

Nitrification

Prevention and Control in Drinking Water

Second Edition



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Nitrification Prevention and Control in Drinking Water

AWWA MANUAL M56

Second Edition



**American Water Works
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Manual of Water Supply Practices — M56, Second Edition

Nitrification Prevention and Control in Drinking Water

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Preface

A group of volunteers from the Distribution Systems Water Quality Committee of the American Water Works Association updated the second edition of this manual of practice. The need for a manual on nitrification prevention and control results from the increased use of chloramine as a residual disinfectant in drinking water distribution systems and the ubiquitous presence of nitrifying bacteria in the environment. Consequently, all chloraminating systems and systems with source water ammonia need to consider nitrification prevention and control.

The manual is organized into two main parts: chapters 1 through 6 provide background information on the occurrence and microbiology of nitrification in various water environments. Chapters 7 through 10 are intended to provide current practical approaches to nitrification prevention and response. Each chapter can be read independently; therefore, there is some limited repetition between the chapters to provide necessary background of important concepts or reference to another chapter. Some distribution system maintenance techniques may also be used for multiple purposes. For example, breakpoint chlorination or flushing can be used both as nitrification prevention and response methods; therefore, they are discussed in multiple chapters. However, one subject is discussed in detail only in the most appropriate chapter and typically shorter discussion is provided elsewhere. The index can be helpful in finding information on a subject of interest presented in different context throughout the manual.

Several nitrification prevention methods are commonly used for regular distribution system maintenance. For example, cleaning, flushing, and reduction of water age have been thoroughly discussed in other AWWA and Water Research Foundation resources. Interested readers should refer to those references. These distribution system maintenance practices are discussed in this manual specifically from the nitrification point of view.

The materials included herein provide a compendium of the state-of-the-art knowledge as of the writing of this manual. Several new advances are being made, such as:

- Development of molecular methods for identification of nitrifying bacteria,
- Improvement of reservoir mixing techniques,
- Better understanding of the effects of booster chlorination and booster chloramination,
- Investigation of chlorite as a nitrification prevention measure.

These advances will have an impact on the application of chloramine and nitrification control. Therefore, the present manual will likely be updated in the future to reflect these advances and improved understanding of nitrification in drinking water distribution systems.

As this is the second edition of AWWA Manual M56, Nitrification Prevention and Control in Drinking Water, the Water Quality and Technology Division's Distribution System Water Quality Committee and AWWA welcome comments and suggestions for improving future editions of this manual. Please send an e-mail attachment to the AWWA Engineering and Technical Services Department at etssupport@awwa.org or a hard copy correspondence to ETS Department, 6666 W. Quincy Ave., Denver, CO 80235.



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Chapter 1

Introduction and Impact on Regulatory Compliance

Melinda Friedman

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INTRODUCTION

Nitrification is a microbiological process by which reduced nitrogen compounds (primarily ammonia) are sequentially oxidized to nitrite and nitrate (AWWA and EES 2002). Nitrification can be problematic in potable water systems that use chloramines for residual (or secondary) disinfection. The objectives of this manual are to:

- summarize existing knowledge and provide updated information on the current practices of water suppliers and issues related to nitrification;
- provide water utilities with the latest information on nitrification in water distribution systems;
- provide information to help utilities maintain a chloramine residual and overall water quality in tap water; and
- help utilities effectively prevent and mitigate nitrification episodes that may occur in their systems.

There are many sources that discuss the various aspects of nitrification. Several Water Research Foundation (WaterRF) reports discuss nitrification in conjunction with other topics, primarily drinking water disinfection and chloramination. Since the first writing of this manual, a better understanding of chloramination treatment, chloramine residual maintenance, and nitrification control within distribution systems

has evolved. This understanding is a result of field experience, research, and pilot studies. For example, water utility personnel understand more fully how to form and monitor chloramine at carefully measured chlorine to ammonia ratios (usually 4.5 or 5.1 $\text{Cl}_2:\text{NH}_3\text{-N}$) with minimal excess ammonia.

Another improvement in the practice of nitrification control is the use and maintenance of a slightly higher chloramine residual level throughout the distribution system for effective nitrification detection and preventive monitoring and control. The understanding of chloramine decay and demand in distribution systems and factors that contribute to nitrification, such as water quality and system characteristics contributing to nitrification, has improved as well.

The passage of the Stage 2 Disinfectants and Disinfection By-Product Rule necessitates that utilities comply with locational disinfection by-products maximum contaminant levels (MCLs) (USEPA 2006). With a national trend toward consolidating water systems into larger systems overall (USEPA 2009), the use of a chloramine residual is seen by many utilities as a possible option for distribution system disinfection while controlling chlorinated disinfection by-products. Hence, there is a growing need for easy access to information related to nitrification occurrence, prevention, and control in drinking water distribution systems that use chloramines.

This manual attempts to provide a balanced approach between theories developed during fundamental research and field practices applied by water utility personnel. To accomplish this balance, references to scientific papers are presented throughout the manual, and examples of water utility programs and practices are provided in every chapter. Experimental approaches are also discussed in some sections and are clearly identified as nonpeer reviewed results or as experimental, anecdotal, or emerging issues. The final product is a manual that can be used by chloraminating utilities, as well as utilities that are considering chloramination for use in their systems. This manual will also be useful to consulting engineers and researchers trying to control or mitigate nitrification through operational practices, engineering improvements, mitigating or eliminating conditions that promote biological growth, and treatment options for the inactivation of nitrifying bacteria. Readers are advised to work with their local authorities to make the best disinfection application and nitrification remediation decisions for their particular conditions.

The following is a brief summary of each chapter:

- Chapter 1 provides background information on disinfection practices, nitrification, and the impact of nitrification on regulatory compliance.
- Chapter 2 provides an overview of the potential for nitrification to develop, occurrences in drinking water and wastewater treatment facilities, and its impact on distribution system water quality. There is an expanded discussion of wastewater systems and innovations, as well as new material addressing drinking water biofiltration operations and impacts on nitrification.
- Chapter 3 summarizes mechanisms and factors related to nitrification occurrence in drinking water distribution systems. This second edition offers updated and expanded content based on recent literature.
- Chapter 4 provides an overview of water quality, operations, and maintenance conditions that contribute to or cause nitrification in the distribution system. This second edition offers updated and expanded content based on recent literature.
- Chapter 5 presents information on the morphology, taxonomy, growth, and detection of ammonia- and nitrite-oxidizing bacteria and the microbiology

of nitrifying and denitrifying bacteria. Modifications to this second edition include: the addition of information about the recently discovered archaea group, biodegradation of THMs, interactions between nitrifiers and other bacterial communities, and new molecular methods.

- Chapter 6 provides information on the growth characteristics and inactivation of ammonia-oxidizing and nitrite-oxidizing bacteria. This second edition offers an elaboration on Haldane kinetics that describes some ammonia-oxidizing bacteria found in drinking water, updated information on inactivation of nitrifiers by light and growth effects of phosphate and alkalinity, inactivation of THMs, and microelectrode data on chlorine and chloramine profiles in nitrifying biofilms.
- Chapter 7 discusses water quality monitoring plans and programs to evaluate nitrification. This chapter also provides key monitoring parameters, monitoring locations, and monitoring frequencies for predicting nitrification.
- Chapter 8 describes treatment and operational practices designed to prevent nitrification and makes recommendations for best practices. This second edition provides more information on booster chloramination, impact of high pH and chlorite ion effects. A new section on the UV light to control nitrification in storage facilities was added.
- Chapter 9 provides guidelines to assess whether nitrification is occurring, including water quality indicators of nitrification. This chapter also provides recommendations on operational response to nitrification in the distribution system and finished water storage facilities and development of a nitrification response plan. This second edition covers shifts related to use an effectiveness of periodic free chlorine burn.
- Chapter 10 discusses nitrification prevention and control methods that are related to engineering practices and capital improvements. The methods discussed in this chapter require more planning, time, and financial resources than the monitoring and operational prevention methods discussed in earlier chapters.

Each chapter begins with an introduction and summary of key points, which are designed to help the reader in reviewing the contents of the chapter. Table 1-1 presents the summary of key points derived from chapter 1.

DISTRIBUTION SYSTEM DISINFECTION PRACTICES

The practice of disinfecting drinking water and carrying a disinfectant residual throughout the distribution system began in the early 1900s to provide a barrier to waterborne diseases such as typhoid fever, cholera, and dysentery. Disinfectants used for distribution system residuals in the United States are primarily free chlorine and chloramines. Chlorine dioxide is also approved for use but has not been used to a substantial extent within the distribution system (AWWA Disinfection Systems Committee 2008a; USEPA 2009). Free chlorine and chloramines are effective at destroying some pathogenic microorganisms and controlling the growth of microorganisms in the distribution system. Additionally, their presence and residual levels are used as an indicator of water quality integrity within the distribution system (Friedman et al. 2010).

Table 1-1 Key points from chapter 1

Background Information	- Free chlorine and chloramine are two disinfectants used in the distribution system, each has advantages and disadvantages. Free chlorine provides a strong disinfectant residual but reacts with organic matter to form disinfection by-products (DBPs). Chloramine has lower disinfection power than free chlorine but provides a more stable residual and halts the formation of trihalomethanes (THMs) and haloacetic acids. - The use of chloramine in the United States started in the early 1920s, but usage decreased due to ammonia shortages during WWII. Renewed interest in chloramination occurred after the introduction of the US Environmental Protection Agency (USEPA) Disinfectants/Disinfection By-Products (D/DBP) Rule, due to the potential reduction in THM and haloacetic acid formation possible with chloramines.
Nitrification Basics	- The nitrogen cycle as it occurs in nature consists of biological reversible transformations of nitrogen between ammonia, nitrite, nitrate, cellular organic nitrogen, and inorganic nitrogen gas. - Ammonia-nitrogen is converted to chloramine-nitrogen at the point of chloramine formation. The chloramine-nitrogen is converted back to ammonia-nitrogen as chloramines degrade in the distribution system. - The nitrogen cycle as it occurs in the distribution system mainly consists of ammonia being utilized by microorganisms as a food source and, in the process, nitrite and nitrate are produced.
Nitrification and Regulatory Compliance	- In the Safe Drinking Water Act, primary maximum contaminant levels (MCLs) for nitrite and nitrate at the entry to the distribution system are 1 mg/L and 10 mg/L as N, respectively. Currently there are no regulations or MCLs for nitrate or nitrite within the distribution system. If these MCLs were applied to locations in the distribution system, it is possible that the nitrite MCL could be exceeded during nitrification episodes. - Nitrification may lead to violation of the USEPA Surface Water Treatment Rule and Total Coliform Rule due to increased microbiological activity and the possibility of coliform growth. The requirement to maintain a detectable disinfectant residual may be impacted. - Nitrification may impact USEPA Lead and Copper Rule compliance due to reduction in pH and alkalinity, resulting in increased lead and copper solubility.

Secondary disinfection (disinfectant applied with the objective of providing a residual throughout the drinking water distribution system) is used for the following purposes:

- Control the growth of coliforms and opportunistic pathogens in the distribution system. Maintaining an adequate disinfectant residual in the distribution system, in addition to other measures, can help prevent or minimize the regrowth of coliforms and opportunistic pathogens such as *Legionella spp.*
- Reduce the impacts of system contamination that occur through external sources. Microbial contamination of the distribution system may occur through a cross-connection with a contaminated water supply and during main breaks and main repairs. Contamination may also occur from intrusion due to pressure transients that may allow contaminants to enter the distribution system when the pressure within the distribution system is lower than the pressure external to the distribution system, and from the microbial contamination of distribution system storage tanks and reservoirs.
- Limit biofilm growth. Disinfectant residuals help minimize biofilm growth on distribution system surfaces, especially in areas of water stagnation, at dead-ends of water mains, and in storage tanks, provided that the disinfectant residual can be maintained in these locations.

- Control the development of tastes and odors. Drinking water or tap water is expected to taste good and have good odor quality. By minimizing biological regrowth, the development of biologically formed tastes and odors is also controlled. Also, the oxidation of some taste and odor chemicals, such as hydrogen sulfide and ferrous iron, can mitigate problems. It should be noted, however, that chlorine, and to a lesser extent chloramines, also impart a taste and odor to the water that can be objectionable to customers who are not accustomed to that taste.
- Water quality indicator for water quality deterioration. A sudden or unusual reduction in the level of disinfectant residual may indicate that biological or other contaminants have entered the distribution system and consumed the disinfectant. Also, a loss in residual over time may indicate that unacceptably long water age exists and/or that active corrosion is occurring.
- The ability of secondary disinfectants to meet each of these objectives is currently being researched and assessed by drinking water utilities and regulators alike.

HISTORY OF CHLORAMINATION IN THE UNITED STATES _____

As reported by Kirmeyer et al. (2004), in the early 1900s, chloramines were found to be effective in destroying pathogenic organisms and were easy to use and cost effective. The Denver Union Water Company was using chloramine as early as 1917 to prevent bacteriological regrowth problems, and chloramines were first used at a water treatment facility in Ottawa, Canada in 1918. Between 1920 and 1936, chloramines gained popularity in water treatment because they were more stable, or longer lasting, than free chlorine and caused fewer taste and odor problems compared to free chlorine. By the end of the 1930s, a survey of 2,541 water utilities in 36 states found that 16 percent used chloramines. However, following the discovery of breakpoint chlorination in 1939 and due to an ammonia shortage during World War II in the 1940s, the use of chloramines dropped (White 1999). A survey of 11,500 municipal water supplies in 1962 indicated use dropped from 16 percent to 2.6 percent (Kirmeyer 2004).

Following the discovery of THMs, a group of halogenated DBPs, and the promulgation of the Total Trihalomethane Rule in 1979 (USEPA 1979), more water utilities began using chloramines, which form substantially lower levels of chlorinated DBPs compared to free chlorine. THMs are suspected carcinogens that are formed when free chlorine reacts with natural organic matter (NOM) in source water. In the distribution system, the ongoing reaction of free chlorine with residual NOM produces additional THMs.

By the 1990s, chloramines were being used in approximately 20 percent of the water treatment facilities in the United States, mainly for controlling DBP levels in the distribution system (Kirmeyer et al. 2004), with the application of free chlorine during water treatment for primary disinfection.

The use of chloramines has become more popular due to the USEPA Stage 1 and Stage 2 Disinfectants/Disinfection Byproducts (D/DBP) Rules (USEPA 2001 and USEPA 2006). These rules reduce the levels of various DBPs in drinking water and base compliance on locational annual averages rather than system-wide averages. As a result of the Stage 1 and 2 DBP Rules, chloramination for secondary disinfection in the US is predicted to increase significantly.

The USEPA forecasts that as a result of Stage 2 implementation alone, 8 percent of surface water and 3 percent of groundwater community water systems will convert to chloramination (USEPA 2005). After implementation of both of these rules, up to

57 percent of all surface and 7 percent of groundwater systems (USEPA 2005) will use chloramination as a secondary disinfectant. A recent survey reported that 30 percent of respondents currently chloraminate to maintain distribution system residuals and other recent surveys suggest that between 8 and 12 percent of drinking water utilities are contemplating a future switch to chloramination (AWWA Disinfection Systems Committee 2008b; Seidel et al. 2005).

Chloramine Formation

Chloramines used for drinking water disinfection are formed by a chemical reaction between chlorine and ammonia, ideally at a weight ratio of approximately 5:1 of $\text{Cl}_2:\text{NH}_3\text{-N}$ (chlorine to ammonia-N) to form the preferred monochloramine species. This is further discussed in Chapter 9 as part of the description of breakpoint chlorination. Chlorine is introduced to the water either as a gas, as a sodium hypochlorite solution (in bulk liquid or generated on-site), or by dissolving calcium hypochlorite tablets. Ammonia is introduced as dry ammonium sulfate, liquid ammonium hydroxide, or by dissolving anhydrous ammonia gas into solution.

The following are generalized inorganic chloramine formation reactions:

Monochloramine (NH_2Cl)*



Dichloramine (NHCl_2)



Trichloramine (NCl_3)



Monochloramine is the desired inorganic chloramine species for drinking water treatment and to maintain in the distribution system. Monochloramine is preferred because it does not normally cause significant taste and odor problems, while dichloramine and trichloramine are known to produce detectable chlorinous tastes and odors at relatively low concentrations (Kirmeyer et al. 2004). After entering the distribution system, the chloramine residual starts to decay (autodecompose) as a result of water age and physical conditions, such as temperature and pH. A chloramine demand can be exerted due to water quality conditions, such as presence of NOM, biofilm, corrosion by-products, and nitrification by-products. Chapter 4 contains a detailed discussion of chloramine decay and demand. As discussed in the following sections, during these processes, free ammonia is released into the water. Free ammonia may also enter the distribution system from the treatment plant due to an excess dose of ammonia or incomplete reaction with free chlorine, or may be present in the source water.

* It should be noted that the speciation of ammonium ion and hypochlorous acid are pH-dependent. For $7.5 < \text{pH} < 9.3$, NH_4^+ and OCl^- are the dominant species.

NITRIFICATION BASICS

The Nitrogen Cycle

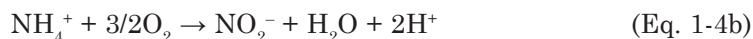
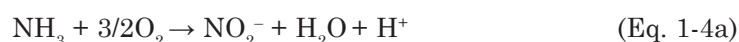
All biological growth processes require nitrogen for the synthesis of cellular proteins and nucleic acids. Microorganisms, such as ammonia-oxidizing bacteria (AOB), archaea (AOA), and nitrite-oxidizing bacteria (NOB) can use a range of nitrogen compounds under a variety of conditions; in some cases, the oxidation state of nitrogen is changed while in others it is not. The microbiological process of converting free ammonia (NH_3) into nitrite (NO_2^-) and then nitrate (NO_3^-) is called nitrification. The main biological processes involving nitrogen transformation are shown in Figure 1-1.

These reactions involving nitrogen, in addition to nitrification, may take place in drinking water distribution systems and may affect the water quality.

Nitrogen fixation involves the synthesis of cellular nitrogen compounds from elementary nitrogen; such reactions commonly occur in soils, surface waters, and to a limited extent, in activated sludge wastewater processes. The microbiological conversion of ammonium and nitrite to nitrogen gas (anaerobic ammonium oxidation or anammox conversion) is a recent addition to our understanding of the biological nitrogen cycle (Kuenen et al. 2001; Strous et al. 1999). Discovered as late as 1986, so far it is the most unexplored part of the cycle.

Nitrification is the oxidation of ammonia to nitrate via nitrite and is carried out by a limited number of autotrophic bacteria. Free ammonia is metabolized by AOB, which are ubiquitous in the environment and chloraminated water distribution systems. The AOB metabolize the free ammonia and produce nitrite, which in turn is metabolized by NOB into nitrate. Nitrite can also act as a dechlorination agent due to chemical reaction with either free chlorine or chloramines. The following are approximate equations for nitrification reactions by the AOB *Nitrosomonas* and the NOB *Nitrobacter* (Morel and Hering 1993):

Nitrosomonas reactions:

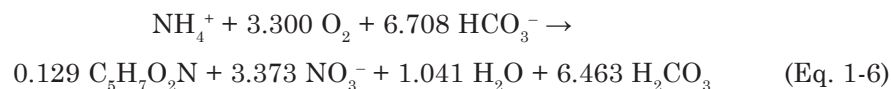


Nitrobacter reaction:

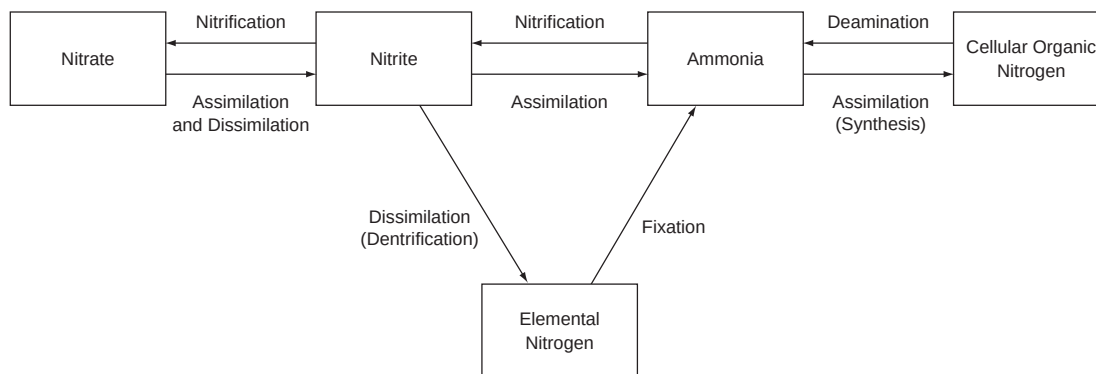


More information on nitrification reactions by AOB, AOA, and NOB is provided in Chapter 5. The overall reaction for complete nitrification (Grady et al. 1999) is shown in equation 1-6.

Overall reaction for complete nitrification:



As shown by the reactants and products of equation 1-6, complete nitrification results in the consumption of alkalinity (HCO_3^-), reduced pH (formation of carbonic acid, H_2CO_3), production of nitrate (NO_3^-), and increased biomass ($\text{C}_5\text{H}_7\text{O}_2\text{N}$). Both complete (to nitrate) and incomplete (to nitrite) nitrification have been frequently observed in chloraminated drinking water distribution systems. Nitrate metabolism



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Figure 1-1 Main biological processes involving nitrogen transformation

can occur either through assimilation (conversion of nitrate to cellular organic nitrogen via ammonia) or dissimilation (oxidation of carbon compounds at the expense of nitrate, which acts as the alternative electron acceptor to oxygen). Denitrification, an important wastewater treatment process, is a special case of dissimilation in which gaseous N_2 and/or N_2O are the end products.

Deamination and lysis of the cell wall occurs in the dying cells and ammonia is formed from organic nitrogen compounds by various deamination reactions (Painter, 1970). The growth of heterotrophic bacteria (as evidenced by high heterotrophic plate counts [HPCs] during nitrification), as well as AOB and NOB, will result in eventual cell lysis and increased chloramine demand. The soluble microbial products produced by active bacteria also exert a chloramine demand. A decrease in relative concentrations of inorganic chloramines and an increase in organic chloramines have been reported in storage reservoirs subject to long detention times; whether cell lysis could contribute to this transformation is currently poorly understood.

Nitrification in the Environment

Nitrifying bacteria are found in soils, compost piles, wastewater, fresh water, marine habitats, and in most other aerobic environments. Many environments with suboptimal conditions still support the growth of nitrifying bacteria. For example, nitrifying bacteria are strict aerobes, yet they can be isolated from wastewater aeration tanks that are extremely low in dissolved oxygen (see Chapter 6 for more information).

The highest concentration of nitrifying bacteria is found in the upper 10-cm layer in soils, at the sediment–water interface in rivers and streams, and attached to the sides of the aeration tanks in wastewater treatment plants (Watson et al. 1981). Sustained high levels of ammonia in the water column of some lakes and deeper rivers would indicate that nitrification is a sediment-based process in these environments. Sediment resuspension might transport the nitrifiers into the water column and macrophytes may serve as surfaces for nitrifiers in the water column.

Nitrifiers are localized in the toxic sediments of the lake, and their activity is likely inhibited by anoxia during summer stratification and by low temperatures in winter (Pauer and Auer 2000). Considerable amounts of nitrifying bacteria (as well as ammonia and nitrate, and other nitrogen-based compounds) are brought into rivers through the discharge of treated and untreated urban wastewater. This discharge produces seeding of the receiving water body with nitrifying bacteria that can be more

pronounced with untreated sewage than with treated effluents. For example, non-nitrified secondary wastewater effluents result in high levels of ammonia that remain present for a longer time in the river environment (Brion and Billen 2000).

Ammonia, nitrate, and nitrite can typically be found in surface water supplies as a result of natural processes. Other sources of nitrogen can include agricultural runoff from fertilization or livestock wastes, contamination from sewage, and industrial discharges. Ammonia also occurs naturally in some groundwater supplies, and groundwater can become contaminated with nitrogen as agricultural runoff percolates into aquifers. The concentration of nitrite nitrogen in surface water and groundwater is normally far below 0.1 mg/L (Sawyer and McCarty 1978). As described in Chapter 8, the level can vary rapidly and dramatically due to watershed events. Seasonal highs in surface water ammonia concentrations typically occur in winter when nitrification rates decline. Groundwater generally contains relatively low concentrations of ammonia because of the cation exchange capacity of soil, unless there have been anthropogenic inputs (Bouwer and Crowe 1988).

No organism has been identified that is capable of fully oxidizing ammonia to nitrate. Consequently, the classification of nitrifying bacteria is based primarily on oxidation of either ammonia or nitrite. Even though several species of heterotrophic bacteria are able to produce nitrates and nitrites, their contribution to total nitrification seems to be insignificant in comparison to autotrophic processes (Kihn et al. 2002). A new ammonia-oxidizing microorganism belonging to the *Archaea* domain has recently been identified and is described in Chapter 5.

NITRIFICATION AND REGULATORY COMPLIANCE

Nitrification and the Safe Drinking Water Act

Nitrification can lead to chemical and biological degradation of water quality and can potentially impact compliance with the following Safe Drinking Water Act (SDWA) requirements:

- Surface Water Treatment Rule (SWTR)
- Total Coliform Rule (TCR)
- Lead and Copper Rule (LCR)
- Primary MCL for nitrate and nitrite and Phase II Inorganic Contaminant Rule
- Stage 1 and Stage 2 DBP Rules

Over the past few years, federal efforts to identify and prioritize distribution system water quality issues have included considering nitrification as a possible public health issue. The National Research Council's (NRC) assessment of distribution system risks determined that "the most important problem exacerbated by both nitrification and by long retention times is loss of disinfectant residuals (NRC 2006)." The NRC designated this to be a medium priority concern due to indirect health impacts caused by the loss of residual, which may result in compromising biological integrity of the system and promoting microbial regrowth (NRC 2006). Additionally, the NRC (2006) determined that the formation of nitrate and nitrite by nitrification is a low priority relative to other distribution system water quality issues. Finally, the NRC also noted that nitrification in low alkalinity waters can cause pH to decrease and affect lead and copper release.

The Total Coliform Rule/Distribution System Advisory Committee recommended that a Research and Information Collection Partnership (RICP) be formed to inform

and support the drinking water community in developing future national risk management decisions pertaining to drinking water distribution systems. The advisory committee recommended seven priority areas for research and information collection, one of which was nitrification. The document entitled *Priorities of the Distribution System Research and Information Collection Partnership* (USEPA 2010b) states that adverse public health impacts are suspected to be associated with nitrification, although available information is more anecdotal in nature and additional research and information collection are necessary to better define public health risks. Several projects identified in the *Priorities* document propose to further explore the occurrence and impacts of nitrification and associated public health risks (USEPA 2010b).

Table 1-2 provides a summary of water quality and possible compliance issues as they relate to nitrification.

Surface Water Treatment Rule

Disinfectant depletion and heterotrophic bacteria increases are examples of water quality impacts associated with nitrification that are addressed under provisions of the SDWA. The loss of a disinfectant residual does not necessarily pose a direct public health threat; however, disinfectant loss can allow growth and/or survival of microbial organisms within the bulk water, in biofilms, or on accumulated sediments.

The Surface Water Treatment Rule establishes maximum contaminant level goals (MCLGs) for viruses, *Legionella*, HPC, and *Giardia duodenalis* (formerly *G. lamblia*). It also includes treatment technique requirements for filtered and unfiltered systems that are specifically designed to protect against the adverse health effects of exposure to these microbial pathogens. The SWTR requires that a “detectable” disinfectant residual be maintained in at least 95 percent of samples collected throughout the distribution system on a monthly basis (or HPC measurements not

Table 1-2 Water quality and compliance issues caused by nitrification

	Water Quality Issues	Compliance Issues
Chemical	Disinfectant depletion	Surface Water Treatment Rule, Total Coliform Rule, Revised Total Coliform Rule
	Nitrite/nitrate formation	Elevated nitrite/nitrate levels*
	Dissolved oxygen depletion	Lead and Copper Rule
	Reduction in pH and alkalinity	Lead and Copper Rule
	DBP formation due to mitigation techniques	Stages I and II Disinfectants/Disinfection By-products Rule
Biological	HPC increase	Surface Water Treatment Rule, Total Coliform Rule
	Coliform occurrences	Total Coliform Rule, Revised Total Coliform Rule
	Increase in AOB and NOB	Cause increases in nitrate and nitrite, can contribute to disinfectant demand
Aesthetic	Taste and odor, color, turbidity	

Note: AOB, ammonia-oxidizing bacteria; HPC, heterotrophic plate count; MCL, maximum contaminant level; NOB, nitrite-oxidizing bacteria.

* Compliance with nitrate and nitrite MCLs is required at the point of entry to the distribution system, not within the distribution system. Systems with elevated source water nitrate/nitrite concentrations could experience levels within the distribution system that exceed the MCL during nitrification events.

exceeding 500 cfu/mL). In general, the minimum detectable residual may be considered the detection limit of the field test analysis employed. This is assumed to be 0.01 mg/L for chlorine and chloramines (APHA et al. 2005; Harp 2002). A system that fails to comply with this requirement for any two consecutive months is in violation of the treatment technique requirement.

Public water systems must monitor for the presence of a disinfectant residual (or HPC levels) at the same frequency and locations as total coliform measurements taken pursuant to the TCR. It should be noted, however, that systems relying on chloramines for secondary disinfection typically aim to maintain a distribution system residual of 1.0 to 4.0 mg/L (USEPA 1999) due to the relatively weak disinfecting properties of chloramines and to prevent nitrification from developing. As discussed in Chapter 8, residual levels of >1.5–2 mg/L are recommended for reducing risk of nitrification. Thus, operating a chloraminated distribution system with only a “detectable” disinfectant residual level is not advisable.

Total Coliform Rule and Revised Total Coliform Rule

Nitrification can have an indirect effect on TCR compliance in three ways: disinfectant depletion caused by nitrification can theoretically allow for the increased growth of coliform bacteria; implementation of nitrification control practices can cause biofilm detachment from pipes and introduction to the bulk water; and the resulting growth of heterotrophic bacteria during nitrification events can interfere with the detection of coliform bacteria. However, as described in several case studies presented in Friedman et al. (2009), the presence of disinfectant residual level alone is not an adequate indicator of microbial water quality or TCR compliance status.

The Revised Total Coliform Rule (RTC) Draft Assessments and Corrective Actions Guidance Manual (USEPA 2010a) recommends that systems collect and evaluate disinfectant residual levels, as well as nitrite/nitrate levels at high water-age locations (for chloraminating systems) during level 1 & 2 assessments. The manual identifies water-age management as a potential corrective action for systems experiencing nitrification and coliform occurrences. Additionally, maintenance of a disinfectant residual, which can be affected by nitrification, is identified as a treatment technique. Although nitrification is not specifically identified as a cause of coliform treatment technique violations, systems with coliform problems will likely need to resolve nitrification should it be identified during the assessment process.

Lead and Copper Rule

Nitrification can have an indirect effect on LCR compliance. Nitrification can cause reduced pH, alkalinity, and dissolved oxygen, which can result in lead and copper release from lead and/or copper bearing materials in the distribution system (Boyd et al. 2008; Zhang et al 2010; Zhang et al. 2009a & b, USEPA 2007; NRC 2006). Although reductions in pH and alkalinity may not pose a direct public health threat, such reductions could theoretically result in enough elevated lead or copper samples to contribute to an exceedance of the LCR action levels (USEPA 2007) as well as failure to maintain optimal water quality parameters such as pH.

As reported by Zhang et al. (2009b), the magnitude of the change in pH depends on the initial alkalinity, extent of nitrification, and acid production. The researchers used lead pipes that were exposed to synthesized water for one year without nitrification and then exposed to water with ammonia (and resulting nitrification) for 15 months. No disinfectant had ever been added to the pipes.

Groups of pipes were exposed to different levels of orthophosphate-P and some of the pipes were exposed to high levels of free chlorine (to a final concentration of

10 mg/L total chlorine), which reacted with existing ammonia (2 mg/L ammonia-N) and formed monochloramine; some were exposed to 1 mg/L chlorite; and some were used as the control (no nitrification inhibitor such as chloramine or chlorite). At 100 mg/L alkalinity as CaCO_3 and pH values of greater than 7.5, complete nitrification did not reduce pH or increase lead release from lead piping, but at 15 mg/L alkalinity as CaCO_3 , nitrification at the elevated chloramine dosages used in the study caused a pH reduction of approximately 1.5 units (to less than 6.5) and an increased soluble lead release by 65 times.

These results are presented in Figure 1-2. Note that nitrification was abruptly halted in the control when no alkalinity was present. The researchers concluded that although some inorganic carbon is required for autotrophic nitrification, it will not be a limiting factor unless alkalinity is below 15 mg/L as CaCO_3 (Zhang et al. 2009b). The researcher's overall conclusion is that serious problems with lead leaching from nitrification are not expected at the alkalinities and initial pHs encountered at a majority of water utilities. However, utilities operating at pH values near 8.3 (the point of minimum buffering capacity) and low alkalinities could be very susceptible to elevated soluble lead from nitrification.

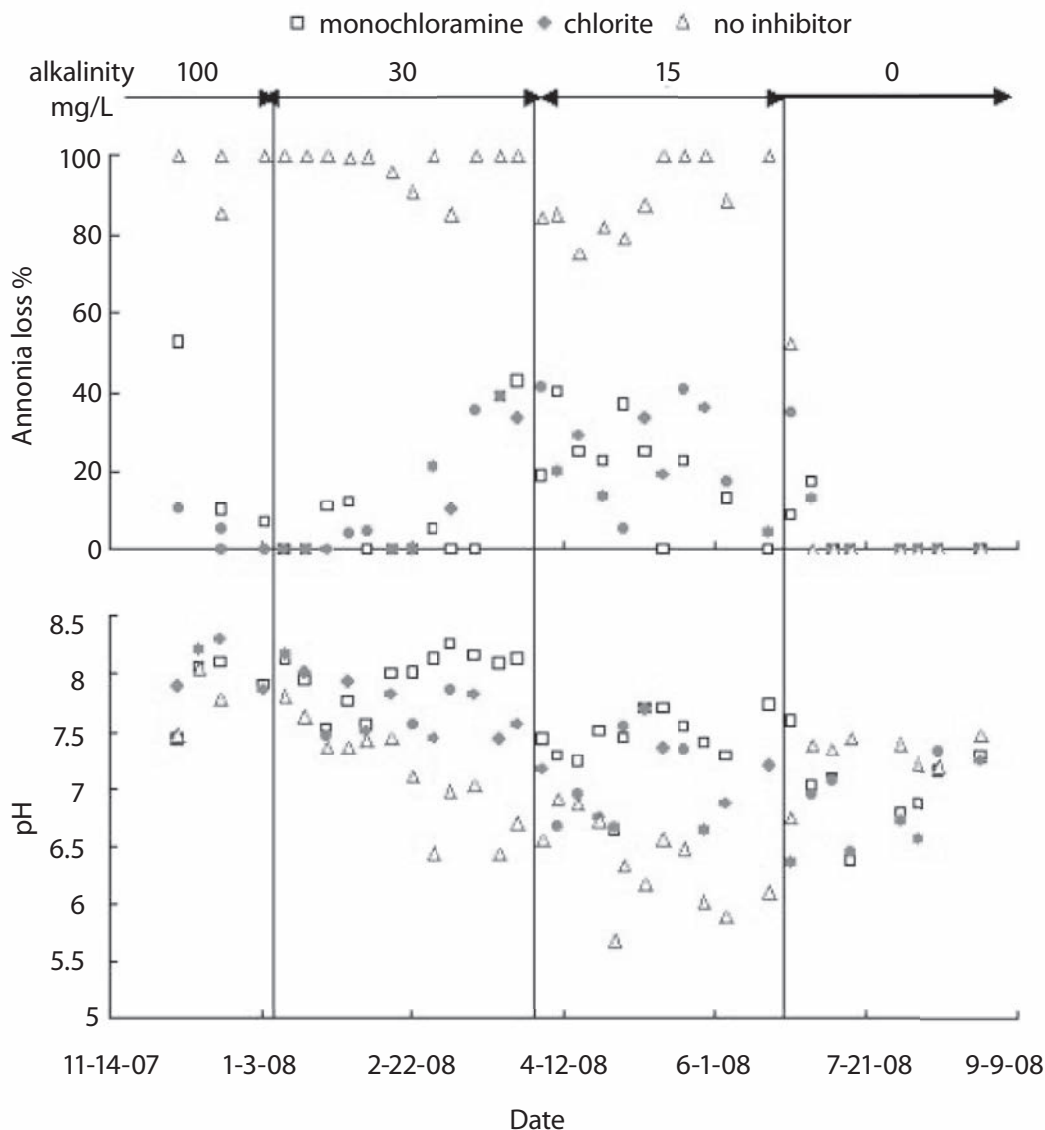
Zhang et al. (2009b) also conducted a bench-scale test to investigate nitrification in PVC pipes in comparison to copper piping. For the experiment, the researchers simulated homes with PVC or copper plumbing and leaded brass faucets by placing a brass rod inside a PVC or copper pipe. In the test, complete nitrification occurred in the PVC pipes after two months. However, this did not occur in the copper pipes, presumably due to copper toxicity. At an alkalinity of 35 mg/L as CaCO_3 , nitrification resulted in a pH drop of 1.4 units (pH 8.2 to 6.8) and leached 45 percent more lead and 81 percent more zinc from leaded brass inserted into the PVC pipes compared to the leaded brass inserted into the copper pipes. The pH increased in the copper piping (8.15 to 9.96) possibly due to corrosion.

The researchers pointed out that the results indicate that when nitrification has occurred and leaded brass faucets are in use, homes with PVC piping will demonstrate higher lead levels in first-draw samples than homes with copper piping. The researchers also determined that lead leaching was not significantly impacted by production of nitrite and nitrate, or by reductions in inorganic carbon or dissolved oxygen via nitrification (Zhang et al. 2009b).

Zhang et al. (2009b) pointed out that copper can have both an inhibitory effect on nitrification or may promote nitrification. The researchers indicated that this may depend on the free copper concentration, copper complexes present, and purity of the nitrifying culture. Zhang et al. (2008) found that levels of copper of 1–10 ppb might stimulate nitrification while levels greater than 100 ppb might prohibit nitrification.

Nitrite and Nitrate Primary MCLs and Phase II Inorganic Contaminants

According to SDWA, primary MCLs have been established for nitrite, nitrate, and the sum of nitrite plus nitrate. The MCLs are 1 mg/L for nitrite (as N), 10 mg/L for nitrate (as N), and 10 mg/L for total nitrate/nitrite (as N). The USEPA Phase II Inorganic Contaminant regulations require water systems to sample for nitrite and nitrate at each entry point to the distribution system at least annually. Additional monitoring is required on a quarterly basis for at least one year following any one routine sample in which the measured concentration is greater than 50 percent of the MCL (USEPA 2001). It should be noted that the nitrite and nitrate MCLs apply at the point of entry to the distribution system, and therefore, elevated nitrite/nitrate levels measured as a result of nitrification within the distribution system are not currently regulated.



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Figure 1-2 Ammonia loss % and final pH, versus time when initial alkalinity was reduced in stages from 100 to 0 mg/L as CaCO₃.

However, in some states if any drinking water sample is analyzed using certified methods and a regulated contaminant is found above its MCL, the result must be reported to the regulator/state, even though it might not be reported as part of the official regulatory compliance program.

Ammonia can be released from chloramine through a series of complex reactions, as shown in Table 1-3. Reactions 2 through 6 describe five mechanisms of ammonia release presented by Wooschlager et al. (2001) and Vikesland et al. (1998). Three of the demand reactions (3, 4, and 6) essentially release one mol of ammonia for one mol of monochloramine. Comparatively, the autodecomposition reaction (major pathway) releases only one mol of ammonia for every three mols of monochloramine. This

Table 1-3 Overview of nitrification and chloramine reactions

Reaction Description	Overall Reaction
1. Ammonia and nitrite utilization	$\text{NH}_3 + \text{O}_2 \rightarrow \text{NO}_2 + 3\text{H}^+ + 2\text{e}^-$ $\text{NO}_2 + \text{H}_2\text{O} \rightarrow \text{NO}_3 + 2\text{H}^+ + 2\text{e}^-$
2. Release of ammonia through chloramine decay (autodecomposition)	$3\text{NH}_2\text{Cl} \rightarrow \text{N}_2 + \text{NH}_3 + 3\text{Cl}^- + 3\text{H}^+ \text{ (Major)}^1$ $4\text{NH}_2\text{Cl} + 3\text{H}_2\text{O} \rightarrow 4\text{Cl}^- + 3\text{NH}_3\text{NO}_3 + 5\text{H}^+ \text{ (Minor)}^1$
3. Release of ammonia through oxidation of organic matter by chloramine	$1/10\text{C}_5\text{H}_7\text{O}_2\text{N} + \text{NH}_2\text{Cl} + 9/10\text{H}_2\text{O} \rightarrow 4/10\text{CO}_2 + 1/10\text{HCO}_3^- + 11/10\text{NH}_4^+ + \text{Cl}^-$
4. Release of ammonia through reaction of chloramine with corrosion products at pipe surfaces*	$1/2\text{NH}_2\text{Cl} + \text{H}^+ + \text{Fe}_2^+ \rightarrow \text{Fe}^{3+} + 1/2\text{NH}_4^+ + 1/2\text{Cl}^-$
5. Release of ammonia through catalysis reactions of chloramine at pipe surfaces	$3\text{NH}_2\text{Cl} \rightarrow \text{N}_2 + \text{NH}_3 + 3\text{Cl}^- + 3\text{H}^+$
6. Release of ammonia through oxidation of nitrite by chloramine*	$\text{NH}_2\text{Cl} + \text{NO}_2 + \text{H}_2\text{O} \rightarrow \text{NH}_3 + \text{NO}_3 + \text{HCl}$

Adapted from Wooschlager et al., 2001

* These equations may not be significant in distribution systems, especially in situations of biologically accelerated chloramine decay.

¹ Source: Vikesland et al. (1998).

indicates that some demand reactions release significantly more ammonia than the major autodecomposition (decay) reaction.

Nitrite and nitrate are produced during nitrification through ammonia utilization by nitrifying bacteria. According to Table 1-3, reactions 1 and 2, for every mol of ammonia-N produced through chloramine decay, a 1-mol equivalent of nitrite-N is produced by AOB. Subsequently, for every mol of nitrite-N produced by AOB, a 1-mol equivalent of nitrate-N is produced by NOB. Figure 1-3 shows that by using reaction 3 in Table 1-3, which is the most conservative chloramine demand reaction in terms of quantity of ammonia produced per mol of monochloramine present, the nitrite-N MCL of 1 mg/L as N could theoretically be exceeded if the chloramine dose greater than approximately 3 mg/L (as total chlorine) and the $\text{Cl}_2:\text{NH}_3\text{-N}$ ratio is less than 5:1. As the chloramine dose increases and the chlorine to ammonia-N ratio decreases, more ammonia becomes available for the nitrification process. In a pilot-scale distribution system, Harrington et al. (2002) measured nitrite concentrations equal to 1 mg/L as N with an average total chlorine concentration of 4.6 mg/L and a $\text{Cl}_2:\text{NH}_3\text{-N}$ ratio of 3.9:1. The pilot train represented conventionally coagulated Lake Mendota (Madison, Wisc.) water at a pH of 9.

Valentine et al. (1998) conducted a series of mass and redox balances on solutions of varying pH, NOM concentration, and initial chloramine concentration. For all conditions that were studied, the amount of nitrate formed as a percentage of monochloramine decay was less than 15 percent, and for all but three cases the amount was less than 10 percent. The authors concluded that although nitrate is an important decomposition product of monochloramine decay, it is not the major nitrogen-containing species of decomposition.

Using data from a survey of 40 utilities that use chloramine as a disinfectant and an previous survey by Hack (1984), Wilczak et al. (1996) indicated that nitrite-N and nitrate-N levels may increase by 0.05 to 0.5 mg/L as N although increases of greater than 1 mg/L are possible. Nitrite levels during nitrification episodes have been reported ranging from 0.005 to 0.5 mg/L as N, with levels more frequently ranging from 0.015 to 0.1 mg/L as N (Wolfe and Lieu 2001). As discussed previously, the amount of nitrite produced is a function of the amount of ammonia present.

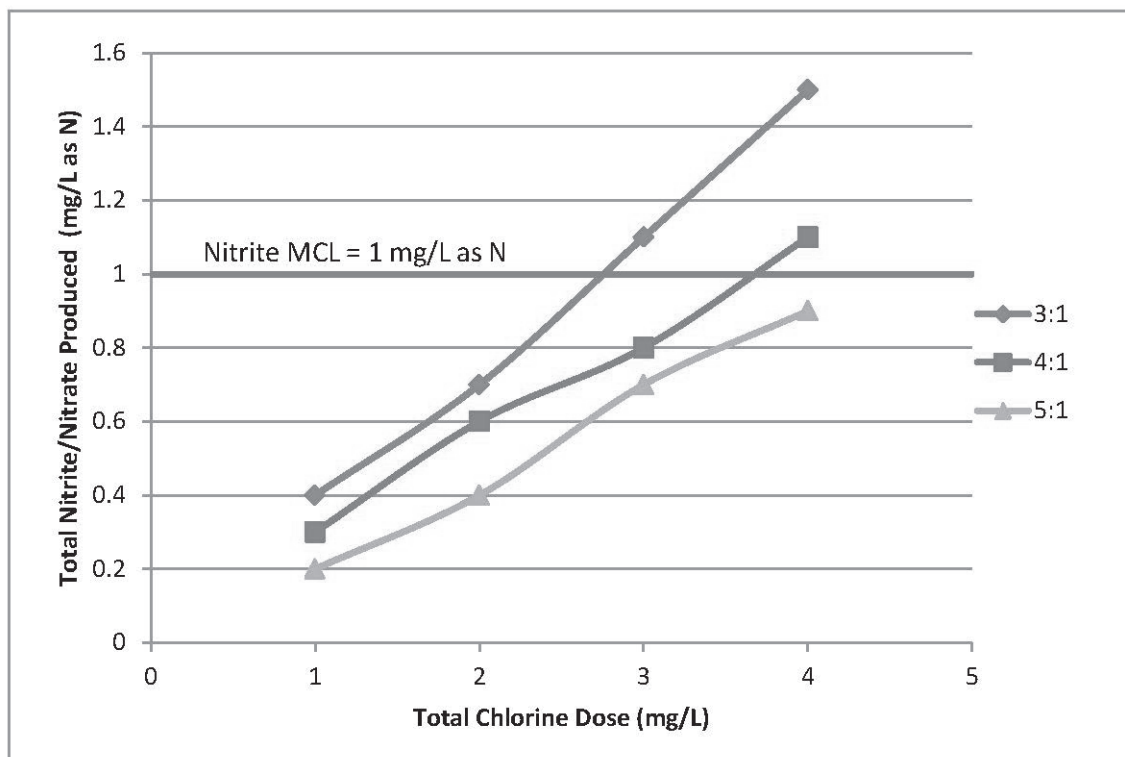


Figure 1-3 Theoretical nitrite/nitrate production based on chloramine decay stoichiometry as a function of chlorine:ammonia-N ratio. Assumes: (1) 100% decay according to Reaction 3 in Table 1-3; (2) 100% conversion of ammonia to nitrite/nitrate-N.

High nitrite levels could be observed in systems with high source water ammonia concentrations, regardless of whether or not disinfection is practiced. Rezania (2010) reported ammonia levels ranging from 1.7 mg/L up to 7 mg/L in a study of 39 public water supply wells in Minnesota. In addition, it was reported that samples taken in the distribution system had nitrite levels ranging from 0.0 to 2.1 mg/L. Lytle et al. (2007; 2009) reported that levels of nitrite greatly exceeded the maximum contaminant level (MCL) of 1.0 mg/L in the distribution systems of several communities in Illinois that had elevated ammonia in their source water.

As previously discussed, the NRC's assessment of distribution system health risks determined that formation of nitrate and nitrite in the distribution system was a relatively low priority compared to other potential distribution system health risks (NRC 2006). The report indicates that an increase of nitrite in the distribution system due to nitrification would likely be less than 10 percent of the MCL. Therefore, nitrite and nitrate formation from nitrification could be of most serious concern in systems with high source water ammonia concentrations. The additional nitrate and nitrate could cause an MCL exceedance.

Disinfection Byproducts

Nitrification in and of itself is not known to form DBPs, however certain practices to prevent or respond to nitrification, such as breakpoint chlorination, can form increased levels of regulated DBPs. As previously described, utilities often use chloramines for secondary disinfection to substantially reduce the formation of regulated DBPs

present in the drinking water. However, other DBPs have been linked to the use of chloramines. Nitrosamines, including *N*-nitrosodimethylamine (NDMA), are another type of DBP that can form during chloramination of waters containing some organic nitrogen precursors and are considered to be carcinogenic (Schreiber and Mitch 2006).

For utilities monitoring NDMA as part of the second Unregulated Contaminant Monitoring Rule, one in four detected the presence of NDMA (Roberson 2010) and two-thirds of those utilities detecting NDMA reported the use of chloramines (Russell et al. 2012). Other nitrosamines monitored as part of this effort appear to occur less frequently and in smaller quantities. It appears likely that the USEPA will propose regulation of nitrosamines in the near future with a Maximum Contaminant Level Goal of zero due to their carcinogenic nature (Roberson 2010). California has established notification levels for NDMA and two other nitrosamines of 10 ng/L.

Speitel et al. (2010) conducted an assessment of how THMs can impact nitrification in chloraminated distribution systems. AOB can biodegrade THMs through cometabolism, an enzymatic process in which a specific microbial enzyme causes biodegradation of a chemical with no significant benefit to the microorganism. With respect to AOBs, this cometabolism produces by-products that are toxic to the microorganism (especially THMs with bromine). The research suggested that THM concentration and length of exposure could delay nitrification. In reactors fed with the concentrations of 400–600 µg/L of THM (much higher than levels that would be found in drinking water distribution systems), nitrification was delayed by 5–8 days in comparison to reactors receiving no THMs. The researchers also found that THMs in the presence of monochloramine also delayed the onset of nitrification in comparison with reactors that received only monochloramine.

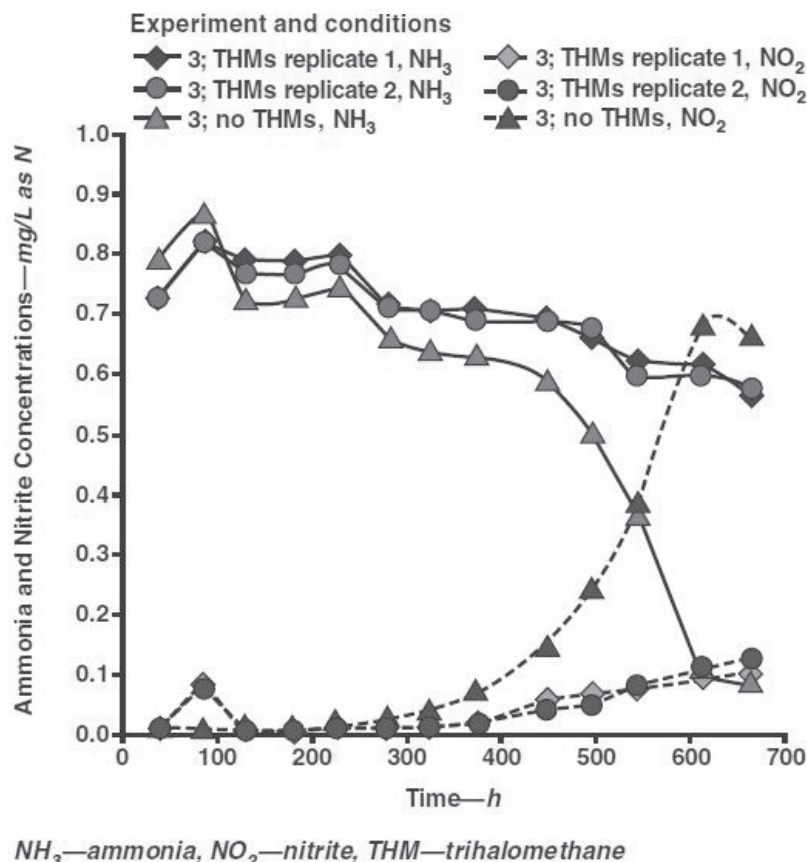
Figure 1-4 presents the results of an experiment conducted by Speitel et al. (2011) in which reactors were fed with a water quality meant to promote nitrifying conditions. The THMs concentration for the two reactors receiving THMs was 56 µg/L and the control reactor had no THMs present (note: one of the two control reactors failed). For the control reactor, nitrification was initially indicated at about 300 hours as shown by the increase in nitrite. Over the next 350 hours, the ammonia concentration decreased steadily with complete nitrification. Reactors receiving THMs started to show signs of nitrification, but full nitrification was not observed even after 650 hours.

CONCLUSIONS

Nitrification is a microbiological process by which reduced nitrogen compounds (primarily ammonia) are sequentially oxidized to nitrite and nitrate. The use of chloramine as a secondary disinfectant (and resulting presence of free ammonia from dosing procedures and/or chloramines decomposition) and the presence of nitrifying bacteria in the distribution system are the main causes of nitrification in water distribution systems. Because nitrifying bacteria are ubiquitous in the environment and the distribution system, nitrification will occur if the physical and chemical conditions that support the growth of these organisms exist and sufficient levels of free ammonia are present in the water.

Ammonia can be present in drinking water through naturally occurring processes, through ammonia addition during secondary disinfection to form chloramines, and when it is released into the water due to chloramine degradation (e.g. decay and demand reactions). Because the use of chloramine is expected to increase in the near future as a response to more stringent DBP MCLs associated with the Stage 2 D/DBP Rules, the occurrence and impacts of nitrification are expected to increase.

There are several impacts of nitrification that can affect distribution system water quality. Of the water quality issues identified in the literature and summarized



Source: Speitel et al. 2011

Figure 1-4 THM inhibition of nitrification during bench-scale studies

in Table 1-2, most impacts on regulatory compliance are indirect, such as loss of disinfectant residual or reduction in pH. Only the formation of nitrite and nitrate within the distribution system poses a potential direct public health threat, but nitrite and nitrate are not addressed within the distribution system through current provisions of SDWA. It should be noted, however, that a nitrite or nitrate MCL violation within the distribution system is unlikely in most chloraminated distribution systems, unless treated water nitrite and nitrate levels are already near their respective MCLs or source waters contain elevated levels of ammonia. Although compliance with current regulations may not be impacted directly by nitrification, it is a good practice to minimize the adverse effects of nitrification in the distribution system, especially the loss of a disinfectant residual, the growth of HPC bacteria, the formation of nitrite and nitrate, and any decrease in pH and alkalinity.

The causes of nitrification, its prevention, and mitigation are discussed in detail in the following chapters. Prevention of nitrification should be considered an essential water quality maintenance practice because it helps to provide a more stable chloramine residual in the distribution system and water that is biologically stable in terms of limiting AOB and NOB growth as well as the growth of heterotrophic bacteria prevalent during nitrification episodes. Maintenance of a disinfectant residual and the biological integrity of the distribution system are important from a customer point of view as well as from a distribution system security point of view, because a disinfectant residual is commonly used as one *indicator* of distribution system integrity and

overall water quality (National Research Council Committee on Public Water Supply Distribution Systems 2006; Friedman et al. 2010).

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Chapter 2

Nitrification in Water and Wastewater Treatment

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INTRODUCTION

This chapter discusses available information on the occurrence of nitrification in water treatment plants and its potential impact on distribution system water quality. Nitrification as part of the water treatment process can occur whenever ammonia is present in or added to the source water, and whenever water is not initially free chlorinated to achieve breakpoint. Nitrification can be either controlled or uncontrolled. Controlled nitrification may be conducted, for example, when the concentrations of source water ammonia are high and the desire is to remove the ammonia partially or completely. Nitrification has been used and modeled extensively in wastewater treatment, and it is useful to understand the lessons learned there and draw certain comparisons and parallels that may improve the understanding of nitrification in water treatment and distribution. The key findings of this review are summarized in Table 2.1.

DRINKING WATER NITRIFICATION AND IMPACT ON DISTRIBUTION SYSTEMS

A key issue that causes nitrification is the amount of ammonia in water sources. Although there is no ammonia drinking water standard in the United States, the European community (EC) has established a maximum limit of 0.5 mg/L and a guide

level of 0.05 mg/L (EC Drinking Water Directive 1980). Although there are no immediate indications that ammonia will become regulated within the United States, there are benefits for utilities to reduce the amount of ammonia that is able to move into the distribution system. This can be achieved in different ways, but the most cost effective way is by using a biologically active filtration process (biofiltration). During the biofiltration process, it is possible to achieve nitrification to reduce ammonia levels.

The reduction of source water ammonia before entering the distribution system has important benefits. The presence of ammonia in drinking water distribution systems has been correlated to increased biological activity (Servais et al. 1995; Wilczak et al. 1996), corrosion (Murphy *et al.* 1997), formation of nitrite and nitrates (Odell et al. 1996; Wilczak et al. 1996), and adverse impacts on taste and odor (Bouwer and Crowe 1988, Rittmann and Huck 1989). In addition, the presence of ammonia can interfere with the effectiveness of some water treatment processes.

For example, the oxidation of arsenic (III) by free chlorine and subsequent removal from water may be reduced as a result of the formation of chloramines that are less effective at oxidizing arsenic than is free residual chlorine (Lytle et al. 2007). When free chlorine is the desired residual disinfectant in the distribution system, the removal of ammonia naturally occurring in the raw water is beneficial to reduce chlorine demand and to avoid formation of chloramines. Such treatment is more common in countries where chloramination is not practiced, e.g., France and Germany.

Table 2-1 Key points from chapter 2

Nitrification in Water Treatment	• Nitrification can occur whenever ammonia is present in or added to the source water and water is not initially free chlorinated to achieve breakpoint, and necessary levels of oxygen are present. • Biological reduction of ammonia is more cost effectively achieved using biological rather than nonbiological methods. Controlled and complete nitrification (to nitrate) can be accomplished biologically in various types of filter beds and media contactors. • Uncontrolled and incomplete nitrification (to nitrite) in filter beds is not desirable because it increases chlorine demand and can lead to serious nitrification in the distribution system. Furthermore, the drinking water standard for nitrite (1 mg N/L) is much lower than the nitrate (10 mg N/L) standard. • Temperature is one of the major factors impacting nitrification in a filter bed. • Low loading rates, long solids retention times, and steady-state operation are needed to achieve a high degree of nitrification in a filter bed. Nitrification can occur in very cold waters at 1°C (34°F) if long detention time and appropriate nutrient concentrations are available. • Very porous filter media are better for achieving nitrification in a filter bed, especially in cold water, in order of decreasing effectiveness: macroporous carbon > microporous carbon > anthracite > sand. • Preammoniation or passing chloramines through a GAC bed can lead to nitrifying bacteria colonization of the treatment train. Monitoring of the biofiltration process should be routinely conducted so that nitrification in the distribution system is avoided. • A nitrification or biological organic carbon removal barrier should be placed at the end of the treatment train. Postdisinfection can be used to block passage of undesirable organisms and growth products to the distribution system. If UV light is used to disinfect a biological filter effluent, additional protection from a residual disinfectant (chlorine and/or chloramines) should be considered to control downstream regrowth.
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Table continued on next page.

Table 2-1 Key points from chapter 2 (continued)

Nitrification in Wastewater Treatment Versus that Occurring in Drinking Water Treatment and Distribution Systems	• AOB growth in the wastewater treatment process is mostly independent of ammonia concentration and is an “all-or-none” phenomenon; it either proceeds at 100 percent ammonia removal or near zero when bacteria grow too slowly and are washed out. Nitrification in wastewater treatment is typically modeled as Monod, or dual Monod kinetics for low DO systems, which is independent of ammonia at concentrations somewhat higher than a half-saturation constant (K), approximately 1 mg/L as N. • Although the nitrification rate may be zero order (independent of substrate concentration, i.e., ammonia) at the head of a plug-flow system, e.g., attached growth reactor, this is not the case near the outlet where the substrate concentrations decrease substantially. • The growth rate of AOB in drinking water systems may depend on free ammonia concentration, except that <i>Nitrosomonas oligotropha</i> (<i>N. oligotropha</i>) requires very little substrate and its growth rate may become substrate-limited at very low NH_3 concentrations (below 0.1 mg/L as N or less). This is an issue of debate as few studies have evaluated <i>N. oligotropha</i>. Nevertheless, keeping free ammonia levels low in the distribution system should be considered key to nitrification control. • Nitrification in drinking water distribution systems can occur over a wider range of temperature, pH, and DO than in wastewater, given very long residence times for the nitrifying bacteria in drinking water distribution systems and low levels of ammonia. This may also be due to the difference in microbes between the two systems. The drinking water systems have <i>N. oligotropha</i> while the wastewater systems have different species. • Based on experience gained from water and wastewater treatment practices, optimal nitrification control in drinking water storage facilities should occur in completely mixed, clean reservoirs with minimal deposits, having high water turnover to surpass AOB growth rate and cause their washout. A completely mixed tank provides the highest possible chloramine concentration and lowest possible free ammonia concentration (it also provides more uniform concentrations throughout the tank). • There have been developments over the last decade that have significantly increased the efficiency in nitrification and denitrification processes for wastewater treatment. Use of technologies such as an integrated fixed-film activated sludge system (IFAS), biological aerated filters (BAFs), and the single-reactor, high-activity ammonia removal over nitrite (SHARON) processes can dramatically improve efficiency while also minimizing process footprint.
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NOTES: AOB = ammonia-oxidizing bacteria

DO = dissolved oxygen

GAC = granular activated carbon

Nonbiological Ammonia Removal for Drinking Water

Both physicochemical and biochemical processes can be applied for ammonia removal. The physicochemical processes (e.g., ion exchange, reverse osmosis, and chemical oxidation) have certain disadvantages. For example, there is no selective cation exchanger for the ammonium ion and, therefore, the sorption capacity of the cation exchanger is also reduced by cations other than ammonium ion. The ion exchange process also produces liquid waste brine that must be addressed. Breakpoint chlorination may require high chlorine doses and possibly more advanced treatment, such as granular activated carbon (GAC) adsorption to remove resulting taste and odor compounds as well as chlorination by-products. In the United States, breakpoint chlorination has been the most prevalent method for removing source water ammonia.

Drinking Water Biofiltration for Ammonia Removal

Based on an average raw water ammonia concentration of less than 0.2 mg/L $\text{NH}_3\text{-N}$, as discussed in Chapter 1, a 1.5-mg/L dose of chlorine will achieve breakpoint and free chlorine residual. Currently, the practice of raw water chlorination is becoming less popular and utilities are evaluating moving the point of chlorination further down the treatment train, i.e., after filtration. In that case, biological nitrification could not only develop in the holding or sedimentation basins, but especially within the filter beds. Biological nitrification within the treatment plant can be an effective approach to reducing ammonia levels.

Biological nitrification consisting of water aeration and a subsequent (or simultaneous) biological step has been generally recommended in Europe as preferable to chemical methods (Janda and Rudovsky 1994; Rogalla et al. 1990) for reducing source water ammonia. Bouwer and Crowe (1988) documented the use of various biological ammonia reduction methods throughout Great Britain, France, and Germany, including fluidized beds, rapid sand filters, biologically active granulated active carbon (GAC), and soil-aquifer treatment. The use of biologically active filtration to oxidize ammonia as a full-scale drinking water treatment process has not been thoroughly considered in the United States. Many concerns with biological water treatment exist including the potential release of excessive numbers of bacteria into finished waters, sensitivity of bacteria to changes in water chemistry and operating conditions, and a lack of long-term documentation of the effectiveness and reliability of biological water treatment processes.

The effectiveness of biological ammonia oxidation treatment to reduce source water ammonia levels is dependent on a number of source water and engineering design factors including: temperature, dissolved oxygen concentration, TOC, pH, biomass quantity and population, media type, and surface area, as well as hydraulic loading rate and contact time (Uhl and Gimbel 2000; Janda and Rudovsky 1994; Andersson et al. 2001; Lytle et al. 2007; Rittmann and Snoeyink 1984). Canadian researchers (Andersson et al. 2001; Kihn et al. 2002) investigated the impact of GAC and temperature on biological nitrification using pilot and full-scale water treatment filters. Full-scale studies were conducted with Mille-Iles River (Quebec, Canada) water (dissolved organic carbon between 4 and 10 mg C/L and alkalinity 20 to 50 mg $\text{CaCO}_3\text{/L}$) that contained 0.02 to 0.12 mg/L $\text{NH}_3\text{-N}$. Full-scale evaluations were performed using loading rates of 3.9 to 5 m/h (empty bed contact time, EBCT, of 20 to 30 minutes). At 15°C, an average of 94 percent ammonia oxidation was achieved, and at 5°C, ammonia conversion was insignificant. Pilot-scale studies were conducted using Mille-Iles River water enriched with 0.4 or 1.2 mg/L $\text{NH}_3\text{-N}$ of ammonia. Ammonia removal capabilities in pilot studies ranged from 40 to 90 percent at temperatures above 10°C. Ammonia removal dropped to 10 to 40 percent at moderate temperatures of 4 to 10°C.

Lytle et al. (2007) monitored the performance of nitrification in a full-scale 1.2 mgd iron-removal filtration plant for more than a year. The treatment train consisted of a forced draft aerator followed by a basin located at the bottom of an aerator that provided a retention time of approximately 12 minutes, and three dual media (sand, anthracite) filters operated at a hydraulic loading rate of 2 gpm/ft². Source water contained 1.2 mg/L $\text{NH}_3\text{-N}$, 2.3 mg/L iron, and had a pH of 7.48. The treatment plant consistently reduced ammonia levels by greater than 95 percent, and nitrification was complete (nitrite was not detected in filtered water). Seasonal variations in ammonia oxidation effectiveness were not observed because yearly changes in water temperature and other water quality parameters were minimal in the groundwater source.

Filter media (sand, anthracite, or GAC) have been used as a support for the growth of nitrifying bacteria. Sand covered with manganese dioxide has also been

reported to be an effective support for the attachment of nitrifying bacteria (Janda and Rudovsky 1994). Kimura et al. (2001) described operation of an experimental rotating biofilm membrane (polysulfone, 750,000 Dalton molecular weight cut-off) pilot reactor treating coagulated and settled river water for ammonia removal.

Nitrifying bacteria are recognized as slow growers and, therefore, subsequent colonization of filters will take time. During this time, ammonia will pass through the filter and as the system transitions, finished nitrite levels will likely be elevated for a brief amount of time. Lytle et al. (2007) conducted pilot tests using dual anthracite/sand filters to predict the amount of time required to achieve complete nitrification of 1.3 mg/L $\text{NH}_3\text{-N}$ by three different seeding methods of new filters. Simply running aerated raw water through new dual filter (sand, anthracite) media promoted bacterial regrowth and nitrification just as quickly as efforts to seed the new filters by either adding some biologically active full-scale media or by adding backwash solids collected during backwashing of full-scale nitrifying filters. The pilot filters achieved greater than 95 percent reduction of ammonia in less than 80 days.

During the colonization period, nitrite levels as high as 0.75 mg $\text{N-NO}_2\text{/L}$ were measured. Pilot data was used to predict the time necessary to regain nitrification abilities of associated full-scale biologically active filters. Following filter rebidding with dual granular media (sand, anthracite), complete nitrification of 1.3 mg $\text{mg/L NH}_3\text{-N}$ in the source water was achieved by 150 days, and much longer than pilot predictions (Lytle et al. 2009). The difference was attributed to the fact that the full-scale system was only operated for approximately 12 hours per day and the pilot was operated 24 hours per day.

Impact of Operations and Media on Biofiltration

Laurent et al. (1997) stated that temperature was one of the major factors impacting nitrification in a filter bed with the optimum between 25°C and 30°C. When temperature dropped below 7°C, nitrification in biological activated carbon (BAC)/sand filters was more effective than in the sand filters. Filter media porosity, the filtration rate, and EBCT are the key parameters for the removal of biodegradable organic matter (BOM) and ammonia at filtration rates varying from 5 to 16 m/h. Macroporous carbon was colonized much faster by the nitrifiers than microporous carbon (Laurent et al. 1997). More porous filter media are better for achieving nitrification in the filter bed, especially in cold water, with the following order of preference:

macroporous carbon > microporous carbon > anthracite > sand

Rittmann and Snoeyink (1984) described four types of nitrification process units and conditions for filtration: biological filters (flooded or trickling), fluidized bed filters, rapid sand filters, and GAC beds. Low loading rates, long solids retention times, and steady-state operation are needed to achieve effluent ammonia concentration less than 0.1 mg/L $\text{NH}_3\text{-N}$. Gravel filters treated Thames River water at rates of 0.8 to 2.8 m/h with a raw water ammonia concentration of 2 to 3 mg/L $\text{NH}_3\text{-N}$. Despite reduced efficiency, the biological filter maintained about 50 percent efficiency at 5°C and a loading rate of 2.4 m/h and maintained good nitrification, even at 3°C. In the fluidized filter reactor, nearly 100 percent removal was achieved at temperatures from 4°C to 21°C, as long as the fluidized solids concentration was at least 30 percent by volume. About 50 percent removal of 1.3 mg/L ammonia nitrogen occurred during rapid sand filtration of Illinois groundwater at 4.9 m/h. Activated carbon columns loaded at 4.84 m/h (9 min EBCT) removed 78 percent of ammonia across the bed. No information was

given if nitrification was complete (to nitrate) or incomplete (to nitrite) in any of these cases (Rittmann and Snoeyink 1984).

Filters filled with GAC provided better nitrification than sand or anthracite filters, especially when external conditions were unfavorable (low ammonia concentration, low temperature). On a volumetric basis, GAC appeared to be three times more effective than sand (Kihn et al. 2002). It was found that a period of colonization is required following startup of biological filters before constant ammonia removal is reached. During this period, ammonia breakthrough and nitrite formation can alter the quality of filtered water. The GAC filters at 20°C showed that nitrification occurred two weeks after the filters were in service (Kihn et al. 2002).

The effectiveness and reliability of nitrifying biological filters for ammonia reduction are also dependent on other factors including hydraulic loading rate; backwash frequency, duration and quality; the presence of iron and other water quality variables.

Monitoring Biofiltration Performance

Methods of monitoring biological filters include measurement of surrogate parameters that can help engineers and operators quantify the fate of the biodegradable fraction of organic matter (BOM). Such surrogates could be either based on chemical or biological assay techniques, or a combination both. Assays that have been used in the past include various methods for measuring biodegradable dissolved organic carbon (BDOC), assimilable organic carbon (AOC), or direct measurement of various aldehydes, aldo-keto acids, or filter media biological growth.

A study summarized by Mofidi (1996) compared BDOC measurement methods that included continuous-flow and batch methods. The continuous-flow method was used somewhat like an on-line monitoring instrument while the batch method was run with duplicate samples. Continuous-flow monitoring allowed water to pass through chromatography column bioreactors (2.3-hr detention time) filled with porous borosilicate glass media. This method was adapted from work conducted by Kaplan et al. (1995). Batch BDOC methods were modeled after those developed by Servais et al. (1987) and Joret and Levi (1986). Results from this study showed that the bioreactors allowed for more than a 10-fold reduction in BDOC measurement time, although bioreactors must be continuously fed water under thermal protection while meeting strict hydraulic conditions. After a one-year study was completed, it was determined that there was no statistical difference between the BDOC determined by the bioreactors and the BDOC measured by a batch method. While this study evaluated the use of bioreactors to monitor BOM levels following ozonation, the same bioreactor configuration has been used to monitor much lower levels of BOM produced from less reactive oxidants such as chlorine dioxide (Mofidi et al. 2004).

When implementing biofiltration downstream of ozonation, Mofidi and co-workers (2005) indicated great success in the monitoring of the biological process by measuring short-chain carboxylic acids. While this report did not evaluate use of biofiltration for reduction of ammonia, it did outline five biofiltration goals, which included the following:

1. **Biological Activity.** Biological performance of biofilters should take place in the “background” so that there are no additional work demands placed on plant managers or operators. The bioactivity should not impact the physical, aesthetic, or operating characteristics of the filters.
2. **Particle Removal.** Biofiltration should be able to perform such that all turbidity and particle count measurements meet and exceed regulatory requirements and utility water quality goals.

3. **Hydraulic Performance.** To maintain proper treatment operations and meet all flow demands, bioactivity should not detract from the hydraulic performance of the filters.
4. **Microbial Water Quality.** Finished water, both at the end of the water treatment plant and within the distribution system, must remain free from microbial compromise either from BOM or biomass/bacteria that may slough from the biofilters.
5. **BOM Removal Guidelines.** The biofilters should maintain capability to remove at least 70 percent of the BOM measured at the filters influent.

Case Studies in Ammonia Removal and Water Treatment Biofiltration

Following startup of the previously documented biofilters, there have been studies reporting that the addition of ammonia (either alone or in combination with chlorine to form chloramines) upstream of ozonation can significantly reduce formation of bromate. Based on early work conducted by Pinkernell and von Gunten (2001), full-scale evaluations are being conducted to determine if ammonia addition can assist in the reduction of bromate—in combination with downstream biofiltration operations to reduce any remaining residual ammonia. The following is a summary of some select pilot- and full-scale work recently completed on the subject.

At the Southern Nevada Water Authority (SNWA), pilot studies have demonstrated that a biologically active nitrifying filter is a viable alternative to breakpoint chlorination for ammonia removal. Following use of the chlorine–ammonia process for bromate control, ammonia residual can pass through the ozone contactor and then be removed by nitrification on downstream biofilters. The conversion of ammonia to nitrate takes place in conjunction with the capability of the biofilters to reduce BOM from ozonation (Wert 2011). The pilot work consisted of pretreatment (ozone, coagulation, flocculation) followed by both shallow (21 inches of anthracite over 10 inches of sand) and deep bed (72 inches of anthracite over 8 inches of sand) dual media filters. Three phases of pilot studies using the previously discussed processes are described below, illustrating that lower filtration rates and higher filter EBCTs enhanced nitrification.

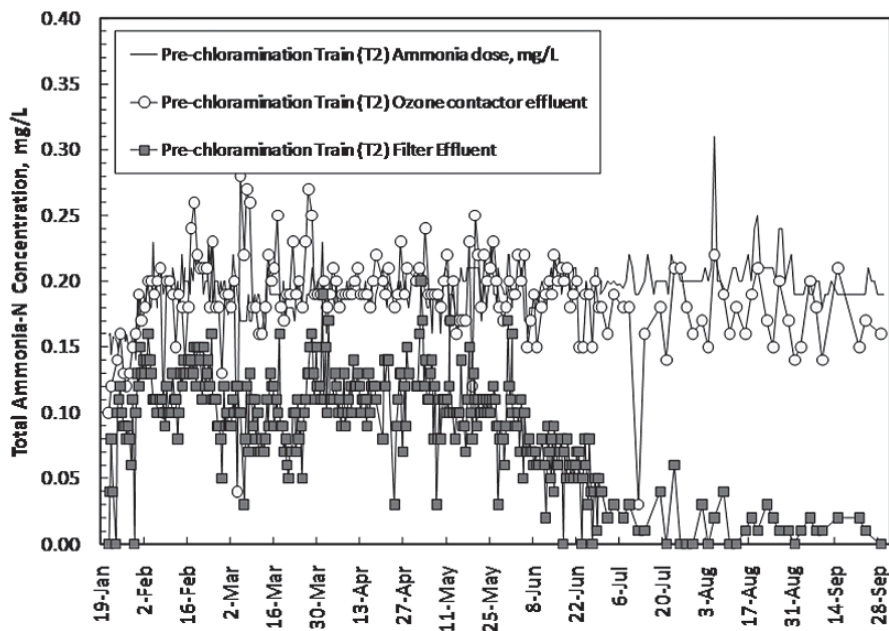
In the first SNWA study, conversion of ammonia to nitrite was observed between 76 and 99 days of operation with no conversion to nitrate through the end of the study at day 165 (Neemann et al. 2004). Chlorinated backwash water was shown to be more detrimental to the nitrifying bacteria than backwash water with chloramines.

In the second SNWA study, full nitrification was achieved while operating at a rate of 6 GPM/sf at approximately day 170 (Wert et al. 2008). This was achieved by using chlorinated backwash water for the first 6 weeks and then dechlorinated water for the remainder of the evaluation. On day 130, both shallow filters showed conversion of ammonia to nitrite. On day 155, residual ammonia was below the method detection limit while nitrite was measured at a level equivalent to the upstream added ammonia concentration. At day 170, nitrate in the finished water exceeded raw water nitrate. At the end of this study (day 290), half of the residual ammonia was converted to nitrite and the other half was converted to nitrate. Most of this study was conducted at an EBCT of 3.2 min, which was found to have equivalent nitrification development to biofilters that were operated at 8.3 min EBCT.

In the third phase of the SNWA study, nitrification was examined using different filter media types over the entire depth of media (Stoughtenger and Wert 2009). This testing consisted of a slightly different filter arrangement with a shallow bed filter (21 inches of anthracite and 10 inches of sand), a deep bed filter (50 inches of

anthracite and 10 inches of sand), and a GAC filter (32 inches of GAC and 13 inches of sand). Sample taps were installed every 10 inches on the filter column, and dechlorinated backwash water was used for all three filters. Results showed that nitrification occurred throughout the depth of media, although most removal was completed within the first 28 inches of filter media. The GAC filter was more efficient at converting ammonia to nitrate than the anthracite filters.

A similar study was conducted at Alameda County Water District (ACWD) to evaluate use of prechloramination to control bromate formation during ozonation and control of residual ammonia through downstream biofiltration (Teefy et al. 2008, 2010). Testing was conducted at ACWD's 21-mgd Water Treatment Plant No. 2 that operated dual-media anthracite/sand biofiltration. Ammonia operations, illustrated in Figure 2-1, included a dose of 0.2 mg/L ammonia that was present going into biofiltration. From a period between January through May, filter effluent ammonia remained at approximately 0.1 mg/L. In June, biofilter effluent ammonia decreased to less than 0.05 mg/L through the remainder of the study (completed in September). To determine whether full nitrification was occurring is very difficult as 0.2 mg/L nitrate is very close to the detection limit for nitrate (all values were <0.3 mg/L as N). However, biofilter effluent HPC counts were high (ranging from 10^2 to 10^4 colony forming units per milliliter, CFU/mL), and aldehydes were being removed during filtration, showing that the filters were definitely operating biologically. During the first five months of the study, nitrite levels rose up to 0.07 mg/L in the biofilter effluent and—after postchlorination—levels were controlled to less than 0.01 mg/L nitrite-N.



Source: Teefy et al., 2010

Figure 2-1 Ammonia levels before and after biofiltration at ACWD.

Cases of Incomplete Nitrification

Seasonal variations in temperature lead to periods of process instability and incomplete removal of ammonia or increased nitrite production, both of which are undesirable from a process control and water quality point of view. Other potential issues with the application of biological processes as part of the treatment process include biomass shedding and generation of microbial growth by-products, which may lead to an increased disinfectant demand.

Andersson et al. (2001) reported 45 to 90 percent ammonia removal in full-scale filters at temperatures between 4°C and 10°C, while this removal efficiency decreased to below 30 percent in cold water below 4°C. Uhl and Gimbel (2000) discussed the periodic inadequacy of the treatment process (ozonation, clarification, ozonation, dual-media filtration, GAC adsorption, groundwater infiltration) in Germany treating ammonia in river water. Ammonia concentration in the river water varied between 0.05 mg/L in summer and up to 4.2 mg/L in winter. During most of the year, effective ammonia removal by the treatment train was achieved. At temperatures above 20°C, up to 1 mg/L of ammonia was already removed during the clarification step. When water temperature dropped below 5°C, ammonia concentrations in the raw water increased very fast, and the removal capacity of nitrifying bacteria in the filters was not sufficient to treat these high concentrations.

Kors et al. (1998) discussed a case of incomplete nitrification in the Netherlands under extreme cold-water conditions below 4°C. Typically, the treatment plant (coagulation, sedimentation, holding reservoir with 100 days detention, rapid sand filtration, ozonation, softening, BAC filtration, slow-sand filtration) can efficiently remove between 1.5 and 3.0 mg/L of the raw water ammonia. The holding reservoir usually accounts for 50 to 90 percent of ammonia oxidation to nitrate and the remaining ammonia (between 0.3 and 1.2 mg/L) is nitrified during passage through the rapid sand filters. Nitrite is generally not present in either reservoir or sand filter effluents.

The temperature decreased in winter 1995–1996 to below 2.5°C for 100 days and was less than 1°C for 35 days. The water production was reduced by 20 percent; the backwash frequency of the filters was reduced; and low-ammonia raw water was blended to help maintain nitrification. These steps improved the removal, but the rapid sand filter effluent ammonia concentration increased to 0.8 mg/L. A phosphate salt solution was dosed ahead of the rapid sand filters at 0.1 to 0.15 mg/L PO_4^{3-} and resulted in an improvement of nitrification efficiency after 7 to 10 days lag time.

Positive results in restoring nitrification by phosphate were known from the literature and other utilities' experiences. The authors indicated that a full restoration of nitrification was possible at 1°C with phosphate addition and other treatment changes previously discussed.

The oxygen demand of nitrification is also significant. For complete nitrification (complete conversion to nitrate), 4.57 mg O_2 is required per mg $\text{NH}_4^+\text{-N}$ oxidized (USEPA 1975). If source water contains ammonia in excess of the stoichiometric amount of oxygen necessary to biologically oxidize it, the water will likely experience incomplete nitrification. The resulting finished water may contain elevated nitrite or little reduction in ammonia.

Need for Posttreatment Following Nitrifying or Biological Filters

As early as 1935, Feben observed that filters receiving ammoniated water were a near ideal environment for growth and multiplication of nitrifying bacteria and that the cells tended to clump into zoogeal masses. The tendency of sand grains to become coated with aluminum hydroxide floc allows ammonia-oxidizing bacteria (AOB) growth to occur. Organic matter, if present, is highly diluted, and the bacteria receive

an uninterrupted supply of nutrients as the waste products are washed away. The filter effluent contains these organisms and, if they survive postchlorination, they will enter the distribution system. Feben (1935) discussed nitrification in the filter beds and argued against preammoniation prior to the filters and chlorination after: "Sand filtration after ammoniation promotes their growth to a point where the chloramine process defeats its own purpose and becomes costly due to wasting of both ammonia to feed the bacteria and chlorine to oxidize their products." He also demonstrated the relative difficulty in disinfecting nitrifying bacteria with chlorine.

Bacterial aggregates from biological processes or bacteria attached to sand or GAC particles are more resistant to disinfection than individual bacterial cells. Thus, a higher chlorine dosage and a longer contact time may be necessary following a biological process to effectively disinfect the product water (Bouwer and Crowe 1988).

Skadsen (1993) and Tokuno (1999, 1997) reported cases of detrimental impacts of GAC filtration on distribution system chloramine stability and nitrification. In both instances, raw water was chloraminated and high doses of free ammonia were applied to fresh GAC filters. Skadsen (1993) reported the first incident of chloramine loss and nitrification in the distribution system to coincide with the change from sand to GAC filter medium. Predisinfection with monochloramine and 1.3 mg/L excess ammonia applied to the GAC filters was thought to be responsible for seeding the system with nitrifying bacteria growing in the filter bed.

Possible release of GAC fines was also implicated in increased heterotrophic plate counts (HPCs) in the system. Tokuno (1997) compared operation of the newly installed GAC filter to anthracite/sand filters fed with 4 mg/L total chlorine and >0.5 mg/L free ammonia. Nitrification in the GAC filter was evidenced within two months of installation with complete removal of total chlorine and ammonia, pH drop of 0.4 units, 3 mg/L nitrite-N, and loss of residual in the distribution system.

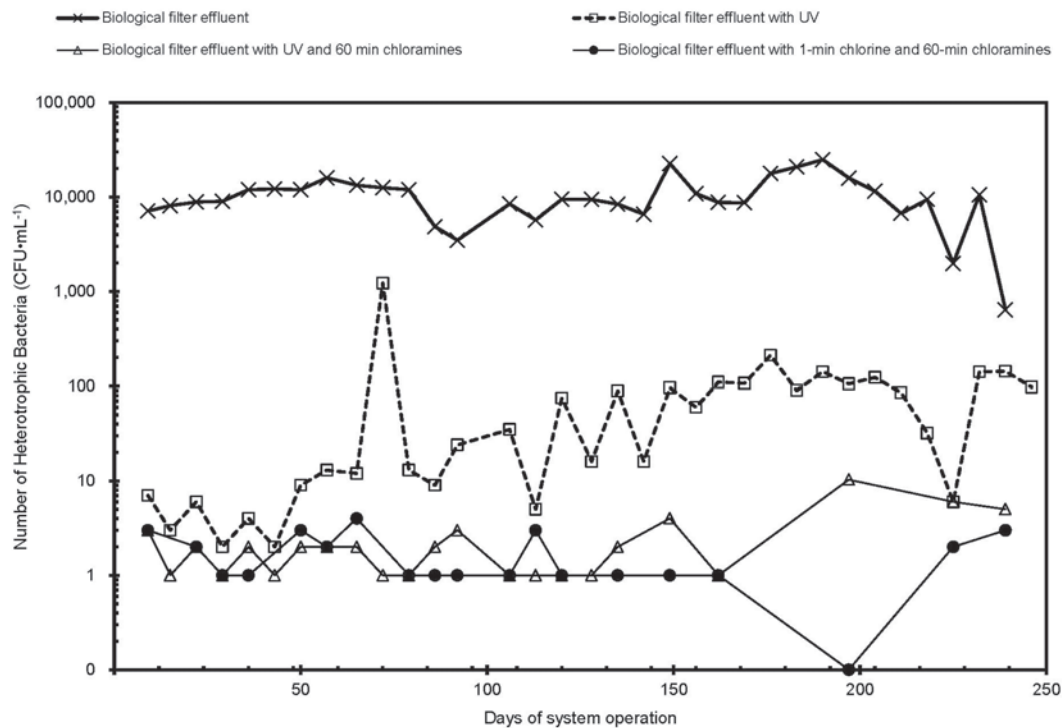
Rittmann and Snoeyink (1984) stated that a single barrier (i.e., postdisinfection) between a process in which microbiological growth is encouraged and the consumer may be considered insufficient in the event undesirable organisms are produced by the process. A good alternative is the use of a biological process, such as a fluidized filter bed, as the first process in the treatment train and to follow biological process with a conventional treatment sequence. Wilczak *et al.* (2003) observed that a free chlorine contact time of one hour or more was necessary after ozone/BAC filtration to avoid severe nitrification in the distribution system. Some modern applications of nitrifying anthracite filters in France are followed by membrane filtration, which ensures that neither undesirable organisms nor growth by-products pass into the product water (pers. commun., A. Wilczak 2004). The impact of treatment processes on chloramine stability and subsequent nitrification is discussed in detail in Chapter 4.

Lytle *et al.* (2007) performed biological analysis (HPCs and AOB) across the treatment train of a full-scale nitrifying biological filtration plant. Raw water contained 4.1×10^2 CFU/mL HPCs and only 7.9×10^{-1} MPN/mL AOBs. The lower AOB counts are most likely due to the lack of oxygen in the raw water. Treatment plant effluent (chlorinated water) bacteria levels were less than detectable, which demonstrated that chlorine effectively killed bacteria that were presumably shed from the filters. Full-scale filter media contained a large amount of HPC bacteria (4.5×10^4 CFU/mL) and a large number of AOB (2.9×10^3 MPN/mL). In a pilot study (White, 2010), two pilot-scale biologically active filters were designed to oxidize approximately 1.3 mg N-NH₃/L. The filter effluent was tested for heterotrophic bacteria throughout the duration of the 120 day study. The filter effluent HPC counts were 31,389 CFU/ml and 34,915 CFU/ml for anthracite/sand and small gravel filters, respectively.

Mofidi and Linden (2004) showed how UV disinfection can be effective in treating biological filter effluent water, although care needs to be maintained in monitoring

disinfection effectiveness and long-term performance degradation issues. Testing with UV was compared along-side chlorine and chloramines contact at both the bench and full-scale (3 mgd). Bench-scale results showed that UV was comparable to the chlorine alternatives, reducing HPC counts from 10^4 CFU/mL to 10^0 CFU/mL. Full-scale testing downstream of a 500 ft² anthracite/sand biofilter was conducted over a period of nearly 250 days. The challenge with UV treatment (as shown in Figure 2-2) was that HPC counts slowly climbed in the UV effluent during this nearly one-year duration of testing (sample taps in concrete mortar-lined steel pipe were back-flushed with chlorine so as not to skew results). For up to 50 days, UV and chlorine disinfection options showed similar performance. From 50 to 150 days, UV effluent counts increased from 10^1 CFU/mL to 10^2 CFU/mL while chlorine disinfectants kept HPCs to $<10^1$ CFU/mL. Disinfection-effluent counts remained at these levels for the remainder of the study.

Very little information is available on the microbial ecology within a full-scale nitrifying biological water treatment system. White *et al.* (2009 2010) reported on the microbial community within a biologically active, full-scale filtration water treatment plant in Ohio (details described in Lytle *et al.* 2007) using molecular microbiological approaches. A culture independent analysis of the filter media using 16s and *amoA* molecular markers revealed a complex microbial community. Specifically, the major ammonia oxidizers were identified as members of the genus *Nitrosomonas*, the majority being closely related to *Nitrosomonas oligotropha*. Other ammonia oxidizing bacteria identified were *Nitrosomonas europaea* and *Nitrospira multiformis*. The nitrite-oxidizing bacteria guild ecology was more diverse. The majority (greater than 50 percent of clones) of identified nitrite oxidizers were of the genus *Nitrospira* of the phylum Nitrospirae. Although many clones were closely related to the order Rhizobiales,



Courtesy of IWA Publishing, Journal Water Supply, Research and Technology—AQUA (53:8)

Figure 2-2 Heterotrophic plate count bacteria released from biologically active drinking water filter and treated with UV, chlorine, and chloramines.