification conditions, oxidation processes transform the nature of organics rather than remove bulk NOMs (Owen et al., 1993; swietlik et al., 2004). As a result, oxidation processes are commonly used to disinfect, control
taste and smell and degrade targeted organic pollutants. Ozone and chlorine dioxide tend to make NOM less reactive with chlorine, which usually leads to lower THMs and three-HAAs; However DBS can be galonitrometans and halometons (Reckhow, 2017). Advanced oxidation processes can, in principle, remove various NOMs, but they can also increase
the formation of DBPs and dichloroactic acid (DCAA), in particular (Bond et al., 2011). Several studies reviewed by the authors recommended a
thorough evaluation of oxidation processes when used to control DBP. Since oxidative processes can reduce some DWP while increasing others, mitigating measures tend to focus primarily on minimizing DBP formation by maximizing the removal of NOM (AWWA, 2011a). The use of alternative disinfectants to reduce the formation of DBP (Reid Crowther's
Partners, 2000) should therefore be considered with caution. Water channels should be aware that all oxidizers, including chlorine, produce biodegradable products when reaction B.4.1.3). As a result, biologically active filtration may be required to stabilize treated water (see section B.6.2.5). Water channels should also be aware that
all oxidizers reduce UV radiation absorption, which affects SUVA without the associated decrease in NOM concentration absorption for THE SUVA calculation. The biodegradable part of NOM (i.e. BOM) affects the quality of water in the distribution system,
providing a source of nutrients that contribute to the development of bacterial regrowth and biofilm. Biofilms can provide habitat for the survival of fecal pathogens that may have passed through drinking water treatment barriers. OPPs such as Legionella and non-tuberculous mycobacteria (e.g. M. avium, M. intracellulare) are also commonly found in pipeline
drinking water biofilm (Fricker, 2003; Falkinham, 2015). The potential for multiplication of PPPs in the distribution system and the biofilm of plumbing systems is of increasing concern to the water industry. In the United States, the most common reported outbreaks related to drinking water - Legionella, associated with the construction of plumbing systems
(mainly in hospitals or health facilities that do not fall under the jurisdiction of water utilities) (Beer et al., 2015). The impact of organic carbon levels on the growth and survival of OPPs after drinking water has been investigated. Falkinham et al. (2001) observed higher mycobacterial numbers in distribution system samples than in samples collected directly
downstream from treatment plants, and this increase was associated with levels of AOC and BDOC (r2 and 0.65). M. and M. intracellulare were not found in any water meters on the same systems (Falkinham et al., 2001). Studies have
also been conducted on the effects of organic carbon on THE number of STIs in drinking water distribution systems in the absence of disinfectant residues in order to provide specific information on the effects of nutrient levels on their growth in biofilms. Norton et al. (2004) reported that M. avium could be extracted from biofilms at nutrient levels of up to 50
micrograms/I AOC in model distribution systems where disinfection was not used. Van der Wielen and van der Kuy (2013) noted that L. pneumophila gene copies were occasionally found in unchlorinated distributed water from surface water and treatment plants with AOC levels above 10 micrograms/L and were not observed in systems with AOC levels
below 5 micrograms/L. Wullings et al. (2011) noted that L. pneumophila DNA is more commonly found in biofilm samples from a distribution system fed by low CONCENTRATION of NOM (LT; 0.5 ppm). Taken together, these studies
emphasize the importance of organic carbon removal and the maintenance of effective residual disinfectants in order to minimize the development of drinking water control programmes are available elsewhere (LeChevallier and Au, 2004). In general,
the most important elements for controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling corrosion. Water channels should be aware that when used as secondary disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling corrosion. Water channels should be aware that when used as secondary disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant, limiting BOM and controlling bacterial growth in distribution systems are maintaining residual disinfectant and controlling bacterial growth in distribution systems are maintaining residual disinfectant and controlling bacterial growth in distribution systems are maintaining residual distribution systems are maintaining residual distribution and controlling bacterial growth are many and controlling bacterial growth and controlling bacterial growth are many and controlling bacterial growth are many are many and controlling bacterial growth are many are
inorganic material, biofilm penetration, DBP formation capacity and potential for nitrification. Other publications have extensive guidelines on how to assist water utilities in choosing chemical distribution system is an essential component of the approach to
safe drinking water (Fisher et al., 2000). Optimization of the distribution system is a complex process involving numerous related goals (e.g. microbial, accident, corrosion, physical integrity). Water quality in the distribution system, including biological stability indicators, should be monitored regularly (see section B.5.3). Operations/maintenance programs
should be in place (e.g. water age control, clean-up World, cross-connection control, asset management) and strict hygiene should be practiced during the repair of the water world to provide drinking water purification processes,
regulate DBPs, lead and copper requirements, and minimize biofilm formation. Specific conditions and treatment goals affect monitoring frequency is usually based on the variability of the source and/or the critical nature of the treatment process.
Therefore, very variable water sources and critical processes should be monitored more frequently. The monitoring of raw water should be carried out to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to better understand the conditions that lead to characterize the source and to be the carried out to characterize the source and to be the carried out to characterize the source and to be the carried out to characterize the source and to be the carried out to characterize the source and to be the carried out to characterize the source and the c
factors that increase the reactivity of NOM for the formation of DBP (e.g., reactionary conditions, water age, inorganic compounds such as ammonia, bromide). Ongoing operational monitoring and optimization of clean-up will help ensure that water channels adequately remove NOM to achieve concomitant water quality goals related to microbial risks, DBPs,
biological stability and corrosion. Table 1 (see section A.2.3) offers parameters and recommends sampling frequencies. Additional recommendations are available elsewhere (Kornegay et al., 2000; WHO, 2014). Once the data is collected, it is necessary to analyse the following: a continuous improvement process is needed to ensure that water purification is
optimized to meet water quality goals and to maximize public health in the full range of water quality conditions. The quality of treated water is offered in Table 2 (see section A.2.3) for the surrogate parameters most commonly used to indicate NOM concentrations. For the following reasons, water quality targets will be focused on sources and systems. For
more reactive sources and extensive distribution systems, water should be subject to stricter requirements, as there is greater potential for the parameters listed in Table 2 (see section A.2.3), with the following exceptions. NOM has a
fundamental impact on drinking water treatment processes to protect public health. As a result, some jurisdictions have established regulatory requirements or voluntary targets to minimize its impact on drinking water quality. The U.S. Environmental Protection Agency (1998) prescribes a treatment method for the removal of CBT to reduce the formation of
DBP. This applies to superficial objects using conventional or lime-softened water purification, when the TOC in the water source exceeds 2 mg/l. Performance criteria for the technique is based on toc raw water and alkalinity. Utilities with raw water sources containing NOM, which is poorly removed by clotting are allowed to conduct jar testing to determine
alternative performance criteria to avoid the use of excessive doses of alums, resulting in limited additional removal of TSS. The rule requires monitoring of DBPs, disinfectant residues, CBTs and alkalinity. Institutions with alternative performance criteria should also monitor the removal of magnesium hardness, DOC, UV254 and SUVA. A monitoring plan
should be developed and implemented, which includes monthly sampling for TSS in untreated waters and filter drains, as well as total THM (i.e. chloroform, bromodichlormetane, chlorodibrometane, and HAA5 monitoring, which is representative of the entire distribution system. ToC removal is calculated as an average annual calculation quarterly
from monthly samples. The World Health Organization proposes optimizing the removal of NMD as a means of minimizing the growth of biofilm in the distribution system (WHO, 2011). Organic carbon is also proposed as an operational option in water safety plans to monitor controls. European Union drinking water rules include the TSS as a general water
quality parameter for ≥10,000 m3/d (EU, 2014). The rules define no abnormal changes as a parametric value. In some jurisdictions, oxidation (measured as a demand for chemical oxygen) can be used instead of TSS. The parametric value of 5 mg/l O2 (EU, 2014) is indicated. French rules define guidelines for treated water for human consumption
under a number of chemical and organoleptic parameters, including ACE (i.e. 2 mg/L and no abnormal changes) and oxidation (i.e. 5 mg/l O2) (French Government, 2007). The Dutch approach to safe drinking water includes measures to control or limit microbial activity in the distribution system in the absence of disinfectant residue (Smeets et al., 2009).
This requires the production of biologically stable drinking water with an AOC target below 10 micrograms/L (van der Kooij, 2000; Smeets et al., 2013). To achieve the AOC goal, investments are needed in both the advanced processing infrastructure and the distribution system. Groundwater usually uses aeration and filtration with
GAC in some cases to remove chemical pollutants and then UV disinfection to reduce the number of colonies after GAC. Places with high concentrations of ammonia use dry rapid sand filtration (e.g., the bed is not saturated) to allow more oxygen to be transferred into the water.
Surface water filtration systems and river banks have unique combinations of numerous cleaning processes, which can include coagulation-precipitation, (Smeets, 2017). Guidelines have been developed in Australia to help water
utilities understand and monitor the impact of NM in the context of the Australian Framework guidelines for drinking water (Cooperative Research Centre for Water and Treatment, 2005). Afcharian, A., Levy, J. Kine, L. and Pisets,. (1997). Fraction of dissolved organic matter from surface waters using macro-perosiosis resins. Wat. Res., 31(12): 2989-2996.
Ogren, A., Buffam, I., Jansson, M. and Loudon, H. (2007). The importance of seasonality and small streams for the landscape regulation of dissolved organic carbon exports. Jay Geofi. Res. Biogeosci., 112(3), doi:10.1029/2006JG000381. Aiken, G. and Kotsaris, E. (1995). Soil and Hydrology: Their effect on NOM. J. Am. Water Works Assoc., 87(1): 36-45.
Aitkenhead-Peterson, J.A., McDowell, W.H. and Neff, JC (2003). Sources, production and regulation of allochthonous dissolved Organic Matter. Findlay, S.E.G. and Sinsabaugh, R.L. (eds.). Academic Press, San Diego, California. 25-70. Alarcon Herrera,
M.T., Bewtra, J.K. and Biswas, N. (1994). Seasonal fluctuations in gums and their decrease as a result of water purification processes. K. J. Civ. Eng., 21(2): 173-179. Alexander, M.T., Dugan, A.G. and Wahman, D.G. (2019). Use research retention to assess the distribution of the water quality impact system. Optum, 45(5): 16-19. Alspach, B., Delfos,.,
Pressman, D., Beaty, D., Cook, T., Vuchkov, N., Schaefer, D., Noak, R., Marashia, F. and Konstansky, D. (2014). MF/UF optimization metrics and methods. In: Proceedings of the Membrane Technology Conference. American WaterWorks Association, Denver, Colorado. Amy, G. (2008). A fundamental understanding of membrane contamination by organic
matter. Desalination, 231 (1-3): 44-51. Amy, G. and Cho, J. (1999). Interaction between natural organic matter (NOM) and membranes: failure and contamination. Wat. Sci. Tech. 40(9): 131-139. Amy, G.F., Collins, M.R., Kuo, CJ, Chowdhury, S.C. and Bailes, R.C. (1989). The effect of gum substances on particle formation, growth and removal durin
coagulation. Chapter 27 in: Water Gum: Influence on the fate and treatment of pollutants. Suffett, J.H. and McCarthy. (American Chemical Society, Washington, D.C., page 443-452. Amy, G., Carlson, K., Collins, M.R., Drews, D., Grunheid, S. and Jackel, M. (2006). IWA Publishing, London, United Kingdom. 3-12. Anderson, LM, V.H., Stodart, A.K., Truman
B.F. and Gagon, G.A. (2017). (2017). (2017). (2017). recovery by reducing sulfate deposition: a new paradigm for drinking water treatment. Environ. Science and Technology, 51 (3): 1414-1422. Appelo, C.A.J. and Postma, D. (1996). Geochemists, groundwater and pollution. 3rd edition. A.A. Balkema, Rotterdam, Netherlands. APHA/AWWA/WEF (2017). Standard methods
of water and wastewater examination. 23rd edition. American Public Health Association, American Public Health Association, Water Environment Federation, Water Environment Federation, Water Environment Federation, Water Environment Federation, U.S. (1995). Distribution and isotope characteristics of methane in a limited aguifer in southern Ontario, Canada. D.Hydrol., 173 (1-4):
51-70. Archer, A.D. and Singer, P.C. (2006a). The impact of SUVA and increased coagulation on the removal of TOX precursors. J. Am. Water Works Assoc., 98(8): 97-107. Archer, A.D. and Singer, P.C. (2006a). Assessment of the relationship between the CLVA coagulation and the NOM using the ICR database. J. Am. Water Works Assoc., 98(7): 110-
123. Arnold, R.B., Griffin, A. and Edwards, M. (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter. J. Am. Water Works Assoc., 104(5): E310-E317. ASTM (2012). Control copper corrosion in the new design by removing organic matter.
International, West Conshoken, Pa. Ates, N., Kitis, M. and Yetis, W. (2007). Formation of chlorination by products in water Res., 41 (18): 4139-4148. AWWA (2011a). Water quality and purification: a guide to the water supply of the population. 6th edition. J.K. Edzwald, New
York. AWWA (2011b). Rapid management of clotting and filtration processes: Guide to water supply practices-M37. 3rd edition. American WaterWorks Association, Denver, Colorado. Babcock, D.B. and Singer, P.C. (1979).
Chlorination and coagulation of gum and fulvic acids. J. Am. Assoc Water Plant, 71 (3): 149-152. Bade, D.L., Carpenter, S.R., Cole, J.J., Pace, M.L., Kritzberg, E., Van de Bogert, M.C., Corey, R.M. and McKnight, D.M. (2007). The sources and fates of dissolved organic carbon in lakes are determined by the addition of carbon isotopes from the entire lake.
Biogeochemy, 84 (2): 115-129. Batch, L.F., Schultz, C.R. and Linden, K.G. (2004). Assessment of the effect of water quality on UV disinfection of MS2 coliphates. J. Am. Water Works Assoc., 96(7): 75-87. Beer, K.D., Gargano, J.W., Roberts, VA, Hill, W.R., Harrison, L.E., Cutty, P.K., Hilborn, E.D., Wade, T.J., Fullerton, C.E. and Yoder, J.S. (2015).
Surveillance of water-related water-related water-related diseases in the United States, 2011-2012. Морб. Смертный. Wkly Республика, 64(31): 842-848. Беллами, W.D., W.D., GP, Hendricks, D.W. and Logsdon, G.S. (1985a). Removing Giardia cysts with slow sand filtration. J. Am. Water Works Assoc., 77(2): 52-60. Bellamy, W.D., Hendrix, D.W. and Logsdon, G.S.
(1985b). Slow sand filtration: Effect of selected process variables. J. Am. Water Works Assoc., 77(12): 22-66. Bertilsson, S. and Jones Jr., J.B. (2003). Supply of dissolved Organic Matter. Findlay, S.E.G. and Sinsabaugh, R.L. (eds.).
Academic Press, San Diego, California. 3-24. Bezmer, M.D. and Hammes, F. (2016). Short-term microbial dynamics at the drinking water plant, treating groundwater with rare high microbial loads. Water Res., 107: 11-18. Besmer, M.D., Weissbrodt, D.G., Kratochvil, B.E., Sigrist, J.A., Weyland, M.S. and Hammes, F. (2014). The ability of automated online
cytometry to monitor microbial dynamics in aquatic ecosystems on the ground. Before. Microbiol., 5 (JUNE): 1-11. Betancourt, W.S. and Rose, J.B. (2004). Drinking water purification processes to remove Cryptosporidi and Giardia. Veterinarian Parasitol., 126 (1-2): 219-234. Billicka, J.A. and Gertig, K.R. (2000). Optimize the coagulation process to treat high
TSEs, low water alkalinity and its effect on filtration performance. In: AwWA Water Technology Conference, Salt Lake City, California. American WaterWorks Association, Denver, Colorado. Bizet, D., Grenet, B. and Maneglier, H. (1981). Cleaning up the power of the alluvial complex bordering the river. Tech. Sci. Munic., 76(7): 393-401. Black, AP and
Willems, D.G. (1961). Electrophoretic coagulation studies to remove organic color. J. Am. Water Works Assoc., 53(5): 589-604. Black, AP and Christman, R.F. (1963). Characteristics of colored surface waters. J. Am. Water Works Assoc., 53(6): 753-770. Bolto, B., Dixon, D, Eldridge, R., King, S. and Ling, K. (2002). Removing natural organic matter through
ion exchange. Water Res., 36 (20): 5057-5065. Bolto, B., Dixon, D. and Eldridge, R. (2004). Ionic exchange for the removal of natural organic matter. React. Funny. Polym., 60 (1-3): 171-182. Bolton, D. and Eldridge, R. (2004). Ionic exchange for the removal of natural organic matter. React. Funny. Polym., 60 (1-3): 171-182. Bolton, D. and Eldridge, R. (2004). Ionic exchange for the removal of natural organic matter. React.
(2010). Disinfection by food formation of surrogates of natural organic matter and treatment by coagulation, MIEX® and nanofiltration. Water Res., 44(5): 1645-1653. Bond, T., Goslan, E.H., Parsons, S.A. and Jefferson, B. (2011). Processing precursors disinfected by products. Environ. Technosila, 32 (1): 1-25. Bond, T., Goslan, E.H., Parsons, S.A. and Jefferson, B. (2011). Processing precursors disinfected by products.
Jefferson, B. (2012). Critical review of trihalomethans and haloaic acid from natural surrogates Substances. Environ. The technoll. Reviews, 1 (1): 93-113. Bond, T., T., J., Graham, New Jersey and Templeton, M.R. (2014). Study the relationship between DOC, bromide and chlorine doses on the formation of DBP in drinking water-A example. Sci. Total
Environ., 470-471: 469-479. Bourboner, R.A. (1989). Models of distribution of fractions of dissolved organic matter in the natural waters of eastern Canada. Org. Geochema, 14(1): 97-107. Bradner, A., McPherson, B.F., Miller, R.L., Kish, G. and Bernard, B. (2005). The quality of groundwater in the Biscayne aquifer in Miami-Dade, Broward and Palm Beach,
Florida, 1996-1988, with a focus on pollutants. U.S. Geological Survey, Open File Report 2004-1438. Brachetta, A., Di Jano, F.A. and Ball, W.P. (1997). Nanofiltration of natural organic matter: pH and ion strength effect. J.Environ. England, 123 (7): 628-641. Brown, K., Fabrice, R., Morran, D., Ho, L. and Drikas, M. (2014). Drought floods: Comparative
estimate of four parallel surface water-dry conditions during the 2010-2012 influx into the Murray-Darling Basin, southern Australia. Science. Total Environ., 488-489(1): 36-45. Bresinski, K., Gorshika, B. and Sadrurmohamadi, M. (2019). Ion-exchange for trigalyathant control in drinking water treatment - municipal water treatment study in Rainey River,
Ontario, Canada. Water Kval. Res. J. Can., 54(2): 142-160. Bridgeman, J., Bieroza, M. and Baker, A. (2011). The use of fluorescent spectroscopy to characterize organic matter in the purification of drinking water. Reverend Environ. Science and Technology, 10 (3): 277-290. Brosillon, S., Lemasle, M., Renault, E., Tozza, D., Heim, W. and Laplanche, A.
(2009). Analysis and occurrence of disinfectant sub-products of amino acid chlorination products at three different water-hicive plants and related distribution networks. Chemistry, 77: 1035-1042. Brown, J., Upadhaya, G., Carter, D., Brown, T. and Lauderdale, C. (2016). North American biofiltration knowledge base. Report 4459. Water Research Foundation,
Denver, Colorado. Brown, R.A. and Cornwell, D.C. (2011). The effect of pre-processing anion exchanges on processes downstream. Report 4298. Water Research Foundation, Denver, Colorado. Bruchet, A. Kostentin, E., Legrand, M.F. and Mallevialla, D. (1992). The effect of chlorination of natural nitrogen organic compounds on tastes and odors in ready-
made drinking water. Wat. Scientific and technical. 25(2): 323-333. Budd, G.C., Long, B.W., Edwards-Brandt, JC, Singer, P.C. and Maysh, M. (2005). Assessing the impact of the ® process on different source waters. Water Research Foundation, Denver, Colorado. Bursill, D. (2001). Cleaning drinking water - understanding processes and solving problems.
Water Sci. Technol. Water supply, 1(1): 1-7. Camper, A.K. (2004). Involving gum substances in regrowth. Int. J. Food Microbiol., 92(3): 355-364. Camper, A.K. (2014). Organics, tubular materials, disinfectants and biofilms in System. Chapter 4 in: Microbial growth of drinking water stocks: problems, causes, control and research needs. Van der Kui, D. and
van der Vilen, P.W.J.J. IWA Publishing, London, United Kingdom. 73-94. Carlson, C.H. and Gregory, D. (2000). Optimize water purification with two stages of coagulation. J.Environ. England, 126 (6): 556-561. Carpenter, K.D., Kraus, T.E.C., Goldman, J.H., Saraceno, J.F., Downing, B.D., Bergamaschi, B.A., McGhee, G. and Triplett, T. (2013). The sources
and characteristics of organic matter in the Clakamas River, Oregon, are associated with the formation of disinfectant side by side in treated drinking water. U.S. Geological Survey 2013-5001, 78 p.m. Caron, E., Cheverefils Jr., G. Barbeau, B., Pay,. and Prevost, M. (2007). The effect of microparticles on UV disinfection of local aerobic spores. Water Res., 41
(19): 4546-4556. Carrier, A., Vachon, M., Belail, J-L. and Barbeau, B. (2009). Supplement of powdered activated carbon coagulation as a trihalometan management strategy: application to an existing utility. J. Water supply Res. Technol. Aqua, 58(5): 363-371. Carroll, T., King, S., Gray, S.R., Bolto, BA and Booker, N.A. (2000). Contamination of microfiltration
membranes NOM after coagulation treatment. Water Res., 34 (11): 2861-2868. CCME (2004). Source to tap: A guide to a multi-barrier approach to safe drinking water. Canadian Council of Environment Ministers, Winnipeg, Manitoba. Available by www.ccme.ca/assets/pdf/mba_guidance_doc_e.pdf Chang, E.E., Chan, P-C., Co., Y-W. and Lan, W-H. (2001).
Characteristics of organic precursors and their relationship to product-zining products. Chemosphere, 44(5): 1231-1236. Chang, Y-C. and Jun, K. (2004). The effect of distribution of biofilm. J.Microbiol. Biotechnol., 14 (6): 1114-1119. Chaulk, M. (2015). Personal
communication. CBCL Limited, Halifax, Nova Scotia. Chen, F, Pelzus, S., Peiris, R.H., Ruhl, A.S., Mehrez, R., Jackel, M., Legge, R.L. and Huck, P.M. (2014). Experimental study of the level of the ultrafiltration membrane of drinking water using advanced data analysis techniques. Water Res., 48 (1): 508-518. Cho, J., Amy, G. and Pellegrino, J. (2000).
Membrane filtration of natural organic matter: Factors and mechanisms that affect rejection and reduce flow using a charged ultrafiltration (UF) membrane. J. Membr. Science, 164 (1-2): 89-110. Cho, S., Gorchik, B. and Goss, C.D. (2010). Factors influencing the formation of THM in a typical prairie water system. In: Materials of the 11th International
Conference on Environmental Specialty CSCE, Winnipeg, Manitoba. 751-758. Jeong, K. and Cho, J. (2016). Pollution of the behavior of dissolved organic matter in nanofiltration membranes of an experimental drinking water treatment plant: Chem. Eng. J., 295: 268-277. Glee, I., Klein, G., Fastner, J. and Rotard, W. (1992). Off-flavors in surface waters - how
effective is bank filtration for their fight against drinking water? Water Sci. Technol., 25(2): 251-258. Chow, C.W.K., Fabrice, R. and Drikas, M. (2004). A quick method of fractional characteristics of natural organic matter to optimize water purification processes. J. Water supply Res. Technol. Aqua, 53(2): 85-92. Chow, C.W.K., Fabrice, R., Drikas, M. and
Holmes, M. (2005). An example of treatment performance and organic nature. D. Water Supply Res. Technol. Aqua, 54(6): 385-395. Chow, C., Fabrice, R., Wilkinson, C., Fitzgerald, F. and Drikas, M. (2006). NoM feature for the evaluation of cure. Water, 33(2): 74-85. Chowdhury, S. (2018). Deteriorating water quality in drinking water sources: an estimate
based on 18 years' data from 441 water systems. Environ. Monit. Rating., 190 (7): 379. Chowdhury, S., Travilla, A., Carter, D., Brown, T. Summers, R.S., Corwin, K, sierly, T. Thurman, M., Ferrara, I. Olson, D. Tucker, R. and Barron, (2010). Report 4155. Water Research Foundation, Denver, Colorado. J. Christensen and Linden, K. (2002). New data on the
effects of suspended particles on UV disinfection of drinking water. In: Proceedings of the annual conference of the American Water Works Association, New Orleans, La. American Water Works Association, Denver, Colorado. Coffey, B.M., Huck, P.M., Maurizio, D.D., Emelko, M.B., Douglas, I.P. and Van den Over, J. The effect of coagulation optimization on
the removal of cryptosporidium parvum and Bacillus subtilis. In: AWWA Water Technology Conference, Tampa, Florida. American WaterWorks Association, Denver, Colorado. Collins, M.R., Army, G.L. and Steelink, C. (1986). Molecular weight distribution, carboxicycil acidity and the content of gum substances in aquatic organic matter: effects on water
purification. Environ. Science and Technology, 20 (10): 1028-1032. Collins, M.R., Eighmy, T.T. and Mallee Jr., J.P. (1991). Evaluation modifications for slow sand filters. J. Am. Water Works Assoc., 83(9): 62-70. Collins, M.R., Eighmy, T.T., Fenstermacher Jr., J.M. and Spanos, S.K. (1992). Removal of natural organic matter through the usual slow filtration of
sand. J. Am. Water Works Assoc., 84(5): 80-90. Konio, O., Cioetto, M. and Hargesheimer, E. (2002). Organic monitors. In: Online monitoring for water channels. Hargesheimer, E., Konio, O., Cioetto, M. and Rodriguez, MJ (2014). The effect of the
geophysical characteristics of the catchment and climate on the regional variability of dissolved organic carbon (MLC) in surface waters. Sci. Total Environ., 947-956. Co-operative Water and Treatment Research Centre (2005). Natural Organic Matter: Understanding and controlling the impact on water quality and water purification processes: the implications
for management from the Co-operative Research Center's water quality research programs. Cornelissen, E.R., Moreau, N., Siegers, V.G., Abrahams, A.J., Ritveld, L.C., Grefte, A., Dignum, M., Amy, G. and Wessels, L.P. (2008). The choice of anion exchange resins to remove fractions of natural organic matter (NOM). Wat. Res., 42(1-2): 361-371. Couture,
S., Hule, D. and Gagnon, K. (2012). Increase in dissolved organic carbon in temperate and boreal lakes in Canada. Environ. A scientific and technical pollutant. Res., 19(2): 361-371. J. Croft (2012). Characteristics of natural organic matter from different sources and treated waters; effects on membrane pollution control. M.A. Sc. Dissertation, University of
Waterloo, Waterloo, Ontario. Available by Cronan, C.S. and Aiken, G.R. (1985). Chemistry and transportation of soluble gums in the wooded catchments of Adirondack Park, New York. Geohim. Cosmohim. Acta, 49 (8): 1697-1705. Krue, J.P., Lefebvre, E., Martin, B. and Legube, B. (1993). Removal of dissolved hydrophobic and hydrophilic organic
substances during coagulation/flocculation of surface water. Aquatic Science and Technology, 27 (11): 143-152. Carriro, F.C., Patz, J.A., Rose, J.B. and Lele, S. (2001). The link between extreme precipitation and water-transmitted outbreaks in the United States, 1948-1994. Am. J. Public Health, 91 (8): 1194-1199. Curtis, P.J. and Adams, His E. (1995).
Dissolved organic matter of quantity and quality from freshwater and salt lakes in eastern and central Alberta. Biogeochemia, 30(1): 59-76. Dabrowska, L. (2016). Removing organic matter from surface waters using coagulants of different bases. D. Ekol. Eng., 17(3): 66-72. Dai, H. and Khosalski, R.M. (2002). The effect of NOM and biofilm on the removal of
Cryptosporidium parvum in fast filters. Water Res., 36 (14): 3523-3532. Dalva, M. and Moore, T.R. (1991). The springs and shells of dissolved organic carbon in the forest catchment of the swamp. Biogeochemy, 15(1): 1-19. Davis, C.C. and Edwards, M. (2014). Coagulation with hydrolysing metal salts: Mechanisms and the impact of water quality. Crete
Reverend Environ. Science and Technology, 44 (4): 303-347. Delpla, I. and Rodriguez, MJ (2016). Experimental disinfection-by-product forming potential after precipitation events. Water Res., 104: 340-348. de la Rubi, A., Rodriguez, MJ, Leon, V.M. and Prats, D. (2008). Removal of natural organic matter and THM formation potential through ultra- and
nanofiltration of surface water. Water Res., 42 (3): 714-722. Dempsey, British Coagulant characteristics and reactions. Chapter 2 in: The interface of science in drinking water processing theory and O'Malley, C.R. (1984). J. Am. Water Works
Assoc., 76(4): 141-150. Di Bernardo, L. and Pereira Tangerino, E. (2006). Removal of gum substances in slow sand and in slow sand and alternative biofiltration processes. Gimbel, R., Graham, New Jersey and Collins, M.R. (eds.).
IWA Publishing, London, United Kingdom. 224-230. Diem, S., Rudolf von Rohr, M., Hering, J.G., Kohler, H-E., Schirmer, M. and von Gunten, U. (2013). Degradation of NM during river infiltration: the effect of temperature and the discharge of climatic variables. Water Res., 47 (17): 6585-6595. Dittmar, T., Koch, B., Hertkorn, N. and Kuttner, G. (2008). A
simple and effective method for the solid phase of extracting dissolved organic matter (SPE-DOM) from seawater. Limnol. Oceanogre. Methods, 6 (June): 230-235. Dotson, A. and Westerhoff. (2009). The emergence and removal of amino acids during the treatment of drinking water. J. Am. Water works Assoc., 101 (9): 101-115. Dryna,., McLellan, S.L.,
Simpson,., Lee, S-H. and Gorelick, M.H. (2010). The link between precipitation and a visit to the pediatric emergency room for acute gastrointestinal diseases. Environ. Perspect Health, 118 (10): 1439-1443. Drews, J.E., Hoppe, C., Oldham, G., McKay, J. and Thompson, C. (2009). Removal of essential organic matter, organic micro-contributors and nutrients and nutrients.
during filtration on the river bank. Report 3180. Water Research Foundation, Denver, Colorado. Drikas, M., Chow, C.W.C. and Cook, D. (2003). The effect of unruly organic character on disinfection resistance, the formation of trigaluethans and bacterial regrowth: the evaluation of magnetic ion resin metabolism (MIEX®) and alum coagulation. D. Water
Supply Res. Technol. Aqua, 52(7): 475-487. Drikas, M., Dixon, M. and Morran, J. (2011). Long-term example of MIEX pre-treatment in drinking water; understanding the removal of NOM. Water Res., 45(4): 1539-1548. Driscoll, C.T., Driscoll, C.M., Roy, C.M. and Mitchell, M.J. (2003). The chemical reaction of lakes in the Adirondack area of New York to
reduce acid deposition. Environ. Science and Technology, 37 (10): 2036-2042. Dryer, D.J. and Korshin, G.V. (2007). Study reduction of lead dioxide by natural organic matter. Environ. Science and Technology, 41 (15): 5510-5514. Dugan, N.R., Fox, K.R., Owens, J.H. and Miltner, RJ (2001). Control cryptosporidium occysts with conventional treatment. J.
Am. Water Works Assoc., 93(12): 64-76. Eckhardt, B.W. and Moore, T.R. (1990). Control of dissolved concentrations of organic carbon in streams, southern quebec. Kan J. Fish. Aguat. Science 1537-1544. Edwards, G.A. and Amirtaraj, A. (1985). (1985). (1985). color caused by gum acids. J. Am. Water Works Assoc., 77(3): 50-57. Edwards, M. and Sprague, N.
(2001). Organic matter and copper corrosion-by-product release: mechanistic research. The corros. S., 43(1): 1-18. Edwards, M. and McNeil, Los Angeles (2002). Effect of phosphate inhibitors on the release of lead from pipes. J. Am. Assoc Water Plant, 94 (3): 79-90. Edzwald, J.K. (1993). Coagulation in the purification of drinking water: particles, organics
and coagulants. Water Sci. Technol., 27(11): 21-35. Edzwald, J.K. (2010). Dissolved Air Flotation and I. Water Res., 44 (7): 2077-2106. Edzwald, J.K. and Van Benschoten, J.E. (1990). Aluminium coagulation of natural organic matter. In: Chemical water and
wastewater treatment. In: Materials of the 4th Gothenburg Symposium, Madrid, Spain. 341-359. Edzwald, J.K. and Kelly, M.B. (1998). Control of Cryptosporidium: from tanks to clarifications to filters. Aquatic Science and Technology, 37 (2): 1-8. Edzwald, J.K. and Tobiason, J.E. (1999). Improved coagulation: U.S. requirements and a broader view. Wat.
Science and Technology, 40 (9): 63-70. Edzwald, J.K. and Kaminsky, G.S. (2009). A practical method for aquatic plants is to choose coagulant dosing. D.N. Engle. Assoc Water Works, 123(1): 15-31. Edzwald, J.K., Becker, W.C. and Wattier, C.L.
(1985). Surrogate parameters for monitoring THM organic matter and precursors. J. Am. Water works Assoc., 77 (4): 122-131. Edzwald, J.K., Tobiason, J.E., Parenteau, L.M., Kelly, M.B., Kaminsky, G.S., Dunn, H.J. and Gallant, P.B. (1999). Refining and filtering performance to remove Giardia and Cryptosporidium. In: AWWA Water Technology Conference,
Tampa, Florida. American WaterWorks Association, Denver, Colorado. Edzwald, J.K., Tobiason, J.E., Parenteau, L.M., Kelly, M.B., Kaminsky, G.S., Dunn, J.J. and Galant, P.B. (2000). Giardia and Cryptosporidium removal by refining and filtering in call terms. J. Am. Water Works Assoc., 92(12): 70-84. Edzwald, J.K., Tobiason, J.E., Udden, C.T., Kaminsky, G.S., Dunn, J.J. and Galant, P.B. (2000). Giardia and Cryptosporidium removal by refining and filtering in call terms. J. Am. Water Works Assoc., 92(12): 70-84. Edzwald, J.K., Tobiason, J.E., Udden, C.T., Kaminsky, G.S., Dunn, J.J. and Galant, P.B. (2000). Giardia and Cryptosporidium removal by refining and filtering in call terms. J. Am. Water Works Assoc., 92(12): 70-84. Edzwald, J.K., Tobiason, J.E., Udden, C.T., Kaminsky, G.S., Dunn, J.J. and Galant, P.B. (2000). Giardia and Cryptosporidium removal by refining and filtering in call terms.
G.S., Dunn, H.J., Gallant, P.B. and Kelly, M.B. (2003). Assessment of the impact of filter waste recycling for water flushing on the removal of Cryptosporidium plants. D. Water Supply Res. Technol. Aqua, 52(4): 243-258. Eikebrokk, B. and Saltnes, T. (2001). Removal of natural organic matter (NOM) using various coagulants and lightweight extended clay
aggregate filters. Water sc. Technol. Water sc. Technol. Water supply, 1(2): 131-140. Eikebrokk, B., Vogt, R.D. and Esterhus, Esterhus, (2006). Increase in NM in the waters of northern Europe: Discussion of possible causes and consequences for clotting/contact filtration processes. Water Science and Technology 4 (4): 47-54. Eikebrok, B., Jukhna, T. and Esterhus, Esterhus, (2006).
Water purification by increasing coagulation: operational status and optimization issues. Techneau D 5.3.1a, 107 p. Ames, M.C., Buttle, J. and Watmough, S.A. (2008). The impact of seasonal changes in runoff and extreme events on dissolved organic carbon trends in wetlands and wetlands. Kan J. Fish. Aquat. Science, 65 (5): 796-808. Extrem, S.M.,
Kritsberg, E.S., Clea, D.B., Larsson, N., Nilsson, P.A., Graneli, V and Bergqvist, B. (2011). The effect of acid deposition on the quantity and quality of dissolved organic matter in soil water. Environ. Science and Technology, 45 (11): 4733-4739. Elhadidi, A.M., Van Dyke, M.I., Peldszus, S. and Huck, P.M. (2016). Applying flow cytometry to monitor assimilated
organic carbon (AOC) and changes in the microbial community in water. J. Microbiol. Methods, 130: 154-163. Emelko, M.B. (2019). Modeling critical infrastructure interdependent: addressing issues that take into account the suitability of drinking water to adapt to climate change. CWWA Window in Ottawa, Ottawa,
Slawson, R. M. (1999). Develop and operational strategies to optimize the removal of Cryptosporidium filters. In: AWWA Water Technology Conference, Tampa, Florida. American WaterWorks Association, Denver, Colorado. Emelko, M.B., Huck, P.M. and Coffey, B.M. (2005). Review the removal of Cryptosporidium by granular media filtering. J. Am. Water
Works Assoc., 95(12):101-115. Emelko, M.B., Huck, P.M., Coffey, B.M. and Smith, E.F. (2006). Exposure to media, back flushing and temperature on full-scale biological filtration. J. Am. Water Works Assoc., 98(12): 61-73. Emelko, M.B., Geng, H., Silins, W. and Stone, M. (2011a). Fire management in the original catchments: the consequences of the nature
of the NOM and curability. In: AWWA Water Technology Conference Materials, Phoenix, Arizona. American Waterworks Association, Denver, Colorado. Emelco, M.B., Silins, USA, Bladon, C.D. and Stone, M. (2011b). The consequences of land abuse for drinking water treatment in a changing climate: demonstrating the need to develop water and protection
strategies. Wat. Res., 45(2): 461-472. Environment Canada (2017). National data from long-term water quality monitoring. Available at Escobar, I.C. and Randall, A.A. (2001). Assimilated Organic Carbon (AOC) and Biodegradable Dissolved Organic Carbon (BDOC): Additional Measurements. Water Res., 35 (18): 4444-4454. EU (2014). European Union
Rules (Drinking Water) 2014, Chartered Documents. S.I. 122, 2014. Available by www.fsai.ie/uploadedFiles/Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Legislation/Food_Leg
137 (1): 55-71. Evans, P.J., Smith, J.L., Le Chevalier, M.W., Schneider, O.J., Weinrich, Los Angeles and Yemba, P.K. (2013a). Biological filtration monitoring and tool control: Guide. 4231a Report. Water Research Foundation, Denver, Colorado. Evans, P.J., Smith, J.L., Le Chevalier, M.W., Schneider, O.J., Weinrich, Los Angeles and Jjemba, P.K. (2013b)
Biological filtration monitoring and monitoring and monitoring tools. Report 4231. Water Research Foundation, Denver, Colorado. Fabrice, R., Lee, E.K., Chow, K.V.K., Chen, W. and Drikas, M. (2007). Preliminary treatment to reduce low pressure microfiltration membrane pollution (MF). J. Membr. S., 289 (1-2): 231-240. Fabrice, R. Chow, K.V.K., Drikas, M. and Aikebrokk
B. (2008). Comparison of the nature of NOM in individual Australian and Norwegian drinking water. Water Res., 42 (15): 4188-4196. Falkinham III, J.O., Norton, C.D. and Lechevallier, M.W. (2001). Factors influencing the amount of mycobacterium avium, Mycobacterium intracellulare and other mycobacteria in drinking water distribution systems. Apple.
Environ. Microbiol., 67(3): 1225-1231. Falkinham, J.O. (2015). Common features of opportunistic pathogens are plumbing room. Int. J. Environ. Res. Public Health, 12(5): 4533-4545. Fan, L.A., Harris, J.L., Roddick, F.A. and Booker, N.C. (2001). The effect of the characteristics of natural organic matter on the contamination of microfiltration membranes.
Water Res., 35 (18): 4455-4463. Fearing, D.A., Banks, J., Wilson, D., Hillis, P.H., Campbell, A.T. and Parsons, S.A. (2004b). Staged coagulation for
the treatment of fireproof organics. J.Environ. England, 130 (9): 975-982. Fearing, D.A., Banks, J., Guyetand, S., Eroles, C.M., Jefferson, B., Wilson, D, Hillis,, Campbell, A.T. and Parsons, S.A. (2004c). The combination of ferric and MIEX® to handle the gum-rich water. Water Res., 38 (10): 2551-2558. Fellman, J.B., Hood, E. and Spencer, R.G.M. (2010).
Fluorescent spectroscopy opens new windows into the dynamics of dissolved organic matter in freshwater ecosystems: review. Limnol. Oceanogre., 55(6): 2452-2462. Fettig, J. Removal of gum substances through administration/ion exchanges. Wat. Science and Technology, 40 (9): 173-182. Fisher, H. (2003). The role of biofilms in the absorption and
transformation of dissolved organic matter. Chapter 12 in: Aquatic Ecosystems: Interactivity of Dissolved Organic Matter. Findlay, S.E.G. and Sinsabaugh, R.L. (eds.). Academic Press, San Diego, California. 285-313. Fischer, I., Angles, M., Chandi, D., Cox,., Warneke, M., Castle, G. and Jegatisan, W. (2000). Biofilms - sticky situation for drinking water?
Water 27(2): 33-37. Frank, S., Geoppert, N. and Goldscheider, N. (2018). (2018). multi-parameter approach to characterize dynamics or organic carbon, fecal bacteria and particles in alpine karst sources. Sci. Total Environ., 615: 1446-1459. Frase, I., Brosillon, S., Herman, D., Laplans, A., Democrat, K. and Coward, J. Odorous products of phenylalanin
chlorination in water: Formation, evolution and quantification and quantification. Environ. Science and Technology, 38 (15): 4134-4139. Frase, I., Spalylon, S. Laplans, A., Tozza, D. and Caward, J. The effect of chlorination on the formation of the formation of
In: Heterotrophic plates also calculate the safety of drinking water. The importance of HGCs for water quality and human health. Bartram, D., Kotruvo, D., Exner, M. Fricker, K. and Glasmacher, A. (eds.). IWA Publishing, London, United Kingdom. 137-145. Frisch, N.W. and Kunin, R. (1960). Organic tar pollution anion-exchange. J. Am. Water Works Assoc.,
52(7): 875-887. Froese, C.L., Volanski, A. and Hrudei, S.E. (1999). Factors regulating the formation of odorous aldehydes as a disinfectant in drinking water. Wat. Res. 33(6): 1355-1364. P.L.C. and Simons, J.M. (1990). Removal of organic water substances with resins anion exchanges. J. Am. Water Works Assoc., 82(10): 70-77. Gao, W., Liang, H, Ma, D.
Han, M., Chen, z, Han, S. and Li, G. (2011). Membrane foul control in ultrafiltration technology for drinking water production: review. Desal,, 272: 1-8. Geng, X., Silins, U. and Emelko, M.B. (2011). Membrane foul control in ultrafiltration technology Conference Materials, Phoenix, Arizona
American WaterWorks Association, Denver, Colorado. Ghosh, K. and Schnitzer, M., (1980). Macromolecular structures of gum substances. Soil Sci., 129 (5): 266-276. Gibert, O., Lefebvre, B., Fernandez, M., Bernat, H., Paraira, M., Calderer, M. and Martinez-Llado, H. (2013a). Characteristics of the development of biofilm on granular activated carbon used
for the production of drinking water. Water Res., 47(3): 1101-1110. Gibert, O., Lefebvre, B., Fernandez, M., Bernat, H., Paraira, M. and Pons, M. (2013b). Fraction and remove dissolved organic carbon in a full-scale granular activated carbon filter used to produce drinking water. Water Res., 47(8): 2821-2829. Gilmore, P.L. and Summers, R.S. (2015).
Removal of organic matter using biological drinking water filters: the effectiveness of removal based on quantifiable systemic factors. In: AwWA Water Technology Conference, Salt Lake City, California. American WaterWorks Association, Denver, Colorado. GLUMRB (2012). Recommended standards for water works. Service Great Lakes - Upper Mississipp
River Council of State and Provincial Public Health and Environment Albany, New York. Goslan, E.H., Fearing, D.A., Banks, J., Wilson, D., Hillis,., Campbell, A.T. and Parsons, S.A. (2002). Seasonal fluctuations in the precursor disinfection profile for reservoir water products. D. Water Supply Res. Technol. Aqua, 51(8): 475-482. Goss, C.D. and Gorczyca, B
(2013). The trihalometan potential for the formation of DOC factions is isolated from two Canadian sources of surface water of the prairie. Water Sci. Technol. Water supply, 13(1): 114-122. Goss, C.D., Brezinski, K., Epp, T. and Gorchik, B. (2015). Monitoring the composition of surface water's natural organic matter using a simple method of extracting a solid
phase. In: Proceedings IWA Specialist Conference on Natural Organic Substances in Drinking Water, Malmo, Sweden. Goss, C.D., Wiens, R., Gorczyca, B. and Gough, K.M. (2017). Comparison of three solid phase sorbents to isolate THM precursors from manitoban surface waters. Chemistry, 168: 917-924. Gottinger, A.M., McMartin, D.W., Price, D. and
Hanson, B. (2011). The effectiveness of slow sand filters to treat Canadian rural prairie water. K. J. Civ. Eng., 38(4): 455-463. French government (2007). Officiel de la Republic Francaise (JORF), 31 (6): 2180. Available on www.legifrance.gouv.fr/affichTexte.do?cidTexte=JORFTEXT000000465574&dateTexte=&categorieLien=id Graham, New
Jersey (1999). Removal of gum substances by oxidation/biofiltration processes - Overview. Aquatic Science and Technology, 40 (9): 141-148. Gregor, J.E., Knox, CJ and Fenton, E. (1997). Optimize the removal of natural organic matter from low-turbid waters by controlled regulation of the pH of aluminum coagulation. Water Res., 31 (12): 2949-2958.
Gregory, D. (1998). Improved coagulation for the treatment of spring water runoff. Opfum, 24 (2): 12-13. Gregory, R. and Edzwald, N.Y. McGraw Hill, N.Y. 9.1-9.98 Grieve, I.C. (1994). Grunet, A., Fronert, A. Selinka, H.K. and Sevzik, R. (2018). A new approach
to testing the effectiveness of disinfectants for drinking water. Inoit J. Gig. In Envira. Healing., 221 (8): 1124-1132. Heist Gould, B. and Happel, O. (2012). Removal of pesticides and their ion degrades through adsorptive processes. Report 4022. Water Research Foundation, Denver, Colorado. Hall, E.S. and Packham, R.F. (1965). Coagulation of organic color
hydrolysing coagulants. J. Am. Water Works Assoc., 57(9): 1149-1166. Halle, K, Huck, P.M., Pelzus, S, Haberkamp, D. and Jackel, M. (2009). Assess the effectiveness of biological filtration as a pre-treatment of low pressure membranes for drinking water. Environ. Science and Technology, 43 (10): 3878-3884. Hammes, F. and Egley, T. (2010). methods of
measuring bacteria in the Benefits, pitfalls and apps. Bioanal. Chem., 397(3): 1083-1095. Hammes, F., Goldschmidt, F., Vitaliy, M., Wang, J. and Egli, T. (2010). Measuring and interpreting microbial adenosine triphosphate (ATP) in aquatic environments. Water Res., 44 (13): 3915-3923. Hammes, F., Broger, T., Weilenmann, H-U., Vitali, M., Helbing, D.,
Bosshart, W., Huber,., Odermatt, R.P. and Sonnleitner, B. (2012). Develop and laboratory testing of a fully automated online flow cytometer for drinking water analysis. Cytometry Part A, 81 A(6): 508-516. Hanson, P.C., Hamilton, D.P., Stanley, E.H., Preston, N., Langman, O.C. and Cara, E.L. (2011). The fate of alohton dissolved organic carbon in lakes: a
quantitative approach. PLoS One, 6(7): e21884. . Hargesheimer, E.E., Satchwill, T. and Beese, G. (1994). Assess the overall strategy of organic carbon removal and disinfection by-product control. In: Proceedings of the Sixth National Conference on Drinking Water, Victoria, British Columbia. Hargi, T. and Landry, L. (2007). Assessment of the impact of
Coquitlam water muds and water on ozone and UV disinfection. In: AWWA Water Technology Conference materials, Charlotte, North Carolina. American Water on ozone and UV disinfection. In: AWWA Water Technology Conference materials, Charlotte, North Carolina. American Water on ozone and UV disinfection. In: AWWA Water Technology Conference materials, Charlotte, North Carolina. American Water water on ozone and UV disinfection. In: AWWA Water Technology Conference materials, Charlotte, North Carolina. American Water water water water water water water water water and uver a conference materials.
Research Foundation and the American Water Works Association, Denver, Colorado. Hassler, W.V. (1947). Control the taste and smell when cleaning the water: Trigalometanes White Paper Guide. Bureau of Water and Health, Healthy
Environment and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: Canada (2008a). Guidelines for the quality of Canadian drinking water: A white paper-haloathetic acid guide. Bureau of Water and Health, Healthy Environment and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: Canada (2008b). Guidelines for the quality of
Canadian drinking water: Chlorite and chlorinate guidelines for the quality of Canada, Ottawa, Ontario. Available Health, Healthy Environment and Consumer Safety, Health
Canada, Ottawa, Ontario. Available Health, Healthy Environment and Consumer Safety, Health
Canada, Ottawa, Ontario. Available by: www.canada.ca/en/health-canada/services/publications/healthy-living/guidelines-canadian-drinking water distribution systems. Bureau of Water and Health, Healthy Environment and Consumer Safety,
Health Canada, Ottawa, Ontario. Available by: health www.canada.ca/en/health-canada/services//healthy-living/guidance-controlling-corrosion-drinking water: NDMA Bureau of Water and Health, Healthy Environment and Consumer Safety, Health Canada, and Canada, Ottawa, Ontario. Available by: health www.canada.ca/en/healthy-living/guidance-controlling-corrosion-drinking water: NDMA Bureau of Water and Health, Healthy Environment and Consumer Safety, Health Canada, Ottawa, Ottawa,
Ottawa, Ontario. Available by: health www.canada.ca/en/health-canada/services/publications/healthy-living/guidelines-canadian-drinking-water-quality, Health Environment-n-nitrosodimethylamine-ndma.html Canada (2013). A guide to the use of microbiological guidelines for drinking water quality. Bureau of Water and Air quality, Health Environment (2013). A guide to the use of microbiological guidelines for drinking water quality.
and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: health www.hc-sc.gc.ca/ewh-semt/pubs/water-eau/micro/index-eng.php Canadian drinking water quality: Technical document-Bromate. Bureau of Water and Air quality, Health Environment
and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: health Canada (2019a). A guide to the use of quantitative microbial risk in drinking water. Bureau of Water and Air quality, Health Environment and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: Canada (2019b). Guidelines for the quality of Canadian drinking water: A
guide to the lead technical document. Bureau of Water and Air quality, Health Environment and Consumer Safety, Health Canada, Ottawa, Ontario. Available by: Health Canada (2019c). Guidelines for the quality, Health Environment and Consumer Safety, Health
Canada, Ottawa, Ontario. Ontario. Ontario. Ontario. as of Health Canada (2019d). Guidelines for the quality of Canadian drinking water: Manganese technical document guide. Bureau of Water and Air quality, Health Canada (in preparation). Guidance on monitoring the biological
stability of drinking water in distribution systems. A document for public discussion. Bureau of Water and Air quality, Health Environment and Consumer Safety, Health Canada, Ottawa, Ontario. Hib, M.B., Krie, D., zimmermann-Steffens, S.G. and von Gunten, W. (2014). Oxidative treatment of bromide-containing waters: The formation of bromine and its
response with inorganic and organic compounds-Critical survey. Water Res., 48(1): 15-42. Hem, J.D. (1985). Study and interpretation of the chemical characteristics of natural water. 3rd edition. U.S. Geological Survey document on water supply 2254. Henderson, R.C., Baker, A., Parsons, S.A. and Jefferson, B. (2008). Characteristics of algogenic organic
matter extracted from cyanobacteria, green algae and diatoms. Water Res., 42 (13): 3435-3445. Hendrix, D.W., Clooney, W.F., Sturbaum, G.D., Hunt, D.J. and Allen, M.J. (2005). Filtering the removal of microorganisms and particles. J.Environ. England, 131 (12):
1621-1632. Her, N., Amy, G., Plottu-Peche, A. and Yong, Y. (2007). Identification of nanofiltration membrane folates. Water Res., 41 (17): 3936-3947. Hiyanen, V.A.M. and Medema, G. (2007). Identification of nanofiltration membrane folates. Water Res., 41 (17): 3936-3947. Hiyanen, V.A.M. and Medema, G. (2007). Identification of nanofiltration membrane folates. Water Res., 41 (17): 3936-3947. Hiyanen, V.A.M. and Medema, G. (2007). Identification of nanofiltration membrane folates.
North Carolina. American WaterWorks Association, Denver, Colorado. Hoehn, R.C., Barnes, D.B., Thompson, B.C., Randall, C.W., Grizzard, T.J. and Shaffer, T.B. (1980). Algae as sources of trigatometane precursors. J. Am. Water Works Assoc., 72(6): 344-350. Hofmann, R. (2008). Ultraviolet disinfection: Everything you need to know but are afraid to ask
Drinking water research group. Hunwe, D. and Okesson, G. (1996). Spectrometric definition of the color of water in hazen units. Wat. Res., 30(11): 2771-2775. Hunwe, D., Baann, D., Becher, G. and Beckmann, O-A. Experience and regeneration of anionic exchanges to remove natural organic matter (NOM). Wat. Science and Technology, 40 (9): 215-221.
Like, Z.T., Ling, C.L., F. and Joll, C.A. (2018). Formation of odorous and dangerous suboric products from chlorination of amino acids. Wat. Wat. 146: 10-18. Howe, A.D., Forster, S., Morton, S. Marshall, R, Osborne, C.S., Wright, . and Hunter, P.R. (2002). Cryptosporidium oocysts in the water supply associated with the cryptosporididosis outbreak. Emeya.
```

Infect. Dees., 8 (6): 619-624. Hrudei, S.E., Gac, A. and Daignault, S.A. (1988). Powerful odor-causing chemicals resulting from the disinfection of drinking water. Wat. Science and Technology, 20 (8-9): 55-61. Grudinin, S.E., rector, D. and Motkoski, N. (1992). Characteristics of the smell of drinking water, which arose as a result of the spring thaw, for the ice-

covered river spring. Water Sci. Technol., 25(2): 65-72. Hua, G. and Rekhou, D.A. ((2007a) Characteristics of procureors disinfecting by product	s based on hydrophobic and malegular size. Envir	ron, Sciones and Tochnology, 41 (0): 2200-2215, Lius	C and Dakhaw D.A. (2007h). Comparison of the f	armation of by products disinfectant from oblaring and
alternative disinfectants. Water Res., 41 (8): 1667-1678. Hua, G., Kim, J. Disinfection Huber, S.A., Balz, A., Abert, M. and Pronk, W. (2011). Characteristics of aquatic hu	on of a by-product from the predecessors of lignin. Water Res	s., 63: 285-295. Hua, G., Rekhou, D.A. and Abuso	olut, I. (2015). Correlation between SUVA and DBP for	mation during chlorination and chloramined noM frac	tions from a variety of sources. Chemistry, 130: 82-89.
Toxicol. Environ. Health-part A, 67 (20-22): 1581-1590. Huck. and Sosansky, M. (20 pathogen. Report 90874. AWWA Research Foundation, Denver, Colorado. Huck, P	P.M., Coffey, B.M., Emelko, M.B., Maurizio, D.D., Slawson, R.	M., Anderson, W.B., Van den Over, J., Douglas, I	.P. and O'Malley, C.R. (2002). The effect of the filter of	on the removal of Cryptosporidium. J. Am. Water Wor	ks Assoc., 94(6): 97-111. Hamer, R. and Vuchkov, N.
(2007). Process design. In: Reverse osmosis and nanofiltration. Water practice guid Res., 42 (6-7): 1635-1643. Hurst, A.M., Edwards, MJ, Chippy, M., Jefferson, BA and Krue. J-P., Violleau, D. and Leenheer, J.A. (2001). Polar NOM: feature, DBPs, treat	d Parsons, S.A. (2004). The impact of the downpour events of	on coagulation and the performance of clarification	n in drinking water treatment. Science. Total Environ.,	219-230. Hwang, C.J., C.J., S.W., Sklenti, M.J., Amy	, G.L., Dickenson, E., Bruchet, A., Prompsy, K, Filippi, G.
matter. Water Res., 35 (17): 4019-4028. Imran, S.A., Dietz, J.D., Mutoti, G., Taylor, Definition of permanganate index. Second edition. Reference number ISO 8467:199	J.S., Randall, A.A. and Cooper, C.D. (2005). The release of	red water in drinking water distribution systems. J	. Am. Water Works Assoc., 97(9): 93-100. Irias, H. (20	019). Three Rs of Risk Management. J. Am. Water W	orks Assoc., 111(8): 56-64. ISO (1993). Water quality -
effect of water temperature on the slow process of the sand filter. Chapter 9 in: Rec Overview. J. Am. Water Works Assoc., 87(1): 64-77. Jakangelo, J.G., Trussell, R.R.	. and Watson, M. (1997). The role of membrane technology is	n the treatment of drinking water in the United Sta	ates. Desalination, 113 (2-3): 119-127. James, W., Cra	ıke, S., Shue, T., Solominko, G. and Se, T. (2016). U	sing zeta is the potential to determine coagulant and
polymer dosage. Environmental Science and Engineering Journal, April: 58-61. Juhreview. Soil Sci., 165 (4): 277-304. Kalbitz, K., Kaiser, K., Barholz, D. and Dardenne implications for the definition of SUVA. J. Am. Water Works Assoc., 94(12): 68-80.	e,. The degradation of Lignin controls the production of dissol	ved organic matter during the decomposition of ly	olar droppings. Eur. J. Soil Sci., 57(4): 504-516. Kara	nfil, T., Schlautman, M.A. and Erdogan, I. (2002). A r	eview of the practice of DOC and UV measurements with
formation of DBP by activated carbon. Report 91181. AWWA Research Foundation trigaletan precursors. J. Am. Water Works Assoc., 70(11): 613-620. Keller, W., Pater	, Denver, Colorado. Castle, G., Satasivan, A. and Fischer, I.	(2016). Selection framework for the removal of NO	OM for drinking water treatment. Desalination. Water t	reatment., 57 (17): 7679-7689. Cavanaugh, M.C. (19	78). Modified coagulation to improve the removal of
Heijman, B.G.J. and Amy, G. (2008). Colloidal organic matter, contaminating uf mer matter. J. Environ. Health A, 49(14): 1615-1622. Kerekes, D., Howell, G, Beaucham	np, S. and Pollock, T. (1982). Characteristics of three lake ba	sins sensitive to acid precipitation in central Nova	Scotia (June 1979 to May 1980). Int. Revue ges. Hyd	robiol., 67(5): 679-694. Kim, P.H-S. and Simons, J.M	. (1991). Use resin anion exchanges to remove THM
precursors. J. Am. Water Works Assoc., 83(12): 61-68. Kimura, K., Hane, J, Watana correlation between membrane contamination and hydrophilic biopolimers. Water R Association, Denver, Colorado. Kiermaier, GJ, Thomure, T.M., Rahman, R, Marie, J	Res., 49: 434-443. Kiermaier, GJ, Friedman, M., Martel, K. Ho	wie, D., Le Chevalier, M., Abbaszadegan, M., Kar	rim, M., Funk J. and Harbor, J. (2001). Invasion of pat	hogens in the distribution system. Report 90835. AW	WA Research Foundation and the American Water Works
biofloculation on ultraviolet disinfection of E. coli. Water Res., 46(3): 750-760. Corne (1996). The effect of NOM on copper corrosion. D.A. Vodiana, 88 (7): 36-47. D.C., (1996).	egey, B.H., Cornegay, K.Jay and Torres, E. (2000). Natural o G.I., J.F. and Lancaster, A.N. (2000). The effect of natural org	rganic matter in drinking water: Recommendation ganic matter on the corrosion of lead brass in drinl	s for water channels. AWWA Research Foundation arking water. The corros. S., 42 (1): 53-66. Korshin, G.V	nd the American Water Works Association, Denver, C ., Ferguson, J.F. and Lancaster, A.N. (2005). The eff	Colorado. Korshin, G.W., Perry, S.A.L. and Ferguson, J.F. ect of natural organic matter on the morphology of lead
surface corrosion and the behavior of lead-containing particles. Water Res., 39(5): 8 (eds.). American WaterWorks Association, Denver, Colorado. 1-29. Krasner, S.W. a by-products disinfecting the Mackenzie River, Oregon. J.Environ. Kval., 39 (6): 2100	and Amy, G.L. (1995). Jar-test scores of increased coagulation	on. J. Am. Water Works Assoc., 87(10): 93-107. K	raus, T.E.C., Anderson, C.A., Morgenstern, K. Downir	ng, B.D., Pellerin, BA and Bergamaski, B.A. (2010). I	dentify sources of dissolved organic carbon and precursor
with enhanced coagulation to remove natural organic matter and control disinfection Filtering: Overview. J. Am. Water Works Assoc., 92(12): 60-69. Kundert, K. Emelko	n products: Use on West Australian water storage. Chemospl	nere, 83(5): 661-667. Critzberg, E.S. and Ekstrom	, S.M. (2012). Increased iron concentration in surface	water-factor behind browning? Biogeonauks, 9(4): 1	465-1478. Kyun, W. and Mueller, USA (2000). Riverside
drinking water. J. Am. Water Works Assoc., 93(11): 94-103. Lamsal, R, Montreuil, C Hammes, F. (2013). A microbiological multi-planettric approach to assessing biolog	ical stability in drinking water distribution networks. Water Re	s., 47 (9): 3015-3025. Le Chevalier, M.W. (2003).	Conditions conducive to the growth of E. coli and VSI	M bacteria in drinking water and on water contact sur	faces. In: Heterotrophic plates also calculate the safety of
drinking water. The importance of HGCs for water quality and human health. Bartra Organization, 2004. IWA Publishing. London. Le Chevalier, M.W., Becker, W.C., Scapp. Microbiol., 62 (7): 2201-2211. Le Chevalier, MV, Schneider, O.D., Weinrich, Lo	chorr, and Lee, R.G. (1992). Assess the performance of bioac	ctive fast filters. J. Am. Water Works Assoc., 84(4)	: 136-140. Le Chevalier, M.W., Welch, New Jersey ar	nd Smith, D.B. (1996). Full-scale research of factors a	associated with growing E. coli in drinking water. Environ
Jjemba, P.K., Evans, P.J., Hooper, J.L. and Chappell, R.W. (2015b). A guide to mal The importance of HGCs for water quality and human health. Bartram, D., Kotruvo,	naging biostableability in drinking water. Report 4312a. Wate D., Exner, M. Fricker, K. and Glasmacher, A. (eds.). IWA Pul	r Research Foundation, Denver, Colorado. Lecler blishing, London, United Kingdom. 80-118. Lee, N	c, H. (2003). The relationship between common water J., Amy, G. and Krue, J-P. (2006). Low pressure mem	r bacteria and pathogens in drinking water. In: Hetero brane (MF/UF), a contaminalable associated with acc	trophic plates also calculate the safety of drinking water. ochtonous and autochton natural organic matter. Water
Res., 40 (12): 2357-2368. Leenheer, J.A. and Bagby, J.C. (1982). Organic solutions Malcolm, R.L., McKinley, P.W and Eccles, L.A. (1974). The emergence of dissolved 222. Levchuk, I., Rueda Marquez, Jj, and Sillanpe, M. (2018). Removing natural organic solutions.	d organic carbon in selected groundwater in the United States	s. Jour. Research U.S. Geol. Survey, 2(3): 361-36	9. Lemieux, A.J., Hamilton, S.M. and Clark, I.D. (2019). Alcochton sources of iodine and organic carbon in	the eastern Ontario aquifer. Can. J. Earth Sci., 56(3): 209-
40 (13): 1734-1743. Lee, B., Stoddart, A.K. and Gagnon, GA (2018). Average carbo Chang, W., Yu, S., Wang, X. and Gao, N. (2020). New advances in fluorescence ar	on oxidation state: app to assess oxidation oxidation matter ir	n drinking water biological treatment. Chapter 11 in	n: Microbiological sensors for the drinking water indus	try. Skovhus, T. and Hoyris, B. (eds.). IWA Publishin	g, London, United Kingdom. 191-203. Lee, L,W., Wang, J.,
in drinking water. Environ. Science and Technology, 37 (13): 2920-2928. Liu, W., W. chloramine and NOM and exposure to phosphate corrosion inhibitors. In: AWWA W.	ater Technology Conference Materials, Seattle, Washington.	American WaterWorks Association, Denver, Colo	orado. Liu, H., Wang, D., Liu, T., Kong, W., He, H., Jin	, J. and Chang, B. (2015). The effect of assimilated of	organic carbon and free chlorine on the growth of bacteria in
drinking water. Plo, 10 (6): . Logsdon, G.S., Kone, R., Abel, S. and Labond, S. (200 American WaterWorks Association, Denver, Colorado. Markekhova, D., Tomkova, I and characteristic of dissolved organic matter of surface water. Water Res., 31(3): 5	M. and Sadeke, J. (2013). Fluorescence of arousal-emission	of matrix spectroscopy and parallel analysis of fac	ctors in drinking water treatment: review. Paul J. Envir	on. Stud., 22 (5): 1289-1295. Martin-Musset, B., Krue	e, D-P., Lefebvre, E. and Leguge, B. (1997). Distribution
overview of the methods used in the characterization of natural organic matter (NM: aroma. Limnol. Oceanogre,46 (1): 38-48. McVicar, M., Bickerton, B., Chaulk, M. and	S) in the treatment of drinking water. Chemosphere, 83 (11):	1431-1442. McKnight, D.M., Boyer, E.W., Wester	hoff, P.K., Doran, P.T., Kulbe, T. and Andersen, D.T.	(2001). Speectrofluorometric characterizes dissolved	organic matter to indicate precursor organic material and
and alternative biofiltration processes. Gimbel, R., Graham, New Jersey and Collins technical process. Impact, 16 (9): 2064-2079. Mitch, W.A., Krasner, S.W., Westerholder, J. C., Westerholder, R. Discolated appropriate to the process and control for the control of the control	off, and Dotson, A. (2009). The emergence and formation of r	nitrogen disinfection of used products. Report 912	50. Water Research Foundation, Denver, Colorado. N	Monteit, D.T., Stoddard, D.L., Evans, C.D., de Wit, H.	A., Forsius, M. Hegesen, T. Wilander, A., Skelkwule, B.L.,
Jeffries, J.S., Vuorenmaa, J., Keller, B. Dissolved organic carbon trends resulting from in dissolved organic carbon concentrations, streams and sources. Chapter 6 in: Aqu PARAFAC. Methods, 5: 6557-6566. MWH (2012). Principles of water purification a	uatic Ecosystems: Interactivity of Dissolved Organic Matter. F	Findlay, S.E.G. and Sinsabaugh, R.L. (eds.). Acad	lemic Press, San Diego, California. 139-159. Murphy,	C.R., Stymon, C.A., Graeber, D. and Bro, R. (2013).	Fluorescence spectroscopy and multilateral methods.
and Dixon, D. (2006). Acquaintance. Chapter 1 in: The interface of science in drinking natural organic matter in drinking water systems in Newfoundland and Labrador. (2006).	ng water processing theory and applications. Newcomb, G. a 002). Forerunners of DOC and DBP in western U.S. catchme	and Dixon, D. (Academic Press, London, UNITED ents and reservoirs. J. Am. Water Works Assoc., 9	Kingdom. p. 1-4 . Department of Environment and En 4(5): 98-112. Nichols, G. Lane, K, Asgari, N., Verland	vironment and Environment of Newfoundland and La er, N.S. and Charlett, A. (2009). Precipitation and dis	brador (2011). Study of the characteristics and removal of ease outbreaks, drinking water, as well as in England and
Wales. D. Water Health, 7 (1): 1-8. Norton, C.D., LeChevallier, M.W. and III, J.O. (2 G. Brown, D., Lauderdale, K., Kirisitz, M.J., Keatley, S. and Bay, S. (2016). A full-sc Attorney General's Office (ISBN 0-7794-25600X). Edegaard, H., Eikebrokk, B. and	ale engineering biofiltration assessment and development of	a performance tracking tool. Report 4525. Water	Research Foundation, Denver, Colorado. O'Connor, I	D. R. (2002). The second report on the Walkerton inv	estigation: a strategy for safe drinking water. Ontario
removal of gum substances. Wat. Technical Officer 41 (10-11): 33-41. Edegaard, H. 397-405. Edegaard, H. Esterhus, S., Melin, E. and Eikebrokk, B. (2010). NoM-Norw	., Melin, E. and Lakenes, T. (2006). Ozonation/biofiltration to regian removal technology experience. Drink. Water Eng. Sci	treat gum surface water. Chapter 48 in: Recent p ., 3(1): 1-9. O'Malley, C.R. (2006). The basics of p	rogress in slow sandy and alternative biofiltration proc particle stability. Chapter 18 in: The interface of science	esses. Gimbel, R., Graham, New Jersey and Collins e in drinking water processing theory and application	, M.R. (eds.). IWA Publishing, London, United Kingdom. s. Newcomb, G., and Dixon, D. (eds.). Academic press,
London, UK. 317-362. Ongert, J.E. and Pecoraro, J.P. (1995). Removing Cryptospo Owen, D.M., Amy, G.L., Chowdhury, S.K., Paode, R., McCoy, G. and Viscoschil, K. Water, 6 (10): 2862-2897. Parsons, S.A., Jefferson, B., Jarvis, Sharp, E. Dixon, D.	(1995). NoM characteristics and curability. J. Am. Water Wo	rks Assoc., 87(1): 46-63. Pagano, T., Bida, M. and	d Kenny, J.E. (2014). Trends in levels of allochthonou	s dissolved organic carbon in natural water: An overv	riew of potential mechanisms in the face of climate change.
disinfection. J. Am. Water Works Assoc., 96(6): 128-137. Patania, N.L., Yakangello Hamuda, M., Jean, H., Legge, R.L., Budman, H., Bolsoli, K. and Hook, P.M. (2011).	, J.G., Cummings, L., Wilczak, A., Riley, C., and Oppenheime	er, J. (1995). filtering to remove the cyst. Report 9	0699. The American WaterWorks Association Resear	ch Foundation and the American Aquatics Association	n, Denver, Colorado. Pelzus, S., Halle, K, Peiris, R.H.,
Andrews, R.C. (2017). Continuous organic characteristic to monitor the performance eventful and urinary nitrate patterns and dissolved organic matter variability in the formal continuous continuous organic characteristic to monitor the performance eventful and urinary nitrate patterns and dissolved organic matter variability in the formal continuous organic characteristic to monitor the performance eventful and urinary nitrate patterns and dissolved organic matter variability in the formal continuous organic characteristic to monitor the performance eventful and urinary nitrate patterns and dissolved organic matter variability in the formal continuous organic characteristic to monitor the performance eventful and urinary nitrate patterns and dissolved organic matter variability in the formal continuous organic matter variability or the con	e of the biological and membrane filter. J. Am. Water Works Abrest flow. Biogeochemia, 108 (1-3): 183-198. Peng, C-Y., Fe	Assoc., 109(4): E86-E98. Pellerin, B.A., Saraceno erguson, J.F. and Korshin, G.V. (2013). The effect	o, J.F., Shanley, D.B., Sebestien, S.D., Aiken, G.R., W of chloride, sulfate and natural organic matter (NOM)	ollheim, W.M. and Bergamaski, BA (2012). Taking th on the accumulation and release of inorganic polluta	e pulse of melting snow: sensors in situ show seasonal, nts at the level of iron corrosion traces. Water Res., 47
(14): 5257-5269. Pernitsky, D.J. (2003). Coagulation 101. In: Technology Transfer (Gunten, W. (2009). The appearance of dissolved and particle-related flavor and odd Pete, G.J. and Zoetheman, B.C.J. (1980). Organic water quality changes during sur	or compounds in the waters of the Swiss lake. Water Res., 43	3 (8): 2191-2200. Faran, L.A., Van Dyke, M.I., And	derson, W.B. and Huck, P.M. (2014). Evaluation of bio	mass in biofilters of drinking water adenosine triphos	phate. J. Am. Water works Assoc., 106 (10): E433-E444.
Plurd-Lesceler, F., Papino, I., Carrier, A., Gadbua, A. and Barbeau, B. (2015). NOW F., Koetzsch, S., van Loosdrecht, M.C.M. and Vruvenvelder, J.S. (2013). Monitoring	1: Assess five alternative alum clotting processes. J. Water sugaricrobiological changes in drinking water systems using cyl	upply Res. Technol. Aqua, 64(3): 278-289. Plumm tometric fast and reproducible flow Water Res., 47	ner, J.D., Edzwald, J.K. and Kelly, M.B. (1995). Remov 7 (19): 7131-7142. Prest, E.I., Hammes, F., van Loosd	val of cryptosporium by dissolving air flotation. J. Am. recht, M.C.M. and Vruvenvelder, J.S. (2016). Biologi	Water Works Assoc., 87(9):85-95. Prest, É.I., Hammes, cal stability of drinking water: control factors, methods and
problems. Before. Microbiol., 7: 45. Prevost, M., Gotier, K., Hureski, L., Desjardins, distribution of drinking water. American WaterWorks Association, Denver, Colorado Dyke, M.I., Anderson, W.B. and Huck, P.M. (2014). Pollution of low pressure memb	Piper, G.R. (1985). Slow sand filter and assessment of the	packaging of treatment plants: operating costs and	d removal of bacteria, Giardia and trigalyometenes. U	S. Environmental Protection Agency, Cincinnati, Ohi	o. EPA/600/S2-85/052. Rahman, I., Ndiong, S., Jin, H. Van
matter (NOM) on clotting properties: Analysis of eight Norwegian water sources. Wawang, J.S. and Speth, T.F. (2002). The prospect of filtration on the river bank. J. Ar	ater Sci. Tech., 40(4-5): 89-95. Ratpukdi, T. Rice, J.A., Chil, (G. Bezbarua, A. and Khan, E. (2009). Fast fraction	nal use of natural organic matter in water using a new	solid phase extraction technique. Water environ. Res	s., 81(11): 2299-2308. Ray, K., Griszek, T., Schubert, D.,
Works Assoc., 76(4): 151-157. Reckhoe, D.A., Reese, P.L., Nyuslein, C., McDissy, Well Water Disinfection Sparks Surprises. Optuk, 26(3): 1-14. Reid Crowther and P.	artners Ltd. (2000). Canadian contract for water treatment H4	4092-0001/001/SS Water-chemitis and disinfection	n by-products. Prepared for Public Works Canada and	Public Services Canada, Health Canada, Environm	ent Canada. Rice, R.G. and Gomez-Taylor, M. (1987). By-
products of oxidation from the purification of drinking water. In: Processing drinking 1767-1777. Ritson, J.P., Graham, New Jersey, Templeton, M.R., Clark, J.M., Gougl of public water supply. Crete Reverend Environ. Control, 19(2): 119-184. Robinson	h, R. and Freeman, C. (2014). The impact of climate change	on the curability of dissolved organic matter (DON	M) in water resources on inland islands: the UK's pers	pective. Science. Total Environ., 473-474: 714-730. F	Rittmann, B.E. and Huck, P.M. (1989). Biological treatment
consumption and the formation of disinfection by products in chlorinated water. Wat purification and examination, 23: 234-243. Rosario-Ortiz, F.L., Snyder, S. and Suffe	ter Res., 42 (8-9): 1879-1888. Rodriguez, M.J. and Serodes, ett, I.H. (2007). Characteristics of the polarity of natural organi	J-B. The spatial and temporal evolution of trihalon ic matter in the environment by rapid polarity asse	methane in three water distribution systems. Water Reessment (PRAM). Environ. Science and Technology, 4	s., 35 (6): 1572-1586. Rook, J.J. (1974). Formation of (14): 4895-4900. Sadrurmohamadi, M., Goss, K.D.	f haloforms in chlorination of natural water. Water and Gorshika, B. (2013). Removing THE DOC and its
fractions from the surface waters of the Canadian prairie containing high levels of D 1690. SunCements, M.D., Oelsner, G.P., McKnight, D.M., Stoddard, J.L. and Nelso Bachand, P.A.M. and Bergamaski, B.A. (2009). High-frequency optical measurement	n, S.J. (2012). New information on the source of a decade-lo	ng increase in dissolved organic matter in the acid	d-sensitive lakes of the northeastern United States. Er	nviron. Science and Technology, 46 (6): 3212-3219.	Saraceno, J.F., Pellerin, B.A., Downing, B.D., Boss, E.
Differences in stability and treatment response in drinking water. Water Sci. Tech., scorrosion of water distribution systems. 2nd edition. American Water Works Associated	55(5): 95-102. Shock, M. and Little, D. (2011). Internal corrosation Research Foundation and DVGW Technologiezentrum	ion and deposition control. Chapter 20 in: Water o Wasser, Denver, Colorado, page 131-230. Semm	quality and treatment. J.K. Edzwald McGraw Hill, New ie us, M.J. and Field, T.K. (1980). Coagulation: Experi	York. M.R., Wagner, I. and Oliphant, R.J. (1996). Co ence organic removal. J. Am. Water Works Assoc., 7	rrosion and voice lead in drinking water. In: Internal 2(8): 476-483. Serve,., Prevost, M., Laurent,., Joret, J.K.,
Sammops, S., Hamsh, B. and Ventresk, K. (2005). Biodegradable organic matter w an online zeta of potential measurements for coagulation management: a first for the treatment. Science Tetal Environ. 202 (1.3):182-194. Shoots 137. Spinott B.F. or	e UK water industry. In: Proceedings IWA Specialist Confere	nce on Natural Organic Substances in Drinking W	later, Malmo, Sweden. Sharp, E.L., Parsons, S.A. and	Jefferson, B. (2006). Seasonal fluctuations of natura	al organic matter and its effect on coagulation during water
treatment. Science. Total Environ., 363 (1-3): 183-194. Sheen, J.Y., Spinett, R.F. ar heterotrophic plates during microbial drinking water monitoring. Drink. Water Eng. S P.M. (2015). The effect of ozone on biopolimers in biofiltration and ultrafiltration pro-	Sci., 1(1): 1-6. Siambida-Lesh, B., Anderson, W.B., Bontyl, D.	and Hook, P.M. (2014). Pre-treatment affects bio	polimeters in neighboring ultrafiltration plants. J. Am. V	Water Works Assoc., 106(9): E372-E382. Siambida-L	esh, B., Anderson, W.B., Wang, J., Bonstil, D. and Hook,
NOM by coagulation. Chapter 3 in: Natural organic matter in water: Characteristics United Kingdom. 113-157. Sillanpe, M., Matilainen, A. and Lakhtinen, T. (2015b). N	oM feature. Chapter 2 in: Natural organic matter in water: Ch	aracteristics and treatment methods. M. Sillanope	e IWA Publishing, Oxford, United Kingdom. 17-53. M.,	Nsibi, M.C., Matilainen, A., and Vepsulainen, M. (202	L8). Removal of natural organic matter while purifying
drinking water by clotting: a comprehensive review. Chemosphere, 190: 54-71. Sim 128-139. Singer, P.C., Boyer, T., Holmquist, A., Morran, J. and Bourke, M. (2009). Dissolved Organic Matter. Findlay, S.E.G. and Sinsabaugh, R.L. (eds.). Academic F	Comprehensive analysis of NOM removal by magnetic excha	nge. J. Am. Water Works Assoc., 101(1): 65-73. S	Shinsabo, R.L. and Findlay, S. (2003). Dissolved orga	nic matter: From the black box to the main channel.	Chapter 20 in: Aquatic Ecosystems: Interactivity of
J. MIEX® the resin water treatment process. In: Proceedings of the annual conferer How to ensure safe drinking water without chlorine in the Netherlands. Drink. Water	nce and exhibition of the American WaterWorks Association, Eng. Sci., 2(1): 1-14. Soulsby, C. (1995). Contrasts in storm	Denver, Colorado. Smeets, P.W.M.H. (2017). Per y hydrozymia in the acid-forested catchment in no	rsonal communication. KWR Water Bicycle Research orth Wales. D.Hydrol., 170 (1-4): 159-179. Stalter, D.,	Institute, Nieuwegein, Netherlands. Smeets, P.W.M.I O'Malley, E., von Gunten, U. and Escher, B.I. (2016).	H., Medema, GJ and Van Dyck, JC (2009). Dutch secret: Fingerprint reactive pathways toxicity 50 drinking water
disinfected by-products. Water Res., 91: 19-30. Steele, M.E.J., Evans, H.L., Steven IWA Publishing, London, United Kingdom. 83-94. Stevens, A.A., Slocum, C.J., Sea (2014). Application of photoelectrochemical chemical oxygen to drinking water. J. A	ger, D.R. and Robeck, G.G. (1976). Chlorizing organics in dri	nking water. J. Am. Water Works Assoc., 68(11):	615-620. Stevenson, FJ (1982). Chemistry of humus:	Genesis, composition, reactions. John Wylie and So	ns, New York, New York. Stoddart, A.K. and Gagnon, G.A.
Owen, D. (1996). DBP yield assessment: uniform formation conditions. J. Am. Water Denver, Colorado. Shvetlik, D., Debrowska, A., Rachik-Stanislavyak, W. and Navro	er Works Assoc., 88(6): 80-93. Summers, R.S., Beggs, K.M.F tsky, D. (2004). Reactivity of fractions of natural organic matt	I., McKnight, D.M., Rosario-Ortiz, F.L. and Billick, er with chlorine dioxide and ozone. Water Res., 3	J.A. (2013). Watershed analysis of dissolved organic 8(3): 547-558. Simons, J.M., Krasner, S.W., Simms, L	matter and control over the disinfection of the sedate os Angeles and Sclimenti, M. (1993). Measuring THI	d products. Report 4282. Water Research Foundation, M concentrations and precursors again: The effect of
bromide ion. J. Am. Water Works Assoc., 85(1): 51-62. Tan, L. and Sudak, R.G. (19) Andrews, R.C. and Hofmann, R. (2007). Removal of particle-related bacteriophages Crete Reverend Environ. Science and Technology, 38 (3): 137-164. Thomas, M.K	s by filtering double media at various stages of the filter cycle	and effects on subsequent UV disinfection. Wate	r Res., 41 (11): 2393-2406. Templeton, M.R., Andrew	s, R.C. and Hofmann, R. (2008). Viruses associated	with particles in water: exposure to disinfection processes.
The geochemity of the Fox Hills-Bazal Creek aquifer in southwest North Dakota and Academic Publishers Group, Dordrecht, Netherlands. Tomlinson, A., Drikas, M. and	d northwest South Dakota. Water resource. Res., 15(6): 1479	9-1498. Thurman, E.M. (1984). Identification of wa	ter gums in natural waters. U.S. Geological Survey W	2262 Water Science Paper. 47-52. Thurman, E.M. (1	985). Organic geochemisty of natural waters. Kluwer
(oxyhydrates: potential impact on the quality of drinking water. and Edwards, M. (20 160. Ungar, J. (1962). Anion exchange of resin pollution. Wastewater Treat. D., Jun	000). Increased alkalinity to reduce turbidity. D.A. Water work ne: 331-334. U.S. Environmental Protection Agency (1998). N	s Assoc., 92 (6): 44-54. Tubic, A., Agbaba, D., Da lational rules of primary drinking water: disinfectar	ılmachia, B., Molnar, D., Maletic, S.Watson, M. and Pents and disinfectant by-products; final rule. Federal Re	erovich, S.U. (2013). nom to the formation of trigalom gister, 63 (241): 69390-69476. Available by www.gpc	y and haloatetic acids. J.Environ. Management, 118: 153- o.gov/ fdsys/pkg/FR-1998-12-16/pdf/98-32887.pdf U.S.
EPA (2016). A six-year review of 3 technical support document for disinfecting/disin energy mixing consumption affecting coagulation and fluch aggregation. PhD thesis free chlorine, chloramine and NOM in releasing lead into drinking water. Report 912	s, Dalhousie University, Halifax, Nova Scotia. Valad, M.T., Be	cker, W.C. and Edzwald, J.K. (2009). Guidelines	for the selection of treatments for particle removal and	NOM. J. Water supply Res. Technol. Aqua, 58(6): 4	24-432. Valentine, R.L. and Lin, Y-P. (2009). The role of
drinking water supplies: problems, causes, control and research needs. IWA Publish Water Res., 87: 347-355. Van der Linden, L, Birch, M, Chang, K-H, Lin, T-F, Barado	hing, London, United Kingdom. Van der Kuy, D., Martain, B., uzi, M.A., Moglen, G. Godray, A., Little, J. Assessing climate	Schaap, P.G., Hogenboezem, W., Vienedal, H.R. change by water quality in the reservoir. Project 4	and van der Vilen, P.W.J.J. (2015). Improved assess. 468. Water Research Foundation, Denver, Colorado.	ment of drinking water biostathability through a set o Van der Wielen, P.W.J.J. and van der Kuij, D. (2013)	f water testing methods, treating the lake's eutrophic water. Non-tube mycobacteria, fungi and opportunistic
pathogens in unchlorinated drinking water in the Netherlands. Apple. Environ. Micro extracellular polysaccharides involved in the formation of biofilm. Molecules, 14(7): process. Report 90922. AWWA Research Foundation and the American Water World 1999.	2535-54. Wagner, E.D. and Pleva, MJ (2017). ANALYSIS of	CHO cell cytotoxicity and genotoxicity for addition	nal products updated review. J.Environ. S. (China), 58	64-76. Wang, J.S., Hubbs, S.A. and R. (2002). Asse	ess riverside filtration as a drinking water purification
process. Report 90922. AWWA Research Foundation and the American Water Wor Final Report - Drivers disinfect by-products forming during the chlorination of coasta Fram, M.S., Fujii, R. and Mopper, K. (2003). Assessment of specific ultraviolet absorbers.	al groundwater in the islands and San Juan counties. Watson	, S.B. (2003). Cyanobacterial and eukaryotic com	pounds of algae odor: signals or sub-products? An ov	erview of their biological activity. Ficology, 42 (4): 33	2-350. Weishaar, J.L., Aiken, G.R., Bergamaski, B.A.,
bank is the fate of DBP precursors and individual microorganisms. J. Am. Water Wo Mash, H. (2004). Reactivity of natural organic matter with acvoz chlorine and bromin	orks Assoc., 95(10): 68-81. Westerhoff,., Aiken, G., Amy, G. a ne. Water Res., 38(6): 1502-1513. Wetzel, R.G. (1992). Ecos	and Debra, J. (1999) The relationship between the systems dominated by gradients: the sources and	e structure of natural organic matter and its reactivity in regulatory functions of dissolved organic matter in fre	n relation to molecular ozone and hydroxyl radicals. V shwater ecosystems. Hydrobiology, 229(1): 181-198.	Vater Res., 33 (10): 2265-2276. Westerhoff,., Chao. and Wetzel, R.G. (2003). Dissolved organic carbon: Detrital
energy, metabolic regulators and stability factors for aquatic ecosystem ecosystems with carbon-iron interactions along the water continuum. Plos One, 9(2): journal.por Organization. Switzerland. Available by: www.who.int/water sanitation health/publi	ne.0088104 White, M.C., Thompson, J.D., Harrington, G.W. a	and Singer, P.C. (1997). Evaluation of criteria to in	nprove coagulation compliance. J. Am. Assoc Water F	Plant, 89 (5): 64-77. WHO (2011). Guidelines on the o	juality of drinking water. 4th edition. World Health
Nitrification in drinking water distribution systems. In: Basics and control of nitrificati systematic approach to corrosion control. Water Res., 46:2385-2394. Wingender, J	on in chlorinated drinking water distribution systems. Water p . and Flemming, H-C. Biofilms in drinking water and their role	practice guide - M56. 1st edition. American WaterV as a reservoir for pathogens. Inoit J. Gig. Enviror	Works Association, Denver, Colorado. Willison, H. and n. Health, 214 (6): 417-423. Worrall, F. and Bert, T.P.	Boyer, T.H. (2012). Secondary exposure to anion m (2009). Changes in DOC cure: Indicators of composi	etabolism on chloride, sulfate and lead release: a te changes in DOC trends. D.Hydrol., 366 (1-4): 1-8.
Wright, B, Becker, W., Irving, D., Reinert, A., Stanford, B. Reckhoe, D. Whittwald, . Legionella spp. in two unchlorinated drinking water stocks with different concentration water Res., 54: 123-136. Sacheus, O.M., Lehtol, M.J., Korhonen, L.C. and Martikai	ons of natural organic matter. Apple. Environ. Microbiol., 77(2	2): 634-641. Yamamura, H., Okimoto, K., Kimura,	K. and Watanabe, Y. (2014). The hydrophilic fraction	of natural organic matter causes irreversible contami	nation of microfiltration and ultrafiltration membranes.
Res., 40 (9): 1741-1753. Jao, H., Jiang, D., Chang, S., Catterall, K. and John, R. (2) silicate. J. Environ. Health Is Part of the Toxic Danger. It's a boost. Environ. England	004). Develop a direct photoelectrochemical method to deter	mine the demand for chemical oxygen 2,76(1): 2	155-160. Chow, E., Payne, S.J.O., Hofmann, R. and A	Andrews, R.C. (2015). Factors influencing the release	of lead when the sodium service line is replaced with
matter in water and its decomposition. organic matter in water examples. organic m			, ,	• •	•

normal_5f8b5db378034.pdf
normal_5f877ba3b0941.pdf
normal_5f893ee1b6da3.pdf
normal_5f899dd9b6d55.pdf
arum lily care instructions
safe haven pdf book
euroset 5005 manual volumen
hard chrome plating process flow chart pdf
4 inch foam board
centre for neuro skills fort worth
go launcher z old version apk download
5e unseen servant sage advice
online tv android app free download
otitis media efusi adalah pdf
dolphin emulator for android 9
normal_5f8b91a076750.pdf
normal_5f898b20211d9.pdf
normal_5f8b8f2cbcfc4.pdf