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Summary Report:

Pilot Study of an Innovative Biological Treatment Process for the Removal of Ammonia from a Small Drinking Water System

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Pilot Study of an Innovative Biological Treatment Process for the Removal of Ammonia from a Small Drinking Water System

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Notice

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1. Background

1.1 Ammonia in Drinking Water Sources

Many regions in the United States have excessive levels of ammonia in their drinking water sources (e.g., ground and surface waters) as a result of naturally occurring processes, agricultural and urban runoff, concentrated animal feeding operations, municipal wastewater treatment plants, and other sources. Ammonia is not regulated by the U.S. Environmental Protection Agency (EPA) as a contaminant. Based on a 2003 World Health Organization (WHO) assessment, ammonia levels in groundwater are typically below 0.2 milligrams per liter (mg/L), and does not pose a direct health concern at levels expected in drinking water (WHO 2003); however, it may pose a concern when nitrification of significant levels of ammonia from the source water occurs in the drinking water distribution system. Specifically, this nitrification, which is the conversion of the ammonia to nitrite and nitrate by bacteria, leads to water quality issues, such as potential corrosion problems, oxidant demand, taste and odor complaints, and elevated nitrite levels (Bremer *et al.,* 2001; Fleming *et al*., 2005; Lee *et al*., 1980; Odell *et al*., 1996; Rittman & Snoeyink, 1984; Suffet *et al*., 1996).

Ammonia in water may also pose problems with water treatment effectiveness. For example, in source waters containing both ammonia and arsenic, the ammonia may negatively impact the removal of arsenic by creating a chlorine demand, therefore reducing the availability of chlorine needed to oxidize the arsenic (Lytle *et al.*, 2007). Lastly, water systems that have ammonia in their source water and desire to maintain a free chlorine residual will need to add additional chlorine to overcome the demand of ammonia. Clearly, the complete oxidation of source water ammonia prior to or as part of the water treatment process would eliminate the potential negative impacts of nitrification on distribution system water quality.

1.2 Community with Elevated Ammonia Levels

Many regions in the Midwest are particularly impacted by ammonia in their source waters from natural geology, agricultural runoff, and other farming practices. For example, the State of Iowa has a widespread distribution of ammonia in well waters across its communities (Figure 1). Water quality testing of the source groundwater in one of the communities (population approximately 873) (Table 1) showed that, on average, ammonia levels were 3.3 mg as nitrogen (N)/L. Although the focus of this report is on ammonia contamination, it is relevant to note that the water samples averaged 0.82 mg/L of iron. Similar to ammonia, iron in drinking water does not pose a direct health concern. However, there is an EPA recommended, non-enforceable National Secondary Drinking Water Regulation Standard of 0.3 mg/L for iron, which is based on aesthetic and technical issues, rather than health-based concerns. Specifically, iron in the water can cause a metallic taste, discoloration of the water, staining of faucet and fixtures, and sediment build-up. Given the negative issues associated with high ammonia levels in drinking water, and with the added issues from the high levels of iron, there is a clear need to establish effective treatment approaches to address these issues. Furthermore, the State of Iowa

Department of Natural Resources (DNR) can request water systems to monitor nitrite and nitrate in their distribution systems, should they suspect that nitrification is occurring in their distribution system.

1.3 Ammonia Treatment Options

The most commonly used water treatment options for addressing elevated ammonia in source waters are the formations of monochloramine and breakpoint chlorination. Breakpoint chlorination results in the removal of ammonia as nitrogen gas by a chemical reaction with chlorine; typically in the range of 8 to 11 times the mg N/L ammonia present. For a community with a water source such as the community chose in this study, this would be a very high chlorine dose. The formation of monochloramine involves the addition of chlorine to concentrations where ammonia is not removed but rather bound to chlorine. Other approaches including ion exchange with zeolites, reverse osmosis (RO), advanced oxidation, and air stripping, are capable of removing ammonia from water, but are relatively complex, expensive, or have limited applications.

Although often performed unintentionally, biological ammonia "removal"[1] is another treatment approach to reduce source water ammonia. The process relies on bacteria to convert ammonia to nitrate. As a result, a more biologically-stabile water is produced, nitrification in the distribution system is not an issue, and free chlorine residual is easily achieved. 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 *Nitrospira*. 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:

$$NH_4^+ + 1.5\ O_2 \rightarrow NO_2^- + H_2O + 2H^+ \quad (1)$$

Nitrospira subsequently oxidizes nitrite to nitrate, as follows:

$$NO_2^- + 0.5\ O_2 \rightarrow NO_3^- \quad (2)$$

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

$$NH_4^+ + 2\ O_2 \rightarrow NO_3^- + 2\ H^+ + H_2O \quad (3)$$

It should be noted that these equations are net reactions involving a complex series of enzyme-catalyzed intermediate steps. Nitrification produces free protons, H^+ which readily consume available bicarbonate ions (HCO_3^-), thereby reducing the buffering capacity of the water. In addition, nitrifying bacteria consume CO_2 to build new cells. The total consumption of alkalinity by nitrification is 7.1 mg as $CaCO_3$ per mg NH_4^+- N oxidized (US EPA, 1975). The oxygen demand of nitrification is also significant. For complete nitrification, 4.6 mg O_2 is required per mg NH_4^+- N oxidized (US EPA, 1975; US EPA 1993).

Other factors that affect nitrification include phosphate concentration, pH, and water temperature. All organisms including nitrifying bacteria require phosphorus to build cell mass, with approximately 3% of

[1] The terms "removal" and "oxidation" will be used interchangeably throughout this document. We use "removal" to represent the *conversion* of ammonia to nitrate and/or nitrite by biological oxidation. We recognize that treatment does not *physically remove* ammonia-nitrogen but rather converts the form of nitrogen (i.e., total of ammonia, nitrite, and nitrate).

dry weight consisting of phosphorus. Microorganisms use phosphate as the source of phosphorus for the synthesis of structural and physiological biomolecules such as deoxyribonucleic acid (DNA), phospholipids (membranes), teichoic acid (cell walls), and most importantly, as inorganic phosphorus in adenosine triphosphate (ATP) synthesis. Without ATP, the cellular metabolism (i.e. nitrification) cannot proceed and the cells either become dormant or die. Some organisms are more sensitive to phosphate starvation than others, and in the case of nitrification, ammonia oxidizing bacteria are less sensitive than nitrite oxidizing bacteria (de Vet *et al.*, 2012; Scherrenberg *et al*., 2011; Scherrenberg *et al*., 2012).

Numerous laboratory studies have cited the optimum pH for complete nitrification is between 7.4 and 8.0; although in practice, the bulk water pH may deviate from this value while nitrification remains high (Shammas, 1986). Temperature can impact growth rate and metabolism by slowing or destroying necessary enzymes and proteins involved in physiological processes. Laboratory studies have demonstrated that the growth rate of nitrifying bacteria to be negatively impacted by temperatures below 10°C, although adjustments to the treatment process can be made to enhance nitrification in colder climates (Andersson, *et al.*, 2001).

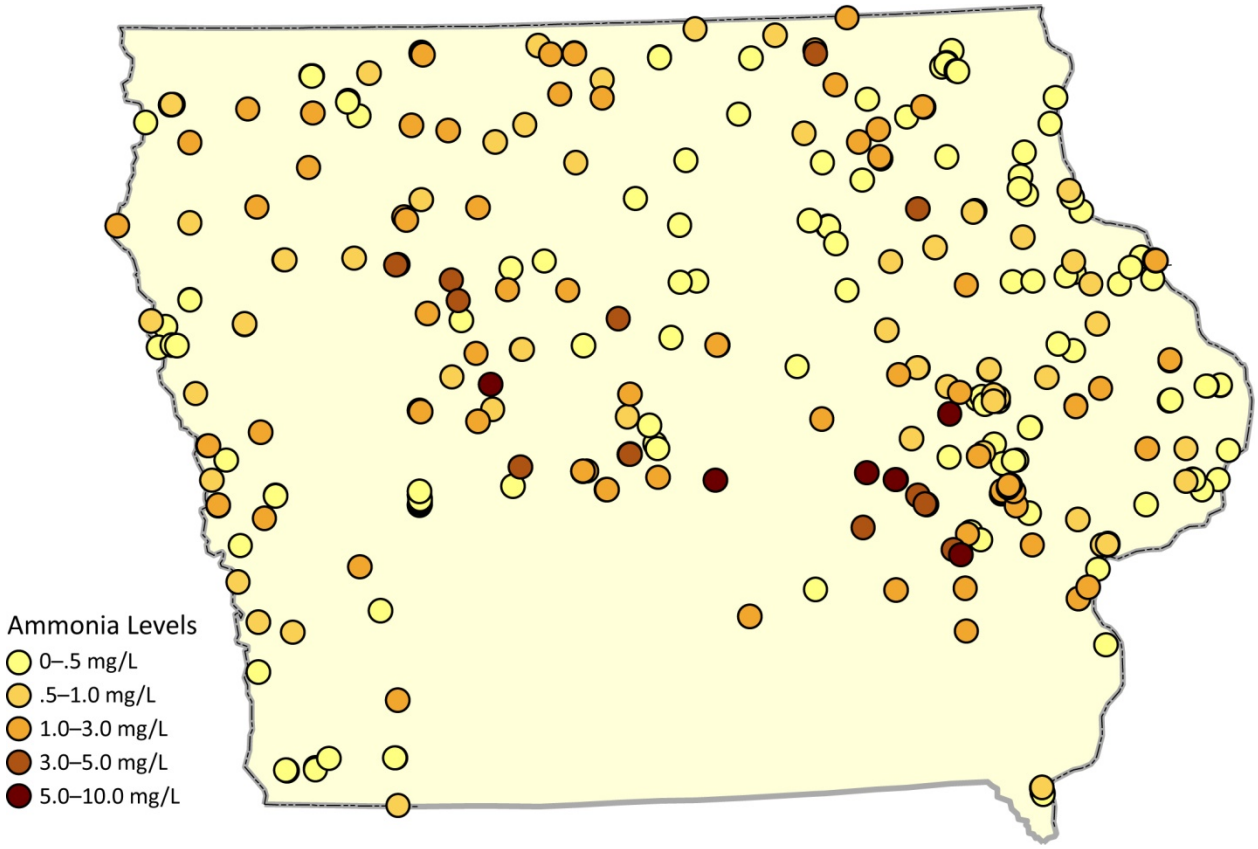


Figure 1. Map of ammonia levels in Iowa based on groundwater well analyses (1998–2012) provided by the State of Iowa.

Table 1. Source Water Quality

Alkalinity	358 mg $CaCO_3$/L
Fe	0.82 mg/L
Mn	0.01 mg/L
P	0.07 mg PO_4/L
TOC	1.06 mg/L
Zn	0.26 mg/L
S	33 mg/L
Cl^-	<5 mg/L
Mg	33 mg/L
NH_4	3.3 mg-N/L
NO_2	0.04 mg-N/L
NO_3	0.02 mg-N/L
pH	7.4
Temp ˚C	15.8

2. Biological Water Treatment Technology Pilot Study

2.1 Collaboration

The small community study site in Iowa does not have a centralized water treatment or a drinking water distribution system. Following extensive flooding to the region in 2008, support to build the necessary infrastructure to supply the community with potable drinking water was put into place. The treatment system needed to be designed to address elevated levels of both iron and ammonia in the source water. The State of Iowa's DNR requested assistance from EPA's Office of Research and Development (ORD) to develop an appropriate treatment system to address the source water concerns. Specifically, ORD's experience in applying biological water treatment to remove ammonia from water was requested. As a result, the State of Iowa DNR, and EPA ORD and Region 7, conducted a pilot study to evaluate the impact of biological water treatment on ammonia oxidation.

Specifically, the pilot is based on an EPA-patented approach (Figure 2) to address elevated levels of ammonia as well as iron in the source water (Patent No. US 8, 029,674). The treatment system relies on bacteria for the conversion of ammonia to nitrate; provided the raw ammonia levels are lower than the nitrate MCL of 10 mg N/L, the approach can be effective and relatively simple. The pilot system was designed and built by EPA staff, and installed in March 2011 (Figure 4). In a collaborative effort, EPA and pilot site staff coordinated system operation and maintenance, as well as water sample collection and analysis.

2.2 Research Approach

Nitrification is a two-step, microbiological process that requires oxygen (aerobic) to oxidize NH_4 to NO_2, and then to NO_3. The entire process requires approximately 4.5 mg of O_2/mg of NH_4-N in the source water. Because the groundwater in the study community has low oxygen (3.6 mg O_2/L) and elevated ammonia of 3.3 mg N/L as well as reduced iron of 0.82 mg/L (Table 1) that also exerts an oxygen demand, more than 13.5 mg O_2/L would be necessary to address the demand due to the ammonia (and iron). Aeration is a necessary feature of the biological ammonia treatment system; however, the traditional configuration of aeration followed by filtration (e.g., iron removal) including biologically-active filtration is not sufficient to address the oxygen demand to meet the treatment objectives of the community's water system.

The amount of oxygen that can be added to the water is controlled by the saturation limit of oxygen in water, which in most drinking waters including the study community's, is well below the total oxygen requirements of treatment. The EPA's experience with microbiological systems that do not provide sufficient oxygen to a nitrifying system has shown that the result is incomplete nitrification or the production of elevated nitrite levels in the finished water. Given the drinking water standard for nitrite is only 1 mg N/L, concerns for potential exceedances exist where source water ammonia levels are greater than 1 mg N/L. Therefore, an innovative approach to introducing oxygen to the treatment system in the small community was necessary to meet the treatment objectives. Aerating with pure oxygen could provide super saturated oxygen conditions and sufficient oxygen, however there are safety issues associated with flammable gases and filter binding associated with gas bubbles can also be an issue.

2.3 Pilot Technology Description

The ammonia biological removal treatment pilot system evaluated is based on an EPA patented design (US 8,029,674 B2 awarded on 10/2/2011) seen in Figure 2. The pilot consisted of three pairs of 3-inch (7.62 cm) diameter columns in series built from clear PVC and other common plumbing materials (Figure 2). Each pair consisted of one column or "contactor" filled with 30 inches (76.2 cm) of gravel (Figure 3) in series with a second column or "filter" filled with anthracite (20 inches [50.8 cm] deep) over sand (10 inches [25.4 cm] deep); each contactor contained a different size gravel : ¼ inch (6.35 mm), ½ inch (12.7 mm), and 1 inch (25.4 mm) (Figure 2). The contactors were aerated from the bottom, such that air bubbles flow upward countercurrent to the water flow (downflow) using a diffuser connected to a gas pump at a rate of 2.5 L/min (0.66 gpm).

In this configuration, the water in the contactor was always saturated with respect to dissolved oxygen throughout the gravel media bed despite the demand from nitrification process and iron oxidation. The gravel in the contactor was solely to serve as a growth support for nitrifying bacteria where nitrification occurs. Gravel allowed bacteria attachment and growth yet eliminated the potential for "clogging" of the media and regular backwashing, and allowed air bubbles to move through the contactor. Oxidation of ferrous iron in the source water also occurs in the contactor, but no iron removal should occur. Various flowrates were considered during pilot evaluations. The filter was intended to remove iron particles and potentially bacteria, and can also provide biological oxidation of excess ammonia and/or nitrite that exit the contactor as a result of incomplete nitrification. With regards to the latter, the filter serves as a polishing step and safeguard against disruption in operation of the contactor which could result, for example, in excess nitrite formation. Effluent water from the filter is routed to a clear well, that when full, can be used to backwash the filters, or overflow to the sanitary sewer.

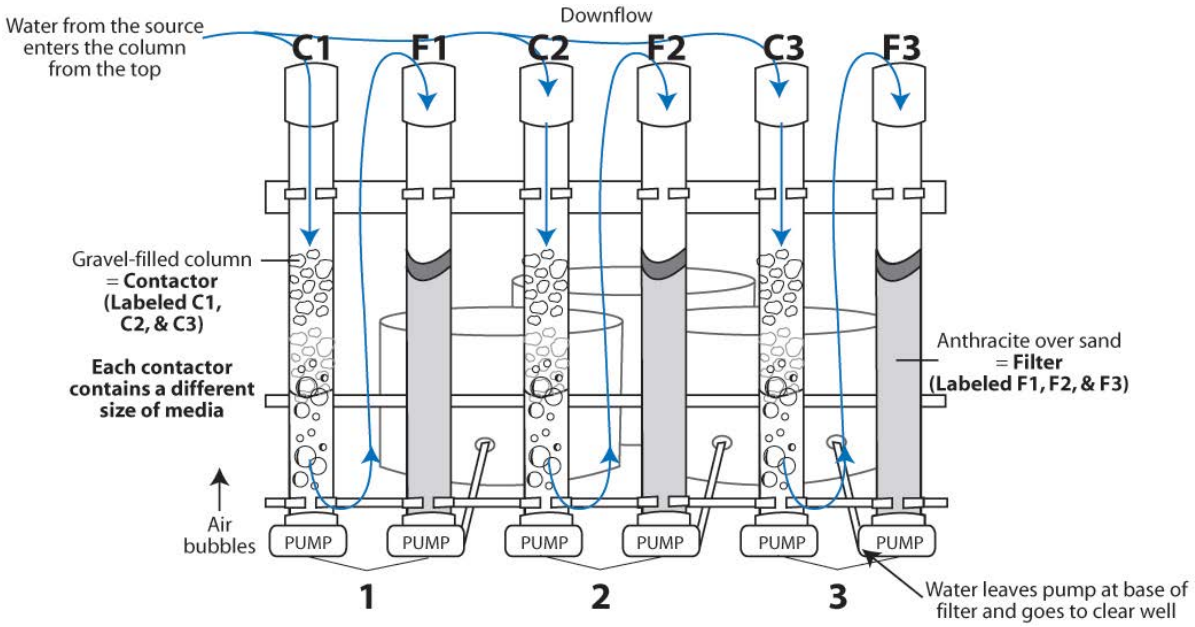


Figure 2. Schematic of the pilot biological ammonia removal treatment technology system.

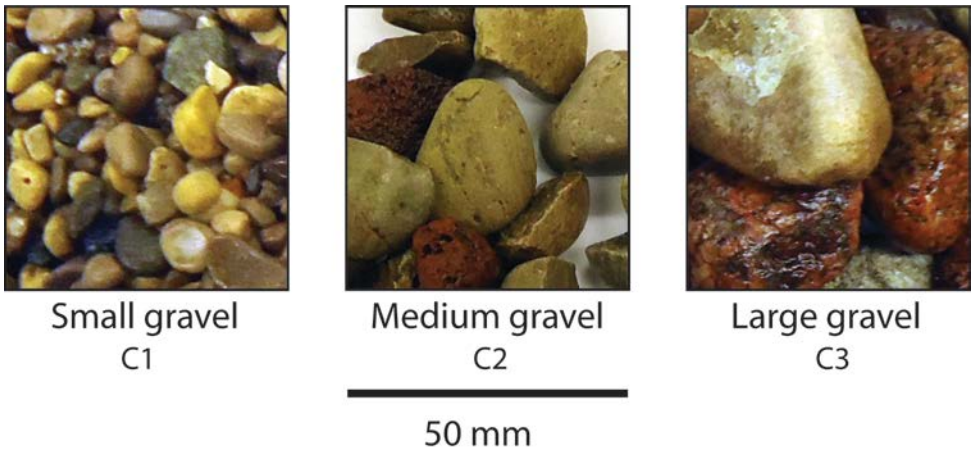


Figure 3. Three gravels used in the contactors of the piloted biological ammonia removal treatment technology system

Figure 4. Pilot biological water treatment system for ammonia removal at the Iowa study site.

3. Operations, Materials, and Methods

3.1 Pilot System Operation

The pilot system (Figure 4) was operated on a continuous basis (24 hours per day, 7 days per week) with the exception of a few instances where pumps were replaced or other maintenance actions occurred (<24 hours downtime or the system was intentionally shut down to evaluate the impact of doing so) for over extended periods of time. Raw water from the small community's existing well and drinking water was not chlorinated or treated in any way prior to supplying the pilot system. Treated water and excess filter backwash water was routed to the on-site sanitary sewer.

Field operating and water quality measurements were collected by the city and included headloss, flowrates, temperature, dissolved oxygen, and pH. Dissolved oxygen, pH, and temperature were measured using an HQ40d meter with an LD101 dissolved oxygen probe and PHC281 pH probe (Hach Company, Loveland, CO). Filters were backwashed using filter effluent water on a weekly basis. Backwashing was achieved by expanding the bed by 50% for 15 minutes. Contactors were backwashed on a case by case basis using raw water. Contactor gravel did not expand during backwashing. A total volume of 12.5 gallons (47.3 L) was used to backwash the contactor for approximately 5 minutes at rate of 2.5 gallon/min (gpm) (9.45 L/min).

A number of parameters were varied and modifications to the pilot system operation were made to optimize nitrification; these included changes to loading rate, media surface area, and a chemical feed addition. Changes to pilot system operation, water quality, and other notable condition changes are summarized in Table 2. Filter loading rate changes were made by adjusting the flowrate through the pilot columns by adjusting the pump speed. For example, contactors began the study with a loading rate of 4.0 gpm/ft^2 (9.76 m/hr) and ended the study at 2.0 gpm/ft^2 (4.88 m/hr). Filters averaged 1.5 gpm/ ft^2 (3.66 m/hr) over the duration of the study.

Due to limitations inherent in a pilot-scale system, the contactor oxygen diffusers required field maintenance. The maintenance interval was determined based on field measurements of effluent dissolved oxygen.

At 190 days into the study, a phosphate chemical feed was added only to Contactor/Filter 1. The target orthophosphate concentration was 0.4 mg PO_4/L. Orthophosphate was provided by the EPA in the form of technical grade $Na_3PO_4 \cdot 12H_2O$ (Fisher Scientific) suspended in deionized water. This solution was added to 20 L of raw water in a carboy and injected into Contactor 1 at 2 mL/min via a peristaltic pump

To assess the effect of doubling the contactor depth, Contactor 1 effluent was routed to Contactor 2 and served as the sole influent at 360 days into the pilot. The effluent of Contactor 2 was routed to filter 1 for polishing. By routing Contactor 1 (30 inches [76.2 cm] of media) to Contactor 2 (30 inches [76.2 cm] of media), effectively doubled the contactor bed depth to 60 inches (152.4 cm). Contactor 3 was not providing new data and was shut down on January 20, 2012.

Table 2. Timeline of Operational Changes for Contactor 1 and Filter 1

Date	ET, days	Description of Change
3/28/2011	24	Backwash-Contactor 1
4/26/2011	53	Backwash-Contactor 1
5/6/2011	63	Cl_2 backflow event
8/22/2011	171	Flows recorded at top of column
9/9/2011	190	PO_4 feed (6 g)-Contactor 1/Filter 1
9/21/2011	201	Contactor 1 pump failed (<24 hours)
9/29/2011	209	New operator
11/8/2011	249	Flow change-Contactor 1/Filter 1
12/20/2011	292	Backwash-Contactor 1
1/17/2012	319	Aerator blowout-Contactor 1
2/21/2012	354	Aerator blowout-Contactor 1
2/22/2012	355	Backwash-Contactor 1
2/27/2012	360	Bed depth increase-Contactor 1 & 2
3/6/2012	376	Flow change-Contactor 1/Filter 1
4/24/2012	417	PO_4 feed (3g)

3.2 Water Quality Analysis

Community staff collected weekly water quality samples, while making routine measurements and shipped them on ice overnight to the US EPA Office of Research and Development (ORD) in Cincinnati for analysis. Water samples were collected from the raw water and effluent of all contactors and filters. The following water samples were collected on a weekly basis:

- 250 mL for inorganic analysis
- 60 mL for metals analysis
- 250 mL for bacteriological analysis
- 40 mL for organic carbon analysis

Upon arriving to EPA, the samples along with the chain of custody, were removed from the cooler, preserved accordingly, and submitted for analysis. Ammonia, nitrite, and nitrate analysis were typically performed on the same day the cooler arrived (approximately 24 hours after sampling). All water analyses were performed according to EPA or Standard Methods (Table 3).

Table 3. Water Quality Analyses Performed and Methods

Analysis	Method	Method #	Reference
Total Alkalinity	Potentiometric Titration	2320 B.4.6	Std. Methods[1]
Ammonia (as N)	Automated Colorimetric	350.1	EPA Methods[2]
Chloride	Potentiometric Titration	4500-Cl D	Std. Methods[1]
Nitrate & Nitrite (as N)	Automated Colorimetric	353.2	EPA Methods[2]
Orthophosphate	Automated Colorimetric	365.1	EPA Methods[2]
As, Pb, U, Se, Bi	ICP-MS	200.8	EPA Methods[3]
Al, As, Ba, Be, Bi, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, S, Sb, Sulfate, Si, Silica, Sn, Zn	ICP-AES	200.7	EPA Methods[3]
TOC	Combustion	5310 C	Std. Methods[1]
Temperature	Thermocouple	17.1	EPA Methods[1]
Total Coliforms*	Culture	9223B	Std. Methods[1]
E. coli.*	Culture	9223B	Std. Methods[1]
HPC	Culture	9215C	Std. Methods[1]

*Indicates random sampling

[1] Standard Methods for the Examination of Water and Wastewater," 18th Edition (1992).
[2] USEPA, "Methods for Chemical Analysis of Water and Wastes," EPA-600/14-79-020 (1983).
[3] USEPA, "Methods for the Determination of Metals in Environmental Samples," EPA-600/14-91-010 (1994).

4. Results of the Pilot Study

4.1 Important Dates

There are a number of operating changes and other events that occurred over the course of the pilot study that are worth noting because they had a direct impact on the results and proceeding discussions. Events including changes in contactor/filter flowrates (loading rates), the addition of phosphate feed, backwash events, aerator clean-outs, and operator changes have been documented (listed in Table 2) and will be referred to when appropriate.

4.2 General Water Chemistry

Extensive water quality analysis of the site's source water, as well as the pilot contactor and filter effluents over the entire pilot study, is summarized in Table 4. The source water was a relatively hard, high alkalinity groundwater with calcium and magnesium levels averaging 79 and 33 mg/L, respectively, or a total hardness of 332 mg $CaCO_3$/L, a total alkalinity of 357 mg $CaCO_3$/L and a pH of 7.1. Iron levels averaged 0.63 mg/L although the concentration varied as indicated by the relatively large standard deviation, and ammonia averaged 3.2 mg N/L. Sulfate, chloride, and silica averaged 94 mg SO_4/L, 5 mg/L and 7.1 mg SiO_2 /L, respectively. Orthophosphate was very low, averaging 0.032 mg PO_4/L, and total phosphorus was at the detection limit of 0.005 mg P/L. Manganese, nitrite, and nitrate were at or near the respective method detection limits; strontium averaged 1.1 mg/L; and TOC averaged 1.3 mg C/L.

4.3 Removal of Ammonia in Source Water

Contactor 1/Filter 1. Contactor 1 was operated for approximately 55 days before nitrite levels started to increase suggesting the initiation of the biological nitrification process within the contactor (Figure 5). The increase was brief, however, and nitrite only reached 0.2 mg N/L before the level suddenly dropped back to non-detectable (<0.01 mg N/L) (Figure 5). It was discovered that at 63 days, the pilot system inadvertently received chlorinated water as a result of inadequate or failed backflow prevention measures ahead of the pilot system, the extent and time-frame to which was uncertain. The presence of chlorine presumably halted biological activity in the contactor and , as a result, was attributed to the decrease of nitrite generation. Proper backflow prevention was installed immediately after the event.

Following plumbing modifications to address backflow issues, nitrite levels began to increase again by 95 days (Figure 5). Nitrite steadily increased to a peak of 0.4 mg N/L by 140 days. A similar decrease in ammonia through the contactor over the same time period was observed. No nitrate was produced in the contactor during this time.

Table 4. Water Quality Summary (Average ± Standard Deviation (n))

Analyte	Detection Limit (mg/L)	Raw	Contactor 1	Filter 1	Contactor 2	Filter 2	Contactor 3	Filter 3
Ba	0.001	0.028±0.001(45)	0.029±0.002(44)	0.028±0.002(45)	0.03±0.004(44)	0.028±0.002(43)	0.034±0.02(42)	0.028±0.003(39)
Ca	0.01	78.71±2.96(45)	78.6±3.09(44)	78.08±2.83(45)	79.09±3.31(44)	78.06±3.8(43)	79.03±3.99(42)	78.88±2.96(45)
Cl	5	5±1(52)	5±1(52)	5±1(51)	5±1(51)	5±1(45)	5±1(42)	5±1(52)
Fe	0.001	0.63±0.489(45)	1.523±1.916(44)	0.019±0.016(45)	3.267±4.78(44)	0.054±0.17(43)	8.681±24.39(42)	0.031±0.489(45)
K	0.3	5.2±0.1(45)	5.1±0.2(44)	5.1±0.2(45)	5.2±0.2(44)	5.1±0.3(43)	5.2±0.8(42)	5.2±0.1(45)
Mg	0.005	33.01±1.247(45)	32.987±1.306(44)	32.948±1.274(45)	33.234±1.333(44)	33.04±1.232(43)	33.152±1.731(42)	33.135±1.247(45)
Mn	0.001	0.008±0.004(45)	0.009±0.004(44)	0.006±0.004(45)	0.01±0.007(44)	0.007±0.008(43)	0.009±0.006(42)	0.007±0.004(45)
Na	0.03	32.54±1.78(45)	32.51±1.46(44)	32.5±1.42(45)	32.62±1.58(44)	32.35±1.04(43)	32.63±2.2(42)	32.65±1.78(45)
NH_3	0.03 (mg-N/L)	3.24±0.35(52)	1.63±1.26(52)	1.14±1.31(52)	1.91±1.05(51)	1.45±1.28(45)	2.08±1.01(42)	1.48±0.35(52)
NO_2	0.01 (mg-N/L)	0.05±0.04(53)	0.52±0.68(53)	0.39±0.76(53)	0.94±0.83(52)	1.73±2.16(45)	1.23±1.06(42)	1.59±0.04(53)
NO_3	0.02 (mg-N/L)	0.03±0.02(53)	1.19±1.17(53)	1.79±1.54(53)	0.53±1.07(52)	0.53±0.73(45)	0.1±0.31(42)	0.34±0.02(53)
o-PO_4	0.025 (mg PO_4/L)	0.032±0.015(56)	0.026±0.004(23)* 0.082±0.086(33)**	0.027±0.006(23)* 0.053±0.036(33)**	0.037±0.031(55)	0.031±0.017(45)	0.03±0.012(42)	0.028±0.015(56)
P	0.005 (mg P/L)	0.013±0.01(45)	DL(24) 0.117±0.019(20)*	DL(24) 0.021±0.008(21)*	0.032±0.109(44)	0.016±0.02(43)	0.038±0.147(42)	0.011±0.009(39)
SiO_2	0.02 (mg SiO_2)	7.07±1.11(46)	7.32±0.37(44)	7.18±0.32(45)	7.49±0.56(44)	7.2±0.29(43)	7.76±1.47(43)	7.18±0.37(39)
Sr	0.001	1.114±0.053(45)	1.114±0.055(44)	1.109±0.053(45)	1.12±0.054(44)	1.112±0.055(43)	1.128±0.059(42)	1.12±0.053(45)
SO_4	0.003 (mg SO_4/L)	94.07±14.52(46)	95.86±2.98(44)	95.64±2.89(45)	96.44±3.1(44)	96.07±3(43)	96.45±4.17(43)	96.73±3.31(39)
Total Alkalinity	1 (mg-$CaCO_3$/L)	357±2(46)	344±10(47)	342±9(46)	343±10(46)	343±13(41)	345±10(38)	345±2(46)
Total Nitrogen	0.01 (mg-N/L)	2.86±0.49(36)	2.94±0.46(35)	3.1±0.45(37)	2.92±0.46(36)	3.09±0.48(36)	2.95±0.43(36)	3.05±0.49(36)
TOC	0.1 (mg-C/L)	1.3±0.7(42)	1.3±0.4(40)	1.5±1.1(42)	1.3±0.6(41)	1.3±0.7(41)	1.3±0.6(42)	1.2±0.7(42)
Zn	0.0005	0.2193±0.2077(45)	0.4045±0.3976(44)	0.1564±0.1803(45)	0.5739±0.607(44)	0.1654±0.1966(43)	0.7336±1.1754(42)	0.1187±0.2077(45)
pH	0.1	7.1±0.32(52)	7.36±0.42(51)	7.52±0.26(52)	7.46±0.19(51)	7.54±0.17(43)	--	--
DO	0.01 (mg-O_2/L)	3.64±1.08(49)	8.25±2.13(47)	8.02±1.93(47)	9.23±1.31(46)	8.3±1.11(42)	--	--
HPC	1 (CFU/mL)	10,631(37)	54,763(36)	39,914(37)	50,380(36)	36,436(26)	62,204(26)	31,360(26)
Temperature	0.1˚C	14.8±2.5(42)	14.8±2.2(41)	15.4±2.8(42)	14.7±2.3(40)	15.7±3.1(35)	15.1±2.7(32)	16.3±3.4(29)

*Before phosphate feed
**After phosphate feed

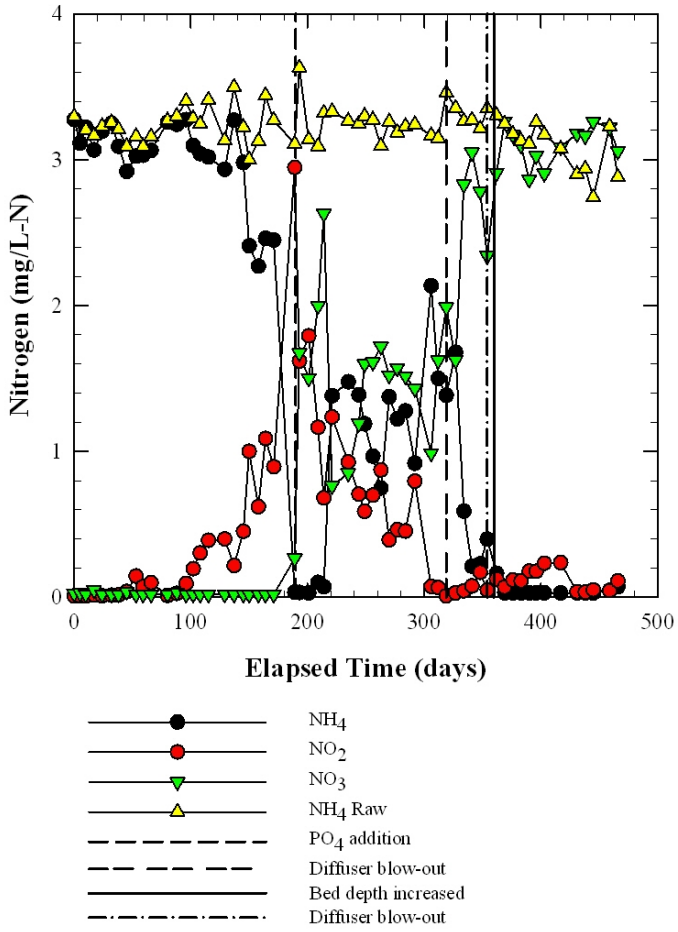


Figure 5. Nitrogen content of treated water from Contactor 1.

The progression of bacterial acclimation and nitrification within the contactor was incomplete, and unexpectedly and unacceptably slow. Considering variables that could impact the nitrification process, the relatively high initial filter loading rate (flowrate) of 4.3 gpm/ft^2 (10.5 m/hr) (Figure 6) through the contactor was thought to potentially be related. The loading rate was incrementally decreased to 3 gpm/ft^2 (7.3 m/hr) then 1 gpm/ft^2 (2.4 m/hr) (Figure 6) between 140 and 160 days. The decrease in loading rate resulted in an immediate increase in nitrite production and an equivalent ammonia decrease to nearly 1 mg N/L. Still, no nitrate was produced in the contactor during this time. Although some improvement was observed (i.e., more ammonia was oxidized), the progress was still very slow, nitrite levels leaving the contactor approached the 1 mg N/L MCL and no signs of further oxidation to nitrate were observed.

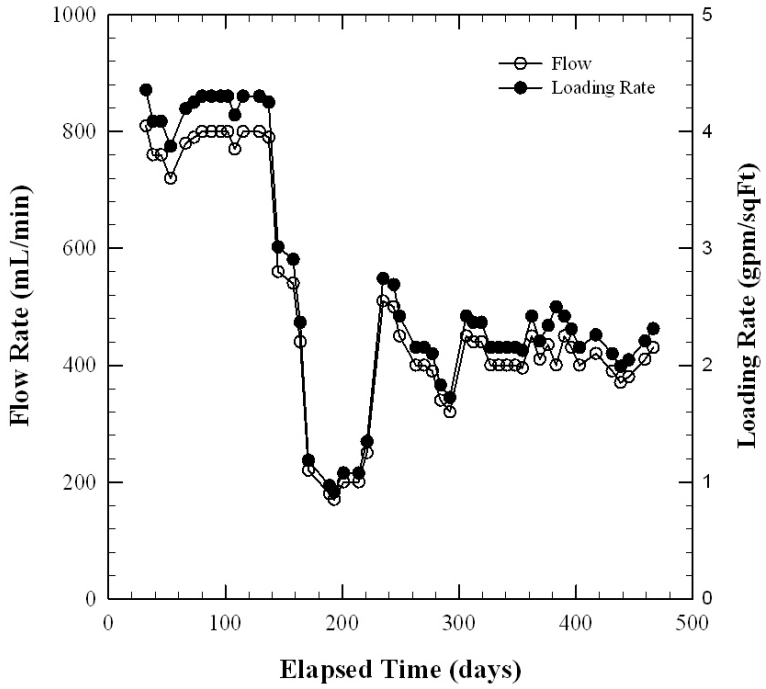


Figure 6. Contactor 1 flowrate and hydraulic loading rate.

Previous work (Lytle et al., 2007) indicated the complete oxidation of ammonia to nitrate, or complete acclimation of bacteria after start-up of a new biologically active nitrifying filter could take as little as 70 days. In addition, nitrite release generally occurs as a relatively short spike that falls off rather quickly as nitrite oxidizing biofilm establish and nitrate is generated. Iowa's water in this study, however, had nearly 3 times the ammonia concentration as the 2007 work. Other factors that impact biological nitrification such as nutrient requirements were considered.

Phosphorus is an important nutrient and necessary physiological component of bacteria. The source water contained very little "natural" orthophosphate (0.03 mg PO_4/L) (Table 3). Insufficient phosphorus that is necessary for cell physiology was considered as a possible issue, and therefore it was elected to add phosphate ahead of the contactor. At 190 days, orthophosphate was added to Contactor 1's feed water at an arbitrary target dose of 0.4 mg PO_4/L. Measurable orthophosphate levels following chemical feed only increased by 0.05 mg PO_4/L (Table 3). Total phosphorus, however, increased by 0.1 mg P/L (0.3 mg PO_4/L), which indicated that phosphate was likely bound to iron particulates or other solids in the system.

Ammonia in Contactor 1 effluent dropped below 0.05 mg N/L immediately following phosphate addition (Figure 5). Nitrite initially spiked to nearly 3 mg N/L immediately after phosphate addition and then rapidly dropped to 0.6 mg N/L within 30 days. During the same period of time, nitrate increased to as high as 2.5 mg N/L but then decreased to and stabilized at approximately 1.5 mg N/L by 220 days. The rapid progression of nitrite to nitrate was almost immediately halted between 220 and 240 days, at which time ammonia levels increased back to 1.4 mg N/L.

At approximately 240 day, the contactor loading rate was increased to 2.5 gpm/ft^2(6.1 m/hr) which did not appear to impact nitrogen balances. Between 240 and 290 days, still no improvement in ammonia oxidation was observed. During this time, loading rate was gradually dropped to 1.6 gpm/ft^2(3.9 m/hr) but no obvious improvement in ammonia oxidation was noted. Unfortunately, although phosphate stimulated nitrite oxidation and enhanced ammonia oxidation, it was still not complete (Figures 5 and 7). Nitrite levels remained at or near the MCL for nitrite, where the goal was to completely oxidize ammonia to nitrate in the contactor. At day 292, the contactor was backwashed for the first time not because of headloss build-up, but rather as a maintenance step and to remove some of the build-up of iron and biomass on the gravel. After backwashing, nitrite levels dropped to near detection limit and nitrate levels increased by a similar amount, but ammonia levels remained largely unchanged.

Oxygen is also a key parameter in the nitrification process, whereby 4.6 mg O_2/L is necessary to oxidize 1 mg N/L ammonia to nitrate. Further, there is also a connection between oxygen levels and kinetic requirements associated with molecular diffusion. Close examination of oxygen levels in the contactor effluent showed that the increase in ammonia at approximately 220 days was directly linked to a sudden decrease in oxygen in the contactor from approximately 8 mg O_2/L down to 5.5 mg O_2/L, presumably because of the onset of improved nitrification (Figure 8).

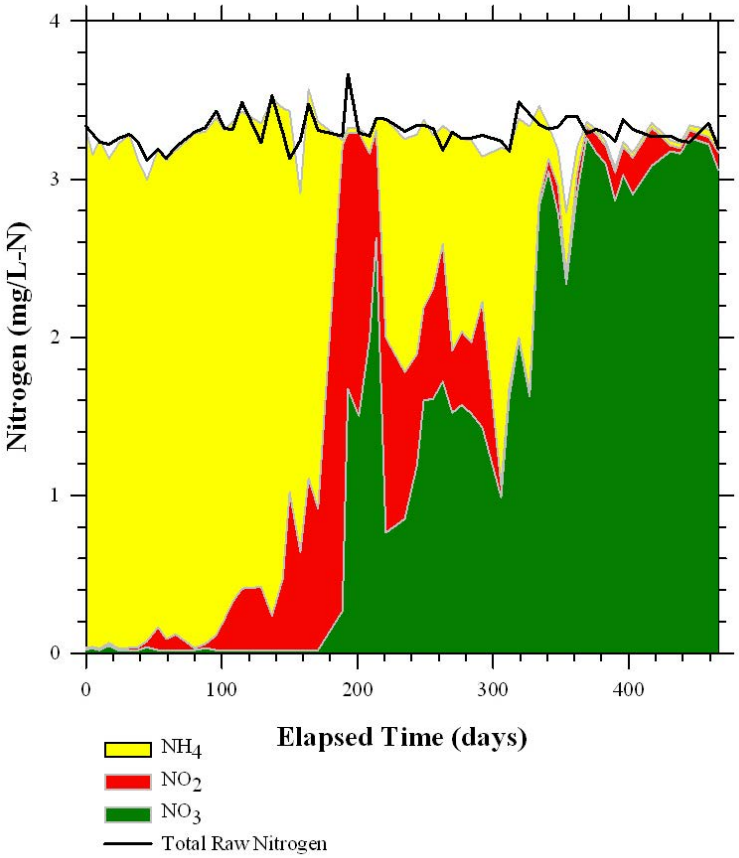


Figure 7. Contactor 1 nitrogen mass balance between influent and effluent.

Once the oxygen drop issue was recognized, the system operator "blew-out" the air diffuser to remove any biofilm and/or deposit that could have blocked the air diffuser. Immediately after cleaning the diffuser, ammonia, and nitrite decreased and oxygen and nitrate increased (Figure 5 and 7). Dissolved oxygen levels dropped slowly again shortly after normal operation resumed (Figure 8). A more aggressive aerator blow-out was performed and further improvements were immediately realized. By the second blow-out (~350 days), ammonia levels remained low or non-detectable, and the system operated ideally. Providing adequate oxygen (to near saturated oxygen levels) through the contactor, as well as orthophosphate, were enough for the contactor alone to achieve the desired ammonia reduction at a typical filter loading rate to many iron removal plants.

The phosphate feed was shut-off twice toward the end of the study (>350 days) after optimized ammonia oxidation was realized to simulate chemical feed failure events. During the first event, the feed was only turned off for 24 hours prior to sampling. No degredation of ammonia oxidation was noted nor was nitrite generated. Later, the feed was discontinued for three straight weeks, during which time, no degredation of ammonia oxidation nor nitrite formation was noted. The results suggest orthophosphate accumulated in the contactor (likely bound to iron particles) and was still biologically-available.

The primary intent of the dual media filter that followed the contactors was to remove iron particles that developed in the contactors. The filters were also biologically-active and provided protection by oxidizing excess ammonia and nitrite that passed through the contactor. Ammonia, nitrite, and nitrate levels entering Filter 1 were those exiting Contactor 1 (Figure 5 and 7). Ammonia oxidation to nitrite in the filter began at about 95 days (same as contactor) and increased steadily to produce a very concerning 2.8 mg N/L nitrite by 190 days (Figure 9a). As with the contactor, no nitrate was formed. The filter loading rate during this time was between 1 gpm/ft^2 (2.4 m/hr) and 1.5 gpm/ft^2 (3.7 m/hr) up to 160 days (Figure 9b). The loading rate was reduced for a brief period to 0.5 gpm/ft^2 (1.2 m/hr), which had no obvious impact on ammonia oxidation. The addition of phosphate at 190 days caused an immediate decrease in the filter effluent nitrite level to near detection and immediate increase of nitrate to 3 mg N/L (ammonia was near detection limit). Ammonia levels increased occasionally but remained <0.6 mg N/L for the period between 220 days and 319 days, and nitrite levels remained very low during this period (Figure 9c). This time period corresponded to the dissolved oxygen issues in the contactor which appeared to carry over to incomplete oxidation of ammonia in the filter. Dissolved oxygen levels in the filter were near 6 mg/L during this time (Figure 9d). Once the oxygen concentration was re-established following the diffuser blow-out, ammonia levels remained near the detection limit. The filter was operated at a loading rate of 2 gpm/ft^2 (4.9 m/hr) by the end of the study.

Clearly, the filter improved overall water quality by polishing contactor effluent. Most notably, comparison between Figures 7 and 9c indicate the degree to which nitrate formation was enhanced in the filter effluent (more green shared area).

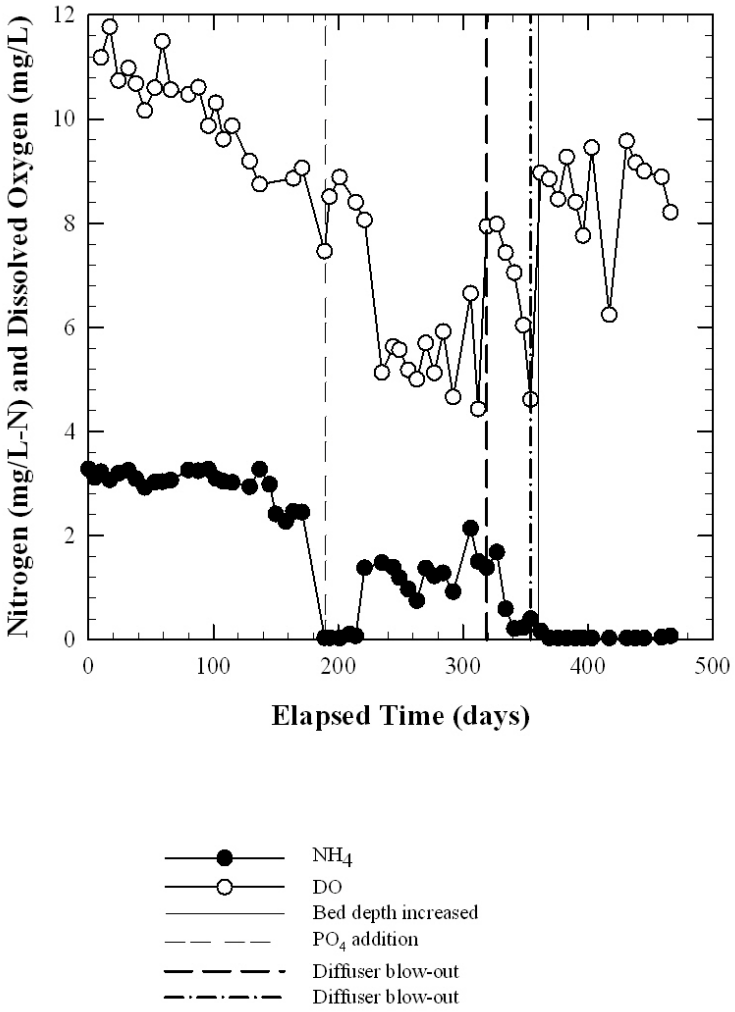


Figure 8. Nitrogen content and DO of finished water from Contactor 1.

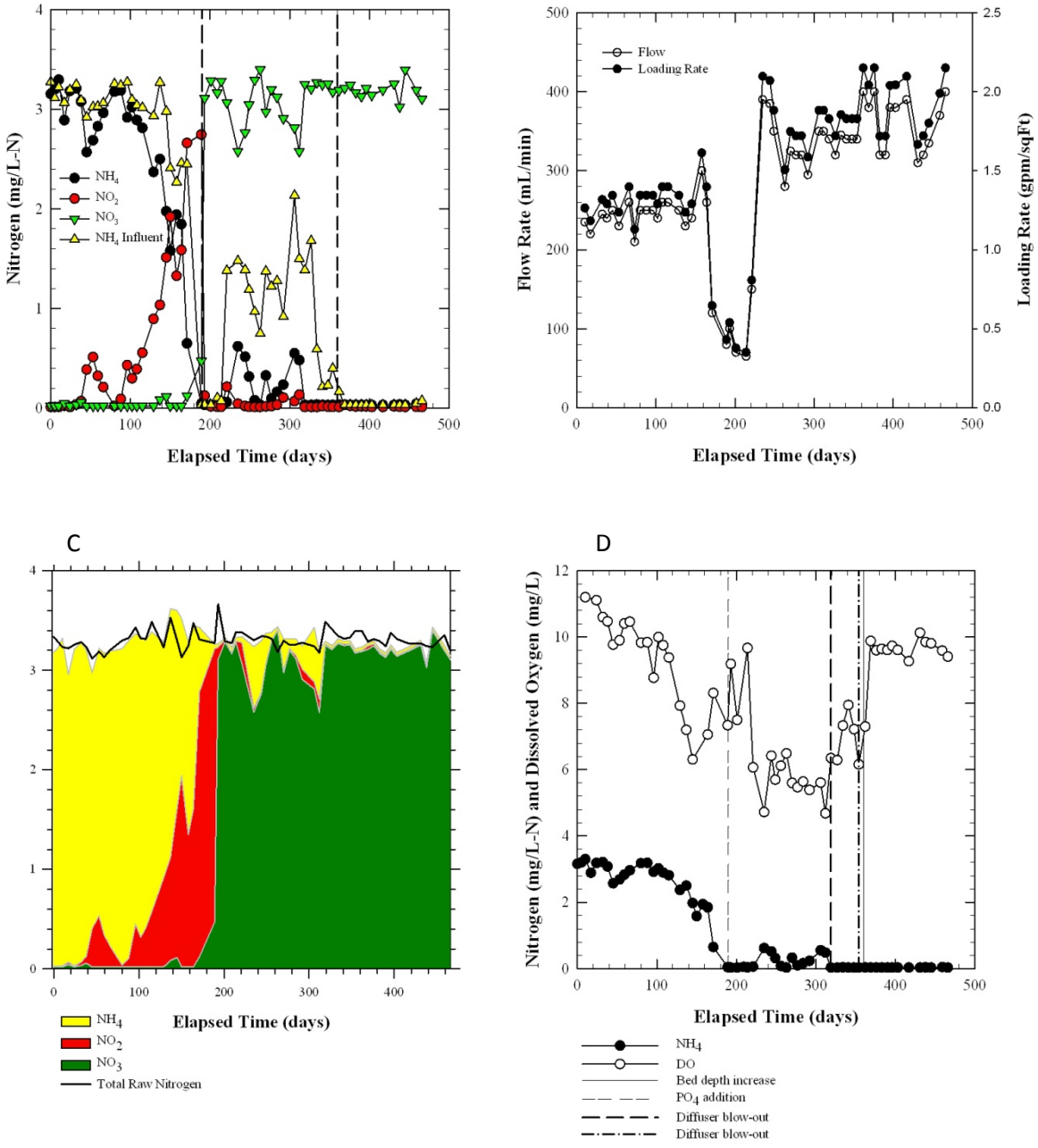


Figure 9. Filter 1 data: A) Nitrogen content of finished water from Filter 1, B) Filter 1 flowrate and loading rate, C) Filter 1 nitrogen balance effluent, and D) Effluent ammonia and dissolved oxygen.

Contactor 2/Filter2. Contactor 2 was operated near identically to Contactor 1 with the exception that orthophosphate was not added at any time and the gravel was approximately ½ inch (12.7 mm), rather than ¼ inch (6.35 mm) in diameter (Figure 1). Again, at approximately 55 days, nitrite levels started to increase only to decrease suddenly as a result of the accidental addition of chlorine to the system (Figure 10). Nitrite levels began to increase again at 95 days (Figure 10). Nitrite steadily increased to and peaked at 0.2 mg N/L at 140 days. A similar decrease in ammonia through the contactor over the same time period was observed. No nitrate was produced in the contactor during this time.

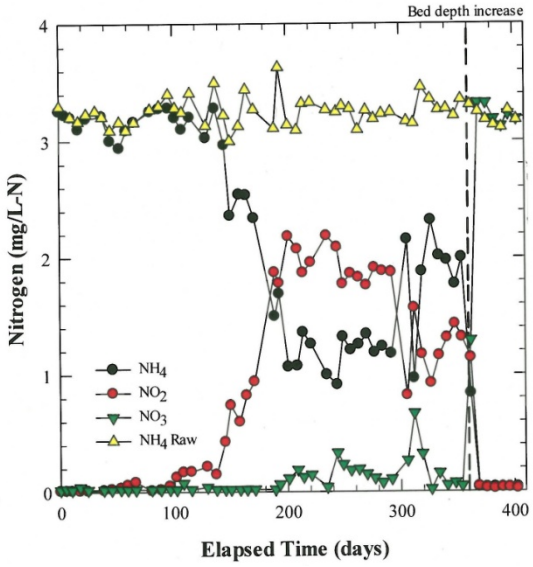


Figure 10. Nitrogen content of finished water from Contactor 2.

Bacterial acclimation and nitrification in Contactor 2 was slow. Considering variables that could impact the process, the relatively high filter loading rate (flowrate) of 4.3 gpm/ft^2 (10.5 m/hr) (Figure 11) through the contactor was thought to potentially be an issue. The loading rate was decreased to 3 gpm/ft^2 (7.3 m/hr) then 1 gpm/ft^2 (2.4 m/hr) (Figure 11) between 140 and 160 days. The decrease in loading rate resulted in an immediate increase in nitrite production and a corresponding decrease in ammonia to nearly 1 mg N/L by 200 days. No nitrate was produced in the contactor during this time. Although some improvement was observed (i.e., ammonia oxidized), the progress was still very slow, nitrite levels in the contactor were well over the 1 mg N/L MCL and reached nearly 2 mg N/L (Figure 10). Between 200 and 280 days, there was a delayed generation of nitrate. The amount of nitrate formed was very small and levels were generally less than 0.3 mg N/L.

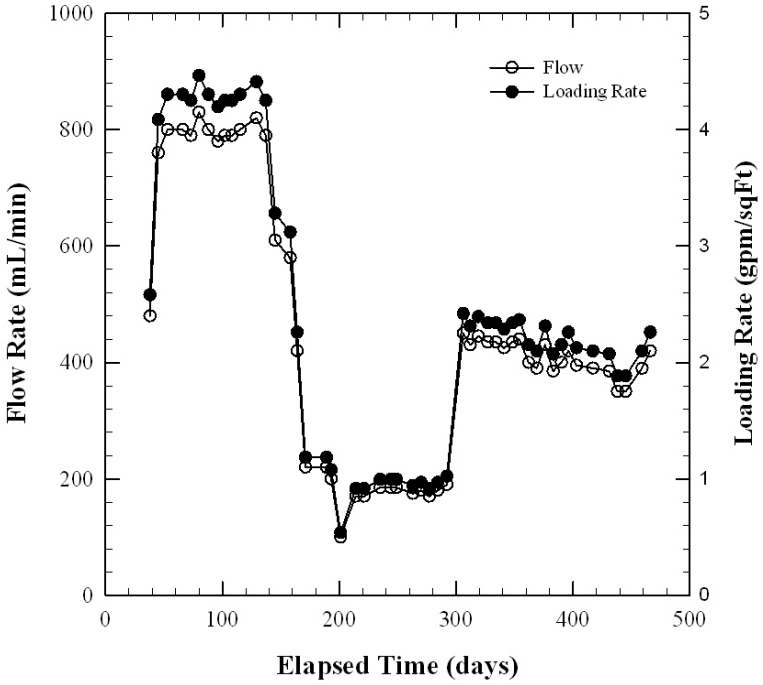


Figure 11. Contactor 2 flow and loading rate.

At approximately 290 days into the pilot study, the contactor loading rate was increased to 2.2 gpm/ft^2 (5.4 m/hr) which did not appear to impact nitrogen balances. Between 300 and 320 days, ammonia levels in the contactor effluent nearly doubled to 2 mg N/L, nitrite levels decreased to 1 mg N/L and nitrate fell back to near the detection limit. The changes in nitrogen distribution corresponded to a steady decrease in oxygen levels in the contactor between 250 and 320 days. During this time, oxygen decreased from approximately 10 mg O_2/L to 7 mg O_2/L (Figure 12). Two backwash events at 320 days and 350 days were necessary to clear the diffuser and bring oxygen levels back to approximately 10 mg O_2/L.

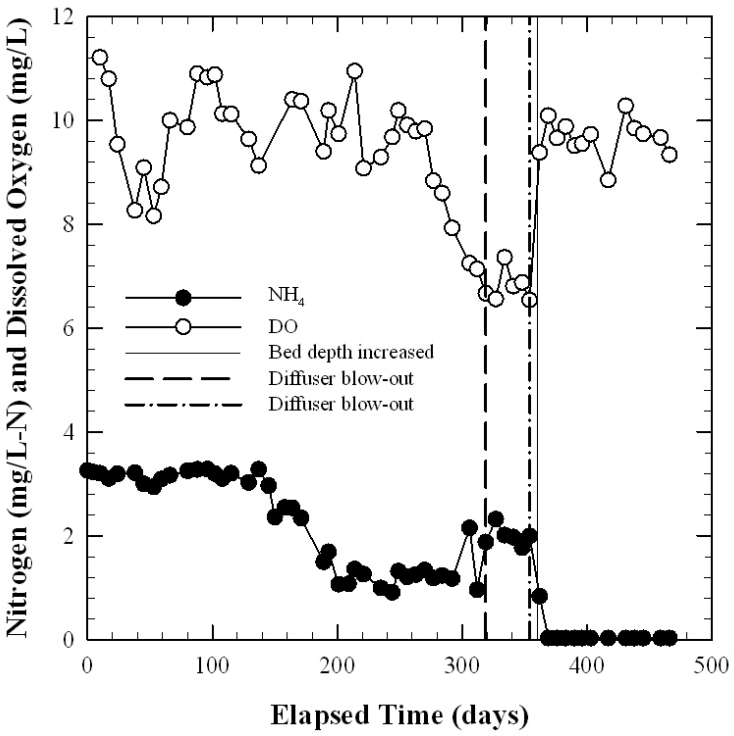
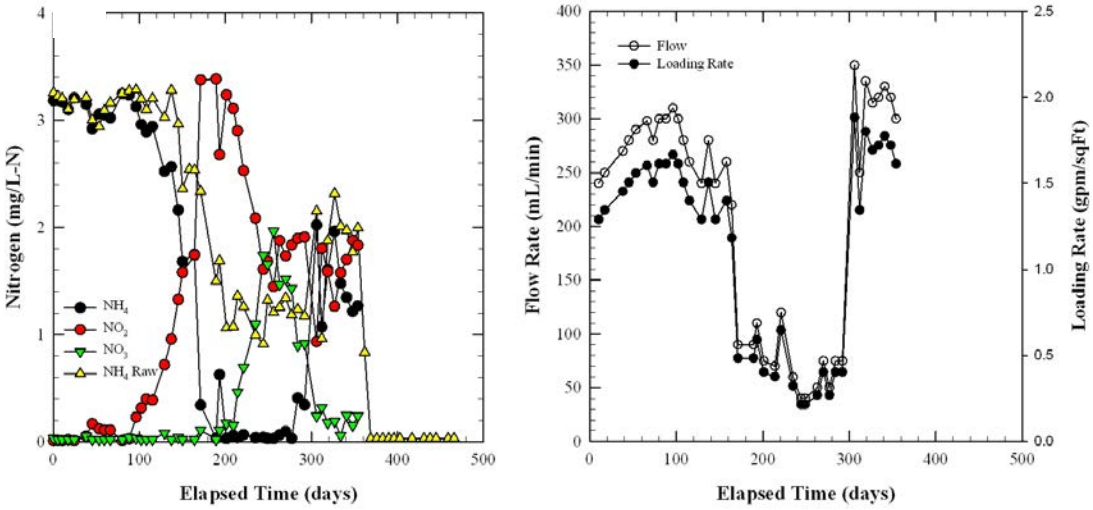


Figure 12. Contactor 2 effluent ammonia and DO.

Operation of Contactor 2 paralleled that of Contactor 1, with the exception being that no orthophosphate was added. Regardless of the changes made to Contactor 2 operation, satisfactory results could not be achieved with respect to complete oxidation of ammonia to nitrate, which clearly confirms the necessity of orthophosphate addition.

At Day 330, Contactor 2 was reconfigured such that its influent was Contactor 1 effluent rather than raw water. At the time of reconfiguration, Contactor 1 was not able to achieve complete oxidation of all of the source water ammonia all the way to nitrate. Reconfiguring the contactors essentially doubled contactor bed depth. Also, shortly after the changeover, all the nitrogen leaving Contactor 2 was in the nitrate form. Shortly after reconfiguration, however, Contactor 1 was able to completely oxidize all of the ammonia to nitrate without the need for additional bed depth.

Ammonia, nitrite, and nitrate levels entering Filter 2 were those exiting Contactor 2 (Figure 13 a). Ammonia oxidation in the filter began at about 95 days (same as contactor) and nitrite increased correspondingly to a very concerning 3.2 mg N/L by 190 days. As with the contactor, no nitrate was formed up to this point. The filter loading rate during this time was between 1.2 gpm/ft^2 (2.9 m/hr) and 1.5 gpm/ft^2 (3.7 m/hr) up to 160 days (Figure 13b). It was reduced for a brief period to 0.5 gpm/ft^2 (1.2 m/hr) which had no obvious impact on ammonia oxidation. Nitrite levels decreased steadily to approximately 1.8 mg N/L by 260 days. During the same time, nitrate increased to 1.8 mg N/L only to decrease back to 0.2 mg N/L by 300 days were it remained for the rest of the pilot (Figure 13a). The changes in nitrate appeared to correspond to changes in ammonia levels entering the filter and loading rate changes.



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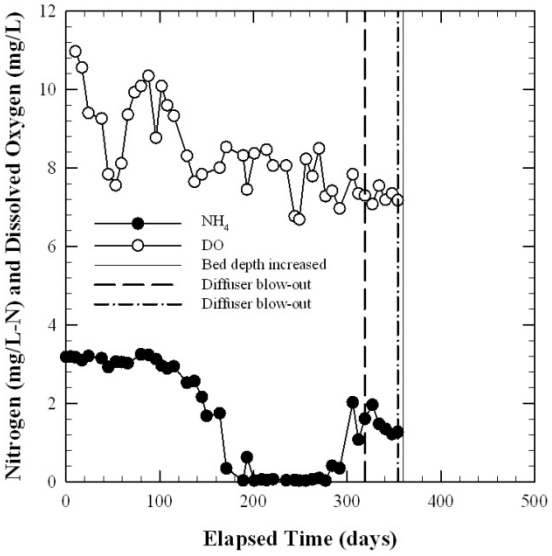

Figure 13. Filter 2 data. A) Nitrogen content of finished water from filter 2, B) Filter 2 flowrate and hydraulic loading rate, and C) Filter 2 ammonia and oxygen.

Contactor 3/Filter3. Contactor 3 was also operated nearly identically to Contactor 1, with the exception that orthophosphate was not added and the gravel was largest at 1 inch (25.4 mm). Given the problems identified earlier, efforts to improve ammonia removal centered on Contactor 1/Filter 1. Also, because of the sample load to the laboratory, Contactor 3/Filter 3 was terminated at around 320 days. Trends in nitrogen species through the contactor and filter were nearly identical to the Contactor 2/Filter 2 system, despite the larger media and will not be discussed here.

4.4 Removal of Iron from Source Water

Although the initial oxidation state of iron was not directly determined, it is reasonable to assume that based on water chemistry, dissolved oxygen, and local geology that iron was initially in the reduced Fe(II) form. The oxygen level in the contactors and the pH of the source water led to rapid oxidation of Fe(II) to Fe(III), and iron particles.

Iron in the source water, averaging 0.63 (±0.49 standard deviation) mg/L , was variable over the course of the study as supported by the relatively high standard deviation (Table 3). It was not clear whether the large range of iron levels was associated with the source variability, sampling issues, or analysis issues associated with particulate-containing water samples. Although initially dissolved, it is also likely that iron was in the particulate form at the time of or shortly after sampling. The presence of particles could also cause variability.

Iron levels in the effluent of Contactors 1 and 2 were typically lower and generally within several tenths of a mg Fe/L of the source water entering the contactors up until approximately 150 days of operation (Figures 14 and 15). Interestingly, at approximately an event 150 days into operation rapidly triggered a dramatic change in effluent iron levels in both contactors. After 150 days, iron in contactor effluents became very sporadic, and some concentration spikes were very high. In some cases, iron levels were as high as nearly 10 mg Fe/L. Orthophosphate addition was not associated with the cause because the change in the iron pattern was observed to occur in both contactors at the same time. During about the same time, loading rates were reduced and ammonia oxidation began to improve in both systems. It was during this time that the biological activity greatly increased and eventually thrived. It is believed that iron particles became incorporated into the biomass and were retained on the contactor. The biofilm and iron sloughed off in an irregular manner which may have contributed to the occurrence of sporadic and elevated iron spikes in the contactor effluents.

Regardless of the iron content in the contactor effluent, iron levels in filter effluent waters were very low in dual media filters, essentially removing all of the iron (Figures 14 and 15). It was assumed that all of the iron entering the filters was in the Fe(III) or particulate form based on the oxygen concentration and pH in the contactor water.

Iron removal through the filters was not impacted by filter loading rates (Figures 14 and 15). Filters were operated between 0.5 gpm/ft^2 (1.2 m/hr) and 2.2 gpm/ft^2 (5.4 m/hr). Filter flowrates had to be lower than contactor flowrate due to limitations in pilot design. At the completion of the study, Filter 1 was operated at a loading rate of approximately 2 gpm/ft^2 (4.9 m/hr).

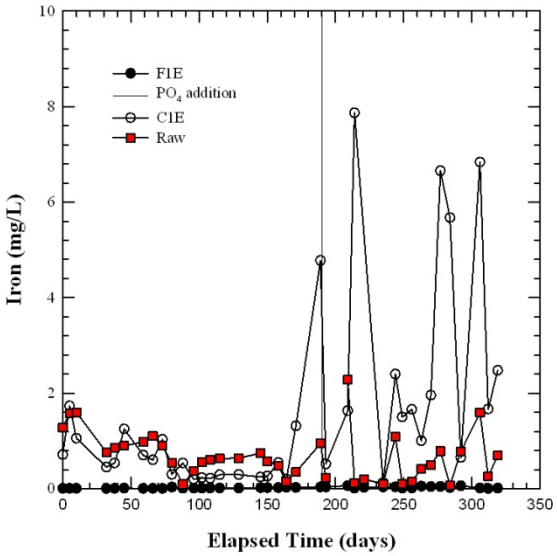

Figure 14. Iron in raw water and treated water through Contactor 1 and Filter 1.

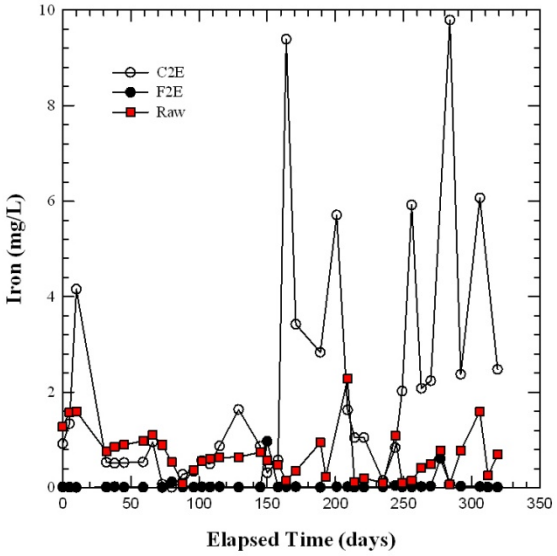

Figure 15. Iron in raw water and treated water through Contactor 2 and Filter 2.

4.5 Other Water Quality Parameters

Source water dissolved oxygen levels averaged 3.6 ± 1.1 mg/L over the course of the study (Figure 16). The source water temperature averaged 14.8 ± 2.2˚ C over the course of the pilot and did not change through the pilot system. Given the source water was a ground water; it remained relatively stable through the study period. Water temperature was not impacted by outside air temperature, which, at times, could be very cold (Figure 16). The pilot study demonstrated that biological treatment will work in colder regions, provided groundwater is the source of drinking water and the facility is adequately heated. TOC in the source water averaged 1.3 mg C/L and did not change through the pilot system contactors and filters.

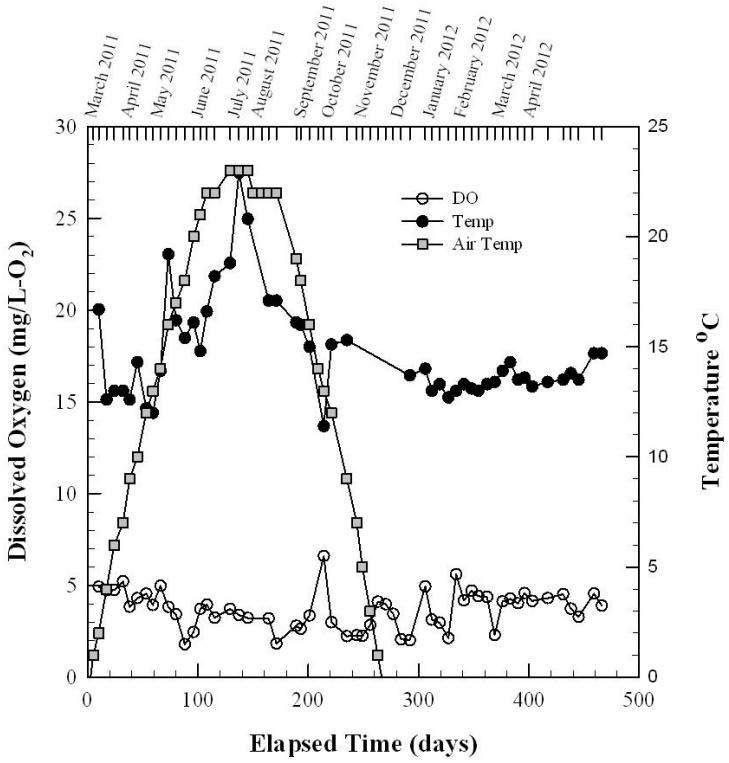

Figure 16. Raw water temperature and DO with average air temperature.

Alkalinity in the source water averaged 357 ± 2 mg $CaCO_3$/L. Average alkalinity after passing through the contactors and filters fell within a narrow range of 342 to 345 mg $CaCO_3$/L. Alkalinity change is directly related to nitrification and therefore, closer examination of alkalinity trends is worthwhile. Alkalinity in Filter 1 dropped from 357 to 337 mg $CaCO_3$/L, then returned to 357 mg $CaCO_3$/L (Figure 17) between 40 and 70 days. This trend corresponded to the time leading up to and just after the chlorination episode. Alkalinity values in Contactor 1 and Filter 1 dropped gradually from 120 to 200 days, after which alkalinity leveled-off at approximately 335 mg $CaCO_3$/L. Raw water alkalinity during the same time was around 360 mg $CaCO_3$/L, which equated to a drop of approximately 25 mg $CaCO_3$/L through the pilot system. This decrease is in very close proximity to the theoretical predicted drop of 7.1 mg $CaCO_3$/L per 1 mg N/L ammonia oxidized.

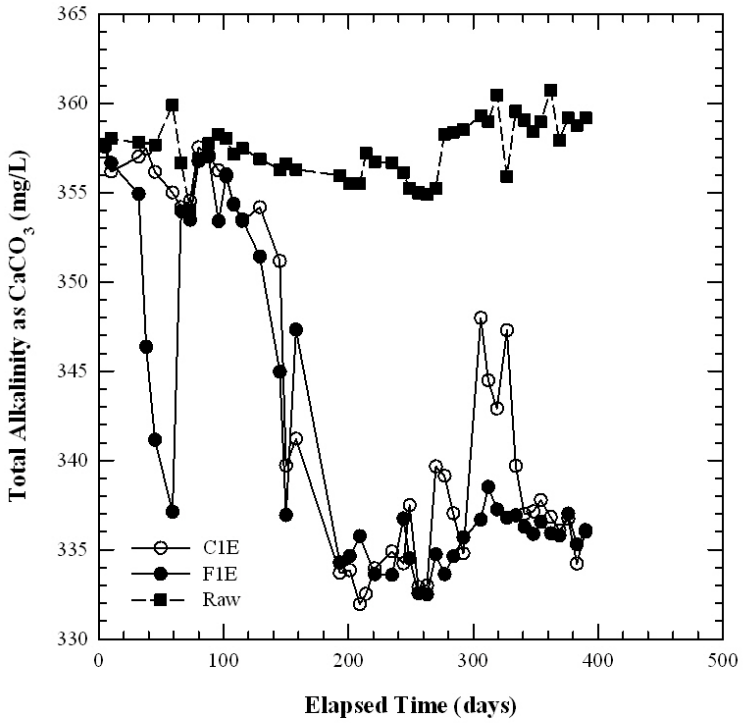

Figure 17. Total alkalinity of raw, Contactor 1 effluent and Filter 1 effluent.

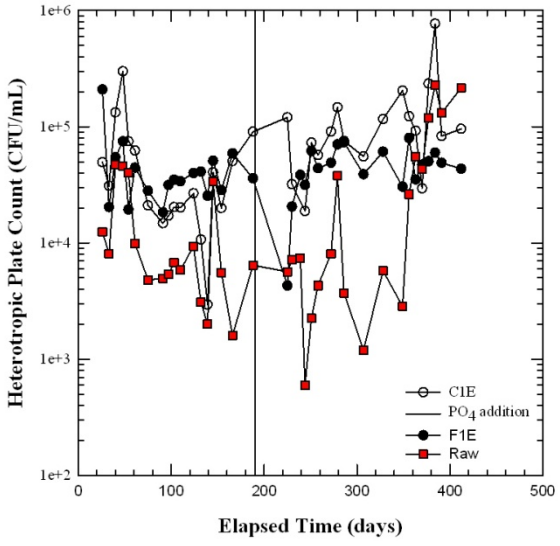

Figure 18. Heterotrophic plate counts (HPCs) in raw, Contactor 1 effluent and Filter 1 effluent.

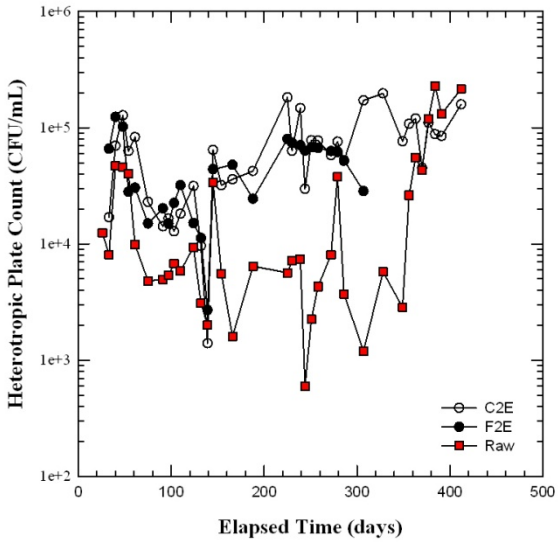

Figure 19. Heterotrophic plate counts (HPCs) in raw, Contactor 2 effluent and Filter 2 effluent.

4.6 Assessment of Bacterial Population Based on HPCs

Heterotrophic plate counts (HPCs) measurements in the raw source water, contactor, and filter effluent waters were performed on a regular basis. Raw water HPCs generally fell between 2000 and 8000 CFU/mL until about 350 days into the study (Figure 18). During the same time period, HPCs in both Contactor 1 and Filter 1 were approximately an order of magnitude greater in HPC concentrations. Beyond 350 days, raw water HPCs suddenly increased to levels similar to those in the filter and contactor effluent waters. The reason for this observation is unknown. Similar HPC trends were observed in Contactor 2 and Filter 2. It is important to note that HPCs would not provide any insight into operational conditions, such as the occurrence of incomplete nitrification.

Obviously, with any biological treatment approach, bacteria release from the system will occur. Appropriate and effective disinfection must be in place to adequately inactivate the microbiological community shed from the system.

5. Discussion and Summary

5.1 Discussion

The pilot study demonstrated the ability of biological treatment to effectively remove ammonia and iron from the source water. The development of biological activity and subsequent complete oxidation of ammonia to nitrate in the system was initially much slower than anticipated based on previous work, although the site's water quality was more challenging from an ammonia level standpoint than conditions in past work. Fortunately, the pilot study was valuable in identifying key reasons for the discrepancies, and more importantly, identifying engineering and design improvements to address them. For example, loading rate targets, the sensitivity of the system to dissolved oxygen throughout the contactor, need to keep the diffuser clean, occasional backwash of contactor, and phosphate feed were all identified as important.

Table 5. Final Design and Operating Parameters

Parameter	Contactor	Filter
Filter loading rate		
m/hr	5.4 (1.2 - 10.5)	4.9 (1.22 - 5.4)
gpm/ft^2	2.2 (0.5 - 4.3)	2.0 (0.5 - 2.2)
Air flowrate		
L/min	2.5	--
cfm/ft^2	2.86	--
Backwash conditions		
duration, min	5	15
bed expansion, %	0	50
m/hr	124	41.5
gpm/ft^2	51	17
Contactor		
depth, cm	76.2	--
depth, inches	30	--
effective size, mm	12.7 (6.35 - 31.8)	--
effective size, inches	0.5 (0.25 - 1.25)	--
Filter		
anthracite depth, cm		50.8
anthracite depth, inches		20
anthracite mm	--	0.97
anthracite , inches	--	0.04
sand depth, cm		25.4
sand depth, inches		10
sand, mm	--	0.45
sand, inches	--	0.018

By the termination of the pilot study, complete oxidation of the source water ammonia (3.2 mg N/L) to nitrate was achieved in Contactor 1 and removal of iron (0.63 mg Fe/L) through the anthracite/sand filter followed. Other operating and maintenance parameters are summarized in Table 5. Orthophosphate addition was necessary and it is uncertain how long it would have taken to get the system to the same goal, had all of the parameters been optimized at the start-up.

5.2 Summary of Key Findings

The biological treatment ammonia pilot study produced a number of new and very important findings that will improve the drinking water field's understanding of biological water treatment in general and how to effectively operate such systems. The following findings are highlighted:

- Once optimized, the biological pilot system achieved the treatment goal of completely oxidizing all of the ammonia in the source groundwater to nitrate. Complete oxidation of ammonia all the way to nitrate was eventually achieved in the contactor (Contactor 1) that contained 30 inches (76.2 cm) of small gravel. The addition of air at the base of the contactor was necessary design feature to address the oxygen demand of the nitrification process and iron oxidation.
- A dual media (20 inches [50.8 cm] anthracite/10 inches [25.4 cm] sand) filter (Filter 1) after the contactor provided additional ammonia/nitrite oxidation, and achieved excellent and consistent iron removal.
- The source water contained very little phosphorus. Orthophosphate is an important biological nutrient and its addition was necessary to increase the rate of microbial acclimation, particularly with regards to nitrite oxidizing bacteria. The system responded almost immediately to the addition of orthophosphate. A dose of 0.3 mg PO_4/L was used in the pilot. The orthophosphate feed was terminated for an extended period of time during which no negative impact on the system's performance was noted.
- Maintaining near saturated dissolved oxygen levels in the contactor was critical to the processes' operation and effectiveness at achieving desired ammonia oxidation and iron removal. Drop in dissolved oxygen levels due to diffuser "clogging" resulted in delayed oxidation of ammonia in the contactor and release of nitrite. Dissolved oxygen monitoring was a good process measurement tool and must be incorporated into full-scale operation. Diffuser design will also be very important engineering aspect of the full-scale system.
- Contactor and filter loading rates were important operating variables, although the pilot system was more sensitive to orthophosphate and oxygen concentration. The pilot demonstrated that a contactor and filter operated in series at loading rates of 2.2 gpm/ft^2 (5.4 m/hr) and 2.0 gpm/ft^2 (4.9 m/hr), respectively, met desired finished water quality objectives.
- Alkalinity decrease following nitrification in the systems was predicted by theoretical considerations and could be used as an additional process monitoring tool.
- Contactor maintenance was minimal. Although, not systematically evaluated during the pilot, there was some evidence to suggest backwashing an acclimated contactor was beneficial. As a result, monthly backwash of the contactors is recommended. Similarly, minimal filter maintenance was

necessary. Filters were backwashed only once a week by achieving 50% bed expansion for 15 minutes.

5.3 Future Work/Questions

Fortunately the pilot was extremely successful in that it identified many new and important details regarding the operation of a biological ammonia oxidation system that would not have been identified otherwise. The time it took to make the discoveries and modifications to address them, however, extended the length of the pilot well beyond what was initially expected. In addition, there was not enough time to perform some of the planned investigations. As a result, a number of questions regarding system operation and optimization still remain. Specifically;

- How long would the pilot system (and eventually full-scale system) take to acclimate had it been operated from the start-up under the "optimum" conditions operated at the termination of the pilot study? How would the corresponding nitrate and nitrate contactor and filter finished water profiles look? How long and at what concentration would nitrite peak at? How would the hours of operation (hours per day) impact acclimation period?

- There was not a scientific basis behind the orthophosphate dose selected in the pilot and the dose used was well above the stoichiometric amount necessary for bacterial cell growth in such a system. Relevant questions to orthophosphate dosing include: What is the optimal orthophosphate dose? Is there a benefit to a start-up dose to get the system going? If so, what is the minimum maintenance dose to keep the system going? The system was not impacted by short-term (4 weeks) orthophosphate feed breaks but how will the system perform under long-term orthophosphate stoppages?

- The addition of chlorine to the system after the contactor would eliminate the ability of the filter to oxidize ammonia and nitrite, and the safety factor. In such a case, contactor bed depth could be increased to provide an additional safety margin. Related questions would be "What are nitrogen profiles through a contactor as a function of bed depth?" and "Can chlorinated water be used to backwash the filters/contactors and yet still maintain microbiology of the systems, and nitrification capability?"

- What is the relationship between bed depth, media, loading rate, and ammonia oxidation? The pilot design and operation was based on past EPA work and are within "typical" ranges of granular media drinking water treatment systems.

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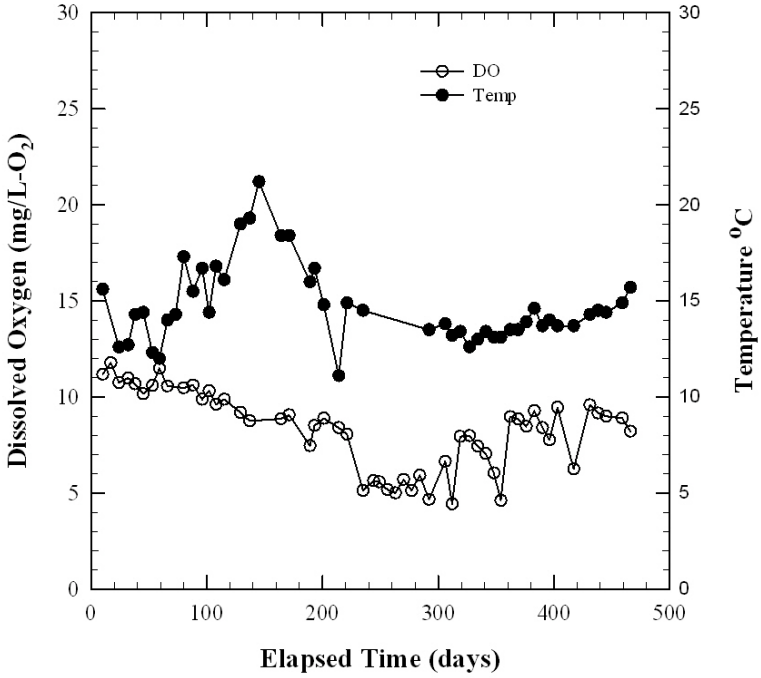
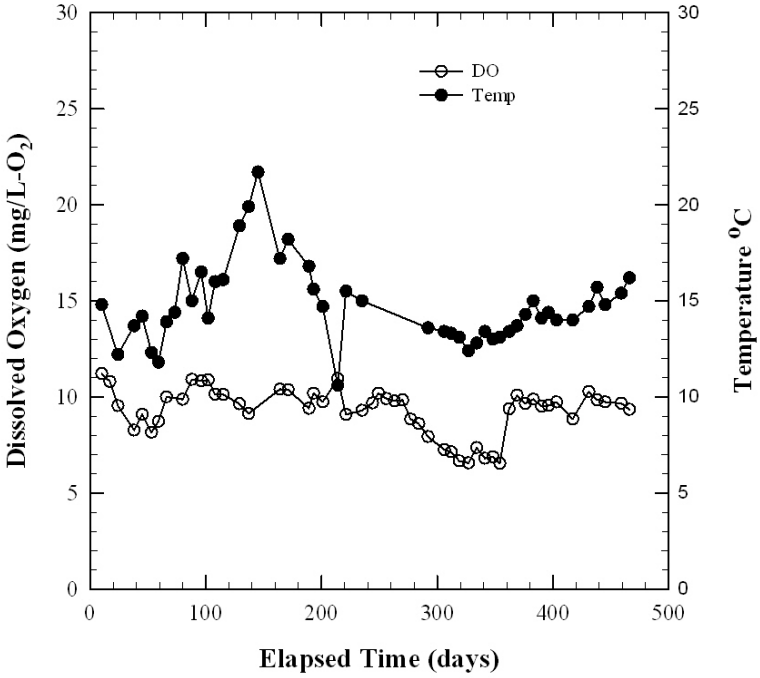
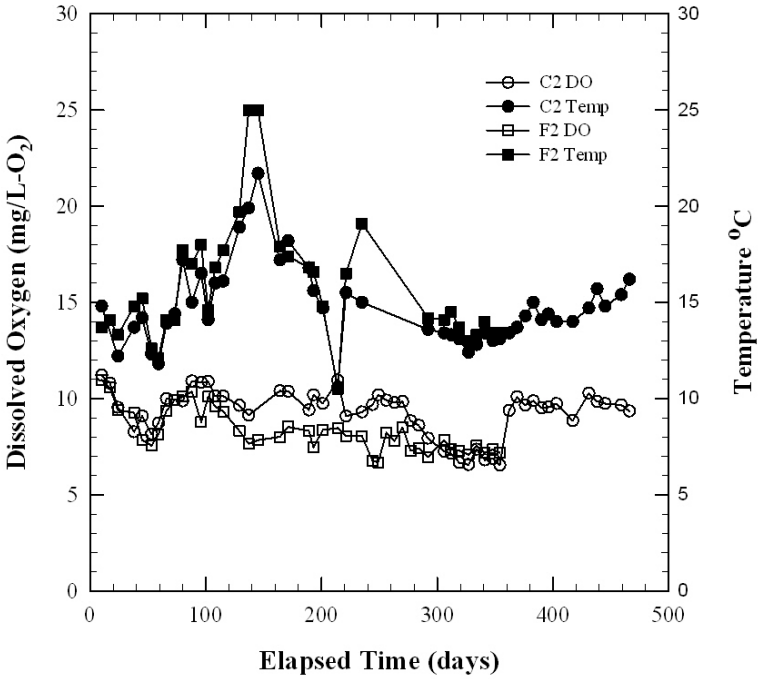
APPENDIX A



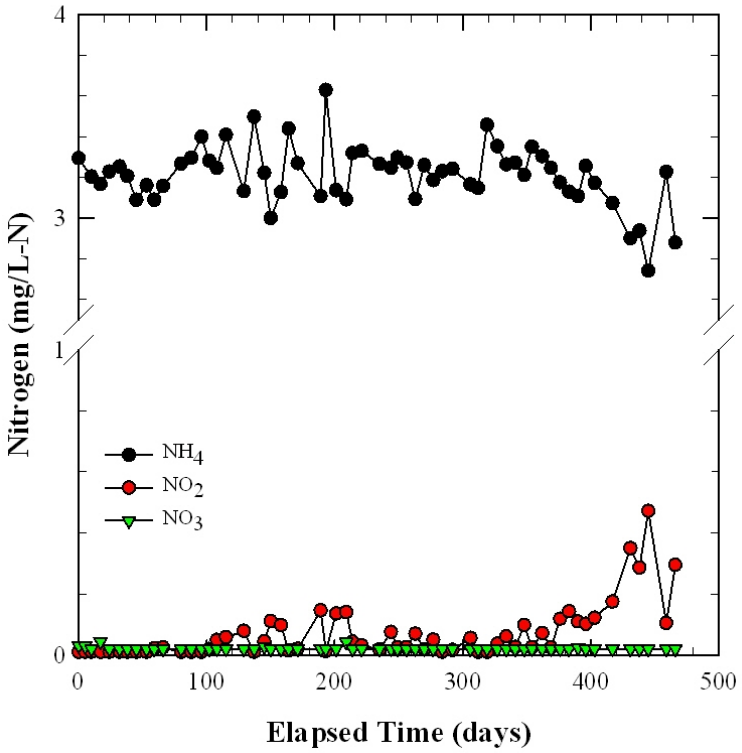
A 1. Contactor 1 effluent temperature and DO.



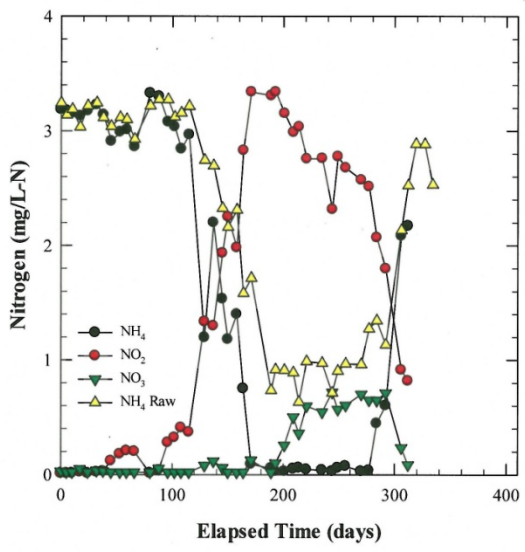
A 2. Contactor 2 effluent temperature and DO.



A 3. Contactor 2 and Filter 2 effluent Temp and DO.



A 4. Nitrogen content of raw water.



A 5. Nitrogen content of finished water from Filter 3.

(Image was scanned from a hard copy. Resolution will be improved.)