

Biological nitrification in a full-scale and pilot-scale iron removal drinking water treatment plant

Darren A. Lytle, Thomas J. Sorg, Lili Wang, Christy Muhlen, Matthew Rahrig and Ken French

ABSTRACT

Ammonia in source waters can cause water treatment and distribution system problems, many of which are associated with biological nitrification. Therefore, in some cases, the removal of ammonia from water is desirable. Biological oxidation of ammonia to nitrite and nitrate (nitrification) is well understood and common in wastewater processes. The biological filtration to convert ammonia to nitrate in drinking water applications in full-scale systems is limited in the United States. The purpose of this study was two-fold: (1) to monitor and evaluate nitrification in a full-scale iron removal filtration plant with biologically active granular media filters located in Ohio, and (2) to determine how to most efficiently regain nitrification following filter rebedding with new filter media. Results showed that the biologically- active filters consistently oxidized all of the 1.2 mg/L NH₃-N to nitrate. Seasonal variations in ammonia oxidation effectiveness were not observed because yearly changes in water temperature and other water quality parameters were minimal. Pilot tests using dual anthracite/sand filters were used to determine the time required to achieve complete nitrification by three different seeding methods of new filters. The results of the pilot tests showed that all three methods took approximately 70 days. Biological oxidation of ammonia is a simple, robust and effective way to convert ammonia to nitrate in full-scale water treatment systems.

Key words | ammonia, bacteria, drinking water, filtration, nitrification

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INTRODUCTION

The drinking water industry has an interest in the fate of ammonia (NH₃) within the treatment and distribution of drinking water. Research on the presence of ammonia in drinking water distribution systems has suggested some correlation between excess ammonia and 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 effects on water's taste and odor (Bouwer & Crowe 1988; Rittmann & Huck 1989). Ammonia occurs naturally in some groundwater, or it is added to water to form chloramines in drinking water distribution systems.

Chloramines (a combination treatment of ammonia and chlorine) are a commonly used disinfectant in the water treatment process, and they are used in place of free chlorine because they produce lower levels of disinfection by-products such as trihalomethanes, haloacetic acids, and other halogenated organic compounds, which are potentially carcinogenic or mutagenic. Free ammonia is produced when chloramines break down within the distribution system (Vikesland *et al.* 2001) or when excess ammonia relative to chlorine is added in the chloramines production process.

The oxidation of ammonia to nitrite and then nitrate is a biological process referred to as nitrification. When

nitrification occurs uncontrollably in the distribution system, the biological stability of the distribution system is disrupted, which can cause a number of water quality problems (Rittmann & Snoeyink 1984). The autotrophic bacteria responsible for nitrification are abundant in many source waters, and can grow readily in distribution systems if ammonia and oxygen substrates are available. The occurrence of nitrification in distribution systems is common and has been well-documented (Rittmann & Snoeyink 1984; Odell *et al.* 1996; Fleming *et al.* 2005). The growth of nitrifying bacteria in distribution systems can cause a number of problems. For example, biological activity has been shown to promote corrosion of some metals (Lee *et al.* 1980; Bremer & Wells 2001). In the case of nitrifying bacteria, the corresponding pH drop associated with the biological oxidation of ammonia directly impacts corrosion of distribution system materials. In addition, the nitrifying bacteria support the development of undesirable heterotrophic biofilms by supplying organic carbon substrates; these biofilms produce metabolic byproducts that adversely affect the taste and odor of the water (Suffet *et al.* 1996). Incomplete nitrification of ammonia can result in increased levels of toxic nitrite (NO_2^-). Because the United States Environmental Protection Agency's (U.S. EPA) maximum contaminant levels (MCLs) apply at the entry point into the distribution system rather than within the distribution system, monitoring of contaminants such as nitrite and nitrate are normally not conducted at the consumer's tap.

Many regions in the United States have excessive levels of ammonia in their source waters. For example, farming and agricultural sources of ammonia in the Midwest contribute to relatively high levels of ammonia in many groundwaters. Although ammonia in water does not pose a direct health concern, nitrification of significant levels of excessive ammonia may. In addition, ammonia in arsenic bearing waters, for example, may negatively impact arsenic removal by creating a chlorine demand and reducing the chlorine's availability to oxidize arsenic. Clearly, the complete oxidation of excess source water ammonia during the treatment process reduces the potential negative impact (nitrification) on distribution system water quality. While physicochemical methods for ammonia removal are possible, such as ion exchange, biological methods appear to be more efficient and cost-effective.

Biologically-active filtration has been used successfully in Europe for years. Bouwer & Crowe (1988) documented the use of various biological methods throughout Great Britain, France, and Germany, including fluidized beds, rapid sand filters, biologically active granulated active carbon (GAC), and soil-aquifer treatment. However, 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. A number of 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 purpose of this study was two-fold: (1) to monitor and evaluate nitrification in a full-scale iron removal filtration plant with biologically active filters and (2) to determine how to most efficiently regain nitrification following filter rebedding with new filter media using a pilot-scale filter study.

BACKGROUND

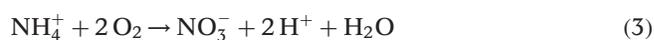
Biological conversion of ammonia (NH_3) to nitrate (NO_3^-) involves a two-step sequence of reactions mediated by two different genera of bacteria, *Nitrosomonas* and *Nitrobacter*. These autotrophic bacteria derive energy for cellular functions from the oxidation of ammonia and nitrite, respectively. *Nitrosomonas* are responsible for the oxidation of ammonia, in the form of ammonium (NH_4^+), to nitrite (NO_2^-) according to the reaction:



Nitrobacter or *Nitrospira* (depending on environment) subsequently oxidize nitrite to nitrate, as follows:



By summing these equations, the overall nitrification reaction is obtained:



These equations are net reactions involving a complex series of enzyme-catalyzed intermediate steps. From Equation 3, it is clear that nitrification produces free protons, which readily consume available bicarbonate ions (HCO_3^-), thereby reducing the buffering capacity of the water. The consumption of alkalinity (as CaCO_3) by nitrification is 7.14 mg/mg $\text{NH}_4^+\text{-N}$ oxidized (U.S. EPA 1975). The oxygen demand of nitrification is also significant. For complete nitrification, 4.57 mg O_2 is required per mg $\text{NH}_4^+\text{-N}$ oxidized (U.S. EPA 1975). The efficiency of biological nitrification is greatly reduced with the lowering of temperature, particularly as the temperature falls below 15°C (Bouillot *et al.* 1992; Kors *et al.* 1997; Uhl & Gimbel 2000).

The application of biological oxidation of ammonia is well-established in the wastewater treatment field (Metcalf & Eddy 2003). In the drinking water field, biological nitrification to oxidize ammonia from source water has not been widely reported and is generally limited to pilot programs. Biological treatment of water in the United States is generally not accepted for a variety of reasons.

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, alkalinity 20 to 50 mg CaCO_3/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 times, EBCTs, between 20 to 30 minutes). At 15°C, an average of 94% 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% at temperatures above 10°C. Ammonia removal dropped to 10 to 40% at moderate temperatures of 4 to 10°C.

METHODS AND MATERIALS

Full-scale treatment plant analysis

The Greene County South Water Treatment Plant (GCSWTP), located in Beavercreek, Ohio, was the study

site (Figures 1 and 2). Average plant production is near 0.6 million gallons per day (MGD), and maximum production is 1.2 MGD. The plant, designed as an iron removal plant, treats raw water from one of three wells. Water is pumped to the top of a forced draft aerator (Figure 2a) and then to a 5000-gallon retention basin located at the bottom of an aerator that provides a retention time of approximately 12 minutes. During the summer months, chlorine tablets are occasionally added to the retention basin to control insects. The aerated water then flows by gravity to a bank of three dual media (sand, anthracite) filters (Figure 2b), each filter having an area of 148 ft². The filter beds are composed of a layer of underbedding gravels and a layer of sand (the actual depth of each layer is unknown), which have been in service since the 1980s. According to the plant operator, at least one of the filters also has some anthrasand (Figure 3). The hydraulic loading rate to each filter is about 2 gpm/ft². The filters are backwashed one at a time, using treated water every 3 days (one filter per day) with a flow rate of approximately 2,100 gpm (corresponding to 14 gpm/ft² hydraulic loading) for approximately 10 to 12 minutes. After filtration, chlorine is added as liquid NaOCl to provide a free chlorine residual of approximately 0.9 mg Cl_2/L .

A total of six sample sets were collected monthly during the study period between 3/10/2004 and 8/26/2004 from various stages of the treatment process. Additional sampling events were conducted between January 2003 and April 2005 to catch seasonal variations in water quality. Changes in important water quality parameters, such as temperature, were small, and trends in water treatment efficiency were not noticeably impacted (i.e. additional measurements fell within the average \pm standard deviation of respective values reported during the study period). Therefore, data discussion was limited to the sequential 6 month sampling effort. The water samples were collected from four locations: (1) the raw water before treatment (Raw), (2) the water before filtration (BF), (3) the water after filtration (AF), and (4) the water after chlorine addition (AC) (Figure 1). Each sample was analyzed for pH, alkalinity, dissolved oxygen (DO), oxidation-reduction potential (ORP), temperature, free and total chlorine, ammonia, nitrite, nitrate, total organic carbon (TOC), dissolved iron, and total iron.

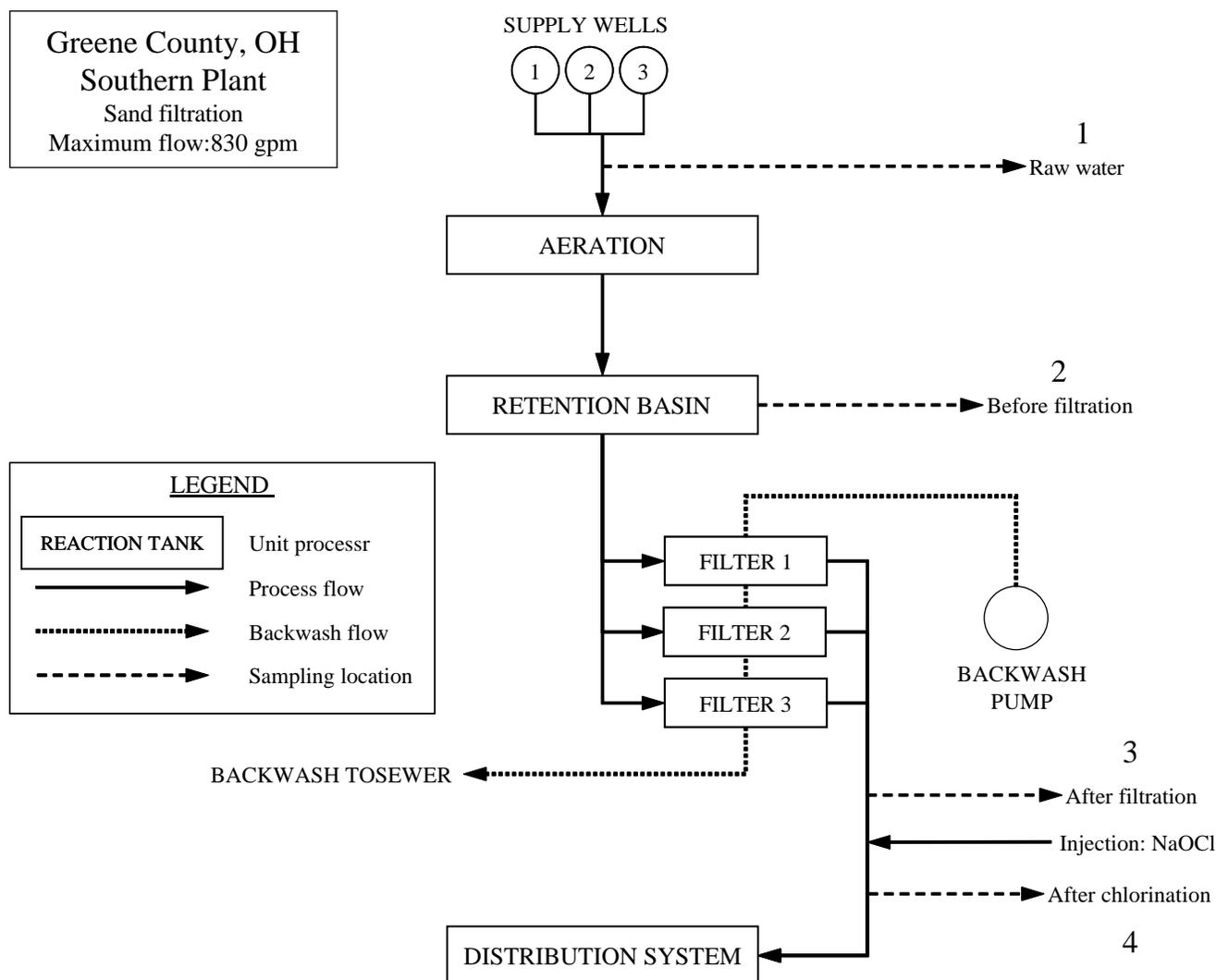


Figure 1 | Greene County South Plant: schematic of treatment process and water sampling points; 1 = raw water (Raw), 2 = aerated water (before filtration, BF), 3 = filtered water (after filtration, AF), and 4 = chlorinated filtered water (water after chlorine addition, AC).

Pilot-scale treatment study

The test apparatus used for the filter pilot study consisted of six individual 8-foot transparent glass columns, 2 inches in diameter (Figure 4) and the design was based on previous U.S. EPA studies. The pilot filters were designed and operated considering the future plans (e.g. type of replacement media, media depth, etc...) of the utility and existing plant conditions (e.g. filter loading rate, filter backwash frequency, etc...). From bottom to top, the media consisted of 4 inches of large gravel, 4 inches of medium gravel, 4 inches of coarse sand, 10 inches of fine sand (effective size of 0.97 mm), and 20 inches of anthracite (effective size of

0.97 mm). The columns were fed with GCSWTP aerated water directly through PVC pipe (point #2 in Figure 1) at a loading rate of 2 gpm/ft² in each column. Initially, the filter columns were seeded with nitrifying bacteria by three different methods. Columns 1 and 2 (C1 and C2) were considered the control, and thus were operated by simply passing the aerated water through the filter media. To seed columns 3 and 4 (C3 and C4), three inches of anthracite from these columns were replaced by three inches of anthracite removed from filter 3 of the GCSWTP. This filter was known to have active biological nitrification. Columns 5 and 6 were seeded with filter backwash water from the GCSWTP, which was passed through the columns

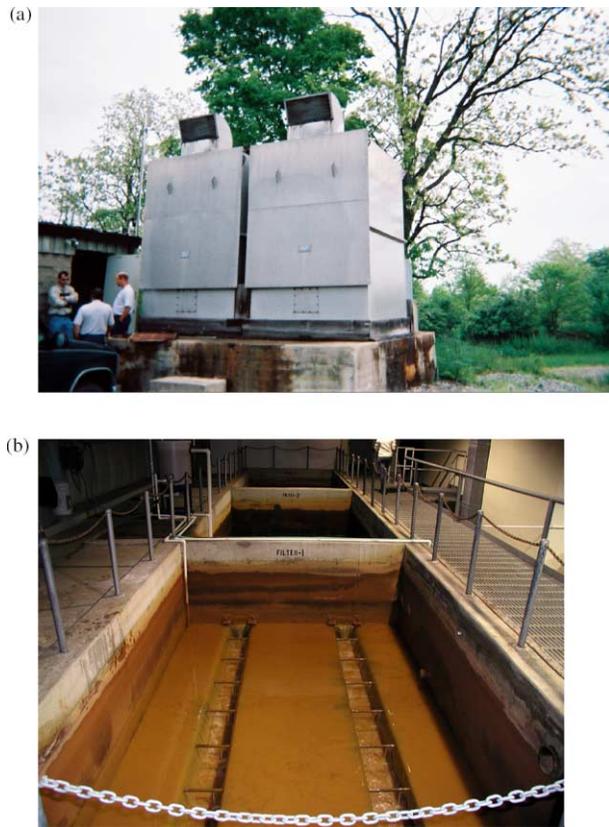


Figure 2 | Photographs of Greene County South water treatment plant (GCSWTP); (a) aeration tower, and (b) gravity filters.

until the head loss in each filter increased from 20 inches to around 35 inches (C5 and C6). Following seeding, the filter columns began operation on 8/5/2005. Backwashing was performed twice weekly, each time for 15 minutes at a 50% bed expansion. On a weekly (or more frequent) basis, the influent and effluent from each column were sampled for ammonia nitrogen, nitrate nitrogen, pH, and dissolved oxygen. Also, the head loss and flow rates of each column were recorded weekly.

Analytical techniques

The pH was measured on-site with a Hach Company (Loveland, CO) EC40 benchtop pH/ISE meter (Model 50125), and a Hach Company (Loveland, CO) combination pH electrode (Model 48600) with temperature corrections. A two-point calibration with pH 7 and 10 standard solutions standardized the instrument daily (Whatman

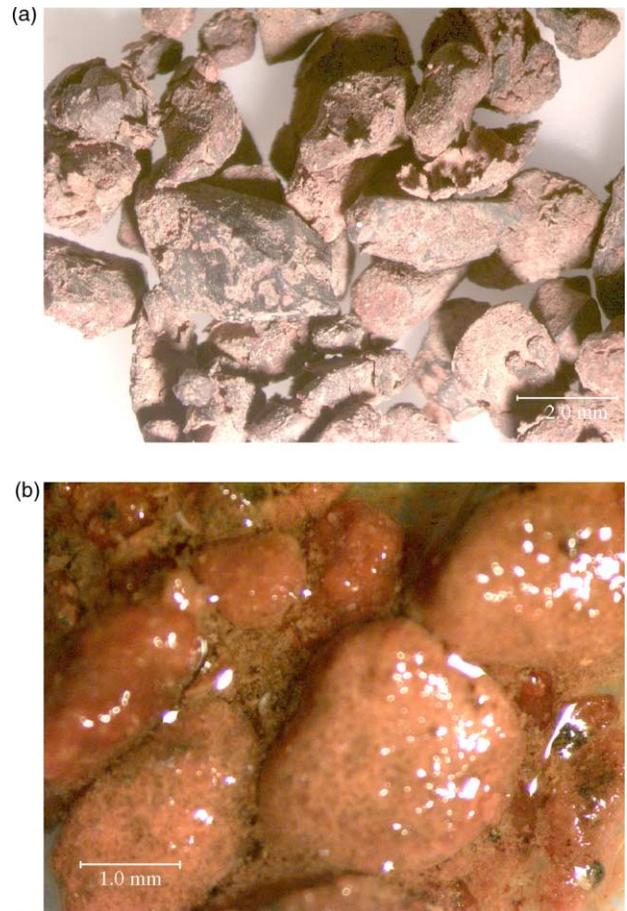


Figure 3 | Stereomicrographs of GCSWTP anthrasand filter media; (a) dried media, and (b) wet media.



Figure 4 | Photograph of pilot scale filtration system.

Hillsboro, OR). Dissolved oxygen (DO) was measured with a Hach Company (Loveland, CO) Model DO175 dissolved oxygen meter and a Model 50180 dissolved oxygen probe. Care was taken to draw DO samples slowly and make DO measurements shortly after sampling to limit the introduction of air to the sample. Free and total chlorine were measured with a Hach DR/2000 spectrophotometer (Loveland, CO), using the DPD method (Standard Method, 4500-Cl G) (Hach Company, 1990). A Thermo Jarrel Ash (Franklin, MA) 61E[®] purged inductively coupled argon plasma atomic emission spectrophotometer (ICP-AES) was used to analyze total iron and other elements including Fe, Mn, S, Ca, Mg, Si, P, etc. (U.S. EPA Method 200.7) (U.S. EPA 1994) during pilot studies and an inductively coupled argon plasma atomic mass spectrophotometer (ICP-MS) (U.S. EPA Method 200.8) (U.S. EPA 1994) was used to measure the same elements in the full-scale study. In the field, ferrous iron was measured using the 1,10 phenanthroline method (Hach Company, 1990; APHA-AWWA-WEF, 2005), and total iron was also measured by the same method, except that a reducing reagent was also included in the reagent powder pillow provided by the Hach Company to convert Fe(III) to Fe(II).

Ammonia (NH₃) was measured in the field with a Hach DR890 colorimeter using method 66. Additionally, nitrate was measured by a Hach DR890 colorimeter using methods 53 and 54. Nitrate and nitrite were further measured by ion chromatography (U.S. EPA Method 300.0) (U.S. EPA 1993) in the laboratory. Nitrate samples were preserved at 4°C and H₂SO₄ to pH < 2, and were analyzed within 28 days and the nitrite was analyzed within 48 hours.

Other water samples were also sent back to either the U.S. EPA or Battelle laboratories for analysis. Dissolved inorganic carbon (DIC) was analyzed by a coulometric procedure on a UIC Model 5011 CO₂ coulometer (Joliet, IL) with Model 50 acidification module, operated under computer control. Syringe filters (0.2 μm) (Anotop 25[®], Whatman, Inc., Clifton NJ) were used to separate colloidal iron from soluble iron.

Biological analysis

The biological counts of the water samples, filter backwash solids and filter media were assayed within hours of

sampling. In the case of solids, biological analysis was performed on the excess water collected with the solids after the sample was briefly mixed. Total heterotrophic plate counts (HPCs) were determined by SM 9215 C using R2A media and incubation at 22°C for seven days. Ammonia-oxidizing bacteria (AOB) were enumerated based upon nitrite/nitrate detection following 30 days incubation at 28°C in Soriano-Walker media using most probable number (MPN) estimates and using 10 tubes/dilution (Soriano & Walker 1968; Koops & Möller 1992; APHA-AWWA-WEF 2005).

RESULTS

Full-scale treatment plant

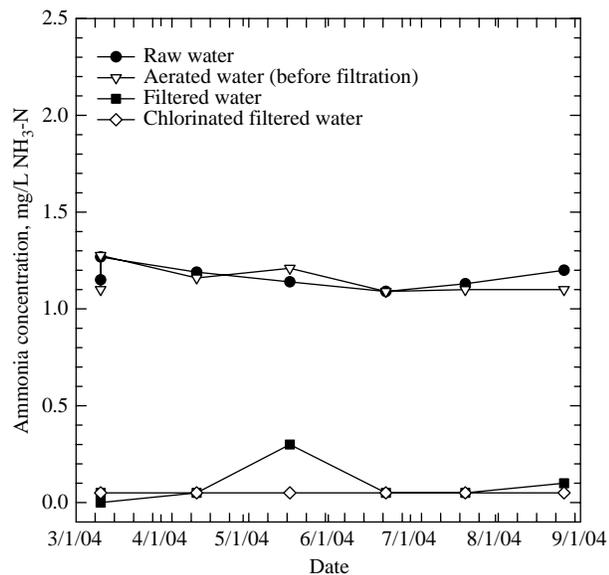
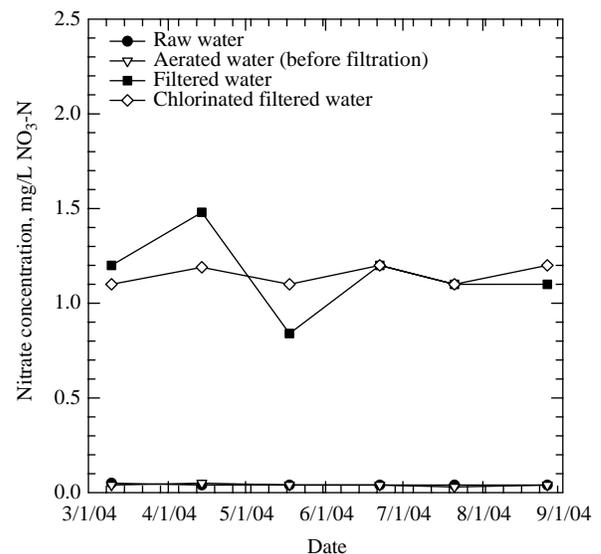
Regularly monitored water quality data from the four sampling locations of the GCSWTP are summarized in Table 1. Reported values represent averages of six sets of water samples collected in monthly intervals between March 2004 and September 2004. Additional raw water quality parameters of interest include: 83 mg Ca/L, 36 mg Mg/L, 16 mg SiO₂/L, and 8 mg SO₄/L.

Most of the influent ammonia was completely oxidized to nitrate during filtration, as indicated by the sharp reduction in the ammonia concentration after filtration at the four sampling points of the treatment process (Figure 5) and an equivalent rise in nitrate-nitrogen concentration after filtration (Figure 6). The ammonia concentrations at the two sampling locations prior to filtration, namely raw water and before filtration locations averaged 1.15 and 1.11 mg NH₃-N /L (Table 1) respectively, while the nitrate concentrations for the same locations were below the detection limit of 0.04 mg NO₃-N/L. Following filtration, the ammonia concentration dropped below the detection limit of 0.1 mg/L at the two sampling locations (AF and AC locations), while the concentration of nitrate increased to an average of 1.11 mg NO₃-N/L before chlorination (AF) and 1.15 mg NO₃-N/L after chlorination (AC). Very little to no nitrite was present at any point in the treatment process (<0.02 mg/L), confirming complete nitrification of ammonia to nitrate. The aeration process did

Table 1 | GCSWTP water quality summary*

Analyte	Raw	Before Filtration (BF)	After Filtration (AF)	After Chlorine Addition (AC)
pH	7.48 ± 0.1	7.76 ± 0.2	7.76 ± 0.1	7.68 ± 0.0
Alkalinity (mg/L as CaCO ₃)	373 ± 11	376 ± 12	365 ± 7	360 ± 10
DO (mg/L)	1.22 ± 0.4	6.74 ± 0.4	5.4 ± 1.0	6.33 ± 0.6
ORP (mV)	-114 ± 19	36 ± 57	187 ± 144	598 ± 36
Temperature (°C)	14 ± 2	14 ± 2	14 ± 2	16 ± 2
Free Chlorine (mg/L)	NA	0.03 ± 0.0	0.04 ± 0.2	0.87 ± 0.3
Total Chlorine (mg/L)	NA	0.04 ± 0.1	0.03 ± 0.2	0.97 ± 0.3
Ammonia (lab) (mg/L as N)	1.15 ± 0.0	1.11 ± 0.1	0.1 ± 0.1	<0.1
Nitrate (mg/L as N)	<0.04	<0.04	1.11 ± 0.2	1.15 ± 0.1
Nitrate (mg/L as N)	<0.01	<0.01	0.02 ± 0.0	<0.01
TOC (mg/L)	1.2	1.1	1.1	1.0
Total Fe (µg/L)	2290 ± 116	2266 ± 110	79 ± 127	<25
Dissolved Fe (µg/L)	2312 ± 159	203 ± 133	<25	<25

*Values listed are averages ± standard deviation of six samples collected between 3/10/04 and 8/26/04.

**Figure 5** | Ammonia concentrations through the GCSWTP.**Figure 6** | Nitrate concentrations through the GCSWTP.

not oxidize ammonia, and chlorination did not impact ammonia levels.

Dissolved oxygen and pH levels were also closely monitored because of the important role they play in the nitrification process (Equations 1–3). Aeration increased dissolved oxygen from 1.2 mg/L to 6.74 mg/L and slightly increased the pH from 7.48 to 7.76, presumably due to the release of CO₂ gas. On average, the dissolved oxygen concentration was 6.74 mg/L after aeration and dropped to 5.40 mg/L after filtration (Table 1). The observed decrease in dissolved oxygen of 1.34 mg/L through the filters was far less than the expected 5.1 mg O₂/L drop predicted by the stoichiometric of the oxidation of ammonia to nitrate (Equations 1–3). The exact reason for the lower than expected drop is not known. However, despite attempting to minimize air introduction to the sample, it is suspected that oxygen was introduced to the samples during or after sampling, or through the plumbing leading to the sampling ports. The pH level remained constant at 7.76 through the filtration process, but dropped slightly to 7.68 after chlorination, that is attributed to the addition of chlorine gas. The change in pH through the system was minimal, likely due to the relatively strong buffer intensity of the high alkalinity source water. The results from changes in dissolved oxygen and pH are displayed in Figures 7 and 8, respectively.

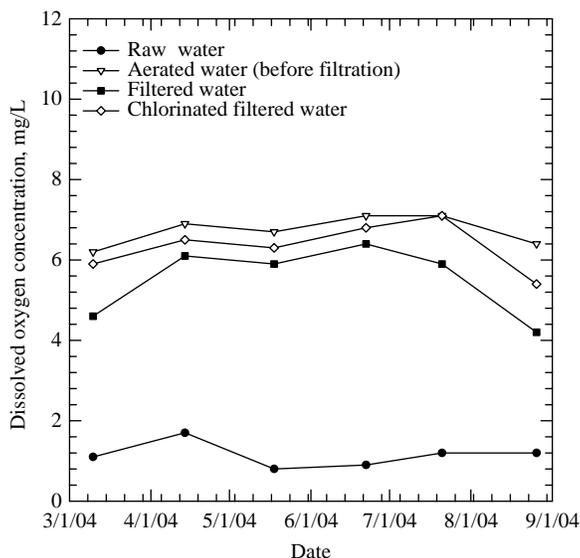


Figure 7 | Dissolved oxygen concentrations through the GCSWTP.

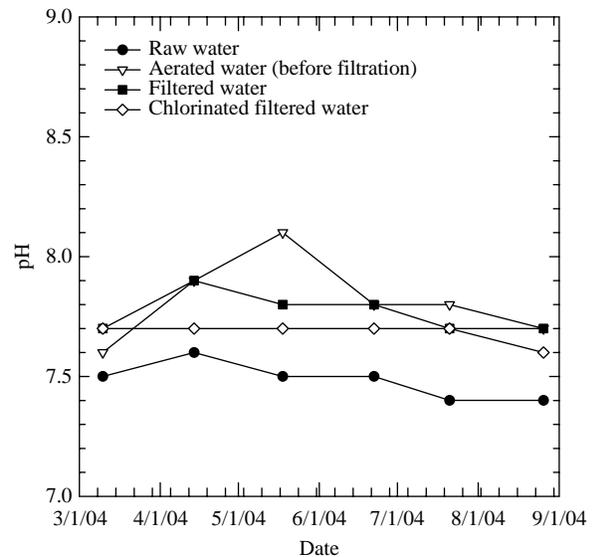


Figure 8 | pH levels through the GCSWTP.

The oxidation-reduction potential (ORP) reflects the presence of oxidants in the water. As seen in Figure 9, the ORP at the GCSWTP of the raw water averaged -114 mV, which reflects the nature of anaerobic ground water. The introduction of oxygen during aeration increased the ORP to $+36$ mV. The ORP also increased through the filters as Fe²⁺ was oxidized. Chlorine injection increased the ORP to $+598$ mV, which reflects the presence of a free chlorine residual.

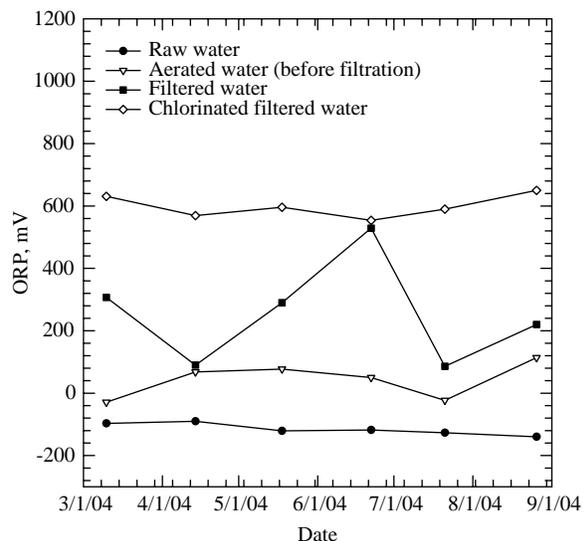


Figure 9 | ORP through the GCSWTP.

The chlorine demand of the filtered water due to ammonia was relatively low as indicated by the small fraction of combined chlorine. After disinfection, the mean residual total chlorine level was 0.97 mg/L. The mean residual free chlorine concentration was 0.87 mg/L. The low fraction of total or combined chlorine is attributed to the effective oxidation of ammonia in the filters. The levels of free and total chlorine of the finished water are plotted in Figure 10.

Iron removal was the intended objective of the GCSWTP. Iron was effectively reduced from an average of 2.3 mg/L in the raw water to 0.08 mg/L after filtration, and less than detection (<0.025 mg/L) following the point of chlorination. Aeration effectively converted most of the soluble (ferrous) iron in the raw water to particulate (ferric) iron based on total and dissolved iron levels before filtration.

Last, temperature is a very important parameter in the biological nitrification process (Andersson *et al.* 2001). Temperature averaged 14°C from the raw water though the filters, and it increased by 2°C following chlorination.

Pilot-scale treatment results

Because Greene County has a need to repair and re-bed their old full-scale filters, a pilot-scale study was conducted

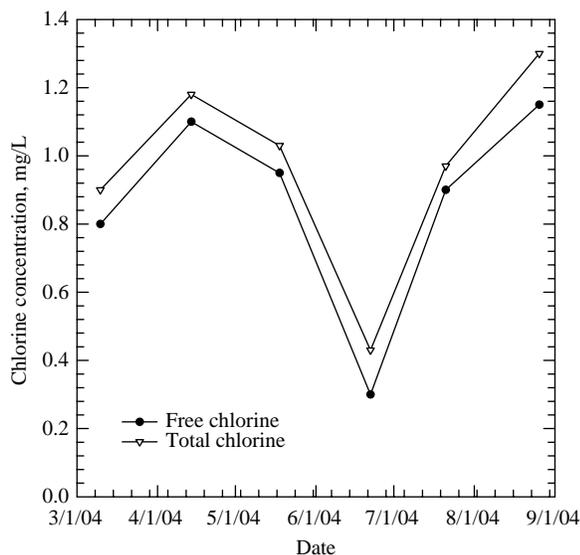


Figure 10 | Free and total chlorine concentrations of chlorinated filtered water through the GCSWTP.

to evaluate the time-frame needed to fully biologically acclimate new dual media filters based on ammonia oxidation measurements. Three “re-seeding” approaches were evaluated using the aerated water, which included (1) simply running aerated water through the filter, (2) adding three inches of full-scale media to the top of the filter columns, and (3) periodically adding backwash solids from the full-scale filters to the filter. The pilot study was monitored over a three month study period, and water quality is summarized in Table 2.

The ammonia concentration in the aerated water averaged 1.3 mg/L $\text{NH}_3\text{-N}$ (Table 2). Over the first 20 days of operation, filter effluent ammonia levels (presented as average of two filter columns, Figure 11) from seeded filters were as much as 0.4 mg/L $\text{NH}_3\text{-N}$ and 0.2 mg/L $\text{NH}_3\text{-N}$ (Figure 11 C3-C4, and C5-C6, respectively) lower than the influent water. The ammonia levels from the columns with the three inches of full-scale media were generally the lowest (Figure 11 C3-C4), followed by the columns containing backwash solids (Figure 11 C5-C6). The differences in ammonia filter effluent values for two sets of columns appeared to be relatively small, and probably not of a practical significance because the real interest is the time of operation needed to reach complete oxidation of ammonia. The columns with only aerated water (Figure 11 C1-C2) experienced very little ammonia reduction during the same 20-day period of operation. Between days 20 to 45, effluent ammonia levels dropped slightly from the previous period in the seeded filters (Figure 11 C3-C6), but they were still relatively high (>0.4 mg/L $\text{NH}_3\text{-N}$). However, ammonia reductions through columns 1 and 2 in Figure 11 dropped greatly during this time period to levels that approached effluent water from the seeded filters. This suggests the beginning of enhanced growth of bacteria within the filter media. Between 45 and 70 days, as seen in Figure 11, effluent ammonia levels dramatically dropped in all six filter columns. There still appeared to be a slight difference between column effluent ammonia levels during this time period. Beyond 70 days, essentially all of the ammonia in all columns was oxidized with no difference in effluent ammonia levels. Ammonia mass balances were performed regularly during the pilot tests.

Nitrogen mass balances through the filters were routinely determined over the duration of the pilot study

Table 2 | Pilot-scale study water chemistry summary *

Analyte	Filter Effluent			
	BF	C1-C2	C3-C4	C5-C6
pH	8.20 ± 0.2	8.01 ± 0.1	8.05 ± 0.1	8.03 ± 0.1
Alkalinity (mg/L as CaCO ₃)	366	362	361	362
DO (mg/L)	5.63 ± 1.2	5.25 ± 0.6	4.86 ± 0.7	5.72 ± 1.1
ORP (mV)	230 ± 25	241 ± 12	246 ± 15	248 ± 14
Temperature (°C)	14.6 ± 1.0	15.4 ± 1.6	15.6 ± 1.6	15.8 ± 1.7
Ammonia (on-site) (mg/L as N)	1.3 ± 0.2	Varies	Varies	Varies
Nitrate (mg/L as N)	0.07 ± .04	Varies	Varies	Varies
Nitrite (mg/L as N)	0.06 ± .01	Varies	Varies	Varies
Total Fe (µg/L)	3079 ± 162	<0.025	<0.025	<0.025

*Values listed are averages ± standard deviation of samples collected between 08/03/05 and 11/22/05.

(Figure 12). Early in the study, filter effluent contained a mix of ammonia, nitrite, and nitrate. As time passed, the results showed that nearly all of the influent nitrogen was ammonia, and nearly complete conversion of ammonia to nitrate occurred in all columns.

Iron removal through all pilot filters was very good (<0.025 mg/L) and was not impacted by nitrification (Table 2). Additional pilot work (not detailed in this manuscript) using the same pilot system seeded with bacteria from Greene County’s treatment plant showed that complete nitrification of 1 mg/L NH₃-N continued when iron was absent from the source water.

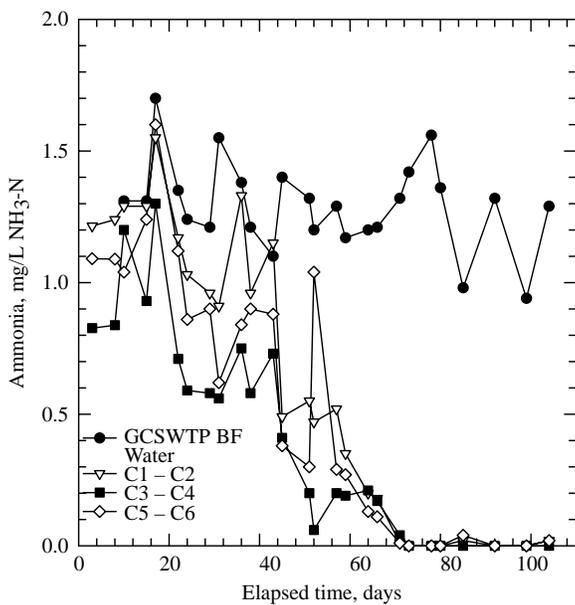


Figure 11 | Ammonia concentrations of effluent from pilot filter tests (C refers to filter column, plotted values represent the average of two filters).

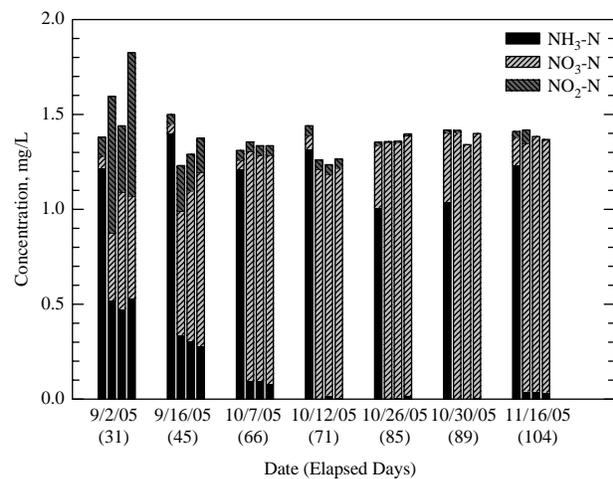


Figure 12 | Nitrogen mass balance of the three pilot filter tests (first bar of each data set represents raw water, second bar represents column 1, third represents column 3, and fourth column 5).

Table 3 | Biological counts in water collected throughout GCSWTP

Location	HPC, CFU/ml	AOB, MPN/ml
Raw	4.1E + 02	7.9E-01
Plant effluent	< 10	< 0.09
Media	4.5E + 04	2.9E + 03
Back wash	6.6E + 04	2.7E + 02
Point 1 well 4, 6, 8	2.2E + 05	9.2E-01
Point 2 well 4, 6, 8	3.2E + 05	1.7E + 01

Biological analysis

Biological analysis (HPCs and ammonia-oxidizing bacteria [AOB]) was performed on raw water, full-scale filter effluent, filter media, and filter backwash solids. The results (Table 3) showed 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). Clearly, the filters are biologically active and a large population of ammonia-oxidizing bacteria was present.

DISCUSSION AND CONCLUSIONS

Analysis of a number of water quality parameters, including NH_3 , NO_2 , NO_3 , pH, dissolved oxygen concentrations, and biological analysis clearly shows that complete biological nitrification is achieved through the filters of the GCSWTP. Ammonia oxidation efficiency was above 95% during the filtration process. Reduction in $\text{NH}_3\text{-N}$ concentration in the filters was balanced by an equivalent rise in the concentration of $\text{NO}_3\text{-N}$ in the filter effluent. Also, no nitrite was detected, demonstrating complete conversion of ammonia to nitrate. The observed decrease in dissolved oxygen through the filters of only 1.34 mg/L was far less than the

expected 5.1 mg O_2 /L drop predicted by the stoichiometric of the oxidation of ammonia to nitrate. The exact reason for the lower than expected drop was uncertain although introduction of air to the sampled waters during or after collection was suspected.

Only a small drop in pH associated with nitrification was observed, which is attributed to the extremely high alkalinity that averaged 376 mg/L (as CaCO_3) upon entering the filters. Alkalinity dropped to an average of 365 mg/L after filtration, which is relatively consistent with predictions. Theoretically, 7.07 mg/L (as CaCO_3) of alkalinity are consumed per mg of $\text{NH}_3\text{-N}$ oxidized (U.S. EPA 1975). The average $\text{NH}_3\text{-N}$ reduction was around 1.05 mg/L that corresponds to a theoretical reduction of 7.42 mg/L (as CaCO_3) of alkalinity. The actual reduction in alkalinity was slightly higher at 11 mg CaCO_3 /L, which correlated reasonably well. Complete nitrification was observed through two calendar years of observations. Seasonal water changes that can impact nitrification, such as temperature, were small because the source water was a relatively stable ground water supply.

Pilot study test results suggest that complete nitrification can be achieved in new filters in less than three months of constant operation. Simply running aerated raw water through new filter media will promote bacterial regrowth and nitrification. Seeding the new filters by adding some old media or by adding backwash solids did not prove to reduce the time to complete nitrification.

Biological oxidation of ammonia is a simple, robust and effective way to oxidize convert ammonia to nitrate. The GCSWTP has clearly demonstrated the effectiveness of using nitrifying bacteria in their filtration process to oxidize ammonia to nitrate. By doing so, the system has reduced both the potential of nitrification in the distribution system and the chlorine demand of the finished water.

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