## ABSTRACT

THOMAS D. JOHNSON. Conversion from Conventional Water Treatment to Microfiltration in Carthage, NC.

(Under the Direction of Dr. Francis A. DiGiano)

The Town of Carthage, North Carolina replaced its 50-year old conventional water treatment plant (WTP) with a 1-mgd microfiltration (MF) plant on May 24, 2002. This is the first MF facility in the state of North Carolina. The objectives of this research were to document and discuss the changes in the water quality and treatment efficiency that were attributed to the conversion from conventional treatment (coagulation-sedimentation-filtration) to MF treatment without chemical coagulation. Monthly samples were collected from the WTP and six locations in the distribution system (DS) from October 2001 to February 2003 with the exception of August 2002. Implementation of MF improved the water quality when measured by turbidity and particle counts. Elimination of coagulation caused the TOC to increase significantly in the DS while HPC data showed that bacterial regrowth occurred when a disinfectant residual could not be maintained. Operating costs after the conversion to MF were higher than preceding the conversion ($ 0.69 vs. 0.47 per 1,000 gal). Given that coagulation is needed ahead of MF to reduce DBPs, this cost analysis should be repeated.

## ACKNOWLEDGEMENTS

I would like to thank Dr. Francis A. DiGiano and Dr. Donald E. Francisco for their guidance during my study at the University of North Carolina. I am also grateful to Dr. Gregory W. Characklis for his support and criticism during the preparation of this report. The assistance of Alan Oldham, Superintendent of the Carthage WTP during this study, John F. Whitlock, plant operator, the Town of Carthage, Mr. Tim Bishop of Heyward Associates, and Mr. Rob Huemer of U.S. Filter with respect to access to plant operational data is gratefully acknowledged.

This work was funded by U.S. EPA's Office of Research & Development, National Risk Management Research Laboratory, Cincinnati, Ohio, under a contract with Shaw E & I INC. (Contract No. 68-C-99-211) for which the University of North Carolina was a subcontractor to Shaw E & I INC.

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## INTRODUCTION

Replacement of granular media filtration by microfiltration (MF) has increased in popularity in the past decade due to an increase in vendors providing equipment and a cost that is competitive with conventional water treatment systems. Recently, the Town of Carthage, North Carolina replaced its 50-year old conventional treatment facility with a 1 million gallon per day (mgd) microfiltration plant. This is the first MF facility in the state of North Carolina. Carthage chose MF for its promise of a lower finished water turbidity and more consistent performance. Its previous 0.50 mgd water system consisted of an upflow combination flocculator and clarifier followed by dual media filtration. Alum coagulation was used to remove particles and total organic carbon (TOC) among other constituents in the raw water. This process has been difficult to operate due to variable raw water conditions and this caused considerable variability in filtration efficiency. In addition, the high alum dosages required equally high caustic dosages to maintain the pH in an optimum range due to very low alkalinity in the raw water. The new microfiltration facility is not using alum coagulation as part of the treatment process. Therefore, there is no need for caustic addition.

The old conventional plant was often plagued by ineffective filtration that produced turbidity in excess of the maximum contaminant level (MCL) of 0.5 ntu; product water turbidity during storm events has been as high as 1.54 ntu. With the inception of the Interim Enhanced Surface Water Treatment Rule (IESWTR) and the new MCL of 0.3 ntu standard, this could mean many violations for the Town of Carthage, that

result in costly fines. Microfiltration has proven that it can keep turbidity at or below 0.1 ntu consistently.

Another problem that could plague the Town is the limits for disinfection byproducts (DBPs). In the past, the use of chloramines as their secondary disinfectant helped reduce DBPs in the distribution system. However, the continued need to use free chlorine as the primary disinfectant coupled with the elimination of alum coagulation is likely to lead to an increased in DBP production. The distribution system in Carthage is potentially unique among many small water systems. Many of the outer edges of the Town distribution system have dead ends, minimal looping, and little demand. Some of these lines are very long which could result in water age that could exceed 30 days. With such a long amount of time, even the more stable residual produced by chloramines can decay, leaving little disinfecting potential. This can cause bacterial regrowth within the system at these points.

The specific objectives of the project were to document and discuss the changes in the water quality and treatment efficiency that were attributed to the conversion from conventional treatment to microfiltration. Removal of turbidity, particles, and TOC were used to measure the plant performance 7 months before the conversion and 7 months after the conversion. Water quality in the distribution system was examined using turbidity, particle counts, chlorine residual, and heterotrophic plate counts (HPC). The efficiency of the plant and the cost of treatment were measured using operational data for power consumption and chemical usage.

# LITERATURE REVIEW

## REGULATORY IMPETUS FOR MF

The implementation of the most recent Environmental Protection Agency (EPA) regulations has tightening the already strict water quality demands for our nation's water suppliers. Current regulations that are already affecting water utilities are the Stage 1 Disinfectant/Disinfection By-products (D/DBP) (USEPA, 1998b) Rule, Filter Backwash Recycling Rule (USEPA, 2001), Interim Enhanced Surface Water Treatment Rule (IESWTR) (USEPA, 1998a), and the Long Term Stage 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) (USEPA, 2002). All of these are now in affect with a minor exception that small systems (<10,000 people) have until 2004 to comply with the Stage 1 D/DBP Rule. Future Regulations include the Ground Water Rule (USEPA, 2000b), the Stage 2 D/DBP Rule (USEPA, 2000a), and the Long Term Stage 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) (USEPA, 2000a). These regulations are forecasted to be in effect by 2003. Among these regulations is the reduction in disinfection by-products (DBPs), the enhanced removal of *Cryptosporidium* and *Giardia*, and lower turbidity standard of 0.3 ntu. These new standards, in addition to anticipated standards, have brought microfiltration (MF) into the forefront of innovative technology. Numerous studies have shown that, MF can replace the conventional granular media filtration process and produce a superior water quality on a more consistent basis (Dharmappa and Hagare, 1998). MF may be very appropriate for small systems (<10,000 people) that need to comply very soon with new EPA regulations. MF technology is

continuing to mature with technological advances that will improve the economies of scale and more competition among manufacturers has lowered the unit cost significantly over the last 10 years. These trends suggest that MF will become more competitive with conventional granular media filtration. As examples, the San Jose Water Company in Saratoga, California (Martinez, 1999) and the City of Marquette, Michigan (Kelley and Olson, 1999) decided to install MF in anticipation of new regulations.

## APPLICATION OF MF TO SMALL SYSTEMS

Small systems are defined as those serving less than 10,000 persons. By this definition, more than 95 percent of the water suppliers in the United States are small systems (Stevie and Clark, 1982). Small systems typically do not have the financial resources of larger systems. Thus, the financial burden of impending EPA regulations on small systems may be very large and this comes at a time when the U.S. economy is suffering as well. Small systems have a much lower average income, higher unemployment rates, larger populations of aging residents, and more difficulty obtaining loans than larger communities (MacDonald, Zander, and Snoeyink, 1997). This can severely limit the tax base from which the small system can gain funds, unless the community is wealthy. Some of the important advantages of membranes in small systems include versatility, lower cost, and easier plant operations. Membrane systems are versatile because a wider range of raw water quality can be treated effectively. For example, nanofiltration (NF) can be used to remove dissolved solids from brackish water, a resource that has heretofore been uneconomical or undesirable to develop (Martinez, 1999). Thus, membranes can be useful for small systems that are located in water scarce environments where finding an alternative water source may be difficult. Small systems

are usually not able to benefit from the economies of scale for the construction of treatment systems (Stevie and Clark, 1982; Logsdon et al., 1990). Membranes may offer an alternative to economies of scale because of their modular design. The compact, modular design of membrane systems offers flexibility in expanding capacity incrementally over a period of years (Martinez, 1999). In this way, small systems do not need to embark upon a single major expansion program with excess capacity in the near term. This approach can avoid a large initial capital cost and thus a large increase in consumer billing rate.

The budgets of small systems are typically inadequate to attract and keep well-qualified, plant operators. Membrane systems may offer a solution because they are easier to operate than granular media filtration and require less maintenance. Operators will shift their daily tasks related to regulatory compliance, maintenance, and treatment. This shift is due to the automation provided by the MF equipment and control system (Gere, 1997). In addition, the complexity and relatively higher cost of conventional treatment requires advanced skill and attention by the operator (Pontius, 1997).

## QUALITY OF MF PERMEATE

The major water quality parameters are turbidity, particle counts, microorganisms/pathogens, total organic carbon (TOC), and inorganic chemicals such as iron and manganese. MF has been proven to consistently produce filtered waters (permeates) with turbidities below 0.1 ntu (Karimi et al., 1999; Taylor et al., 1999; Coccagna et al., 2001; Karimi et al., 2002). The requirement of the IESWTR is that 95 percent of the finished water samples have a turbidity of less than 0.3 ntu for and a maximum of not greater than 1 ntu (USEPA, 1998a). Thus, MF has been proven to be

capable of more than meeting the new standard. In addition, the attainment of ultra-low turbidities can qualify the water utility for a disinfection credit of at least 2.5 log removal of *Cryptosporidium* in anticipation of the LT2ESWTR (USEPA, 2000a).

Even though turbidity is the reporting requirement of EPA regulations, the use of particle counters has gained in popularity because they measure particle concentration directly rather than indirectly. Moreover, the instruments record the particle size distribution which is useful when seeking a surrogate measure of protozoa cysts of health concern such as *Cryptosporidium* and *Giardia* (Mallevialle, Odendaal, and Wiesner, 1996; Jacangelo et al., 1991). Many studies have shown that MF can reduce particle counts in the raw water that are in the thousands per mL to below 100 per mL (AWWA 1999; Mallevialle, Odendaal, and Wiesner, 1996). A pilot study in Manitowoc, WI demonstrated that particle counts from 3,100-54,000 particles per mL in the raw water were reduced to below 40 particles per mL (Taylor et al., 1999). Likewise, Coccagna et al. (2001) reported a 3.5 log rejection of particles in the size range of 2-4 µm. Karimi et al. (1999) also noted 2.5-3.5 log rejection.

The high removal of particles by MF also gives rise to instrumental problems in measuring particles accurately at low concentrations in the permeate water. In particular, the Memcor MF design includes an air backwash at very high pressure (over 90 psi) every 20 to 30 minutes. The air is introduced into the permeate side of the membrane. Upon restart of operation, air bubbles are entrained in the product water and these may be recorded in particle counters as particles in the range of 2 to 100 µm. The number of bubbles may far exceed the number of particles. In a pilot-plant study in Manitowoc, WI (Taylor et al., 1999; Kothari and Taylor, 1998), an air relief tube was added to remove

bubbles. In addition, the particle counter was set to record only the particles in the size range of 2to 5 µm in order to avoid measurement of air bubbles as particles. This revised size range was also chosen to be more consistent with the size of protozoan cysts of regulatory concern such as *Cryptosporidium* and *Giardia*.

The fact that the nominal pore size of MF membranes is in the range of 0.2 µm raises questions as to the usefulness of a particle counter given that particles of less than 2 µm cannot be measured. Because particles are removed by the sieving mechanism, the theory would indicate that no particles greater than 2 µm should be found in the permeate if the pore size is 0.2 µm.

The presence of particles greater than the stated nominal pore size could be due to damaged seals that prevent feed water from entering the permeate stream, damaged hollow fibers or to the presence pores much larger than the nominal size (Chow et al. 1997), large variation in pore size around the nominal size (Sadr-Ghayeni et al. 1999) or biofilm growth on the membrane and sloughing (Jacangelo et al., 1991). The failure of only a few individual hollow-fibers within the modules that contain thousands of hollow fibers may cause particle penetration (Chow et al., 1997). Particle penetration into the permeate may also increase as pressure increases during operation to maintain a constant flux as a cake layer develops on the membrane surface.

Sadr-Ghayeni et al. (1999) offered oversized pores as a possible explanation for poor performance of MF. They found that the presence of oversized pores has been reported frequently for all commercial types of membranes with stated pore size of 0.2 µm. For example, while the nominal pore size for two polyvinylidene fluoride MF membranes were 0.22 µm, the median pore size was 0.45 µm. A similar conclusion was

reached for a polycarbonate UF membrane; the nominal pore size was stated as 0.05 µm, but a median pore size was 0.1 µm. Two other polycarbonate membranes had median pore sizes very close to the nominal pore size. These large pores could result in channeling of a significant volume of raw water without as much particle removal.

Bacteria can also grow on the concentrate side of the membrane surface because substrate is available and backwashing of MF membranes does not usually include the addition of a disinfectant because it may damage the membrane surface. These bacteria may penetrate through to the permeate side or they may also grow on the permeate side (Jacangelo et al., 1991). The sloughing of clumps of bacteria with a size of greater than 2 µm could be measured by particle counters.

The emphasis on efficient and reliable filtration has increased greatly since the 1993 outbreak of *Cryptosporidium* outbreak in Milwaukee, WI. *Cryptosporidium* oocysts range in the size from about 1-5 µm in diameter while *Giardia* cysts are in the range of 3-9 µm (Adham et al., 1998). The typical MF membrane with a nominal pore size of less than 0.5 µm should theoretically provide an absolute barrier. Numerous studies have been conducted to confirm the ability of MF to remove microorganisms from the raw water. Adham et al. (1998) reported that MF rejected *Giardia* and *Cryptosporidium* cysts to below the detection limit which, based on the feed concentration, meant that more than a 5-log removal could be achieved. Similar removals of *Cryptosporidium* by MF were found by Coccagna et al. (2001) and States et al. (2000). A 2-log removal of cyanobacteria was found by Chow et al. (1997). However, this removal was measured under the "worst case scenario" with a transmembrane pressure of 2 bar (29.4 psi) whereas a typical system operates at 0.3-1 bar

(4.4 – 14.5 psi). At typical pressures, a higher removal (3.4 log removal) was reported for micro-algae by Coccagna et al. (2001).

The penetration of bacteria into permeate was evaluated in a wastewater application of MF (Sadr-Ghayeni et al. 1999). A 3-4 log removal of SW8 bacterial was measured. An even higher removal (greater than 6-log) was obtained if measured by a culturable bacterial count (i.e., bacteria that could be grown after filtration). However, these investigators also used direct counting technique to observe that ultra-microbacteria (~1.2 x 0.5 μm) in the membrane feed could not be removed; nevertheless, those found in the permeate by this technique could not be cultured.

The effectiveness of MF in removal of coliform bacteria has also been measured (Coccagna et al. (2001; Taylor et al. 1999; Hillis 1997). Coliform bacteria are used as an indicator of adequate disinfection because of their presence in human faeces and other warm blooded animals (Stevens et al., 2001). Out of all the coliform species present, *Escherichia coli (E. coli)* is the greatest contributor. This species of bacteria is only found in faecal material. Coccagna et al. (2001) found that both *E. coli* and the coliform group were absent in the MF permeate even after spiking of the raw water. Complete removal of these indicator organisms was also reported in an MF pilot-plant study in Manitowoc, MI and also in a full-scale application in North West Water Huntington (Hillis, 1997).

While indicator organisms have been shown to be removed effectively by MF, their usefulness as indicators of pathogen free water has been challenged (Nwachuku et al., 2002; Stevens et al., 2001). This is because coliform bacteria are not always present

during waterborne disease outbreaks, such occurred in Milwaukee, WI in 1993 due to penetration of *Cryptosporidium* through granular media filters (Stevens et al., 2001).

MF has been shown to remove heterotrophic bacteria although the removal is never absolute (Hillis, 1997; Karimi et al., 1999). Heterotrophic bacteria are naturally occurring and more resistant to processes and chemicals that are typically used to remove fecal coliforms but they are not indicative of a public health risk (Lisle, 2003). Even though the Safe Drinking Water Act (SWDA) does not regulate these organisms, the SWDA does recommend that a disinfection residual be maintained such that the heterotrophic plate count (HPC) is consistently below 500 colony forming units (CFU)/mL. HPC was absent in the MF permeate at the 21 mgd North West Water Huntington water treatment plant in the United Kingdom (Hillis, 1997). Karimi (et al., 1999) found that 50-86 percent of heterotrophic bacteria were removed initially by MF in a pilot-plant study, but soon after startup, they noted a higher HPC in the permeate than in the feed water. Regrowth was suggested due to the lack of a chlorine residual in the raw water feed. An experiment was conducted during a pilot-plant study by Jacangelo et al. (1991) to determine whether the presence of heterotrophic bacteria in the permeate was due to penetration through the membrane or to colonization of the membrane piping between the MF module and the sample tap for permeate. The results showed that heterotrophic bacteria were due to the latter rather than the former.

Because the nominal pore size for MF is fairly large (0.2 µm) compared to the size of viruses, it is not expected that MF will remove them effectively. Tests of virus rejection in all membrane processes (MF, ultrafiltration (UF), NF and reverse osmosis) have been conducted with the MR2 bacterial virus, a non-infectious type that is similar in

shape and size (0.025 μm) to polio and hepatitis. Adham et al. (1998) showed log removals in the range of 0.05 to 3.27 for one type of MF unit. The variable nature of removal suggested that MF is not an effective barrier for virus. .

MF can only be effective in removal of total organic carbon (TOC), the precursor material to DBPs, if preceded by chemical coagulation or by addition of powdered active carbon (PAC). This is because TOC is a surrogate measure of natural organic matter (NOM) that ranges wide in molecular size from 0.001 μm (truly dissolved fraction) to about 1 μm (colloidal, macromolecular fraction). In fact, most of the NOM is less than 0.1 μm in diameter, making it very difficult for MF to remove without coagulation. For example, Karimi et al. (1999) found only an average of 10 percent TOC removal by an MF system that was operated at four different permeate flux values. The addition of ferric chloride as a chemical coagulant increased the TOC removal to 22 percent. A follow-up study at the same MF facility (Karimi et al., 2002) showed very little removal of non purgeable organic carbon (essentially equivalent to TOC).

Higher removals of TOC can be achieved by Ultrafiltration (UF) than MF membranes because of their smaller nominal pore size. A UF membrane was capable of removing 40 percent of TOC without pretreatment (Laine et al., 1990). MF can only achieve the same removal or higher if a coagulant is added for pretreatment. When a polyaluminum chloride coagulant was added ahead of UF, 60 percent of the TOC was removed while the sole addition of PAC ahead of UF had an 85 percent removal (Laine et al., 1990). The combination of PAC and polyaluminum chloride provided little additional removal over PAC alone, increasing the removal to only 86 percent (Laine et al., 1990). A similar study of UF using a polyaluminum coagulant resulted in 66 percent

removal of dissolved organic carbon from the feed water (Lahoussine-Turcaud et al., 1990). The combination of coagulation and UF, therefore, has been shown in several studies to remove more TOC than conventional treatment processes that use either aluminum sulfate or iron chloride for coagulation (typical TOC removals range from 30-60 percent) (Mallevialle, Odendaal, and Wiesner, 1996).

MF has been shown to remove inorganic chemicals such as iron, manganese and aluminum from feed waters (Ellis et al., 2000; Hillis et al., 1997; Park et al., 2002; Mallevialle et al., 1992). These inorganic chemicals are generally in the particulate form of a metal hydroxide. Iron and manganese are generally only present in ground waters in their divalent forms (Ellis et al., 2000). They are very soluble in these forms, but when oxidized to their trivalent forms, they become very insoluble and precipitate as metal oxides. These precipitates range in size from 1.5 to 50 μm (Ellis et al., 2000). Thus, they should be removed easily using MF. Ellis et al. (2000) demonstrated that a MF membrane could remove both of these metal oxides to the detection limit. The detection limit for iron was 0.1 mg/L, while manganese was 0.03 mg/L. These results were confirmed by Hillis et al. (1997) at a full-scale MF facility; iron and manganese were not detected in the permeate. Park et al. (2002) showed that MF reduced iron, manganese, and aluminum concentrations by 84, 94, and 89 percent, respectively. A UF membrane was able to achieve even higher removals of aluminum (from 89 to 99.5 percent) (Mallevialle et al., 1992). Removal of iron was also very high for MF as shown in repeated tests in the iron concentration in the permeate was below the detection limit (Mallevialle et al., 1992). Removal of these metals was attributed to their occurrence as colloidal hydroxides.

## MF WATER PRODUCTION

MF production is generally linked to the flux through the membrane, the recovery, and the fouling potential. Flux is defined as the permeate flow rate produced per unit surface area of membrane. Permeate flux is proportional to the applied pressure and inversely proportional to the resistances that are caused by the cake layer and the membrane (Taylor and Wiesner, 1999). Darcy's law is used to describe the relationship among flux, pressure and resistance:

$$\text{permeate flux} = \frac{\text{applied pressure}}{\text{cake resistance} + \text{clean membrane resistance}} \qquad (1)$$

$$J(t) = \frac{\Delta P}{\mu\left[R_c(t) + R_m\right]} \qquad (2)$$

where $\Delta P$ is the transmembrane pressure across the membrane (nearly equivalent to the applied pressure), J is the permeate flux, $\mu$ is viscosity of the water, $R_c(t)$ is cake resistance over time, and $R_m$ is clean membrane resistance. An alternative expression of filtration performance is specific flux, $K_w$:

$$K_w = \frac{J(t)}{\Delta P} = \frac{1}{\mu\left[R_c(t) + R_m\right]} \qquad (3)$$

MF modules are designed to operate either at constant $\Delta P$ or constant J. For constant $\Delta P$ operation, the specific flux declines with time due to the increase in cake resistance as particles accumulate on the membrane surface. For constant J operation, the specific flux decreases because $\Delta P$ increases with time. Because viscosity decreases with increasing temperature, the specific flux increases with increasing temperature. Most MF systems are designed to operate at constant J so as to provide a constant water production rate throughout the day.

Fouling of the membrane surface over time is a major concern. Reversible fouling is a reduction in specific flux, $K_w$, that can be recovered either by chemical or hydrodynamic means. For instance, in dead-end, hollow fiber membrane modules, the rejected particles accumulate on the membrane surface in the form of a cake layer. To maintain constant flux, the pressure typically increases from about 4 psi for a clean membrane to about 15 psi at which time the membrane is backwashed to remove the cake layer. The typical frequency of backwashing is 15 to 60 minutes, depending on raw water quality (Chow et al., 1997; Karimi et al., 1999; Taylor et al., 1999). The concentrate-side of the membrane is flushed with water while typically an air pulse is used from the permeate side to assist in dislodging the filter cake. In general, the time between cleanings decreases with increasing permeate flux but increases with increasing temperature (Taylor et al., 1999; Kothari and Taylor, 1998; Karimi et al., 1999). Complete recovery of specific flux is rarely achieved, i.e., a higher applied pressure is needed after each backwashing than for the initially clean membrane (Chow et al., 1997; Dharmappa and Hagare, 1999; Karimi et al., 1999, Taylor et al., 1999). Usually after several months of operation, the membrane module must be taken offline for chemical cleaning. If complete recovery of the specific flux is not attained, then irreversible fouling has occurred. The buildup of irreversible fouling eventually will force replacement of the membrane elements within the module; a typical useful life is estimate as 3-5 years (Owen et al., 1995).

Net water production is the permeate water that is delivered to the distribution system after subtraction of the permeate water that is used for backwashing membranes. A study in Japan showed that the net water production ranged widely (45 to 308 $m^3/m^2$)

for thirty-five different MF/UF systems (Kunikane et al., 1995). The range is wide because systems operate over a range of pressures and thus a range of permeate flux (Equation 2). Net water production can be used to estimate the membrane surface area, and thus the number of membrane modules, that are needed for specific plant size. Operation at higher flux to increase the net water production required coagulation with polyaluminum chloride as a pretreatment in order to reduce membrane fouling. Water recovery is the ratio of the permeate water to the feed water which is usually expressed as a percentage. The difference between the feed water rate and permeate rate is due to water lost in backwashing of the membranes for dead-end operation or due to feed water that does not permeate through the membrane in cross-flow operation. Kunikane et al., (1995) also reported that recovery values ranged from 80 to 99 percent with no significant difference between MF and UF systems. Recovery in cross-flow systems is typically lower than dead-end filtration. However, recovery can be increased above 90 percent if the recycle stream is treated and reprocessed through the membrane.

Karimi et al. (2002) compared the performance of two MF commercial systems in terms of permeate flux and percent water recovery. Both demonstrated acceptable performance over a flux range of 38 to 86 gfd (gallons per square foot of membrane surface area per day) and the recoveries exceeded 94 percent. Taylor et al. (1999) showed that a fairly high flux (67 gfd) could produce high quality water although the frequency of chemical cleaning was not acceptable. The addition of alum as a pretreatment decreased the frequency by a factor of 2.5 to 3. However, these authors suggest that the accrual of aluminum hydroxide precipitates on the membrane surface could cause irreversible fouling and may offset the advantage of operating at a high flux.

## IMPACT OF MEMBRANE TREATMENT ON WATER QUALITY IN THE DISTRIBUTION SYSTEM

Maintaining water quality within the distribution system is critical for protection of the consumer. The piping, storage, and appurtenances are a "barrier" that needs to maintained and operated to prevent contamination (Kirmeyer et al., 2001). Water quality issues are generally classified as aesthetic, microbiological, and physical/chemical.

The usual aesthetic issues relate to color, taste, and odors. Although both the consumer and the utility place a high value on the microbial and chemical safety of the water, the consumer also places emphasis on the aesthetic aspects. The consumer prefers not to detect a taste or smell of chlorine and typically equates, although incorrectly so, a low health risk with the absence of chlorine in the water (Kirmeyer et al., 2001). Thus the water utility must strive to maintain a disinfectant residual in the distribution system for public health safety, yet keep the residual low enough so as not bring complaints of a chlorinous odor. The addition of NF after conventional treatment offers the possibility to reduce the amount of chlorine needed to maintain a residual by removing NOM and other chemical constituents that would be otherwise oxidized by chlorine. Peltier et al. (2002) observed that a decrease in chlorine demand from a range of 0.1 to 0.2 mg/L before the addition of NF to 0.05 mg/L after the addition. As a result, the decision was made to reduce the chlorine in the finished water from 0.5 to 0.2 mg/L. Customer satisfaction improved because the majority of taste and odor problems were attributed to excessive chlorine. A survey conducted by utility found that 35 percent of the respondents perceived a change in water quality and 98 percent described this change as an improvement; 33 percent of the respondents stated that the water tasted less chlorinous.

Escobar and Randall (1999) also observed that chlorine residual was more stable in the distribution system after addition of NF treatment.

Whether a switch from conventional treatment to MF in contrast to the addition of NF will also improve water quality in the distribution system has not been reported. Unlike NF, with the possible exception of Fe in reduced form, chlorine demanding substances are unlikely to be removed by MF. Nonetheless, a lower turbidity can be achieved with MF than with conventional treatment.

Bacterial regrowth in the distribution system can be minimized by reducing the substrate supply through treatment. A reduction in biodegradable dissolved organic carbon (BDOC) by NF has been found to reduce the potential for bacterial regrowth in the distribution system (Escobar and Randall, 1999; Peltier et al., 2002). By removing NOM, NF removes the associated BDOC fraction to below 0.1 mg-C/L (Peltier et al. 2002; Kopp et al., 1993). Therefore, NF should be able to produce water that is microbiologically stable in the distribution system. According to Peltier et al. (2002), a decrease in the total bacteria abundance was found in the distribution and with use of a significantly lower disinfectant residual. If only the actively respiring bacteria were counted, the reduction in BDOC resulted in a 1-log decrease from an average of 4,200 bacteria/mL during conventional treatment to 420 bacteria/mL after the startup of NF. Escobar and Randall (1999) found a similar result; they determined that the HPC in the distribution system receiving NF treated water was lower than when receiving conventionally treated water.

NF treatment can also reduce the formation of DBPs within distribution systems by removal of a large fraction of the precursors that are generally described being NOM.

Among other factors that increase DBP production is high water temperature (Singer and Reckhow, 1999). Temperature determines the rate of reaction between the DBP precursors and the disinfectant. As the temperature increases, the rate of reaction decreases, thus forming more by-products. Therefore, seasonal changes in temperature can affect DBP production. While the rate of DBP formation increases with temperature, if a DBP is biodegradable, an increase in temperature can also lower the DBP concentration after its formation. This would be true for haloacetic acids (HAA) that have been shown to be biodegradable (Singer and Reckhow, 1999). Water temperature; cannot be controlled and thus, other options must be taken to reduce DBP formation. Removal of NOM is one of the most effective methods of reducing DBP formation. It addition to reacting to form trihalomethanes (THM) and HAA, which are the currently regulated DBPs, NOM removal reduces the chlorine demand. While NF can reject NOM, the pore sizes of MF and UF are too large to achieve NOM removal. Peltier et al. (2001) measured a 50 percent reduction in THM concentration in the distribution system after installation of NF. The reduction was attributed to both a large removal of NOM and use of a lower chlorine dosage. Escobar and Randall (1999) reported a 93 percent reduction in NOM after NF treatment. While they did not measure THMs, it is reasonable to expect that THMs would be reduced if NOM is removed.

## COST OF MEMBRANE TREATMENT

The increase in popularity of membrane applications has led to a more competitive cost market, a more demanding regulatory environment, a broader range of membrane processes, and the availability of new membrane materials. (Wiesner and Chellam, 1999). Capital cost for membrane systems has steadily decreased. The cost for

MF and UF are equivalent or less than the cost of conventional systems for the same applications (Owen et al., 1995; Wiesner and Chellam, 1999; Pickering and Wiesner, 1993). A study by Dharmappa and Hagare (1999) reported a capital cost of 80-90 percent of a conventional system for small systems defined as those treating less than 1 mgd.

Economies of scale are one of the most import aspects of capital cost. .Typically, as the plant capacity increases, the capital cost decreases per unit volume of plant capacity because the cost of manufacturing equipment and the other facilities decreases as the unit size increases (Sethi and Wiesner, 2000). The typical way to express a capital cost equation is given by:

$$C_{capital} = a * (Q_{GPD})^{n} \qquad (4)$$

Where the exponent, n, represents the economies of scale. This exponent is between 0.4 and 0.8 for membrane systems (Wiesner and Chellam, 1999; Sethi and Wiesner, 2000). The lower values are for small systems while the upper values are for larger systems.

The overall economies of scale for a membrane system are dictated by both the equipment and the raw water quality. Non-membrane components, such as pumps, pipes, and valves represent up to 50 percent of the capital costs. These components are known to yield economies of scale of about 0.4 (Wiesner and Chellam, 1999; Sethi, Wiesner, and Dennis, 1999) and are not affected by the raw water quality. Costs that are affected by the raw water quality are mostly associated with the permeate flux. The selection of permeate flux, which determines the membrane area needed, is sensitive to the size and concentration of particles in the raw water feed (Sethi and Wiesner, 2000; Sethi, Wiesner, and Dennis, 1999). Very small and large particles exhibit less fouling tendencies because

of a high rate of back-transport away from the membrane surface. This permits the use of a high permeate flux because even though particles are brought to the surface at a high rate, they do not foul the surface. Intermediate sized particles have a low rate of back-transport and thus, tend to foul the membrane surface. As a result, the rate of particle introduction must be kept low to avoid fouling and consequently, the permeate flux must be low. Given that permeate flux determines the required membrane surface area, capital costs will increase as the concentration of intermediate particles in a raw water increase (Sethi and Wiesner, 2000). The capital and operating costs will also be higher because more frequent backwashing is needed. This lowers the net water productivity meaning more membrane area is needed and increases the operating costs associated with backwashing (Sethi, Wiesner, and Dennis, 1999).

Operation and maintenance (O&M) costs include the cost of chemicals, energy, membrane replacement, labor, and residuals disposal. Dharmappa and Hagare (1999) reported O&M costs were 40-50 % higher for membrane systems than for conventional systems at flow rates of less than 1.1 mgd.

Irreversible fouling eventually leads to chemical cleaning of the membrane. The cost for chemical cleaning in O&M analyses has been set at a fixed cost for chemicals (Dharmappa and Hagare, 1999) or ignored (Owen et al., 1995; Sethi, Wiesner, and Dennis, 1999). Only Chellam et al. (1998) included the chemical costs by increasing the membrane area required so as to produce a specified average design flow. Gere (1997) found that chemical costs at the San Jose Water Company's 5 mgd MF plant were approximately 30 percent less than for direct filtration (23 mgd). The raw data showed that chemical costs were higher for MF, but this was influenced greatly by the higher cost

of sodium hypochlorite delivery ($1.56/gal for MF vs. $0.59/gal for DF) because of the smaller amounts that were needed at the smaller WTP..

Energy costs depend on the applied pressure, required permeate flux, and operating time between backwashings. These costs can be variable depending on the raw water quality. They have been reported as a percentage of the capital cost, a cost per unit volume, or as total energy consumption. Owen et al. (1995) expressed the energy cost as 9-12 percent of the capital cost depending on the transmembrane pressure used. Dharmappa and Hagare (1999) reported a total electrical consumption around 929,500 kWh/y for a 4 Ml/d (1.1 mgd) facility, which was nearly three times that for a similar sized conventional facility (305,000 kWh/y). The resulting energy cost for this facility, using their assumption of $0.04/kWh, would be $37,180/year or approximately 31 percent of the total annual O&M cost. This cost is very close to the estimate by Gere (1997) for a 5 mgd facility in which energy accounted for 28.6 percent of the annual operating costs. Pickering and Wiesner (1993) reported that the energy costs varied with the element diameter. Depending on the element diameter, energy costs could range from $0.006/$m^3$ to $0.018/$m^3$. Recycling costs vary from $0.0020/$m^3$ for a fiber diameter of 0.2 cm to $0.0042/$m^3$ for a fiber diameter of 0.3 cm.

Membrane lifetimes are estimated by manufactures to range from 3-5 years for polymeric membranes and up to 10 years for ceramic membranes (Owen et al., 1995). The cost of replacing a membrane module, which consists of many membrane elements and thousands of hollow fibers in each element, depends on the module design. Costs are usually expressed per unit area of membrane surface and range from $70.62*(Producer Price Index)/$m^2$ (Dharmappa and Hagare, 1999) to $100/$m^2$ (Sethi and Wiesner, 2000).

Dharmappa and Hagare (1999) included all costs associated with the membrane modules (control valves, membrane replacement, membrane module, etc.) while Sethi and Wiesner (2000) based their estimates on only the UF hollow-fiber membranes as provided by membrane manufacturers. The latter were a function of the membrane material, geometry and process.

Plant operations can be controlled from a computer with Supervisory Control and Data Acquisition (SCADA) software. Thus, labor costs are primarily associated with operation and maintenance of all the membrane modules (Dharmappa and Hagare, 1999). Owen et al. (1995) estimated that the labor costs would be approximately 16-21 percent of the capital cost, while Pickering and Wiesner (1993) estimated 40 percent of the non-construction costs. Dharmappa and Hagare (1999) stated that the labor cost for small water systems was a function of the daily wage rate and the feed flow rate. The estimated costs were included within the overall annualized O&M cost. By using their assumptions (4,000 $m^3$/d and $11/hr labor rate in 1982 dollars) and calculating the labor cost from their cost function, the estimated labor costs were $23,045 or approximately 20 percent of the overall annual cost. Gere (1997) estimated that approximately 32 percent ($97.64/MG) of the total operating cost was associated with labor at the Saratoga 5 mgd MF facility. These costs can be compared to the Montevina 23 mgd facility, where operating costs are 39.7 percent ($23.42/MG) (Gere, 1997). Economies of scale are an important factor. Gere (1997) explained that an MF plant with a capacity of 23 mgd would not require proportionally more operator time, which would result in a significant reduction in unit labor costs. Correspondingly, if a DF plant were scaled down to 5 mgd,

unit labor costs would not be reduced because operators spend most of their time on process control which is independent of plant capacity.

Residuals disposal costs can be highly variable depending on what type of method is available. Types of disposal include sewer discharge, deep well injection, surface discharge, and a specific residuals treatment system (Chellam et al., 1998). Cost estimates for each disposal method were approximated by the cost of energy and chemicals invested (Chellam et al., 1998; Pickering and Wiesner, 1993; Sethi, Wiesner, and Dennis, 1999). The waste disposal costs were estimated at $\$0.05/m^3$ for NF systems and $\$0.01/m^3$ for MF/UF (Chellam et al., 1998). For a 5-mgd MF facility, the residuals management was estimated at 0.2 percent of the operations cost or \$0.618/MG (Gere, 1997). A somewhat higher percentage of operating cost (1.4 percent) and cost (\$0.826/MG) was found for a 23-mgd direct filtration plant..

# METHODOLOGY

## WATER TREATMENT IN CARTHAGE, NC

Raw water is supplied from Nick's Creek and from spring water that enters a 20 MG capacity reservoir. The raw water characteristics are shown in Table 1. Both raw waters have a very low turbidity, low pH, low alkalinity, moderate color, moderate iron, low manganese, and moderate TOC. During storm events, the turbidity has been known to reach as high 100 ntu. The current average demand is about 0.2 mgd. Since the plant only operates 12 hours out of the day, the resulting average demand is 0.4 mgd.

Table 1. Raw Water Characteristics

| Parameter | Value |
|---|---|
| Turbidity | <4 ntu |
| pH | 5.8-6.2 |
| Alkalinity and Hardness | <10 mg/L as $CaCO_3$ |
| Color | 12-125 color units |
| Iron/Manganese | 0.5/<0.01 mg/L |
| TOC | 4 mg/L |

Before the conventional water treatment plant (WTP) was converted to MF, it consisted of an upflow combination flocculator and clarifier (Accelerator) followed by a single dual media filter as shown in Figure 1. Alum was added as a coagulant before the clarifier to remove particles and TOC from the raw water. The pH was adjusted with the addition of lime to a value that was considered optimal for good coagulation, typically in the range of 6.5. Chlorine was added before the filter to reduce biological growth in the

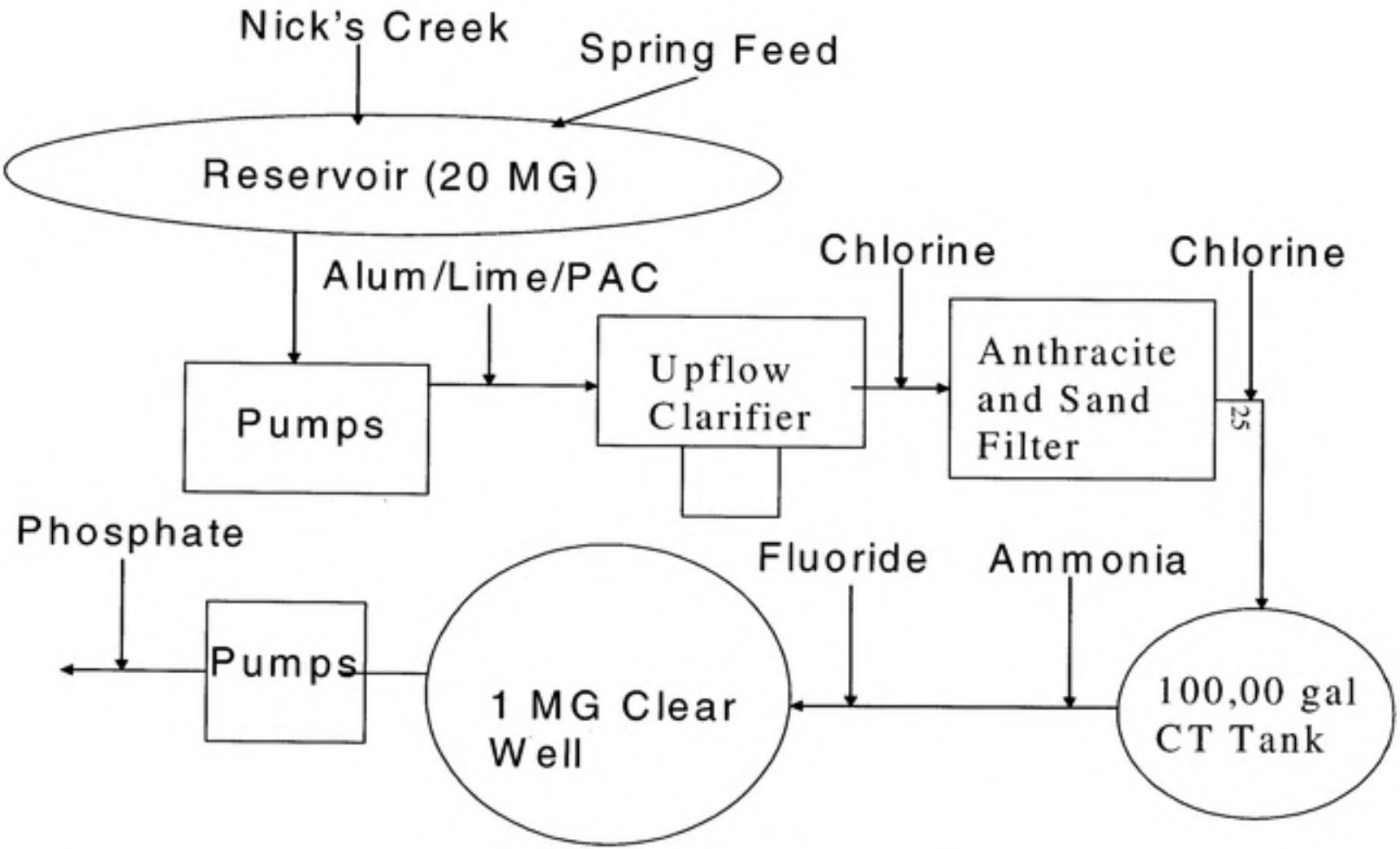



Figure 1. Schematic of Conventional Water Treatment in Carthage, NC prior to May 24, 2002.

filter and again after the filter for disinfection. Filtered water passed into a 100,000 gal tank that is used to meet the CT (chlorine concentration-time) rule for disinfection. After approximately 6 hours of contact time in the tank, ammonia was added to form chloramines and thus to reduce the formation of DBPs.

The cost of a second shift to operate the WTP continuously for 24 hours per day was considered prohibitive. Thus, the Carthage WTP only operates for about 12 hours each day. The chloraminated water is then stored in a 1 MG clear well to feed the distribution system over the 12 hours that the facility is not operating. At the current maximum flow rate through the WTP of 0.4 mgd, the mean residence time of water in the clear well is about 2.5 days and longer during winter months when water demand is lower. While the hydraulics are unknown, the circular shape of this tank would suggest non ideal flow conditions that would approximate complete mixing. Accordingly, the residence time distribution would be rather wide, with 2.5 days in summer months as the average.

The WTP was converted to MF on May 24, 2002. All of the existing treatment facility was abandoned with the exception of the 1-MG clear well. A process flow diagram for the new WTP is given in Figure 2. The MF system consists of two, MEMCOR MF skids, each capable of treating 0.5 mgd. The MF treatment operates at approximately constant permeate flux by continual, automatic regulation of the transmembrane pressure to compensate for membrane surface fouling. Each unit consists of 48 membrane modules and each module contains thousands of hollow fibers with a nominal pore size of 0.2 μm. Water flows from outside to inside and is backwashed every 30 minutes. In this process, raw water passes over the outside surface of the

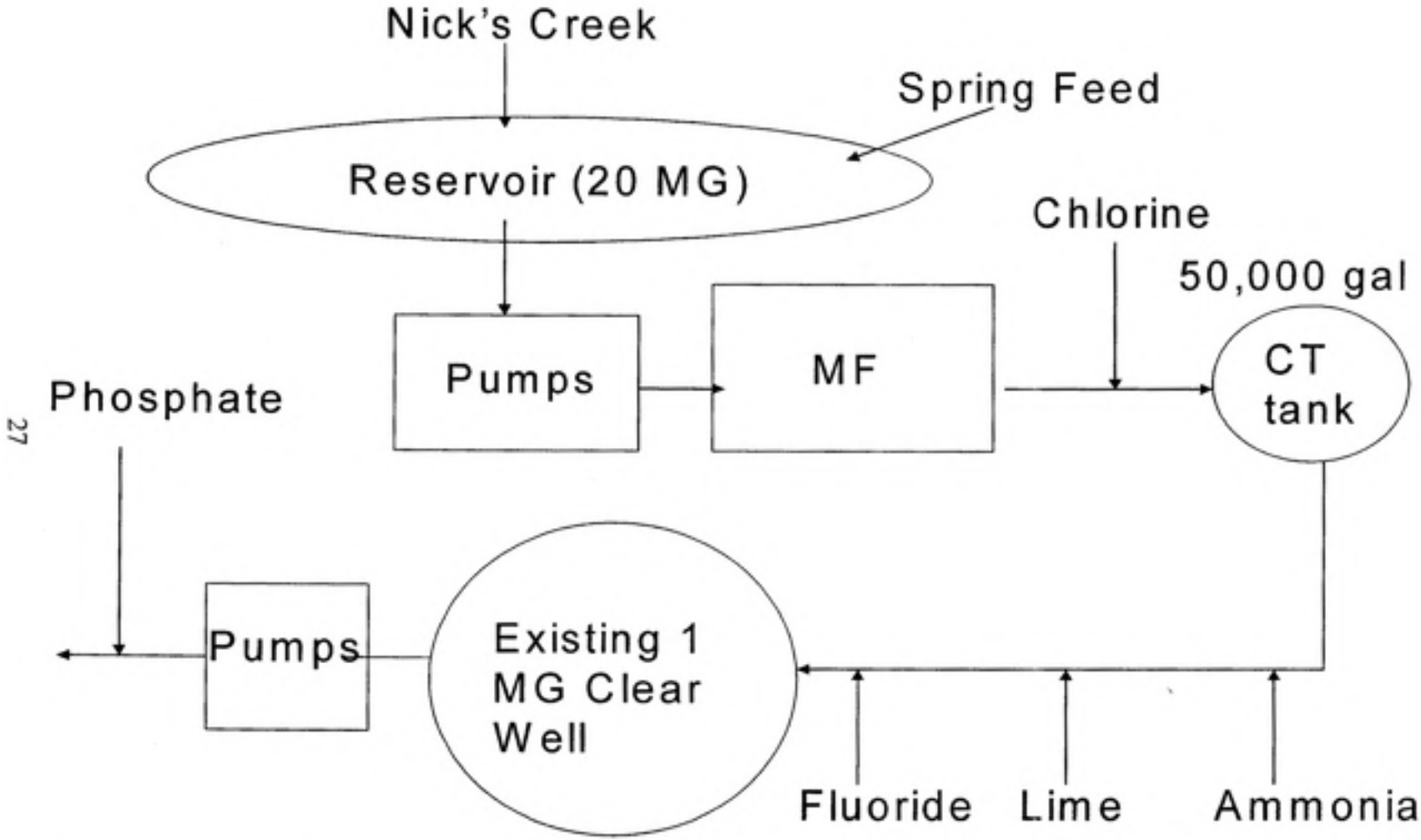



Figure 2. Schematic of New MF Water Treatment Starting May 24, 2002.

hollow fibers while air is pulsed to the inside surface. The air pulsing system operates at a very high pressure (90 psi) to dislodge particles that are on the outside surface. The transmembrane pressure for a clean membrane system to produce permeate at a constant design flow is about 4 psi. The 30-minute backwashing is usually not sufficient to remove all foulants and thus, the transmembrane pressure increases over time in order to deliver the same design flow rate. Once the transmembrane pressure for a MF unit reaches about 15 psi, a clean-in-place (CIP) is performed using citric acid. A CIP begins by taking the MF unit off-line, draining the feedwater from the modules, manually opening the CIP inlet valve, and manually closing the filtration outlet valve. Next, a 2% citric acid and a 2% Membclean EXA2 solution are fed from the chemical tank into the modules. The solution is circulated around both sides of the membrane system for 30 minutes before it goes into a 30 minute soak. After soaking, the cleaning solution is circulated for 60 minutes on the feed side of the membranes. The solution is then purged out of the system with low pressure air (17 psi) back into the CIP tank. Feedwater is reintroduced into the system and the MF unit immediately goes into a backwash cycle to remove any residual cleaning solution. The backwash water (with any residual cleaning solution) is discharged into the town sewer. The cleaning solution in the CIP tank is reused four times and then discharged into the town sewer.

The permeate flow from the two MF units is chlorinated before passing into a new 50,000 gallon CT tank. The residence time in the CT tank is 1.2 hours at design flow rate (1 mgd) but about 3 hours at the current average flow rate of 0.4 mgd. Ammonia is added after the CT tank to form a chloramine residual before the water passes to the 1 MG storage tank.

## DISTRIBUTION SYSTEM

The Carthage distribution system (DS) covers an area of approximately 2.7 square miles with approximately 33.1 miles of pipe. Pipe diameters range from 2 inches to 12 inches. Water leaves the WTP through two lines that enter the same storage tank in the center of Carthage, a distance of approximately 1.2 miles. PVC accounts for most of the pipe material (a length of about 20 miles) in the distribution system, with diameters ranging from 2 to 8 inches. However, there are also about two miles of ductile iron and three miles of asbestos cement with diameters of 6- and 12-inch. A small length (< 1,000 ft) of 6-inch diameter, cast iron remains in one isolated square in the center of town. These lines have not been replaced because they pass under buildings at the site.

Two elevated storage tanks having a combined capacity of 297,000 gallons are located in the town center. These are the only storage tanks in the system. An 8-inch PVC line extends from the center of town about 5 miles to serve only a very few customers. Consequently, the water age is extremely long in this section.

## SAMPLING PROGRAM

Monthly samples were collected from the WTP and the DS from October 2001 to February 2003 with the exception of August 2002 when plant operations were not considered representative of normal conditions due to a combination of the severe drought conditions (which prompted purchase of water from another utility) and difficulty in achieving a disinfectant residual leaving the plant. The WTP samples were taken from the raw water, filtered water, and finished water after the 1 MG storage tank. The filtered water was collected after the dual media filter before the conversion and from the MF permeate after the conversion. The nominal sampling location for raw and

filtered/permeate water samples was at the sample tap in the laboratory. However, additional samples of the permeate water were taken directly from each of the two MF skids to investigate if the bacterial regrowth occurred in the sample line. Performance parameters measured were temperature, turbidity, particle counts, TOC, iron, and heterotrophic plate counts (HPC).

Samples from the DS were taken from October 2001 to January 2003 at five locations. A sixth location was added starting June 2002. The sampling locations were chosen to represent a range of water age and pipe materials. Figure 3 is a map showing the location of the water treatment plant (Site 1) and the sampling sites in the DS (Locations 2 to 7). Station 2 is on a small length (< 1,000 ft) of 6-inch diameter, cast iron that remains in one isolated square in the center of town and a relatively short distance (1.2 miles) away from the water treatment plant. Station 3 is located on an asbestos cement line at a distance of about 1.8 miles from the water treatment plant. Station 4 is approximately 1.1 miles from the WTP on a 6-inch asbestos cement line. Station 5, at about 5 miles, is a relatively long distance away from the WTP. This pipe line is 8-inch PVC. Stations 6 and 7 are both on an 8-inch PVC line. Station 6 is about 4.0 miles from the WTP while Station 7 is approximately 5.5 miles. The water quality parameters measured in the DS were turbidity, particle counts, TOC, iron, HPC, total coliforms, DBPs, (THMs and HAAs) and chlorine residual.

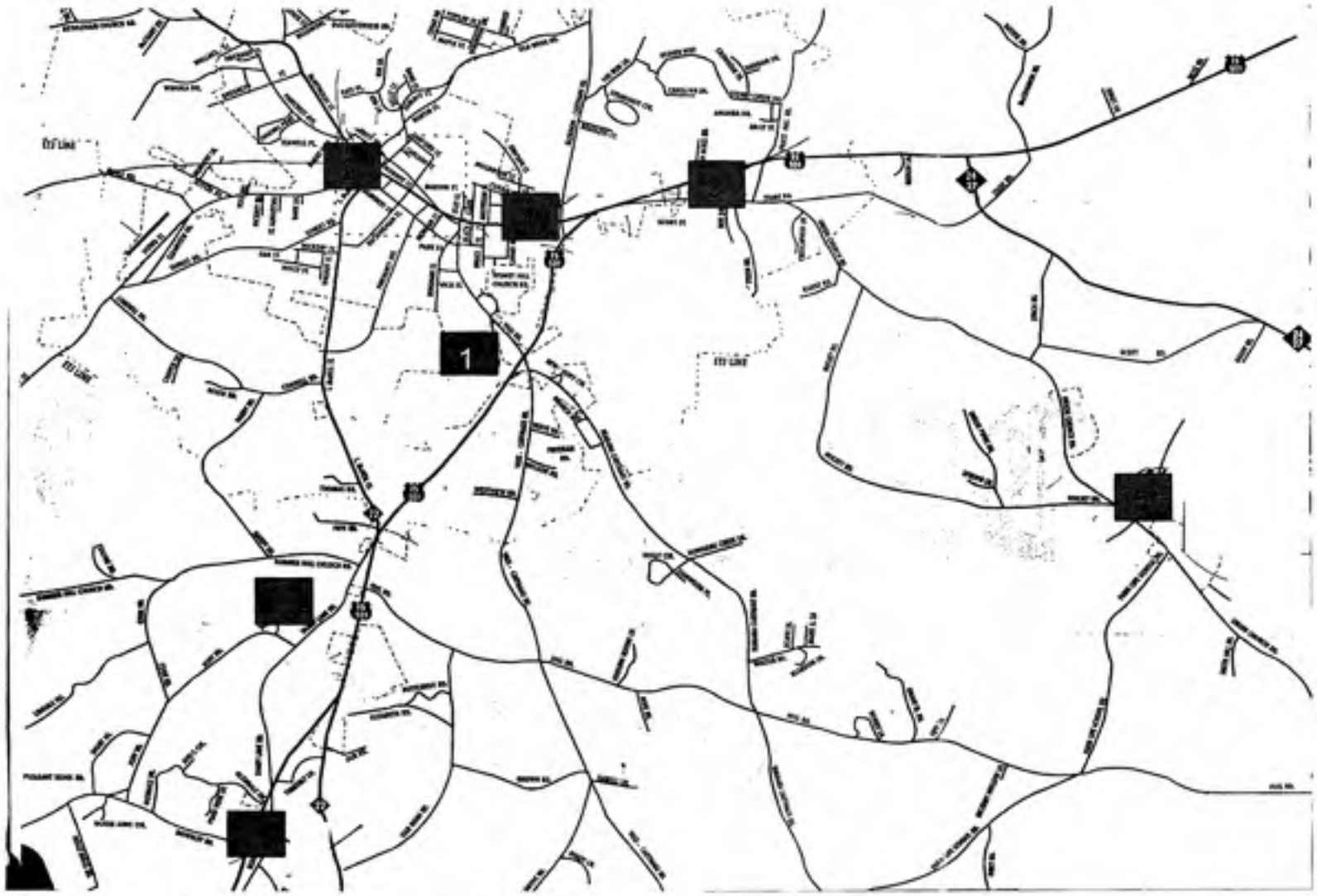



Figure 3. Location of WTP (Station 1) in Carthage, NC and location of sampling stations (2-7) in the DS.

## ANALYTICAL METHODS

The following analytical methods were used:

***Temperature.*** Water temperatures for raw, filtered, finished, and the DS were measured in the field using a Fischer Scientific Accumet Portable pH/mV/Temperature meter (Model AP62). Temperature readings from the meter were calibrated using a thermometer.

***Turbidity.*** Measurements of turbidity were taken using a Hach Ratio Turbidimeter (Model 18900-69). Samples were collected from the raw, filtered, finished, and DS using a 500 mL bottle and were stored in ice until the measurement could be made. The turbidimeter was calibrated using standards of known turbidity.

***Particle Counts.*** Raw, filtered, finished and DS samples were taken and stored in a 500 mL bottle and stored in ice until a measurement could be taken. Particles were counted in a range of 2-100µm using a MetOne WGS 260 Water Grab Sampler. Standard Method 2560D was used.

***TOC.*** Duplicate samples were taken from the raw, filtered, finished, and the DS using 40 mL glass tubes. A Shimadzu TOC 5000 analyzer with an ASI 5000 auto-sampler (Shimadzu Corporation, Columbia, MD) was used to measure total organic carbon. Analysis for TOC content was conducted according to the procedures written in Standard Method 5310B with one revision; 2-N hydrochloric acid rather than phosphoric acid was used to acidify the samples. This substitution was made because the Shimadzu analyzer is sensitive to excess phosphoric acid.

***Iron.*** Total iron was measured using the Ferro Ver Hach Method 8008 on a Hach DR2000 spectrophotometer. Samples from the raw, filtered, finished, and DS were measured immediately on-site.

***HPC.*** HPC was determined by Standard Method 9215B using R2A media incubated for 7 days at 28°C. R2A media was prepared using dehydrated media (Becton Dickinson) and reagent-grade water. Once the media was completely dissolved, 12 mL portions were dispensed into 30 mL glass test tubes using a plastic disposable 25 mL pipette. After dispensing, these tubes were sterilized in an autoclave for 15 minutes at 121°C. After autoclaving, the tubes were placed in a 45°C water bath (Model 66850, Precision Scientific) until used. Triplicate samples were serially diluted using peptone water prepared according to Method 9050C of *Standard Methods* (1998). The peptone dilution water was prepared by completely dissolving peptone crystals (Bacto™) in reagent-grade water to form a 0.1% solution. Next, the solution was dispensed in 9 mL quantities into 20 mL test tubes. Finally, the tubes were sterilized in an autoclave at 121°C and cooled to room temperature. Samples from the DS, permeate (combined and from the individual skids), and finished water were collected in sterilized 250 mL Nalgene bottles. Prior to sterilization, 1 mL of sodium thiosulfate ($Na_2S_2O_3$) was added to the sample bottles to remove any residual disinfectant from the sample water. Once the samples were collected, they were stored in ice until analysis. Samples were diluted from $10^0$ to $10^4$ depending on their anticipated concentration. All plates were inverted and wrapped in plastic bags during incubation in a Precision Scientific incubator at 28°C for 7 days, according to the recommendations in Section 9215A.7 of *Standard Methods* (1998).

Colonies were counted manually using a Darkfield Quebec Colony Counter (American Optical Company). The dilution containing 30 to 300 colonies was selected for counting.

For QA/QC, four controls were tested in duplicate:

1) Air Blank: Media was allowed to be exposed to the atmosphere for 15 minutes to test for possible air contamination.
2) Media Blank: Sterile R2A media was dispensed into a Petri dish and incubated to test for media contamination.
3) Dilution Blank: One mL of sterile peptone dilution water was incubated to test for dilution water contamination.
4) Autoclave Blank: Prior to sterilization, one tube of peptone dilution water was spiked with one mL of primary wastewater effluent. After autoclaving, one mL was incubated to test the ability of the autoclave to sterilize properly.

***Total Coliforms.*** Total coliforms were determined by use of the membrane filter method (SM 9222B) using m-Endo Broth (Difco™), 95% ethanol alcohol and reagent grade water, prepared according to instruction. Glass funnels in sufficient quantity for all samples were sterilized in an autoclave at 121°C for 15 minutes. At least 100 mL of sample was filtered through the membrane filters followed by a sterile reagent water rinse. All samples had duplicates. Filters were rolled onto a sterile pad saturated with the medium in a sterile petri dish. Once plated, the filters were incubated inverted at 35°C for 22 to 24 hours. Any positive colonies were verified using lauryl tryptose broth for fermentation and gas production and confirmed in brilliant green bile broth. To assure broth integrity, a positive blank was prepared by spiking a previously sterilized reagent

water tube with primary wastewater effluent. This was treated as a sample according to the above procedure.

***Total Chlorine Residual.*** Total Chlorine was measured using Hach method 8167 on a Hach DR2000 spectrophotometer. Total chlorine for all intents and purposes is equal to combined chlorine for periods when the WTP was delivering chloramines. When the WTP was delivering free chlorine or when a switch had been made recently between chloramines and free chlorine, it was also necessary to measure free chlorine using Hach method 8021. Measurements were recorded on-site from the finished and DS water.

## PLANT OPERATION RECORDS

Daily plant operational data are collected by the plant personnel every 4 hours and are reported monthly to the State of North Carolina. These data are also used to adjust plant operations. In addition, data were collected on chemical usage and on MF operations from the SCADA system. The SCADA data are from continuous monitoring of particle counts in the MF permeate water and of transmembrane pressure across the MF.

## ANALYSIS OF WATER QUALITY DATA

The monthly samples collected at the WTP and in the DS, before and after the conversion to MF, were plotted over the period of the study. All data points (each water quality parameter at each station) were included so as represent all of the raw data obtained. A student t-test could not be used because fewer than 30 samples had been collected for each sampling location over the course of the study and thus, a Gaussian distribution could not be assumed (Rasmussen, 1992; Devore, 1991).. A mean and standard deviation were calculated for each water quality parameter at each sampling

location. In addition, a box and whisker plot was used to represent the data.. A box and whisker plot is useful because it depicts both the range of data and its variability (Rasmussen, 1992). The following five statistical measures are included: median; first (25%) and third quartiles (75%), maximum, and minimum. The box portion includes the median, and the first and third quartiles, while the whiskers extend out to the maximum and minimum values.

The monthly plant records, which nominally consisted of four daily measurements, along with the SCADA data produced much more data than the monthly sampling performed by UNC. These data were sufficient to conduct a student t-test to determine if the water quality of the finished water (i.e., the water leaving the WTP) had improved after the conversion to MF. The raw water quality was also tested statistically to determine if it remained the same for both test periods so that conclusions about the differences in the finished water could be better interpreted.

## OPERATING COST ESTIMATION

Daily backwash usage, chemical consumption, CIP & disposal, and power consumption were all compared before and after the MF conversion. Operating costs per 1000 gallons of finished water produced were developed from these data so as compare the conventional WTP with the new MF plant. Changes in the distribution of operating costs over the study period were also compared before and after the MF conversion from previously calculated chemical consumption, power consumption, daily backwash usage, and CIP & disposal costs in addition to obtaining an overall budget and payroll from the Town of Carthage. In addition, other costs such as membrane/filter media replacement, capital recovery, and labor costs were compared before and after the conversion to MF.

Finally, a hypothetical comparison between a new 1.0 mgd conventional treatment facility and the new 1.0 mgd MF facility was provided to compare the total costs per 1000 gallons of finished water produced between the two facilities operating at 0.75 mgd.

## RESULTS AND DISCUSSION

### GENERAL ASSESSMENT OF WTP PERFORMANCE

Table 2 provides a summary of the Carthage WTP performance as measured by average values of temperature, turbidity, particle counts and TOC in the raw water, the filtered water (either granular media or MF permeate) and the finished water leaving the plant from the clear well. These data were collected from monthly grab sampling by the University of North Carolina (UNC) (14 samples) and from four-times daily grab sampling by the Carthage WTP personnel (455 samples) before and after the conversion to MF. Particle counts and TOC are not measured by the Carthage WTP personnel. The following sections will discuss these results in more detail.

Table 2. Average Water Treatment Plant Performance Before (October 2001 to May 2002) and After (June 2002 to January 2003) the Conversion to MF from Data Collected by UNC and the Carthage WTP

| Parameter | UNC | | Carthage WTP | |
|---|---|---|---|---|
| | Before Conversion | After Conversion | Before Conversion | After Conversion |
| Raw Water Temp., °C | 13.6 | 18.7 | 13.7 | 19.7 |
| Raw Water Turbidity, ntu | 5.3 | 3.3 | 9.3 | 2.7 |
| Raw Water Particle Count, #/mL | 9,323 | 7,399 | NA | NA |
| Raw Water TOC, mg/L | 4.45 | 4.45 | NA | NA |
| Filtered Water Turbidity, ntu | 0.83 | 0.06 | NA | NA |
| Filtered Water Particle Count, ntu | 2782 | 179 | NA | NA |
| Filtered Water TOC, mg/L | 2.4 | 3.9 | NA | NA |
| Finished Water Turbidity, ntu | 0.603 | 0.380 | 0.254 | 0.097 |
| Finished Water Particle Count, #/mL | 670 | 640 | NA | NA |
| Finished Water TOC, mg/L | 3.21 | 3.92 | NA | NA |

NA = not available

**Variability in Temperature During the Study**

Statistical analysis of both the UNC and the Carthage WTP data indicates that the raw water temperature was significantly higher ($P<0.005$) after the conversion to MF. This was expected because the measurements after the conversion were dominated by the warmers months of the year whereas those before the conversion were dominated by the colder months. The variability in temperature is shown graphically in Figure 4.

**Particle Removal as Measured by Turbidity**

A statistical analysis showed that the UNC data and the Carthage WTP data for turbidity were not statistically different ($p>0.005$) for either the period before or after the conversion to MF. The more important statistical analysis is the comparison of raw water turbidity before and after the conversion as measured by the larger data base collected by the Carthage WTP (four times daily measurements compared to once monthly for UNC).

A statistical analysis of the four-times daily measurements of turbidity by the Carthage WTP showed that standard deviation was very large (13.7 ntu) on the average raw water turbidity (9.28 ntu) before conversion to MF. In strong contrast, the standard deviation was much lower (2.54 ntu) for a much lower average turbidity (2.74 ntu) after conversion to MF. The variability in raw water turbidity is shown in Figure 5. The decrease in raw water turbidity for the period of study after the conversion to MF was statistically significant ($p<0.005$).

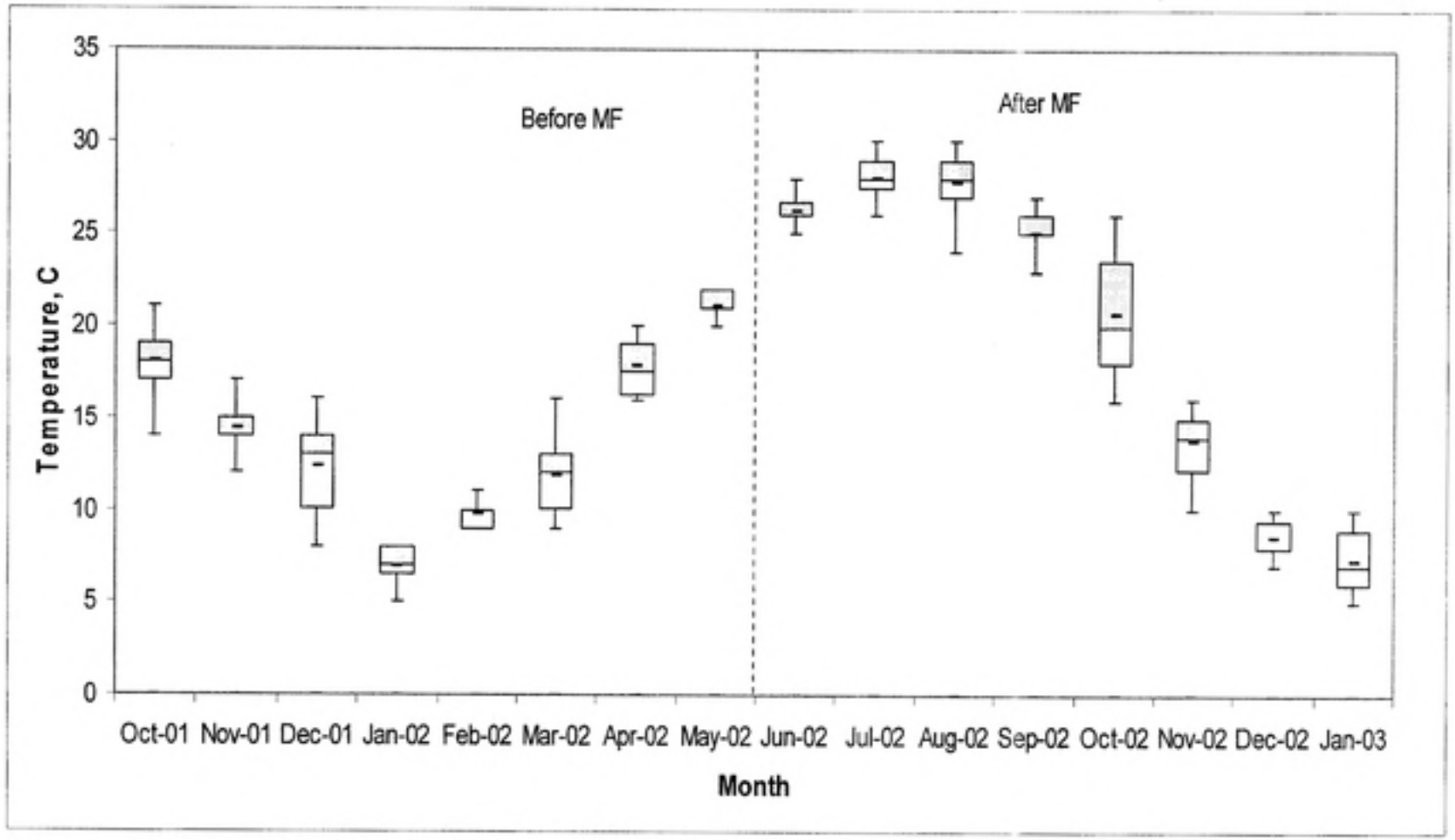



Figure 4. Box and whisker plot of raw water temperature data collected by the Carthage WTP

A detailed analysis of high turbidity values (>10 ntu) showed an occurrence rate of 22 percent (51 days) before the conversion while the rate was only 1.2 percent (3 days) after the conversion. The occurrence rate of low turbidity values (<5 ntu) was 52 percent before the conversion but 91 percent after the conversion. Higher raw water turbidity before the conversion can be explained by numerous large rainfall events during the months of October 2001, January 2002 and February 2002 that may have washed suspended particles into the raw water reservoir. Lower raw water turbidity would be expected after the conversion because these months were dominated by drought conditions. The importance of this statistical finding is in drawing conclusions about whether the conversion to MF led to an improvement in filtered- or finished water quality. To make a fair assessment, the raw water turbidity would have to be identical for the two period of study. Unfortunately, actual plant conditions made this impossible.

The finished water turbidity data as collected by the Carthage WTP are summarized graphically in Figure 6. Inspection shows qualitatively that the average finished water turbidity was much lower after the conversion to MF and, as important, the variability for any month of data is far less after the conversion. Statistical analysis showed that the average values of turbidity after the conversion were 0.15 ntu lower than before the conversion and this difference is statistically significant ($p<0.005$). In addition, the standard deviation of finished water turbidity values after conversion is much lower than before (0.02 vs. 0.14 ntu) showing that MF is a more reliable method to remove particles than granular media filtration as had been operated at the Carthage WTP. Nonetheless, some of the variability in finished water turbidity after the conversion could be explained by difficulties in the lime feeding system that is used for

pH adjustment. Plant operators have reported that if the lime is not fully dissolved, small particles may remain in the finished water, resulting in an increase in the turbidity

The variation in percent removal of turbidity based on the difference between raw water and finished water values of turbidity collected by the Carthage WTP is summarized in Figure 7. In general, the removals are somewhat higher after the conversion to MF. A statistical summary, however, indicates that if all the removal data are pooled for the two periods, the average percent removal only increased from 93.3 to 94.9. The box and whisker plots show qualitatively that the percent removal was less variable after the conversion. In fact, comparisons of percent removal are of limited use to assess whether MF produced a lower turbidity than granular media filtration. The best indicator of performance of filtration processes is the absolute value of the finished water parameter of interest as was given in Figure 6. The process should be robust enough such that a low value of turbidity is obtained regardless of the variations in raw water values. As noted earlier, the raw water turbidity values were higher before the conversion. Therefore, the percent removals would need to be higher for this period in order to maintain the same finished water quality.

The monthly turbidity values measured in the raw water, the filtered water (conventional filter effluent and MF permeate) and the finished water are given in Figure 8. The finished water turbidity, based on measurements made by UNC, was lower after the conversion to MF. However, the mean values and standard deviations of seven monthly values before the conversion and seven monthly values after the conversion were all higher than measured by the Carthage WTP from four-times daily grab sampling. While the Carthage WTP data showed a reduction in turbidity of 0.15 ntu after

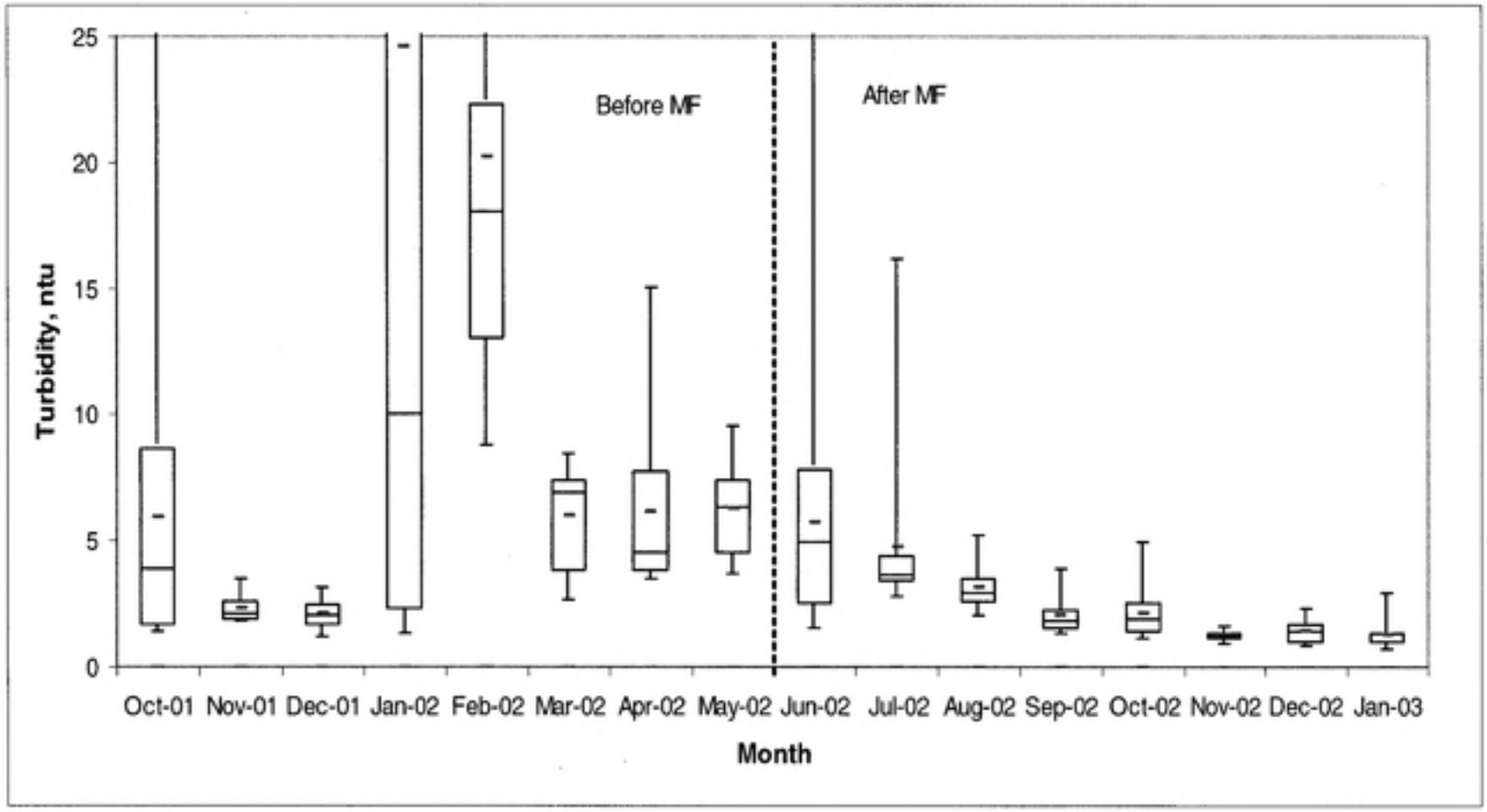



Figure 5. Box and whisker plot of raw water turbidity data collected by the Carthage WTP

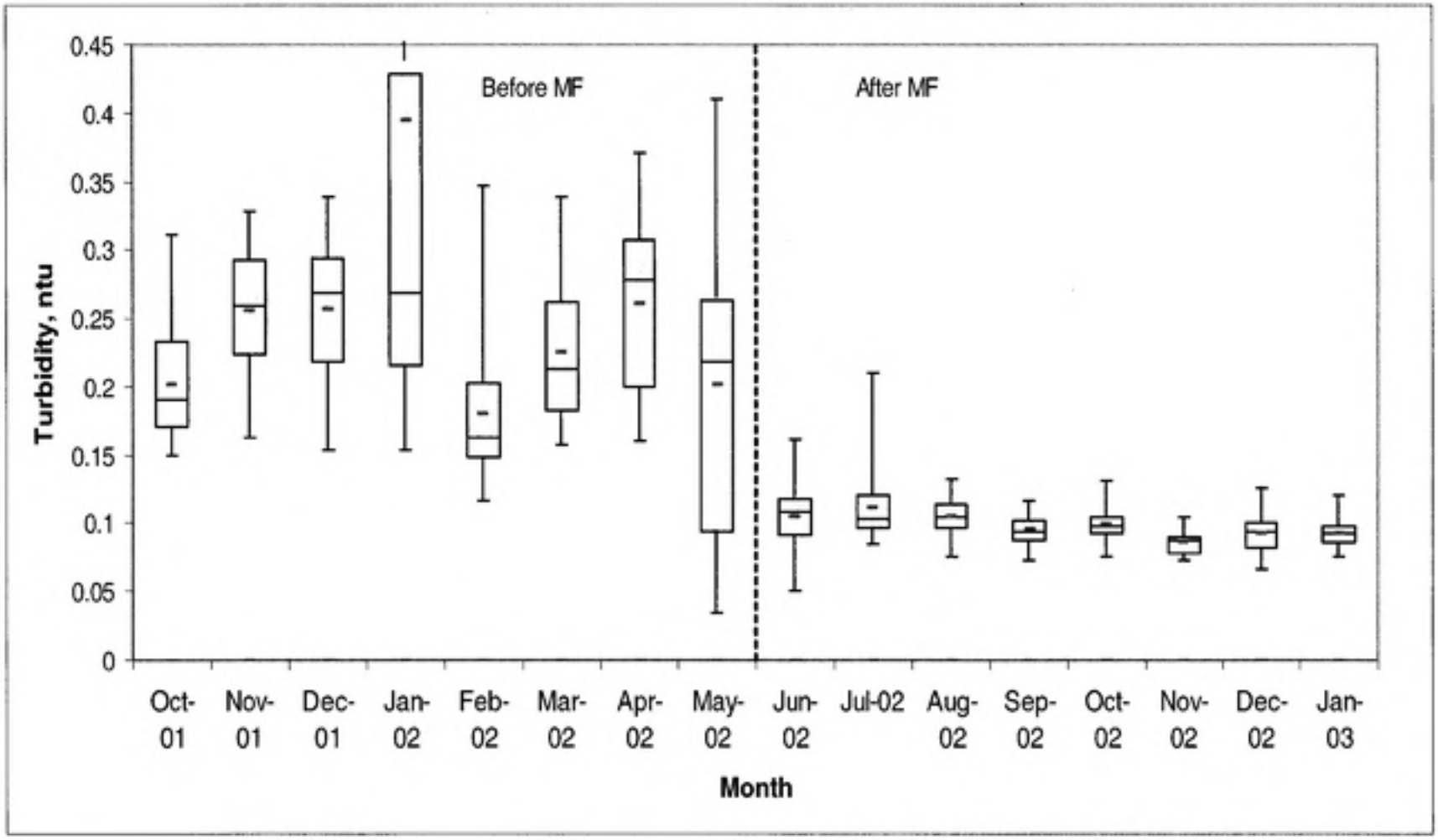



Figure 6. Box and whisker plot of finished water turbidity data collected by the Carthage WTP

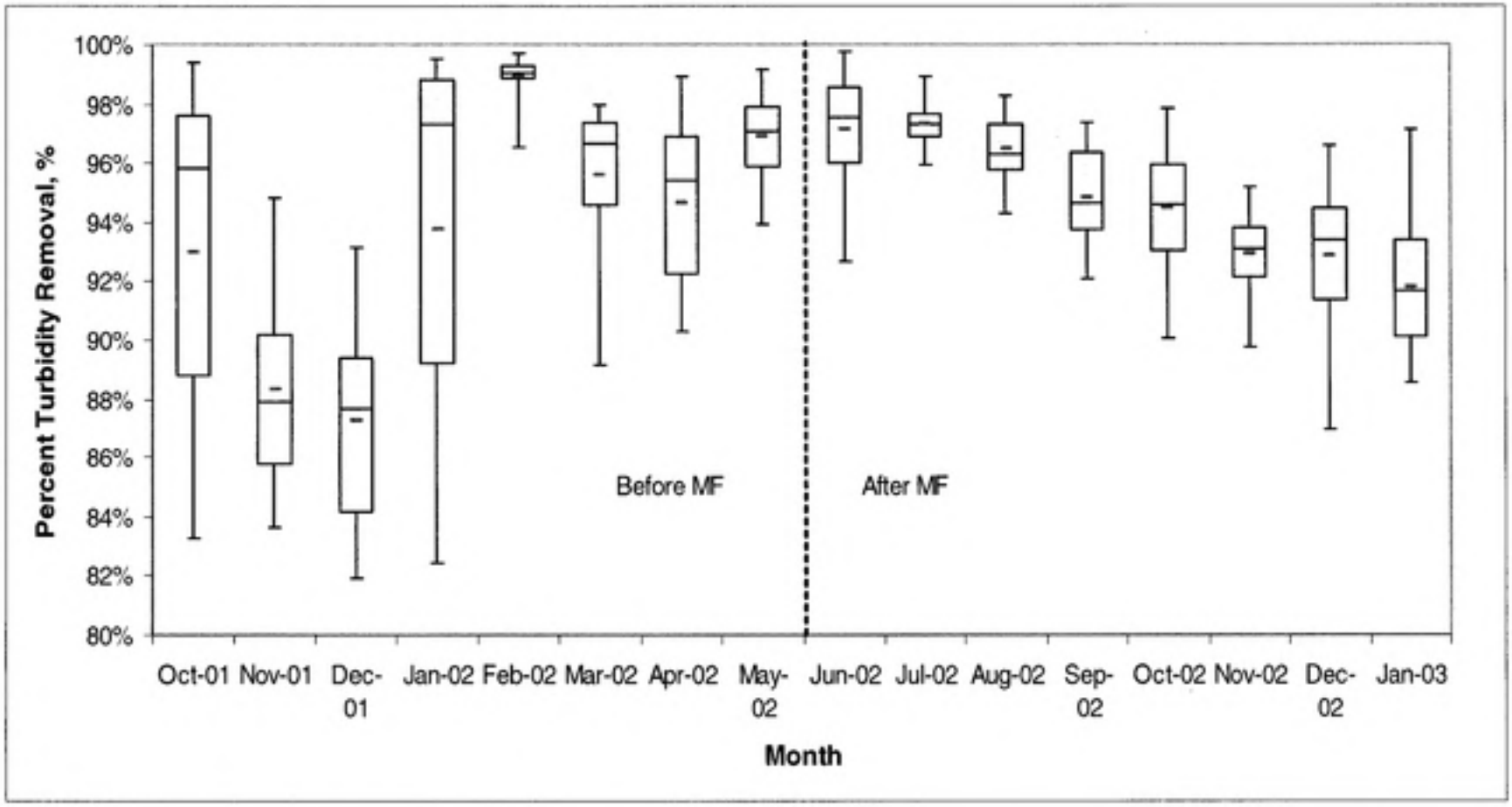



Figure 7. Box and whisker plot of percent removal of turbidity based on raw and finished water samples collected by the Carthage WTP

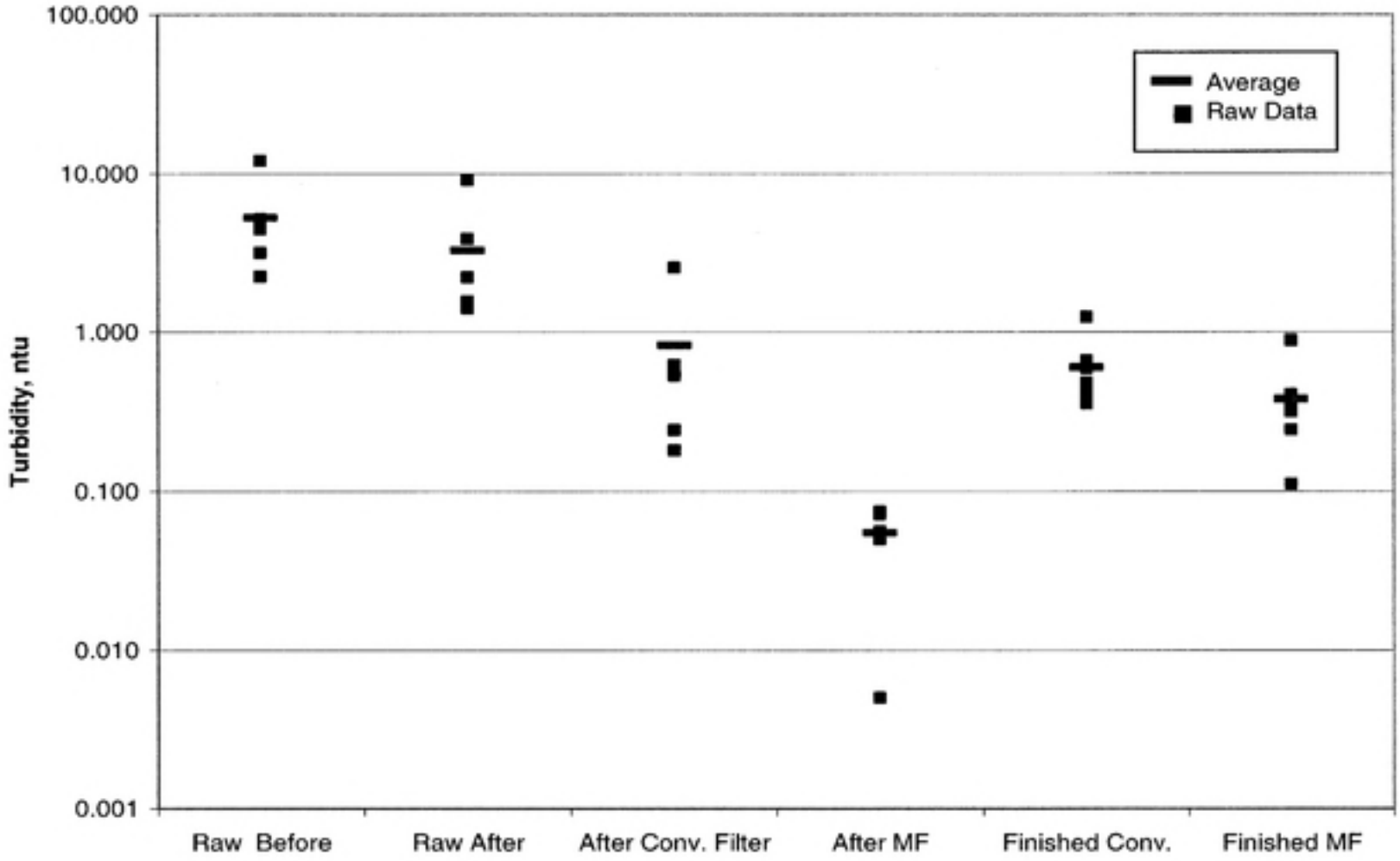



Figure 8. Raw, filtered, and finished turbidity data collected by, UNC before and after the conversion to MF

the conversion, the UNC data showed a reduction of 0.22 ntu. The standard deviation from the Carthage WTP data was reduced from 0.14 to 0.02 ntu whereas the reduction from the UNC data was from 0.3 to 0.24 ntu after the conversion. All but one measurement of finished water turbidity in the UNC database was higher by about 0.3 ntu than the measurements made by the Carthage WTP personnel.

While the finished water turbidity values is important because it describes the water quality entering the distribution system, a more fair comparison of performance change would be between the filtered water from the granular media filter (i.e., conventional filtration) and the permeate from the MF. As Figure 8 indicates, the average filtered water turbidity in the MF permeate was 0.8 ntu lower than in the effluent of the granular media filter. The standard deviation was also less (0.982 for granular media filtration vs. 0.024 ntu for MF). As was noted for the finished water comparison, MF treatment provides a much more stable product that granular media filtration as practiced at the Carthage WTP. These results agree with the findings of other authors (Karimi et al., 1999; Karimi et al., 2002; Taylor et al., 1999; Coccagna et al., 2001).

The difference between filtered and finished water turbidity merits further discussion. During conventional filtration, the finished water turbidity was always about 0.23 ntu lower than filtered water turbidity. However, after the conversion to MF, the reverse was true, i.e., the finished water turbidity was always about 0.32 ntu higher than the filtered (permeate) water. The percent removals of turbidity listed in Table 3 are based on using the results obtained from sampling of the filtered water and those of the finished water. The improvement in turbidity removal caused the conversion to MF becomes more apparent when using the filtered water results. The carryover of alum floc

into the clearwell may offer an explanation. Evidence of carryover was provided during construction of the new MF facility when a diving team inspected the clearwell. Approximately five feet of sediment was found at the bottom of the clearwell. The MF facility was put into service after the dive team removed the sediment layer by a vacuum process. The cleaning operation could have resuspended some of the alum floc which would cause the turbidity to be higher in the finished water after MF

Table 3. Comparison of Percent Removal Calculations of Turbidity Based on Average Percent Removal of Turbidity Based on Raw Water Turbidity and Use of Either Average Filtered Water Turbidity or Average Finished Water Turbidity (i.e., After Clearwell)

| | Before Conversion | After Conversion |
|---|---|---|
| Average Filtered Water Turbidity | 79.2 | 97.4 |
| Average Finished Water Turbidity | 84.9 | 81.7 |

**Particle Removal as Measured by Particle Counts**

The particle counts in the monthly samples of filtered and finished water that was measured by UNC are given in Figure 9. The average particle count in MF permeate was 179 particles per mL whereas the average in the conventional filter effluent was 2,782 particle per mL. These data confirm the better performance of MF with a more sensitive measure than turbidity. As noted for turbidity results, the particle count data also show a reduction from filtered to finished water before the conversion to MF but an increase after the conversion. Whether particles are generated from within the clearwell after the conversion remains an open question. The percent removal of particles was calculated based on the difference between raw- and filtered water samples and between raw- and finished water samples. Both are presented in Figure 10. The average particle removal was 98 percent for MF and only 71percent for conventional filtration. Percent removal is not a fair measurement of comparison, however, given that the raw water values of

particles were lower after the conversion. If the single MF permeate sample with the extremely high particle count (779 per mL) is ignored, the average removal increases from 98 to 99.2 , to about a 2-log removal. This value is still less than the 2.5-3.5 log removals reported by Karimi et al. (1999), Coccagna et al. (2001) and Taylor et al. (1999). However, if performance is based instead on achieving a specific particle count in the permeate, the average value achieved in this study (79/mL) agrees with reports by AWWA (1999) and Mallevialle, Odendaal, and Wiesner (1996) where MF was stated to produce a permeate with fewer than 100 particles per mL. The literature on MF also suggests that air bubbles that are generated during the backwash cycle (every 30 minutes) could be counted as particles (Taylor et al. 1999 and Kothari and Taylor 1998). Given that the data reported in Figure 7 were for grab samples that were returned to the UNC laboratory for analysis, it would seem unlikely that air bubbles could reside for several hours in the sample.

**Removal of TOC**

The TOC concentration before and after MF is illustrated in Figure 11. As expected, conventional media filtration with the aid of chemical coagulation removes more TOC than the new MF system that does not include coagulation or other pretreatment. The average TOC removal through conventional filtration with alum coagulation was 30 percent whereas it was only 11 percent through MF. The fact that some removal was obtained can be explained by that portion of TOC that is in the particulate form. The percent removal of TOC for both MF and conventional treatment were similar to those found in previous studies. Mallevialle, Odendaal, and Wiesner (1996) stated that conventional filtration with the aid of coagulation can remove TOC in

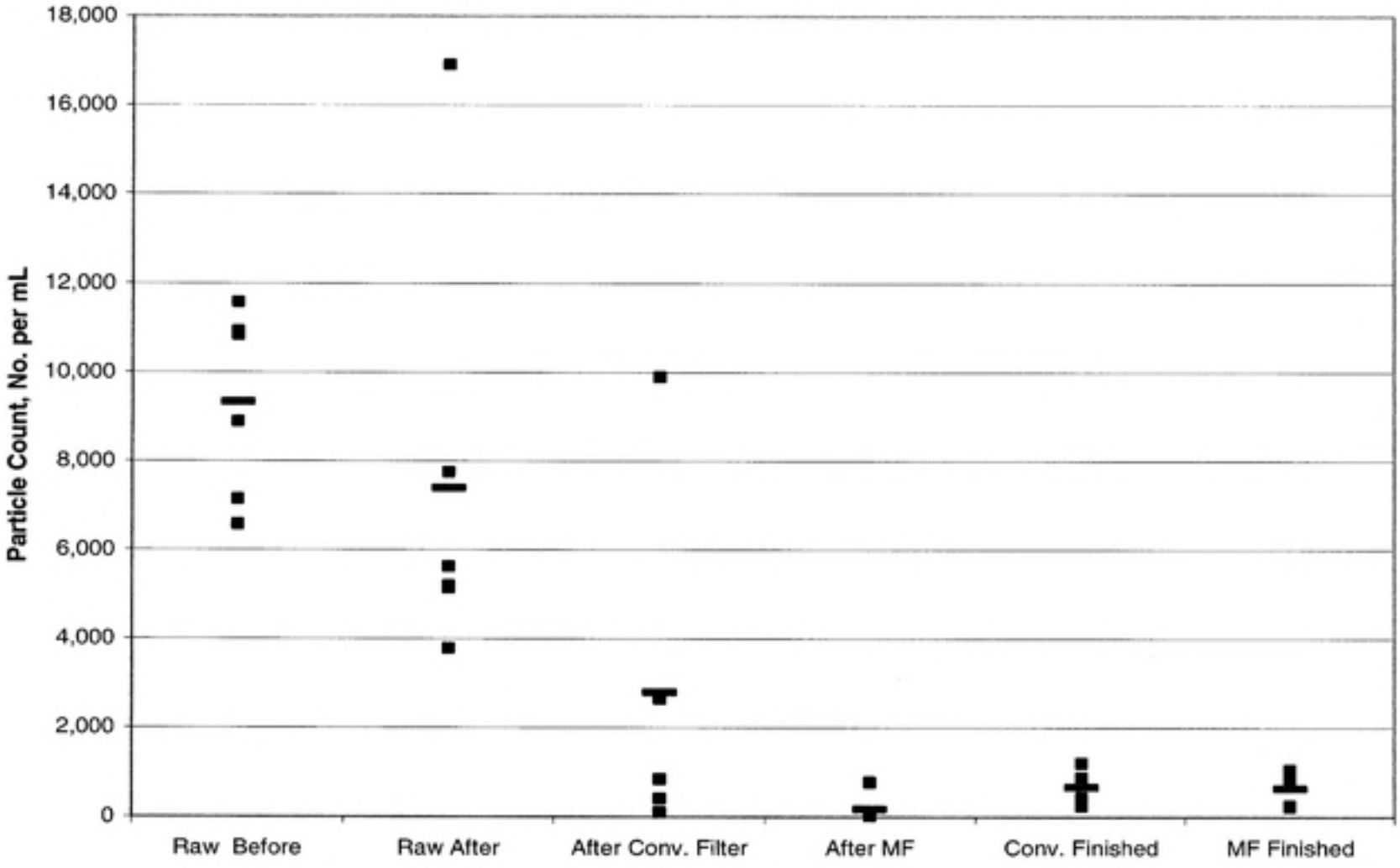



Figure 9. Particle counts measured in monthly samples of raw, filtered and finished water by UNC

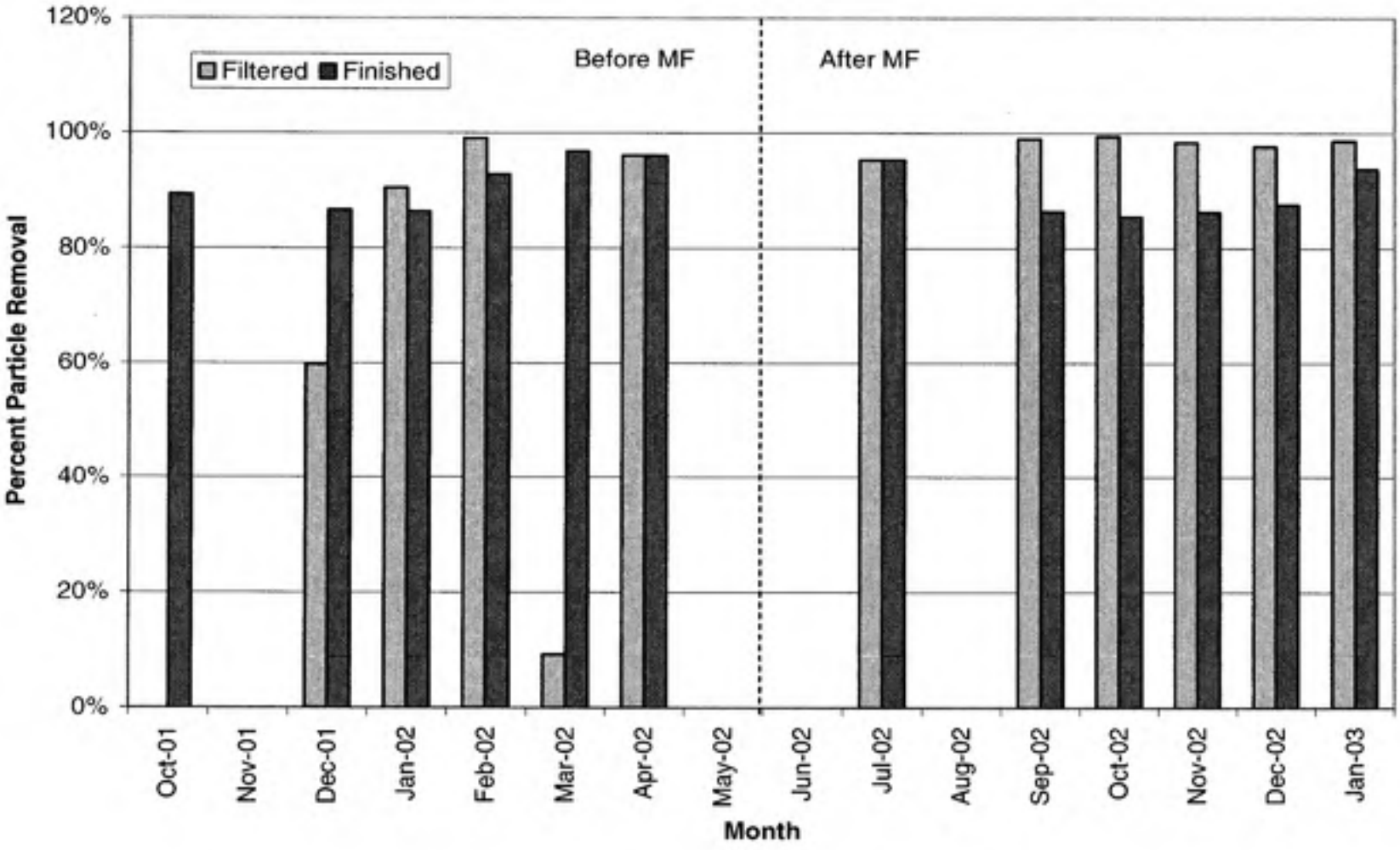



Figure 10. Comparison of percent removal of particles based on data collected on filtered and finished water samples by UNC

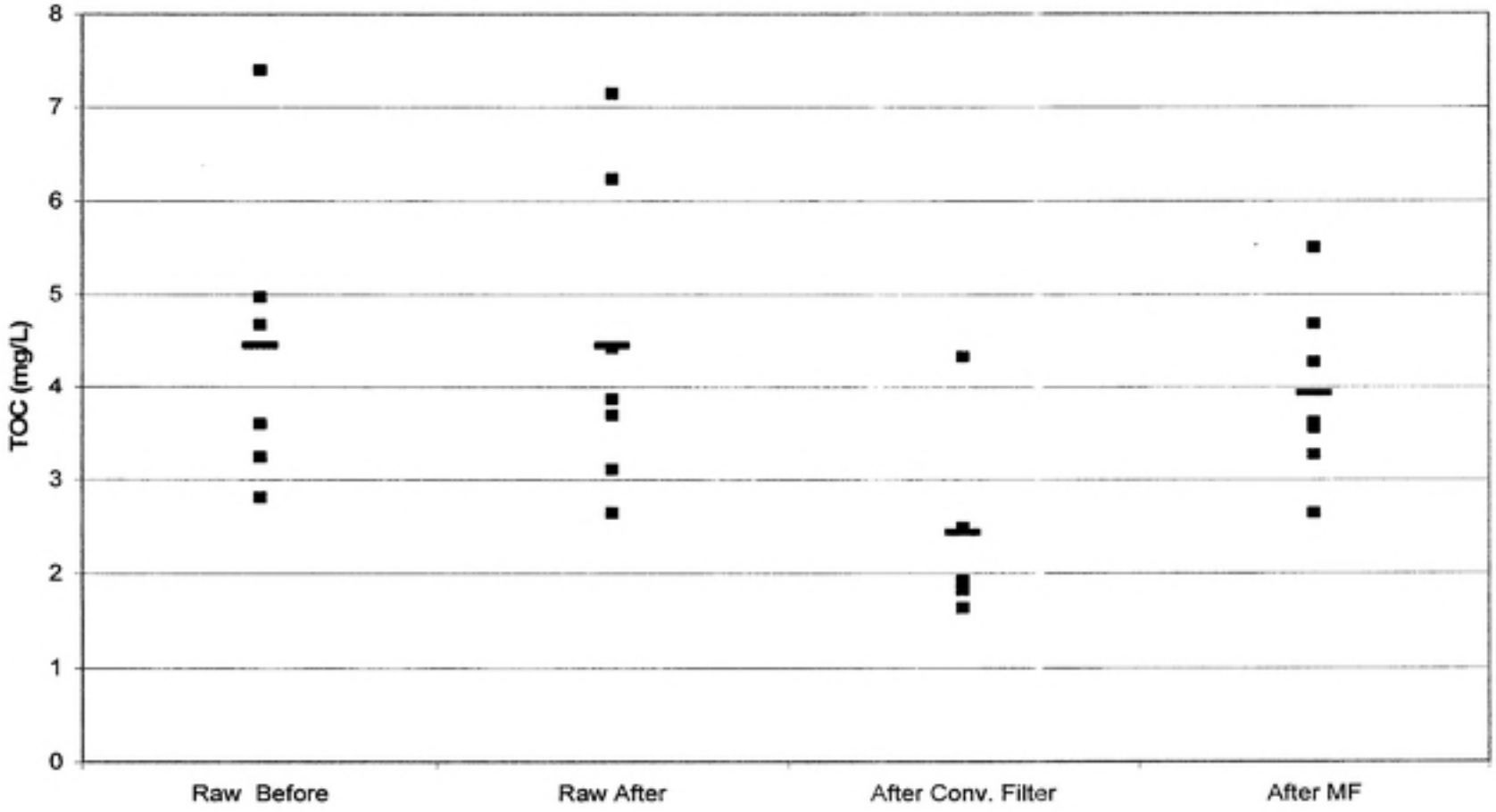



Figure 11. Raw and filtered TOC data collected by UNC before and after the conversion to MF.

the range of 30-60 percent while Karimi et al. (1999) reported an average of 10 percent TOC removal by MF.

## MF PERFORMANCE AS MEASURED BY SCADA SYSTEM

Data are recorded hourly by the SCADA system for each of the two membrane skids and held for 60 days. The data are then lost if plant personnel do not specifically save them. Hourly SCADA data were retrieved for the month of July 2002 and months of November 2002 through early January (1st to the 12th) 2003. Data for August, September, and October had been lost because the operators failed to save these. Figure 12 show turbidity and particle count data for the MF permeate as obtained from the SCADA system for July 2002 while Figure 13 shows these same parameters for November 2002 through early January 2003. Data taken at 8 am and 10 am were chosen to show particle removal efficiency at plant startup and any change that occurred during the first few hours.

The turbidity of MF permeate in July 2002 was consistently below 0.1 ntu with only about 12 percent of the values (12 points) being higher. Turbidity values ranged from 0.026 to 0.143 ntu and the average was 0.046 ntu. These results show that MF is able to produce a consistent, high quality water that meets MCLs listed in the IESWTR (USEPA, 1998a) and LT2EWSTR (USEPA, 2000a). Particle counts in the permeate ranged from 3.8 to 306 particles/mL and the average was 36.7/mL. As was observed for the turbidity data, the particle count data were dominated by low values, with only about 10 percent being higher than 100 particles/mL.

Figure 13 shows results that particle counts were lower in the MF permeate in the later months than in July. The average value was 25.7/mL compared to 36.7/mL in July.

The standard deviation was also lower (21.2/mL compared to 66.6/mL in July). The improvement in permeate could be attributed to the experience gained by plant personnel in reducing the fouling of membranes by optimization of the time between backwashes and the frequency of CIPs. The SCADA system also includes the TMP and the permeate flux from which membrane resistance, $[R_c(t) + R_m]$, is calculated according to Equation 2 As fouling occurs, the membrane resistance increases. The next CIP is determined by the point in time where the TMP exceeds a selected operational value which is typically in the range of 15 psi. The membrane resistance value, which is determined by the SCADA system at 8 am, is shown for each of the two skids in Figure 14 for July 2002 and in Figure 15 for November 2002 through early January 2003.

The resistance in both Figures 14 and 15 increases gradually to about 12-14/m before a CIP is performed after which the resistance then decreases sharply to approximately 4/m. These results demonstrate clearly the effectiveness of the CIP in removal of foulants. In effect, the decrease in resistance corresponds to the ability to once again achieve the desired flux at a low TMP.

Table 4 shows the dates on which the CIP was performed on each skid and the time interval between cleanings, based on plant startup on May 24$^{th}$, 2002. The average time between cleanings was estimated to be 79 days or 1904 hours of operation. These results show that the CIP frequency is fairly constant. While the first CIP on Skid No. 1 took place after only 49 days, the CIP on the Skid No. 2 was not needed until 88 days, which is difficult to explain given that both started on the same date. However, the time interval to the second CIP on both skids was about 97 days. Although the data is fairly

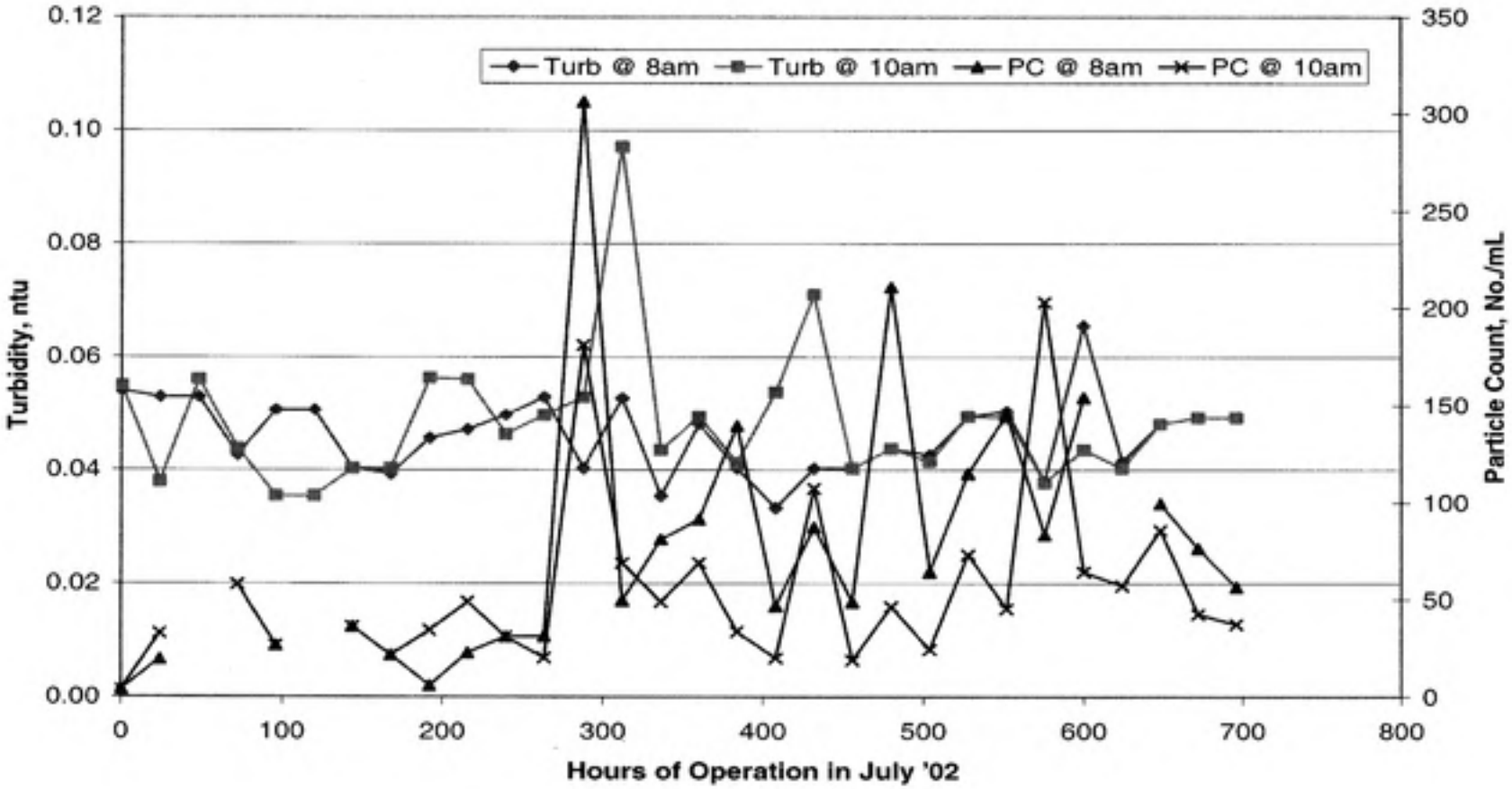



Figure 12. Hourly turbidity and particle count data for MF permeate of two skids taken from the SCADA system for July 2002

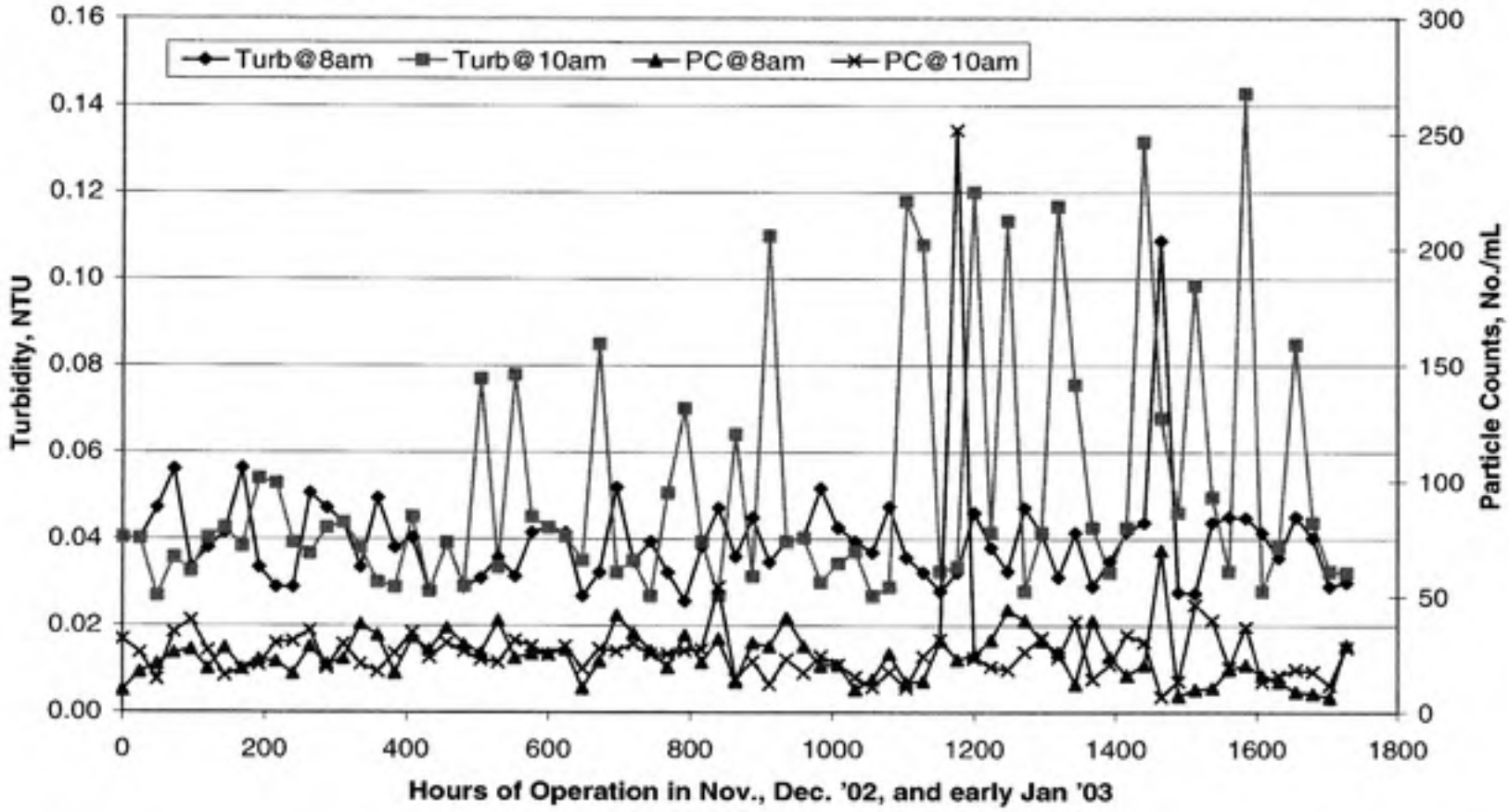



Figure 13. Hourly turbidity and particle count data for MF permeate of two skids taken from the SCADA for November 2002. December 2002 through early January 2003

constant, more data should be collected to insure accurate forecasting of future CIP frequencies.

Table 4. Summary of CIPs

| | Date | Days of Operation Between CIPs |
|---|---|---|
| Skid No. 1 | 7/12/2002 | 49 |
| Skid No. 1 | 10/17/2002 | 97 |
| Skid No. 1 | 1/6/2003 | 81 |
| Skid No. 2 | 8/20/2002 | 88 |
| Skid No. 2 | 11/26/2002 | 98 |
| Skid No. 2 | 1/28/2003 | 63 |

## MF PERFORMANCE AS MEASURED BY OTHER WATER QUALITY PARAMETERS

The effectiveness of MF in rejection of particulate contaminants was also measured monthly measurements of total Fe, HPC and total coliforms in the permeate. While measurements of total Fe were made in the finished water throughout the study period, those in the raw water were made only from September 2002 to January 2003 and those in the permeate water for only the last two months of the study (December 2002 and January 2003). Thus, the ability to assess directly the rejection of total Fe is limited to only the last two months, although inferences can be made from measurements of Fe in the finished water when MF permeate measurements were not taken. HPC and total coliforms were measured monthly in the MF permeate from June 2002 to January 2003. No measurements were made of these parameters in the raw water. Nevertheless, it is quite reasonable to assume that both HPC and total coliforms would be present in any raw water supply reservoir. The goal of the study was not to measure the performance of MF alone, but instead the improvement of finished water quality as compared to the conventional treatment system.

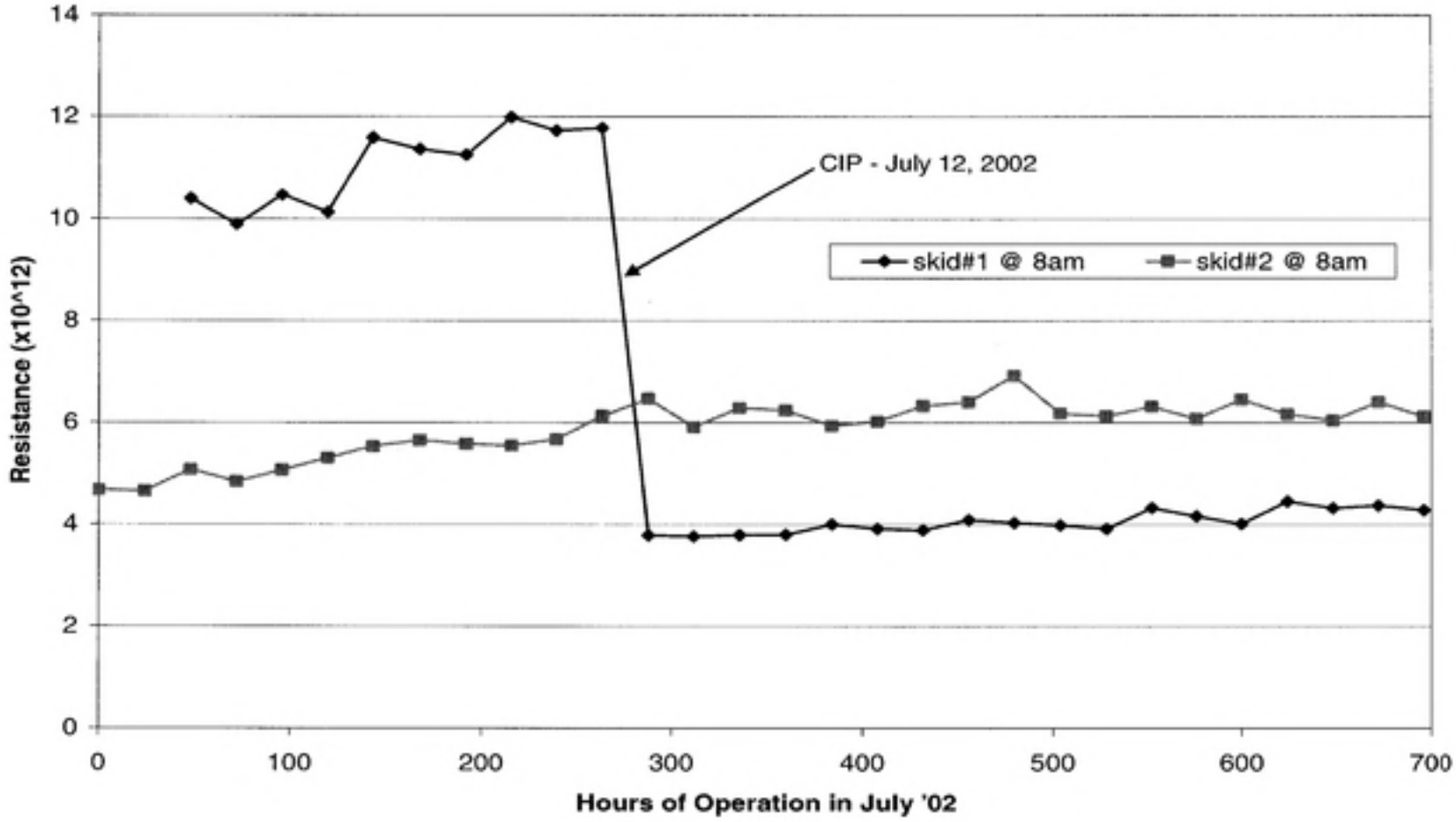



Figure 14. Membrane resistance calculated by the SCADA system for both skids at 8am daily in July 2002

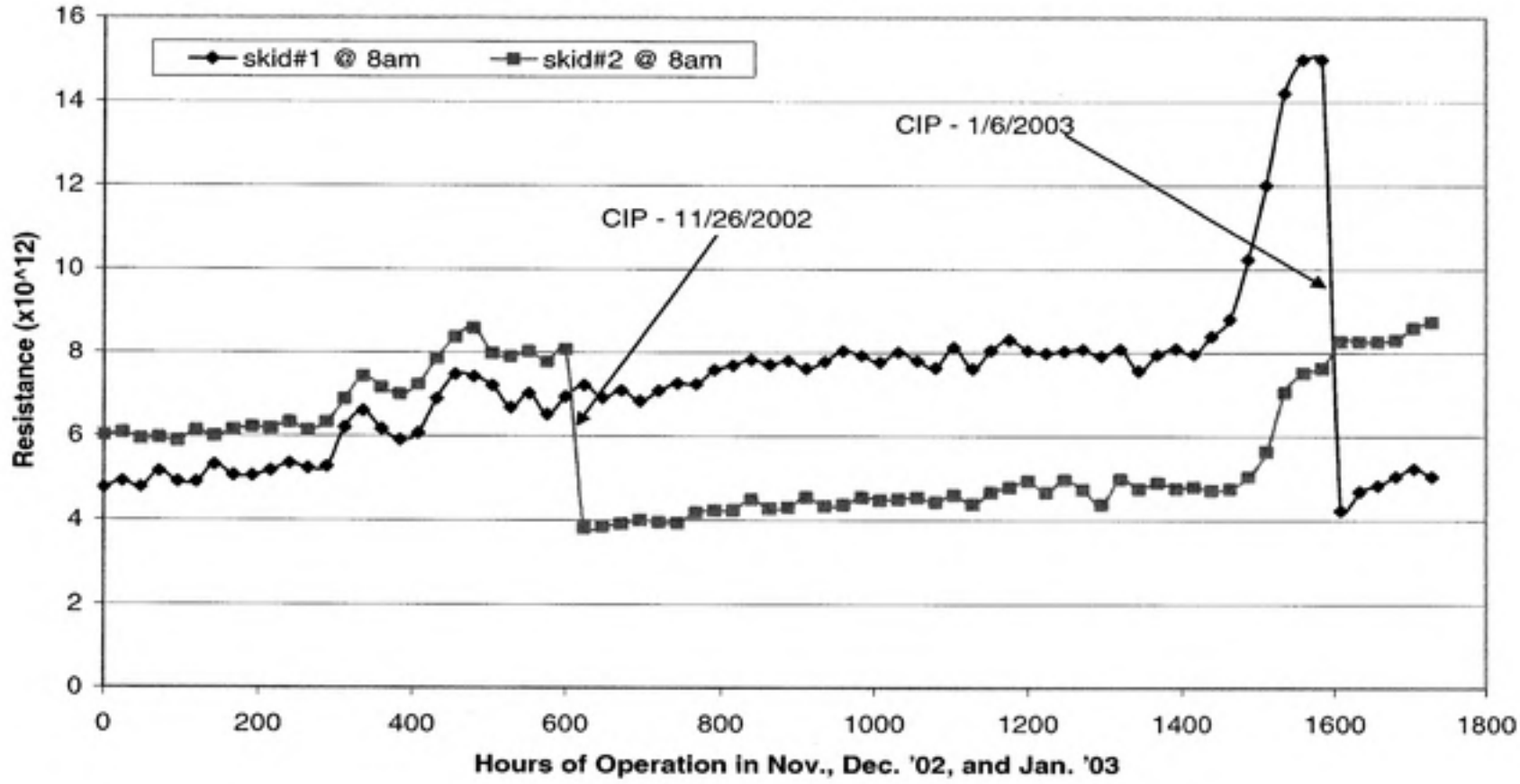



Figure 15. Membrane resistance calculated by the SCADA system for both skids at 8 am daily from November 2002 through early January 2003.

Table 5 gives the average values of the total Fe, HPC and total coliforms in the MF permeate. Averages total Fe in the raw water and finished water and averages of HPC and coliforms in the finished water are also included for reference. Total Fe in the permeate is far below the secondary MCL of 0.3 mg/L. The total Fe in the raw water for the five monthly samples for either measurements of total Fe in the permeate or finished water was 0.41 (±0.24 mg/L). Using all of these data, the average removal of total Fe is 93 percent. Park et al. (2002) reported only 84 percent removal of total Fe by MF with values in the permeate that were below the detection limit of 0.02 mg/L. Ellis et al. (2000) obtained a similar result.

Most of the Fe in surface waters is either in the soluble reduced form ($Fe^{2+}$) or the insoluble oxidized form ($Fe^{3+}$). The distribution between the soluble and insoluble forms of Fe was not measured in this study. It would depend upon the oxidation-reduction conditions that vary seasonally and also upon the depth of the raw water intake. In the case of Carthage, the raw water reservoir receives a significant flow of spring water that could contain $Fe^{2+}$. While most Fe is probably in the oxidized form and thus present as $Fe(OH)_3$, this may not be true in summer months when reduced conditions may exist in the reservoir. The soluble form of Fe cannot be removed by MF. Therefore, the large removal of total Fe found by MF treatment must have occurred by rejection of the insoluble form. One possible explanation for high removals of total Fe regardless of the Fe oxidation state in the raw water could be the role of DO that is introduced by the pressure air pulsing to the MF every 30 minutes. At high pressure, the driving force for mass transfer of DO into water is greatly increased according to Henry's law of gas solubility (solubility is directly proportional to partial pressure of the gas). The rate of

iron oxidation is first-order with respect to DO (Singer and Reckhow, 1999). The average DO concentration in the finished water was 5.5 mg/L and reached as high as 6.9 mg/L. Such high DO concentrations could oxidize $Fe^{2+}$ within the membrane module and thus explain the presence of the particulate form that could be rejected.

Table 5. Average values of Total Fe, HPC, and Total Coliforms in MF Permeate

| Parameter | Permeate Value | Months Sampled | Raw Water Value | Finished Water Value |
|---|---|---|---|---|
| Total Fe, mg/L | N/A | 12/02-01/03 | 0.41 (±0.24) | 0.03 (±0.02) |
| HPC, cfu/mL | 1,393 | 6/02-01/03 | NA | 937 |
| Total coliforms, cfu/100 mL | 0 | 6/02-01/03 | NA | 0 |

NA = not available

The occurrence of HPC in the permeate demands further explanation because these MF membranes have a nominal pore size of 0.2 µm, which should prevent passage of HPC into the permeate. The permeate values shown in Table 5 were for samples taken at the sample tap within the laboratory and not directly from the membrane skids. Although the average HPC was very high in the permeate (1,393 cfu/mL), there was a noticeable trend of decreasing HPC with each month, especially after the CIPs as indicated by the results presented in Figure 16. This figure also includes HPC measurements from samples of permeate taken at each of the two membrane skids. HPC at the two skids was not substantially less than at the sampling tap in the laboratory. HPC in the permeate can originate from penetration of microbes through the membrane (e.g., due to membrane damage or seal damage), from regrowth on the permeate side within the membrane skid, or from regrowth within the pipe line that collects the permeate from the membrane skid as suggested by Jacangelo et al. (1991). The fact that the HPC at two skids were not much lower than at the sampling tap in the laboratory suggests that HPC did not originate from regrowth in the sample line. From discussions with Carthage WTP

personnel, it appears that the permeate lines were not chlorinated before the membrane skids were placed into operation. An alternative explanation is that regrowth could have taken place on the permeate side of the membranes within the skids. The presence of HPC in permeate water has been reported by Karimi et al. (1999) wherein 50-86 percent of HPC were removed by MF initially, but soon thereafter regrowth in the permeate caused a higher permeate HPC.

Also of interest in Figure 16 is the lower HPC in Skid No. 2 than in Skid No. 1 on the sampling day in January 2003. The citric acid solution and flushing used in the CIP just two days before this sample was taken may explain the lower HPC.

Total coliforms were not detected in the permeate samples. Although none of the colonies that grew on the m-Endo agar was confirmed as coliforms, atypical colonies were routinely found from samples taken up until July 2002. The source of these atypical colonies could be from failure to chlorinate the permeate lines before putting the MF skids into service. The CIP that was performed on Skid No. 1 in July could have reduced or eliminated these microorganisms.

**Percent Recovery of Water**

In conventional filtration, the backwash water is taken from the clear well. The estimate backwash volume from discussion with Carthage WTP operators is 5,000 gal. Thus, the recovery of water would be the ratio of the volume of water delivered to volume of raw water pumped to the WTP. In dead-end MF, all of the feed water becomes permeate during the time of filtration. However, raw water is needed for backwashing of the MF skids every 30 min. In addition, water is lost in each CIP because of the makeup and disposal of the chemical solution. Therefore, the ratio of the

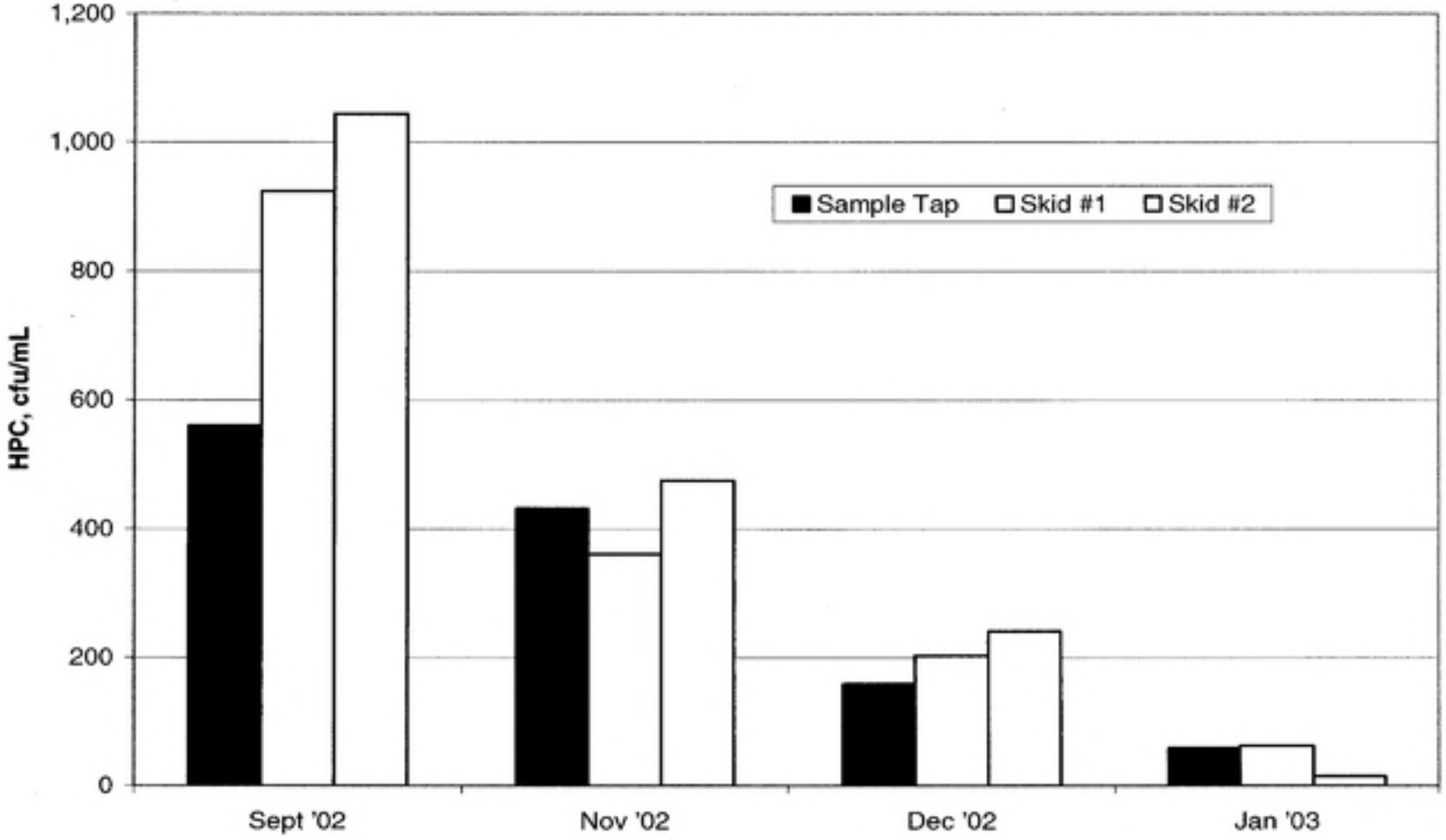



Figure 16. HPC in the MF permeate as measured at the sample tap in the laboratory and at the two membrane skids.

permeate water produced to the raw water pumped to the WTP represents the water recovery. The percent recoveries of water before and after the conversion to MF are presented in Table 6. Recoveries for MF include an annualized CIP volume within the backwash volume. As should be expected, the water recoveries are well above 90 percent, both before and after the conversion. Despite the fact the MF requires 23 times more backwashes than the conventional filtration system, the water recovery is almost 95 percent. The reason for the high water recovery is that each MF backwash uses only 525 gallons of water whereas one conventional backwash uses slightly over 5,000 gallons. Mallevialle, Odendaal, and Wiesner (1996) suggested 93 percent recovery for MF systems that employ an air backwash which is very close to the value achieved at Carthage.

Table 6. Summary of Water Recovery Before and After Conversion to MF

| | Hrs of Operation | Backwashes/Day | % Recovery |
|---|---|---|---|
| Before Conversion to MF | 12.22 | 1 | 98.0 |
| After Conversion to MF | 11.53 | 23 | 94.6 |

**DISTRIBUTION SYSTEM WATER QUALITY**

The average of monthly measurements of turbidity, particle counts, TOC, total chlorine residual, HPC, and total iron measured before (7 months) and after (7 months) the conversion to MF in the finished water (Station 1) and six stations in the DS are presented in Table 7. Five sampling stations were selected initially and an additional station was added in June 2002. The location of these stations was given in Figure 3.

Some general observations are possible from Table 7 before examining the monthly variability in values of each parameter. Station 2 stands out for its exceptionally

high average values of turbidity, exceeding the current MCL of 1 ntu both before and after the conversion; consistent with high turbidity is the high average particle counts.

Table 7. Average Monthly Values of Water Quality in the Finished Water (Station 1) and at Six Stations in the DS, Before (7 months) and After (7 months) the Conversion to MF

| | Turbidity (ntu) | | Particle Counts (No./mL) | | TOC (mg/L) | | Chlorine Residual (mg/L) | | HPC (No./mL) | | Total Iron (mg/L) | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Station No. | Before. | After | Before | After | Before. | After | Before | After | Before | After | Before | After |
| 1 | 0.603 | 0.380 | 670 | 640 | 3.2 | 3.9 | 2.39 | 2.38 | 85 | 937 | 0.05 | 0.03 |
| 2 | 2.000 | 1.514 | 3873 | 2296 | 2.4 | 2.1 | 0.66 | 0.05 | 13,668 | 54,006 | 0.53 | 0.68 |
| 3 | 0.579 | 0.343 | 754 | 541 | 3.0 | 3.7 | 2.04 | 2.65 | 109 | 3,882 | 0.05 | 0.05 |
| 4 | 0.630 | 0.331 | 853 | 367 | 2.9 | 3.7 | 2.08 | 2.00 | 7,204 | 4,105 | 0.07 | 0.05 |
| 5 | 0.489 | 0.462 | 237 | 763 | 2.3 | 3.3 | 0.99 | 0.91 | 129,144 | 103,229 | 0.06 | 0.08 |
| 6 | NA | 0.963 | NA | 463 | 2.5 | 3.7 | 0.14 | 0.98 | 69,000 | 128,715 | NA | 0.07 |
| 7 | 0.667 | 0.554 | 339 | 796 | 2.5 | 3.2 | 0.06 | 0.44 | 69,496 | 61,646 | 0.06 | 0.07 |

NA = Not Available

The conversion did decrease both turbidity and particle counts at Station 2 although these are still very high. TOC is higher at all stations except Station 2 after the conversion. Total chorine residual, which was free chlorine for some months and combined chlorine for other months, is very difficult to interpret without more information. Nonetheless, the levels are quite low at Stations 2 and 7 and the conversion to MF did not increase the residual at all stations. The average HPC is very high at all stations with the possible exception of Station 3 and was not decreased after the conversion; this is directly related to the failure to increase the chlorine residual. The average total Fe was low at all stations except for Station 2, which also is characterized by high turbidity and particle counts. The conversion to MF did not change the total Fe.

Having seven monthly data sets for before and for after the conversion to MF provides a good balance of information to draw conclusions about how DS water quality is affected. However, there was an extreme drought during the months following MF installation (June through October 2002) that could have affected the raw water quality. However, as shown in Table 2, the decrease in raw water turbidity is the only parameter that reflects an influence of the drought.

**Turbidity**

Figure 17 shows the effect of conversion to MF treatment on turbidity levels in the finished water and in the DS. The squares represent a monthly data point at each station. The bar denotes the average value at each station. When all DS stations (except Station 1) are considered together, the conversion to MF resulted in a 22.7 percent decrease in turbidity. For individual stations, the greatest reduction in turbidity (47 percent) was observed at Station 4 while the smallest reduction (5.3 percent) was

observed at Station 5. The turbidity at Station 2 is extremely high and in fact, well above the current MCL of 1 ntu for almost all samples, both before and after the conversion. Unlined, cast iron piping surrounds this sampling station. As will be later shown, the Fe concentrations were also extremely high in all samples at this station because of iron corrosion. These corrosion products are the most likely explanation for the very high turbidity values. Consequently, the conversion to MF would not be expected to lower turbidity because particles are being generated at this site.

The lowest percent reductions in turbidity (5.3 and 18.1 percent) were measured at the Stations 5 and 7, respectively. These two stations are located the farthest distance from the WTP and thus, the cumulative effects of deterioration in water quality within the DS would likely negate the benefits of improved particle removal at the WTP. However, the percent reduction at Station 5 is low partly because of one very high value of turbidity (0.315 ntu) value; excluding this value increases the removal to 35.6 percent. Another high turbidity value was measured at Station 7 on the same day as the high value at Station 5. One possible source of turbidity was DS flushing which was done on the day of sampling and on the previous day.

**Particle Counts**

The monthly data for particle counts in the finished water and in the DS are shown in Figure 18. Because of the anomalous nature of Station 2 (i.e., iron corrosion contribution to particles), the same data with the exception of Station 2 are replotted in Figure 19 for better scale resolution. The reduction in particle counts is minimal in the

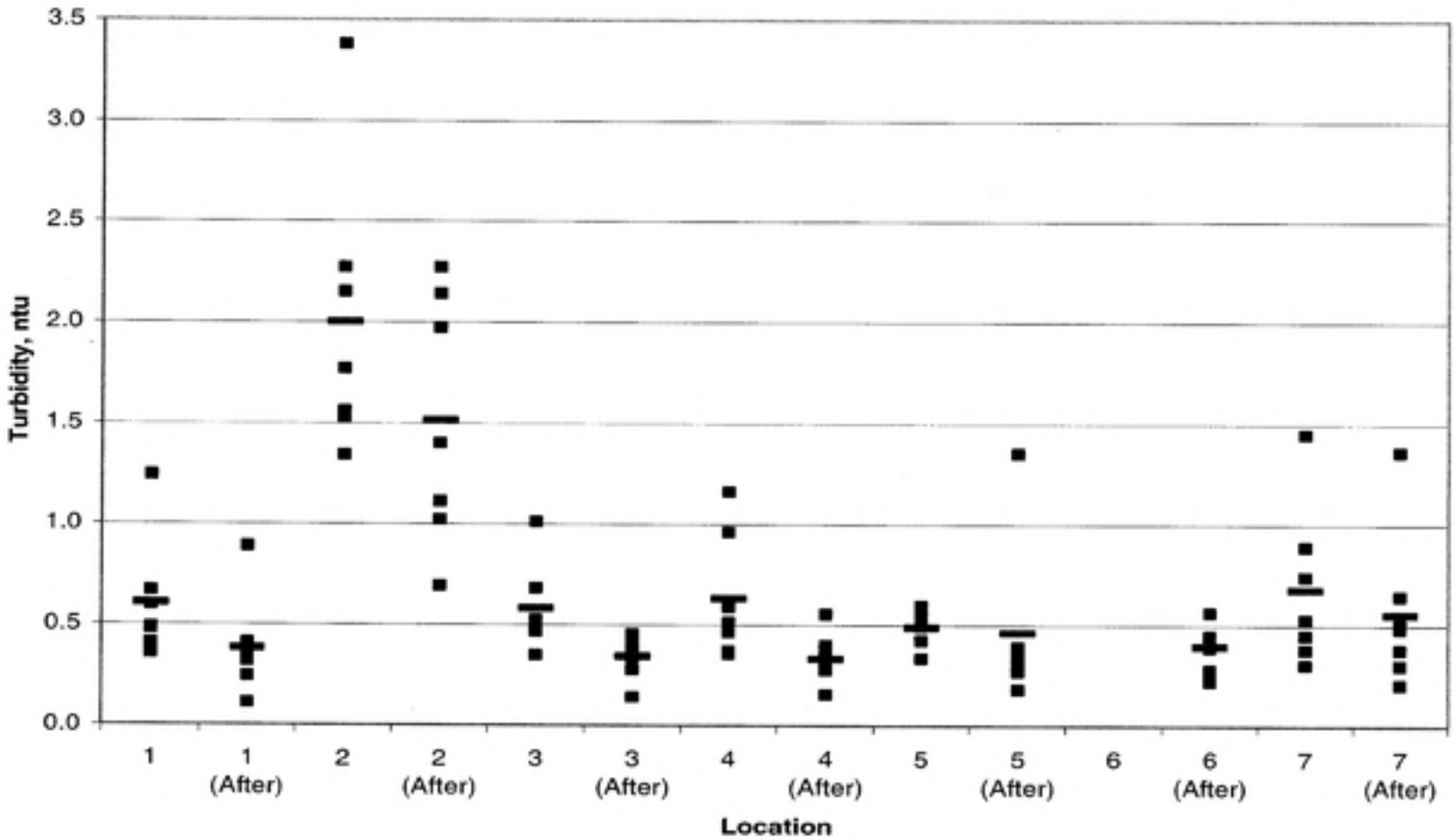



Figure 17. Monthly (squares) and average (bars) turbidity in the finished water (Station 1) and at stations in the DS, before and after the conversion to MF

finished (4.4 percent), probably because of the particles that were generated within the clearwell after the suction cleaning as note previously. The particle count was reduced at four of the six stations. The average reduction in particle counts was 41.5 percent with the maximum reduction measured at Station 4 (57 percent). The particle count increased 221 percent at Stations 5 and 35 percent at Station 7. However, these increases were caused by two monthly values that were exceptionally high. Two of the three high values of particle counts were measured on the same sample date which corresponded to measurement of high turbidity as well. These outlier values were most likely due to the flushing of the DS that took place on this date and previous day. Five of the seven monthly values at Station 5 were below 110 particles/mL but the values were extremely high for the other two months (1,769 and 3,075 particles/mL). If these two monthly samples are excluded, the average particle count is lowered from 763 to 100 particles/mL and this results in a 58 percent reduction in particles as opposed to an increase of 221 percent. One exceptionally high value was measured at Station 7 (4,446 particles/mL). Excluding this monthly sample, the average particle count decreases to 190 particles/mL to yield a 43.9 percent reduction instead of a 35 percent increase. By removing the three outlying data points for Stations 5 and 7, the average reduction is particles increases to 41.5 percent.

**TOC**

The monthly data for TOC in the finished water and in the DS is shown in Figure 20. TOC increased at all locations except Station 2 after the conversion to MF. The reason for the decrease in TOC at Station 2 will be explained later in more detail. An

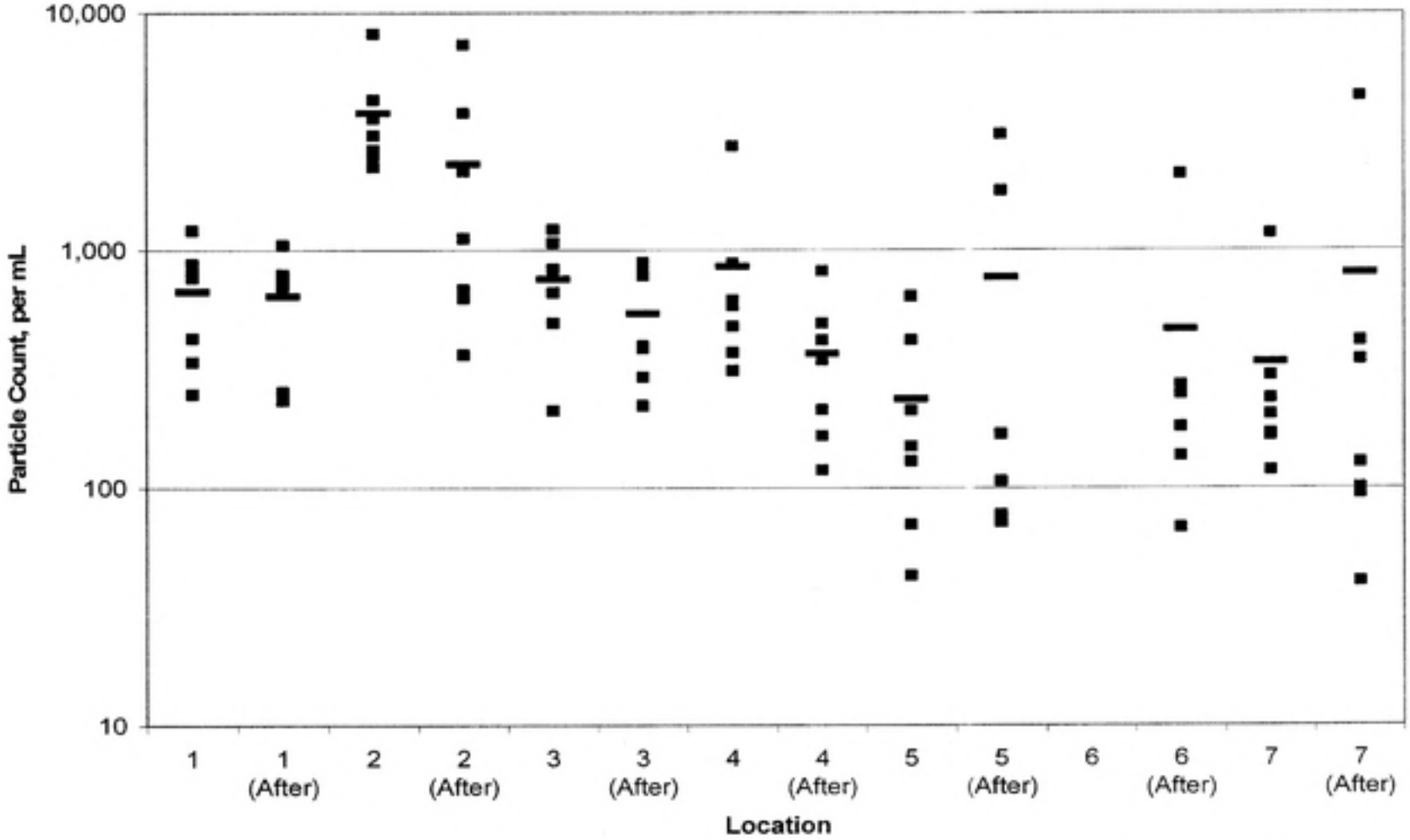



Figure 18. Monthly (squares) and average (bars) particle counts in the finished water (Station 1) and at stations in the DS, before and after the conversion to MF

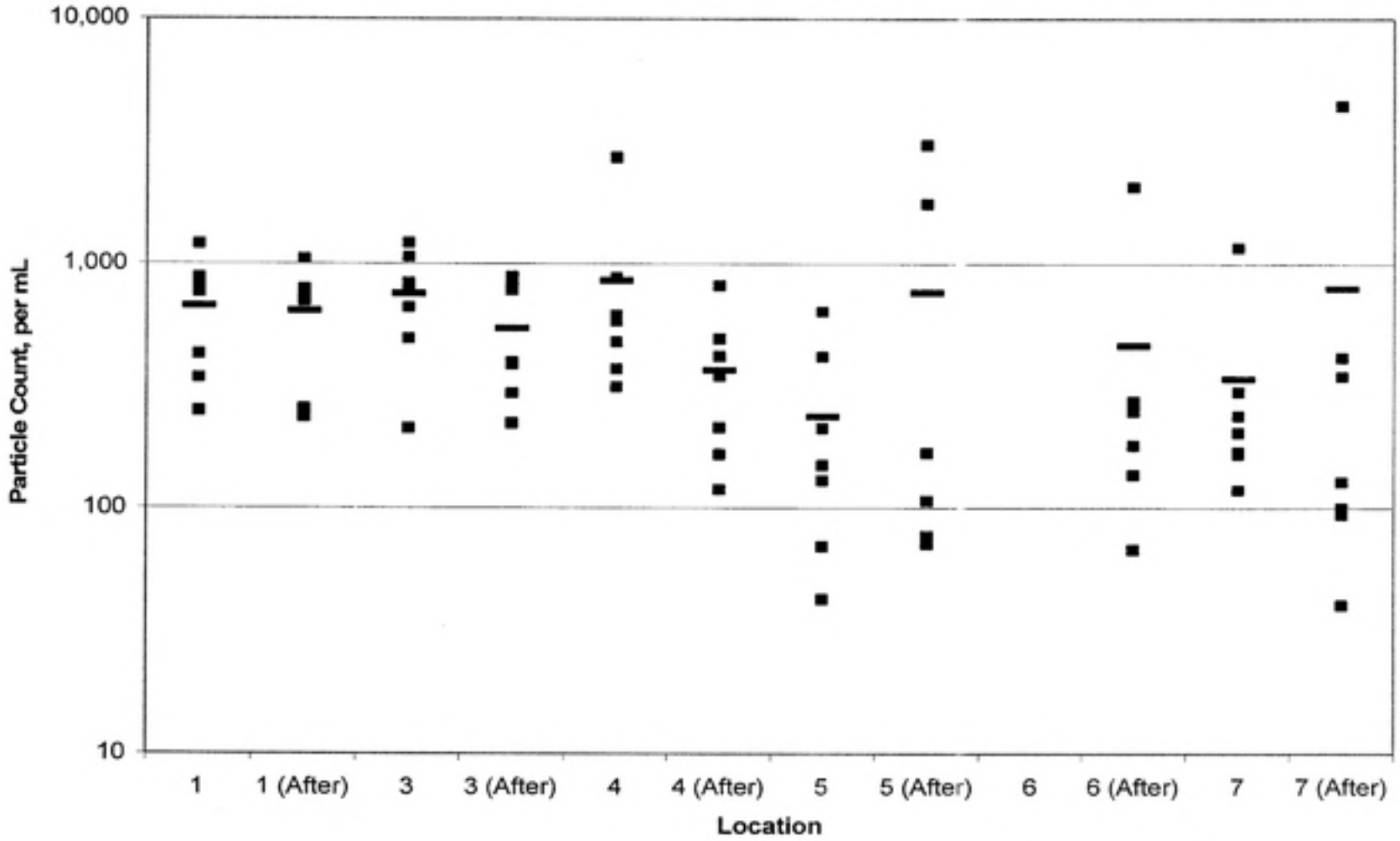



Figure 19. Monthly (squares) and average (bars) particle counts in the finished water (Station 1) and at all stations except Station 2 in the DS, before and after the conversion to MF

increase in TOC was expected at all stations because chemical coagulation had been eliminated upon conversion to MF and thus, only the particulate fraction of TOC can be removed. Excluding Station 2, the average increase in TOC at all stations was 29.1 percent. The largest increase (46.5 percent) was observed at Station 6. If Stations 2 and 6 are excluded, the percent increase in TOC at each of the other stations was about the same. Figure 20 also shows that TOC is generally lower at Station 2 than all other stations in the DS, both before and after the conversion, despite the fact that Station 2 is located only 1.2 miles from the WTP. The loss of TOC can be explained by either production of a biofilm or by adsorption onto corrosion products (Butterfield et al., 2002). The loss in TOC from the finished water (Station 1) to Station 2 is much larger after the conversion. One possible explanation is increased biofilm activity owing to lower chlorine residuals after the conversion.

**Chlorine Residual**

The monthly data for total chlorine residual in the finished water and at the DS stations is shown in Figure 21. The nominal plant operation is for addition of ammonia to form combined chlorine. However, following the conversion, there were several months during which free chlorine was used. This was caused by problems stemming from failure of the ammonia pump and also very high chlorine demands during the summer months. The high chlorine demand made it impossible to generate sufficient free chlorine residual after the CT tank such that combined chlorine could be formed in a concentration sufficient to achieve the target residual of about 3.5 mg/L. The target value for free chlorine, on the other hand, was about 2 mg/L because of its

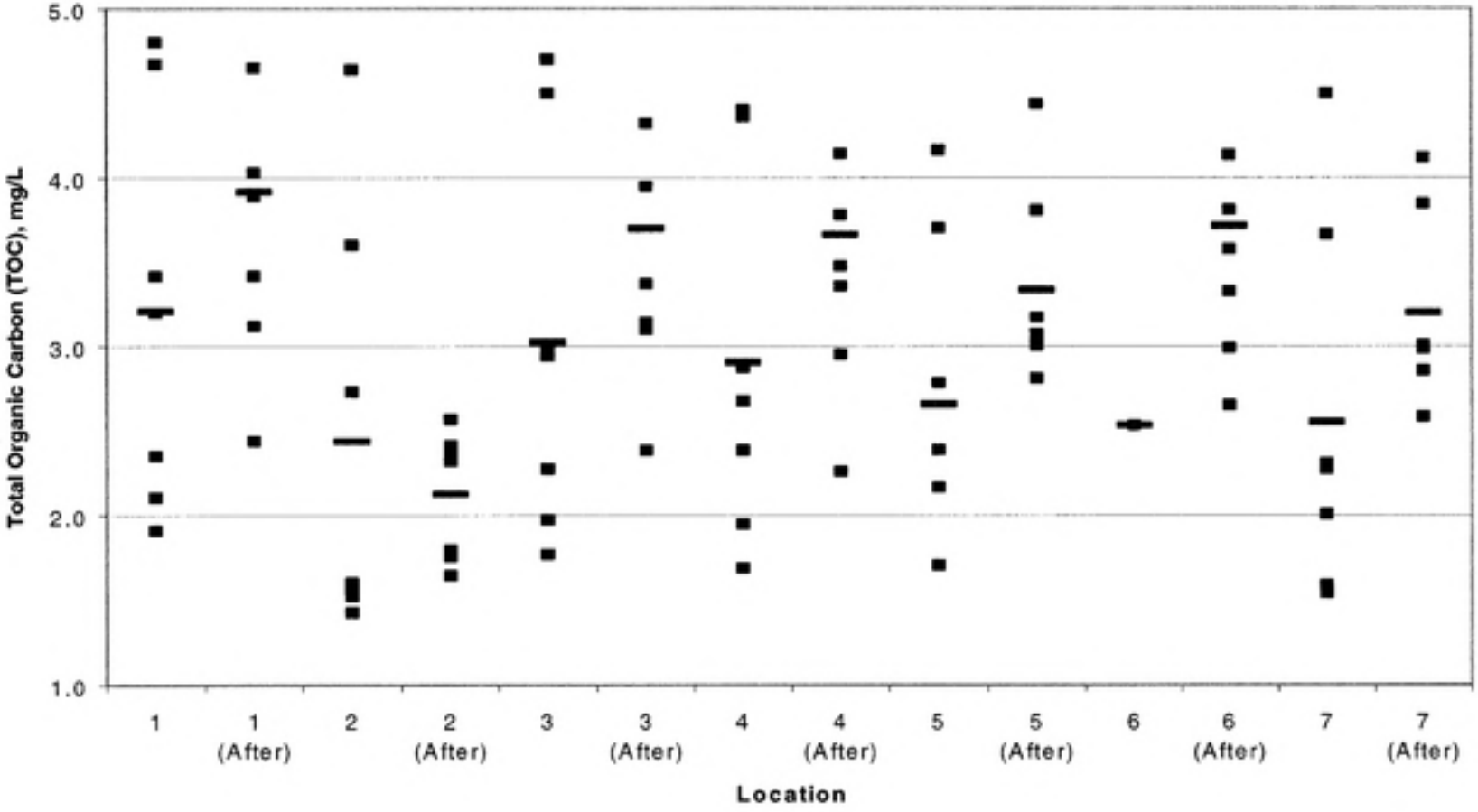



Figure 20. Monthly (squares) and average (bars) TOC in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF

stronger disinfecting power. As shown in Figure 20, the chlorine residual was highly variable from month to month at all stations. This variability is due in part to some values that were free residual (lower residuals in finished water) and others that were combined residual (higher residuals in finished water). As important, the chlorine residual was extremely low in several monthly samplings at many of the stations. Of particular importance are the consistently low residuals measured at Station 2, which is only 1.2 miles from the WTP, and at Station 7 which is at the end of DS and is essentially a dead-end site. The impact of low chlorine residuals at these two sites will be discussed in more detail. In the case of Station 2, the most likely cause is reduction reactions with corrosion products. In the case of Station 7, the cause is extremely long residence time and reaction with chlorine demanding substances, as well as nitrification during chloramination periods that produced nitrite which in turns reduced chloramines (Wilczak et al., 1996).

The chlorine residuals for sampling dates on which combined chlorine was used are given in Figures 22. A similar set of data is provided in Figure 23 for sampling dates when free chlorine was used. Combined chlorine was applied on five of the seven sampling dates before the conversion to MF and on four of the seven sampling dates after the conversion. Residuals increased after the conversion at Stations 5, 6 and 7 but there was little change at Stations 3 and 4 and a decrease at Station 2.

The extremely low chloramine residuals at Station 2 are likely caused by reduction reactions that occur between with iron corrosion products (Vikesland and Valentine, 2002). Reactions with the unlined cast iron pipeline. The same reaction occurs

between free chlorine and corrosion products. Thus, while chloramination is generally considered

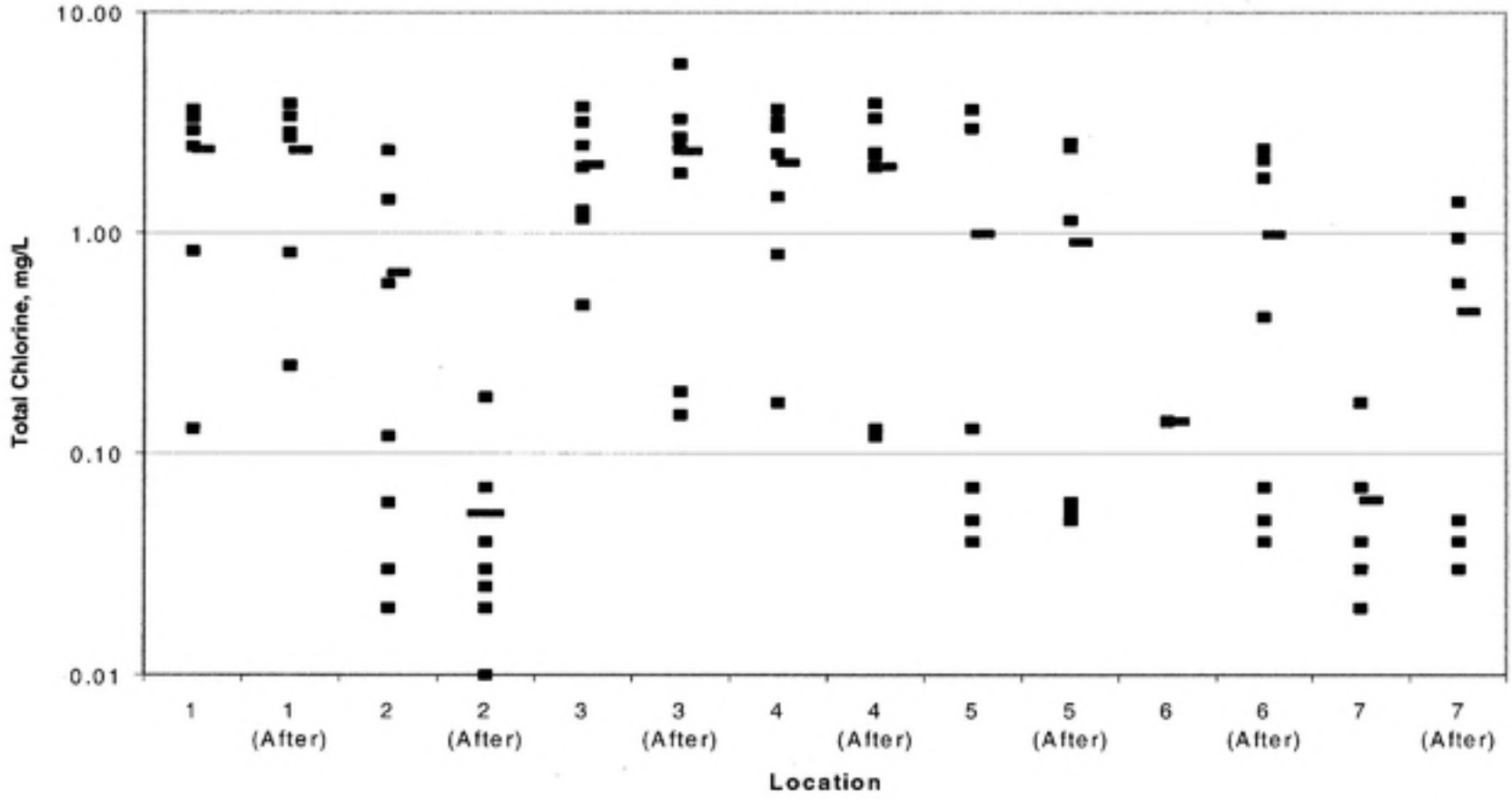



Figure 21. Monthly (squares) and average (bars) total chlorine in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF

a "stable" disinfectant, this is not the case if cast iron pipe is present.

Aside from Station 2, there appears to be a trend at the other stations in the DS toward higher chloramines residuals after the conversion to MF. Inspection of Station 1 (finished water) data in Figure 22 shows that the chloramine residuals leaving the WTP for the months following the conversion were not significantly higher than in months before the conversion; thus, higher residuals in the DS were not caused by higher residuals leaving the WTP. Colder temperatures, which slow the chloramines demand reactions, could possibly explain these higher residuals (i.e., lower chloramine demand) because the data collected after the conversion are heavily influenced by use of combined chlorine in the months of November 2002 through January 2003. An alternative explanation could be that the conversion to MF was responsible; however, the data do not allow further interpretation. The extreme variability in free chlorine residual at each station is apparent in Figure 23. These data correspond to the months of July, September and October 2002 (no sample was taken in August 2002). The sampling date in July is especially important to explain. Combined chlorine was discontinued just four days prior and free chlorine was started. The clear well has a 1 MG capacity and a theoretical residence time of about two days but fractions of the water will spend more time because of the mixing condition. Thus, the free chlorine that enters the clear well can react with chloramines, causing a reduction in the total residual, i.e., moving the system toward breakpoint chlorination, unless sufficient chlorine is added to form free chlorine in the finished water. This could explain the absence of residual in the finished water (Station 1) in the July sample. Because samples are taken with the DS, the age of the water can vary greatly such stations with long water age may still have chloramines residual. In

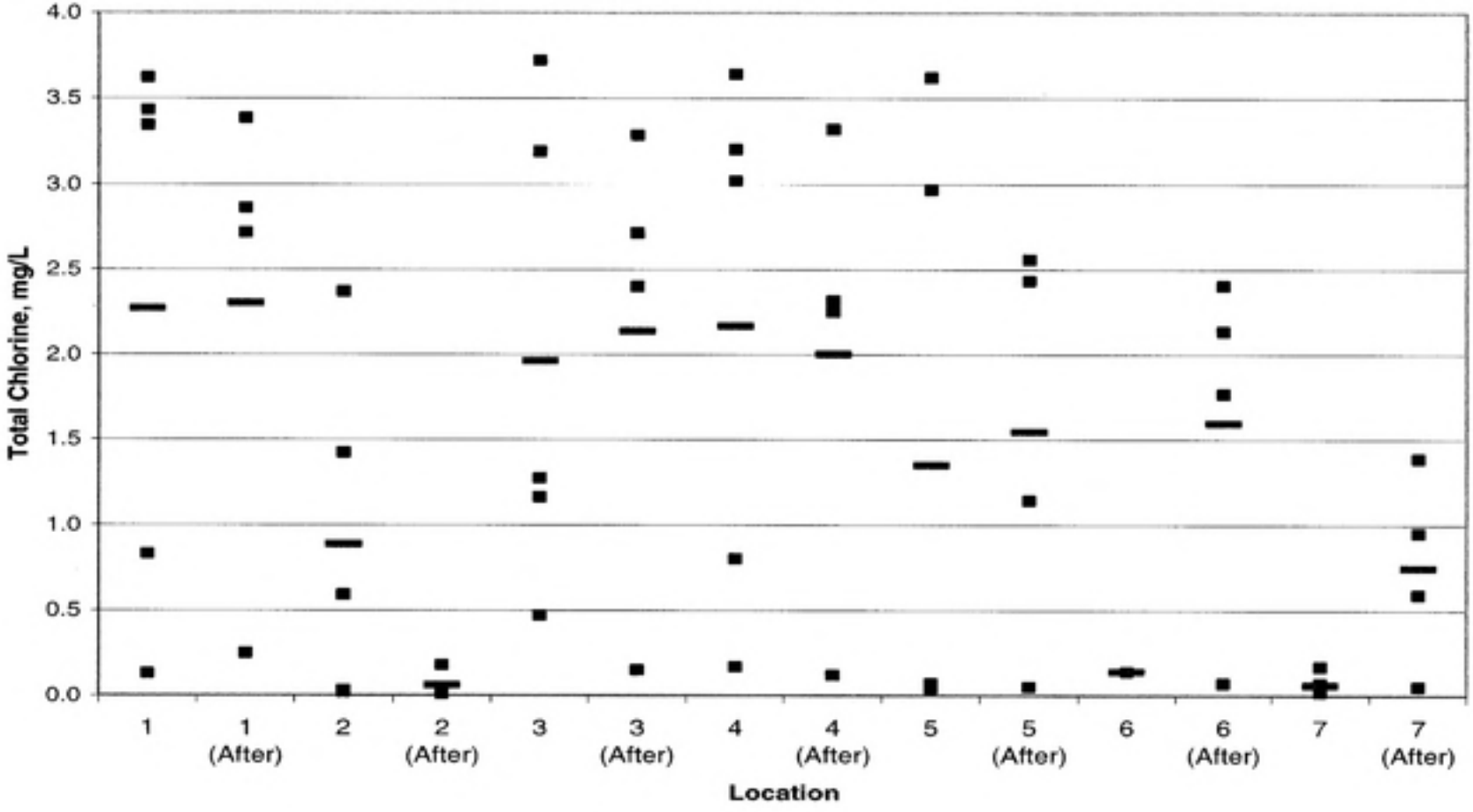



Figure 22. Monthly (squares) and average (bars) combined chlorine in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF

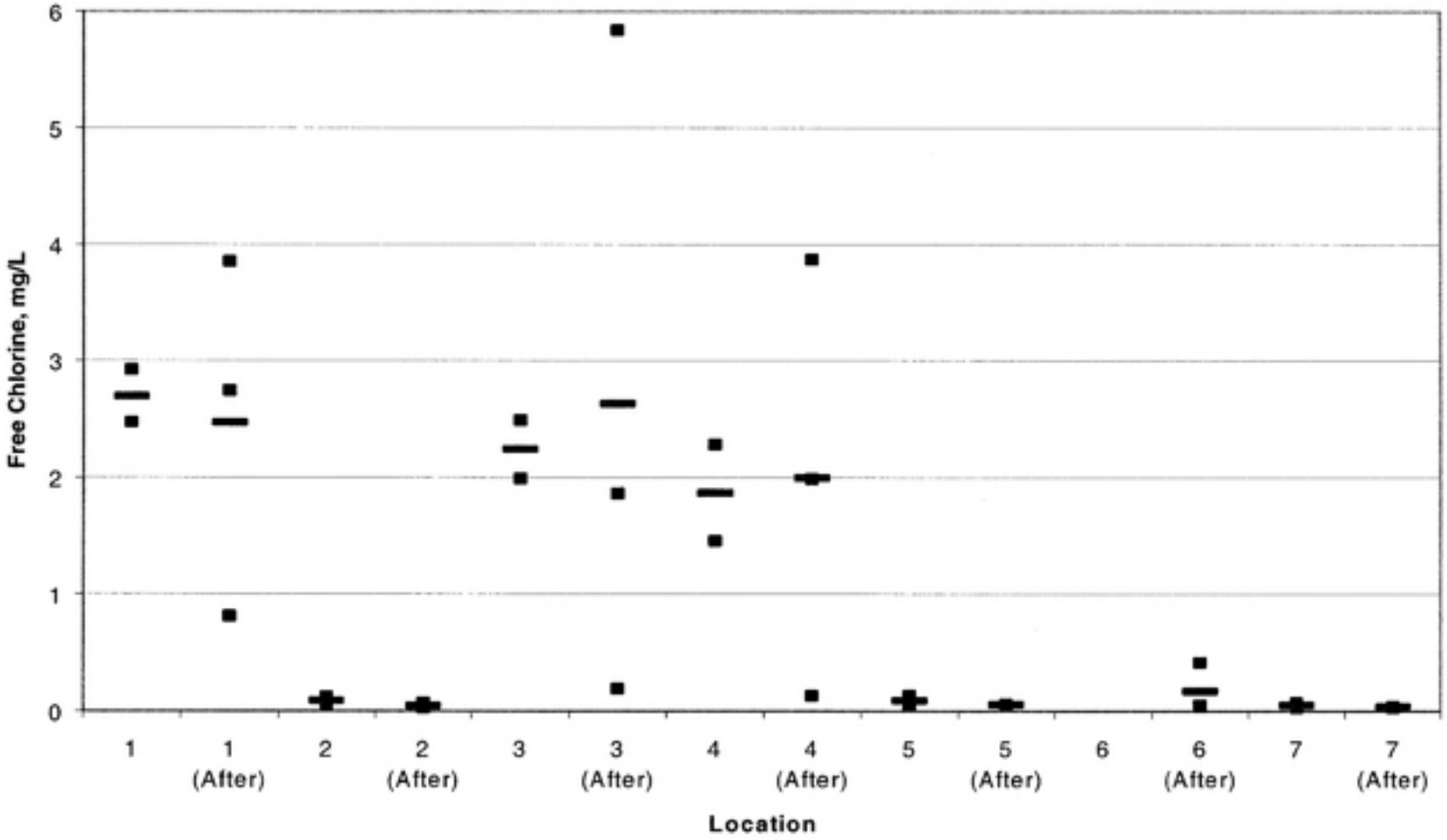



Figure 23. Monthly (squares) and average (bars) free chlorine in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF

addition, as the free chlorine water enters the DS and mixes with the existing water, there is again the "titration" reaction between free chlorine and chloramines that can reduce the chlorine residual. The failure to achieve a free chlorine residual was also due to exceptionally high chlorine demands as is evident from Figure 24 wherein the chlorine dosages applied are shown to increase greatly from July through October. The applied chlorine exceeded 10 mg/L for several months. The decrease in chlorine demand after the summer months, but without a corresponding reduction in the applied chlorine dosage, eventually caused very high free chlorine residuals to appear in the finished water and at Stations 3, and 4 in the DS. In fact, the residuals at Stations 3 and 4 were so high that sample dilution was required in the field in order measure the values with Hach spectrophotometer; dilution

errors were possible due to the lack of the proper dilution glassware in the field. Drought conditions may have caused the high demand owing to oxidation reactions with higher TOC concentrations during the summer months also to the possible withdrawal of water from lower depths, which typically is characterized by a more chemically reduced condition.

**HPC**

The monthly data for HPC in the finished water and in DS are shown in Figure 25. A log-scale is used to show more clearly the wide range of HPC. These data need to be interpreted with the aid of the disinfectant residuals data given in Figure 21. Both before and after the conversion to MF, several stations could be characterized as having extremely high HPC, exceeding 10,000 and on some occasions 100,000 cfu/mL. The highest values were associated with chlorine residuals that were near the limit of

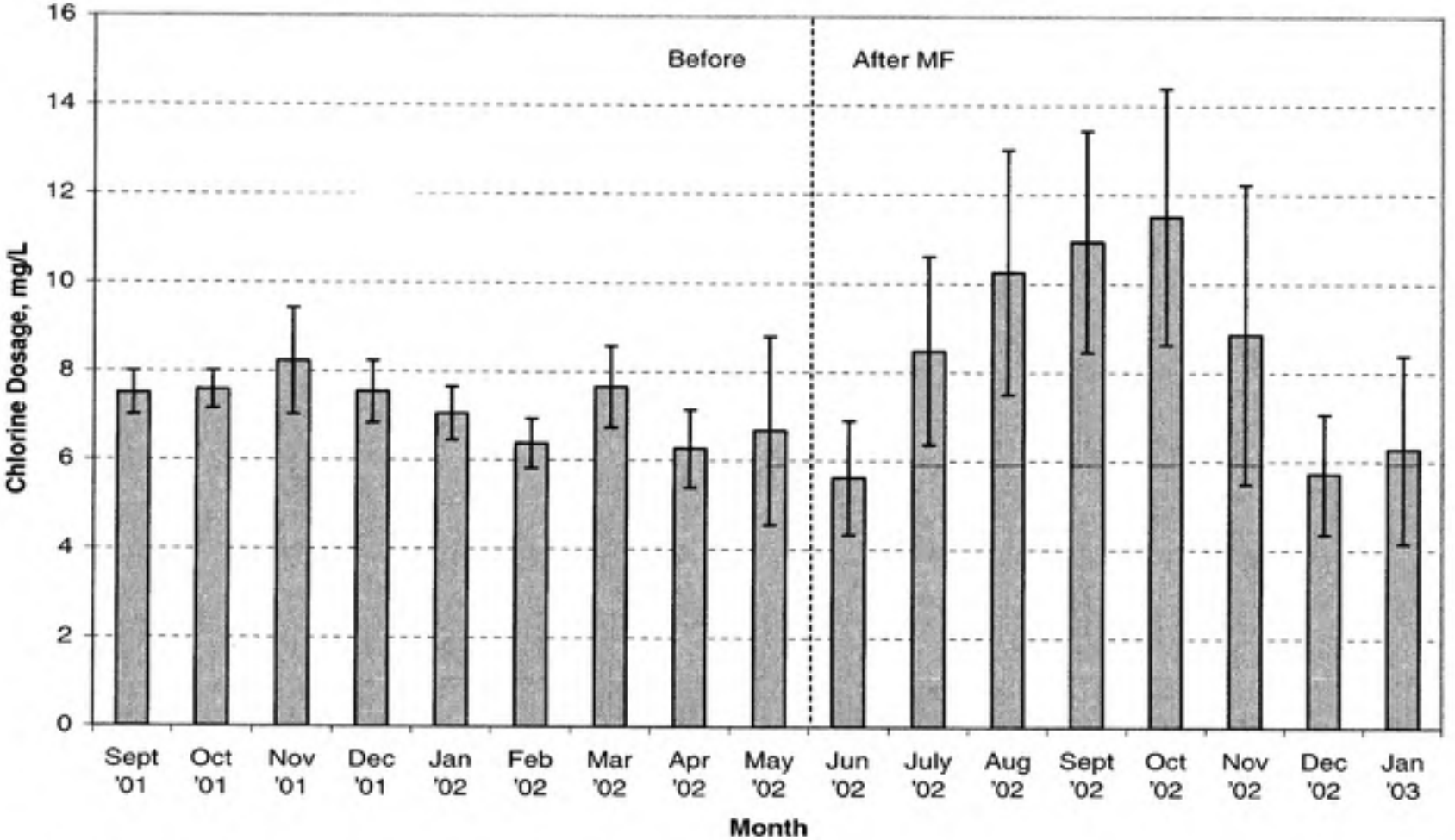



Figure 24. Box plot of the applied chlorine dosage for each month of the study

detection (0.05 mg/L). Many of the high values were observed at Station 7, which is the farthest from the WTP and behaves much like a dead-end. As reference point, the State of North Carolina requires retesting of any site for total coliforms if the HPC exceeds 500 cfu/mL, based on the Standard Plate Count technique. Admittedly, the R2A agar method is known to yield HPC that are up to two orders of magnitude higher than the Standard Plate Count agar. Nonetheless, the exceedance by three orders of magnitude of the 500 cfu/mL level is cause for concern.

The effect of the conversion to MF on HPC is mixed. HPC increased at Stations 2, 3, and 6 in the DS. Inspection of the chlorine residual data (Figure 21) indicates lower residuals at these stations after the conversion. The finished water (Station 1) in July is considered a special case. As noted earlier, the target chlorine residual could not be achieved leaving the WTP because of very high chlorine demand. The measured chlorine residual on this sampling date was 0.82 mg/L which is less than the typical target value of 2 mg/L but nonetheless, not extremely low. However, the WTP plant data revealed that much lower chlorine residuals existed in the days just prior to this sampling and in fact, earlier on the same sampling day. Thus, the high HPC in the finished water in July was probably because of the low chlorine residuals that preceded this sampling date. In addition, a very high TOC (4.7 mg/L) was measured in the finished water that could have also contributed more substrate for microbial growth and also contributing to the high HPC. Excluding the July sampling date, the average HPC in the finished water was 20 cfu/mL which is within expected values after disinfection. The increase in HPC after conversion to MF at Station 2 may be explained by two factors. First, the disinfectant residual at Station 2 was consistently low, with an average free chlorine of 0.045 mg/L

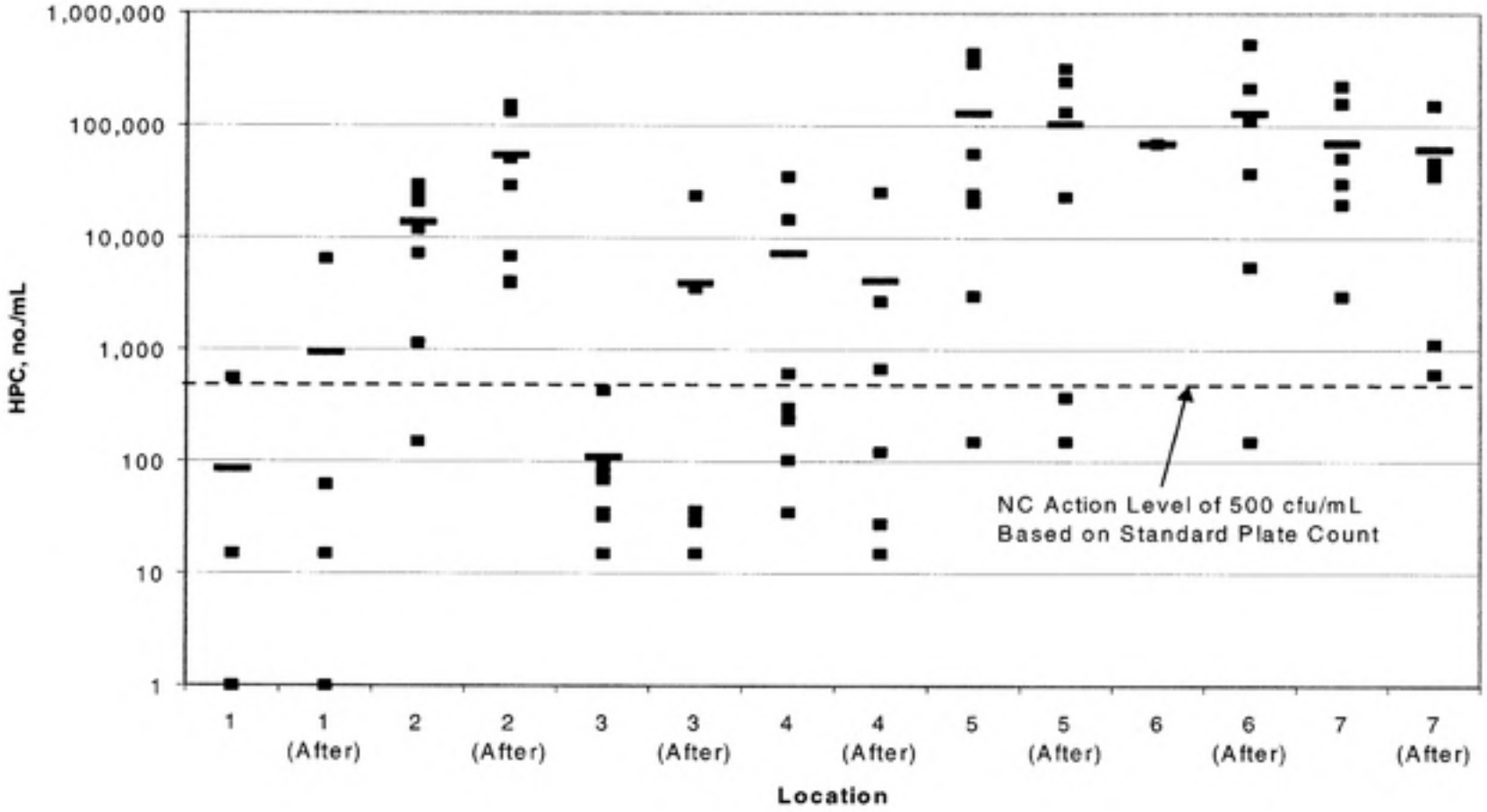



Figure 25. Monthly (squares) and average (bars) HPC in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF

and average combined chlorine of 0.06 mg/L. Before the conversion, the average free chlorine was 0.09 mg/L and the average combined chlorine was 0.89 mg/L. The second contributing factor could be the increase in TOC caused by elimination of coagulation. A fraction of this TOC is most likely biodegradable.

The high HPC measured on several dates at Stations 3, 6, and 7 could be cause for concern from a public health viewpoint. Two high values were measured at Station 3 in June (3,500 cfu/mL) and in July (23,567 cfu/mL). On both of these days, a low total chlorine residual (<0.2 mg/L) was reported. Extremely high values (all exceeding (100,000 cfu/mL) were measured at Stations 6 and 7 where the total chlorine residual was below 0.1 mg/L. All three of these Stations are about five miles away from the WTP and have potentially high water ages. The effect of chlorine residual on HPC during the sampling days when combined chlorine was used is illustrated in Figure 26. A similar set of data shows the effect on HPC during sampling days when free chlorine was used in Figure 27. Inspection of the HPC data suggests that a negative relationship between the disinfectant residual and HPC. Therefore, it could be suggested that the chlorine residual is an important factor in influencing the growth of bacteria in the DS. Pearson's correlation coefficients (r) were found to be in the range of -0.481 to -0.797 for combined chlorine whereas free chlorine was between -0.876 and -0.927. Zhang and DiGiano (2002) obtained similar results with correlation coefficients of -0.74 and -0.63 for Durham and Raleigh, NC, respectively.

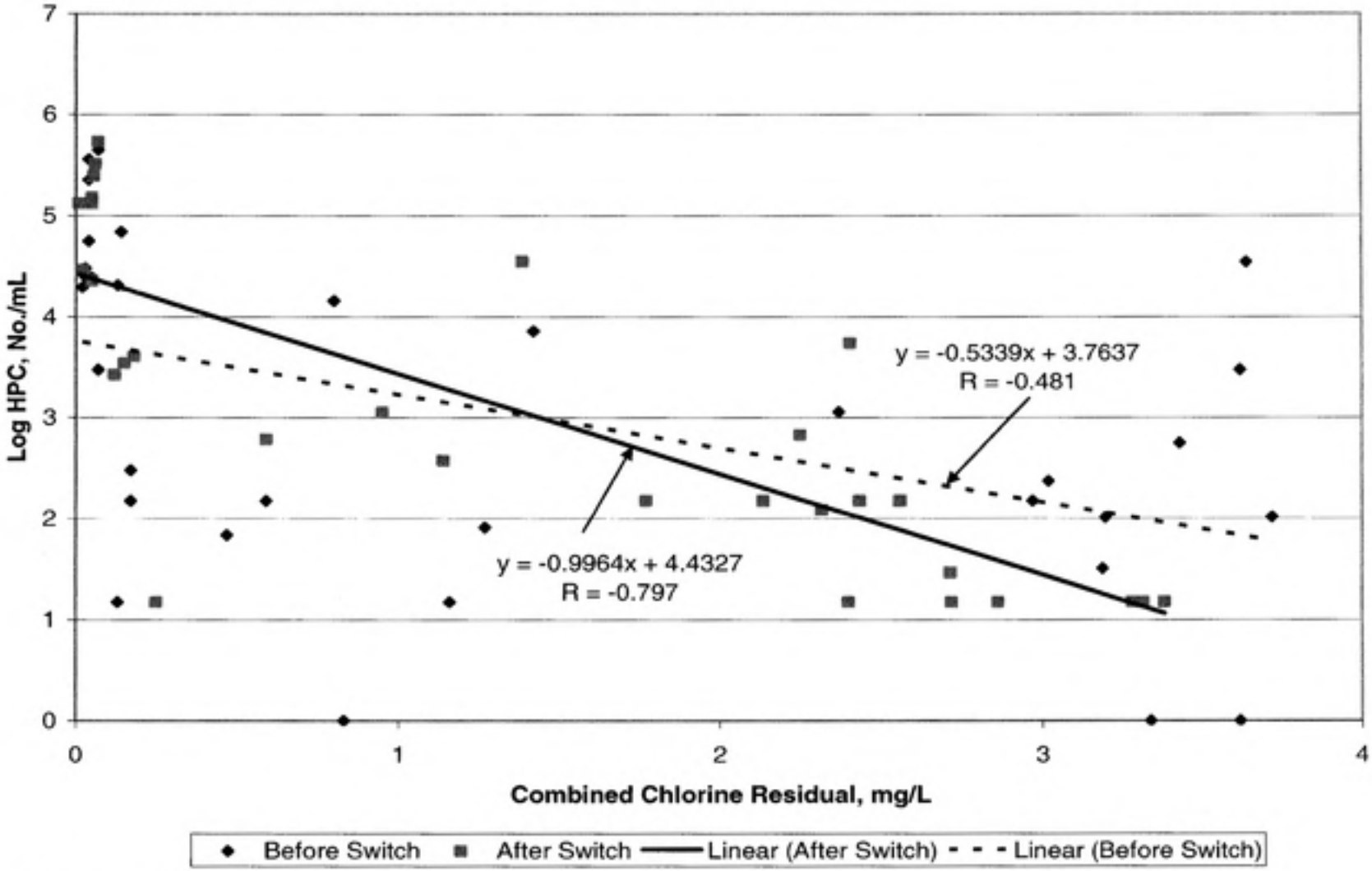



Figure 26. Correlation between HPC and Combined Chlorine Residual in the DS.

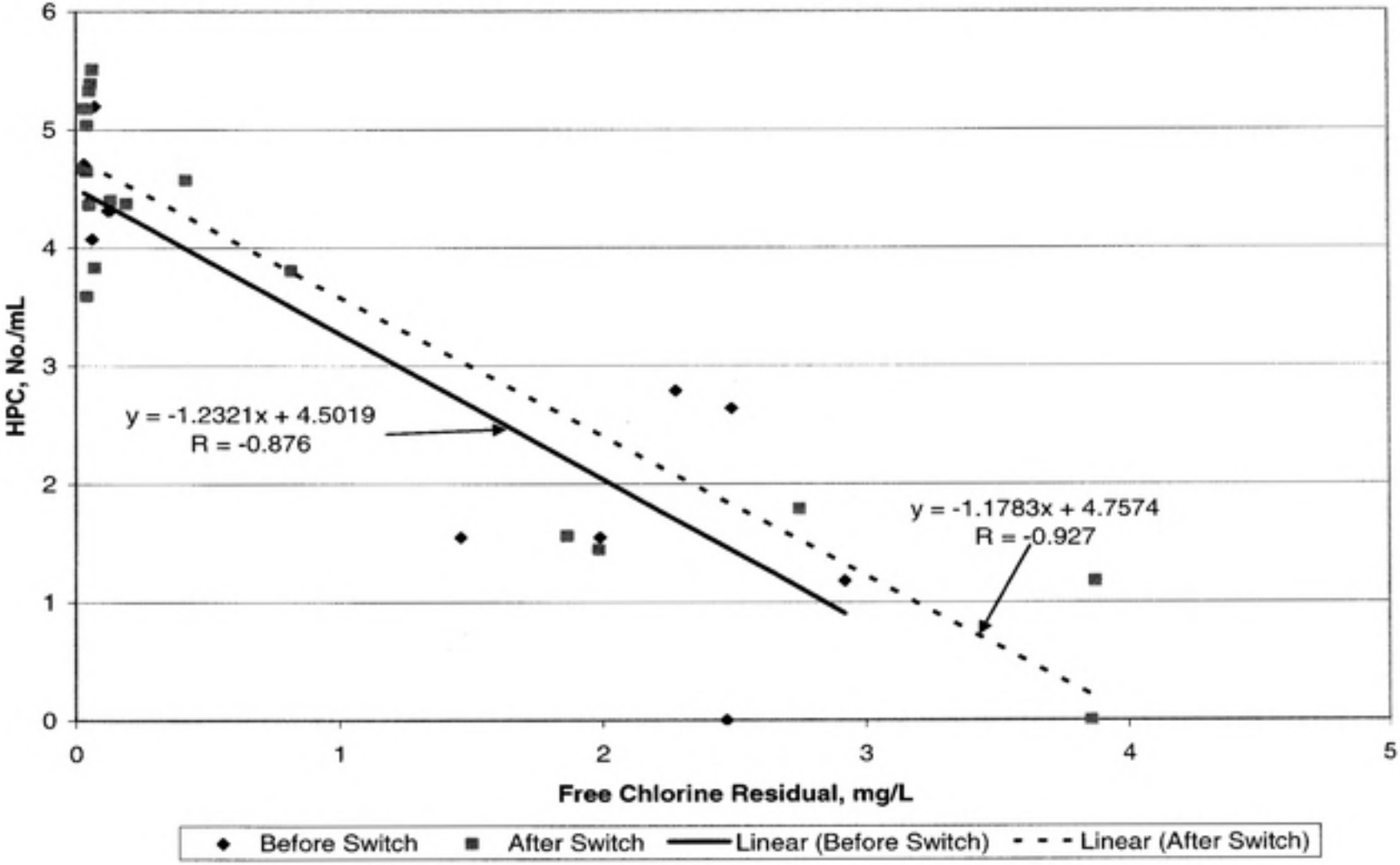



Figure 27. Correlation between HPC and free chlorine residual in the DS.

**Total Coliforms**

During the entire study period at Carthage (14 monthly samples at all stations), the total coliforms test was negative in the finished water and the stations in DS, despite the occurrence of extremely high HPC as noted above. Therefore, this standard indicator of microbial safety of the water would suggest that even though HPC were far above recommended limits, the water would was still within the limits of public health agencies. Atypical colonies were found on the m-Endo medium for many of the samples with long water ages and low disinfectant residuals in the DS up to October 2002. The lack of occurrence of atypical colonies after October 2002 is thought to be from a larger disinfectant residual at the corresponding stations.

**Total Iron**

The monthly data for total Fe in the finished water and at stations in the DS is depicted in Figure 28. The average values of total Fe at all but Station 2 were less than 0.1 mg/L. In contrast, the total Fe routinely exceeded 0.4 mg/L at Station 2 and reached a high of almost 1.5 mg/L. These results, along with the high values of turbidity and particle counts at Station 2 all point toward the problem of iron corrosion in this isolated section of the DS. The average total Fe increased at Station 2 following conversion to MF. One possible factor is failure to add sufficient corrosion inhibitor. Before the conversion to MF, the average concentrations of diphosphoric acid/tetrapotassium salt for corrosion control was 3.7 mg/L and ranged from 1 to 9 mg/L. These concentrations are exceptionally high. Inspection of plant operating records showed that the same mass rate of addition of corrosion inhibitor was used regardless of plant production. Thus, the highest concentrations resulted on days during which the plant flow rate was relatively

low. Upon startup of the MF, the corrosion inhibitor was not added because problems were encountered in integrating the chemical and high service pumps into the SCADA control system. According to plant personnel, some corrosion inhibitor was being added to the finished water beginning in December 2002, but the startup date is unknown. In addition, the daily mass addition of the corrosion inhibitor was not recorded.

## ESTIMATION OF WTP OPERATING COSTS

The estimated operating costs before and after conversion to MF, exclusive of labor costs, are shown in Table 8. These costs were sub-divided into chemical usage, power usage, residuals disposal (i.e., backwash water disposal, CIP chemicals and CIP disposal). The total cost over seven months of operation for each treatment process and the cost per 1000 gallons of water produced are included. Chemical costs were calculated from the daily usage of alum, chlorine, powdered activated carbon (PAC), fluoride, lime, ammonia, and phosphate, which are recorded by the Carthage WTP personnel, and the chemical costs per pound of each chemical as acquired from the Town of Carthage.

Table 8. Summary of Operating Costs (Exclusive of Labor) Before and After Conversion to MF

| | Before MF | | After MF | |
|---|---|---|---|---|
| | Total Cost, $/7 mos. | $/1,000 gal | Total Cost, $/7 mos. | $/1,000 gal |
| Chemical | 16,975 | 0.247 | 8,654 | 0.164 |
| Electrical | 14,015 | 0.206 | 18,531 | 0.343 |
| Backwash disposal | 1,170 | 0.017 | 3,062 | 0.058 |
| CIP & disposal | 0 | 0.000 | 6,609 | 0.125 |
| Total: | 32,160 | 0.47 | 36,856 | 0.690 |

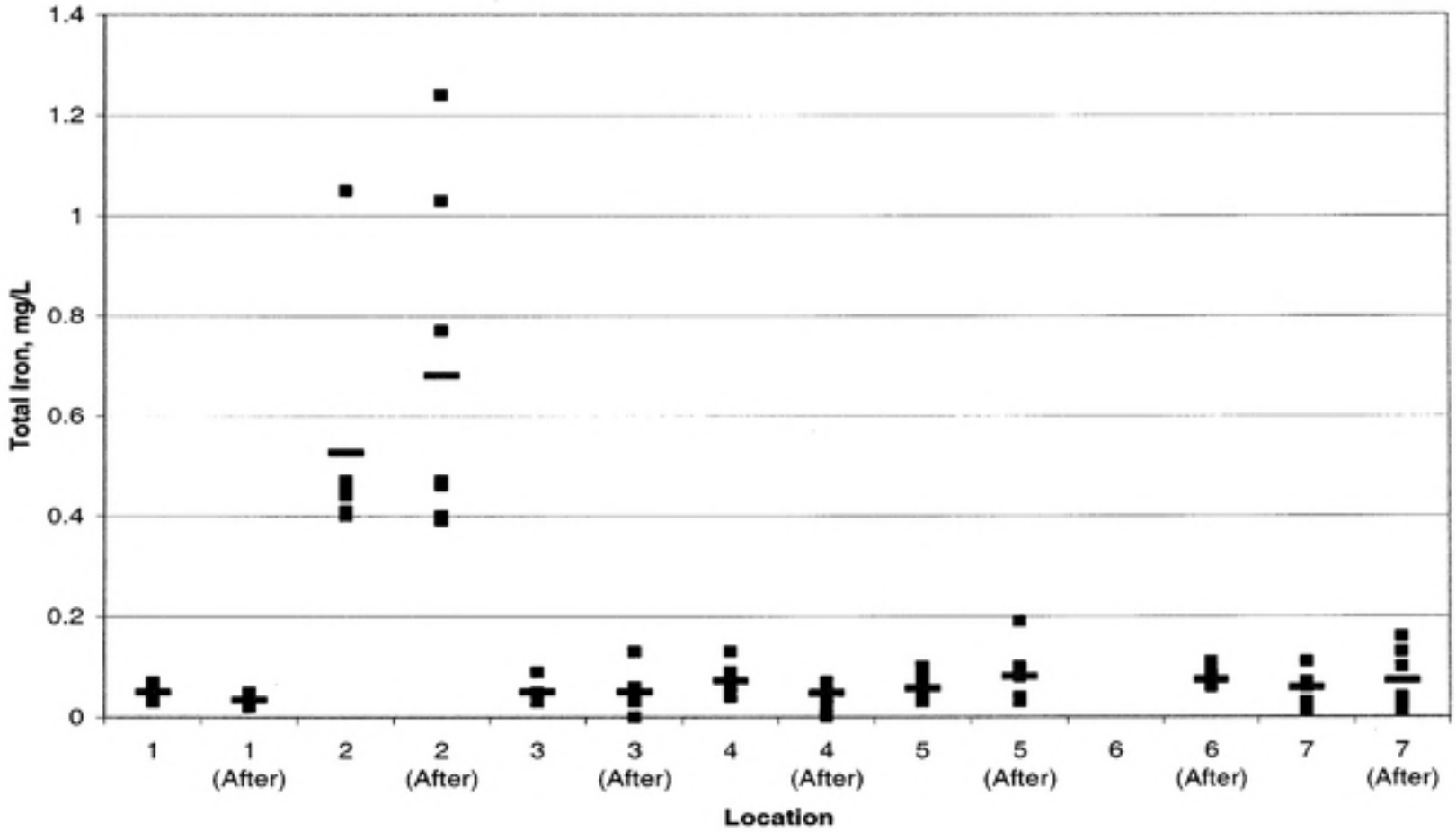



Figure 28. Monthly (squares) and average (bars) total Fe in the finished water (Station 1) and at all stations in the DS, before and after the conversion to MF.

**Chemical Costs**

Figure 27 shows the dosage in pounds per 1,000 gal for each chemical in use before and the after the conversion to MF. Because water production was greater in the months following the conversion to MF, the impact of the conversion on cost must be measured by the cost per 1,000 gal. One possible reason for higher dosing of fluoride and phosphate corrosion inhibitor after the conversion was pump metering rate control. The metering rate is supposed to be integrated into the SCADA system to feed a constant dosage regardless of the flow rate as the high service pumps deliver water to the DS. Problems were experienced at the WTP in integration of the high service pumps into the SCADA system such that the chemical dosing pumps were set at a constant rate instead of a flow proportioned rate. The dosing of phosphate corrosion inhibitor after conversion to MF was not recorded in the daily records. Problems were encountered in delivery of the corrosion inhibitor to the WTP and with the metering pump that prevented dosing for the period following start up of MF treatment system. Because the exact date on which dosing resumed and the dosage were not available, the same dosage as was used before the conversion to MF was assumed for the entire period after the conversion.

Using the dosages shown in Figure 29 and the unit costs for chemicals, the conversion to MF resulted in a 33.6 percent reduction in chemical costs (from $ 0.247/1,000 gal to $ 0.164/1,000 gal). Gere et al. (1997) reported a similar savings (30 percent).

Several changes in chemical usage account for the savings after conversion to MF. The largest reduction in chemical usage was due to the elimination of alum coagulation given that it accounted for 22.7 percent of the chemical costs before MF.

Elimination of alum also reduced the amount of lime needed because it was no longer necessary to adjust pH upward after addition of alum to reach the target range of pH (6.2- 6.5) for optimum coagulation. PAC was also discontinued after the conversion although the reason is not clear because PAC is not needed for alum coagulation. Approximately 4 percent of the overall chemical costs were due to PAC before the conversion. The cost per 1,000 gal for fluoride and phosphate-containing corrosion inhibitor remained the same because their dosages are always in proportion to the flow rate through the WTP.

Although the net result of the conversion to MF was a savings in chemical costs, the chlorine usage increased by almost150 percent. The increase in usage is mainly because of the very large chlorine demand through the summer months (see Figure 22). The demand could have been caused by the increase in TOC when coagulation was eliminated and by the drought conditions; in addition, high water temperatures would increase the demand. The additional chlorine cost by the 150 percent increase in dosage was estimated at $0.005/1,000 gal, which is only 4.1 percent of the total chemical cost after the conversion.

The fact that chloramination was not practiced during several of the seven months following the conversion accounted for lower cost of ammonia which is thus an artifact. If the calculation of ammonia costs is modified to account for only the volume of water treated when combined chlorine was used, then the ammonia cost was 151 percent higher after MF. Thus, the percent increase in ammonia cost was very similar to the percent increase in chlorine costs.

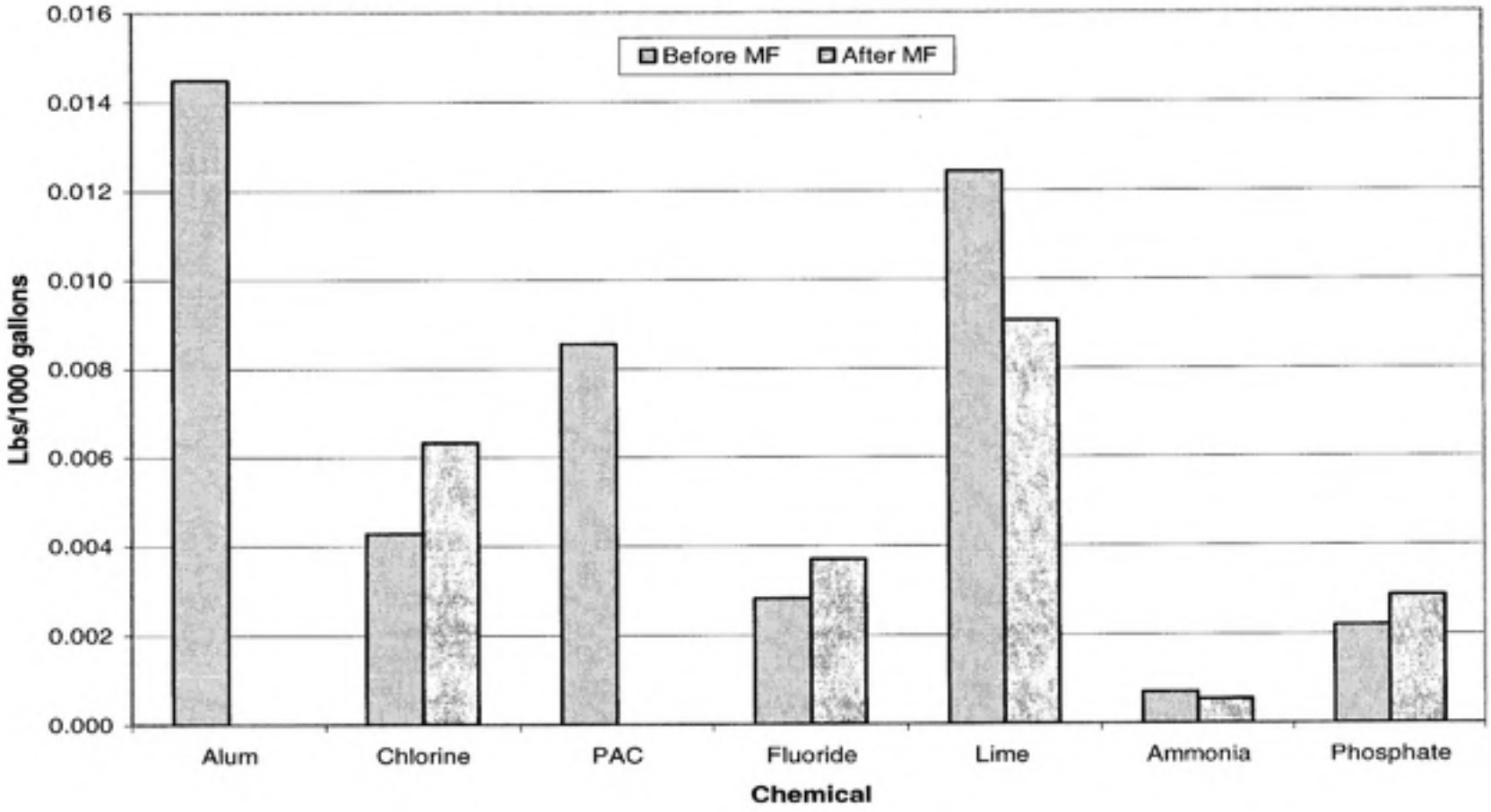



Figure 29. Comparison of chemical dosages before and after the conversion to MF.

**Power Costs**

Figure 30 shows power consumption and average annual costs before and after MF. Power consumption and associated costs were calculated from electrical billings acquired from the WTP. It includes the power needed to pump the water from the reservoir to the WTP and to pump the water from the clear well to the DS. Thus, the power cost associated with MF is the difference between the values obtained before and after the conversion to MF. The increase in power cost is about $ 0.14 per 1,000 gal. Correspondingly, the average power consumption increased from 17,770 to 33,586 kWh, or by 189 percent primarily because of the costs associated with pressurization of the MF system. Based on the information from Dharmappa and Hagare (1999), the cost of power was $0.093 per 1,000 gal based on 1.1 mgd facility. However, power consumption is not linear with flow rate. In addition, the cost per kWh may be higher for lower consumption rates.

As the fouling rate increases with time, the transmembrane pressure must increase which also increases the power costs. The cost of power can vary depending on total power consumption, time of day, and seasons. The average cost per kWh was found from the electrical billing records to decrease from $0.089/kWh before the conversion to MF to $0.067/kWh after the conversion; this may be due to the increase in usage rate.

**Backwash Water Disposal**

Backwash and CIP disposal costs were calculated from the daily operations data. Before the conversion to MF, backwashing only occurred once per day, either at startup or shutdown of the WTP. The backwash water was discharged to the Carthage sewer for treatment at the wastewater treatment plant (WWTP) at a cost of about $ 1.00 per 1,000

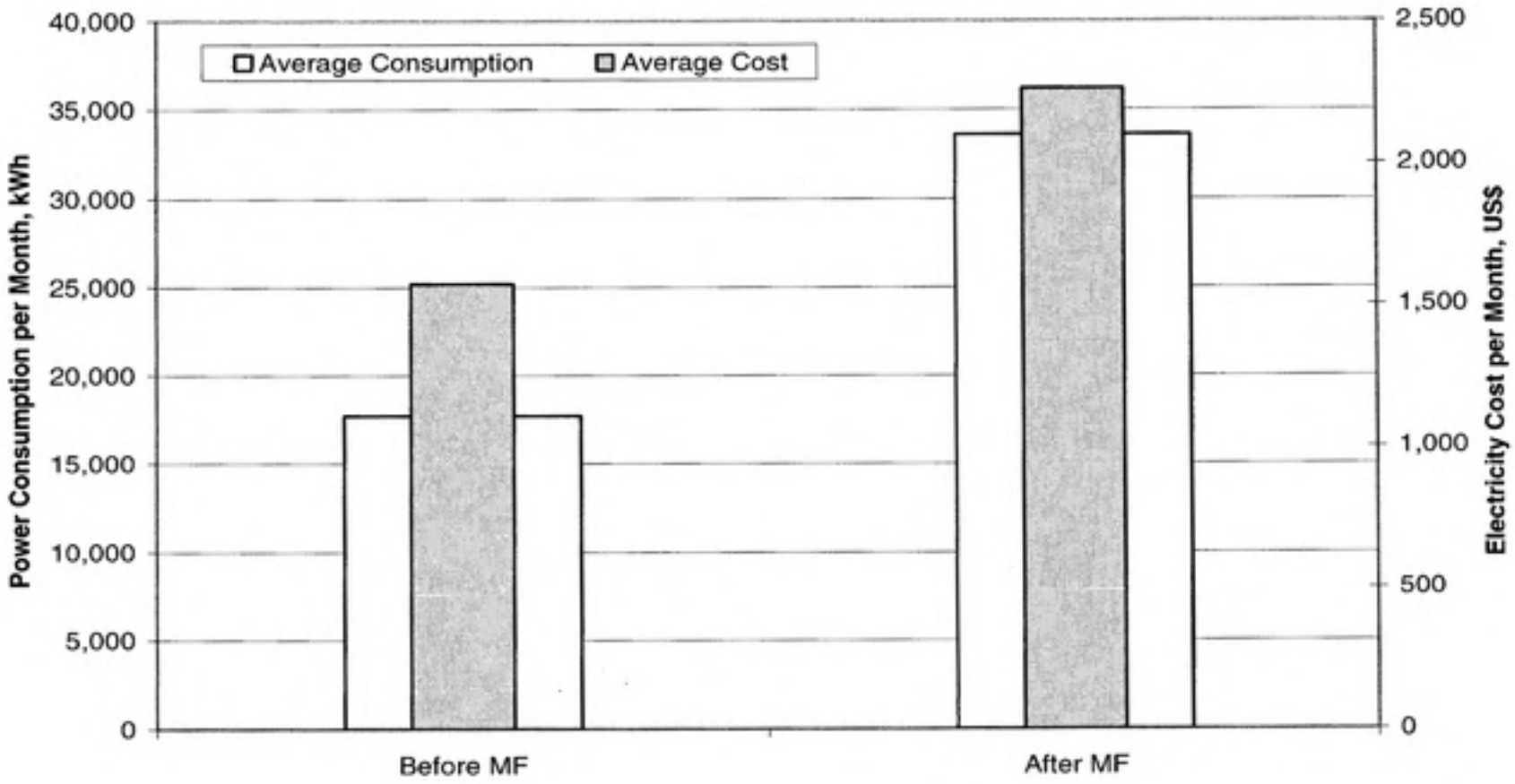



Figure 30. Comparison of power consumption and power cost per month before and after the conversion to MF.

gallons. Carthage plant personnel are not required to record the number times that the MF is backwashed. Therefore, an estimate was made based on the number of hours of operation and 30-minute backwash intervals, including an initial backwash before the MF was placed into serviced. The MF backwash water is nominally discharged into the Carthage sewer as was the case with the backwash water from conventional filtration.

Figure 31 shows the monthly average of the daily number of MF backwashes. The number of daily backwashes is fairly constant from month to month, averaging approximately 23 per day. Approximately 525 gallons of backwash water are discharged to the sewer after each backwash. Thus, the average daily discharge of backwash is 12,075 gallons which is approximately 3.5 times larger than the backwash volume discharged during conventional filtration. Using the same cost per volume discharged as for disposal of backwash from conventional filtration, the disposal cost of MF backwash was $ 0.058 per 1000 gallons ($0.015/m$^3$). This disposal cost is fairly close to $0.01/m$^3$ as reported by Chellam et al. (1998) for MF/UF systems. Although the disposal costs are higher than for conventional filtration, the ability terms of MF to meet the new MCLs for turbidity (<0.5 ntu) should be considered; the old conventional filtration system would not be able to meet this MCL consistently based on the performance records collected in this study.

**CIP and Disposal**

Chemical cleaning costs were calculated from the disposal cost and the chemical. Based on the frequency of CIPs conducted thus far at the Carthage WTP (see Table 4), an average CIP frequency of 80 days was selected for cost estimation. Each chemical requires a 55-gal drum of cleaning agents that is mixed into approximately 1,500 gal of

water. Each 55-gallon drum of chemical solution costs approximately $2,200. The cleaning solution is reused for four cleaning cycles. Given that five CIPs are assumed per year on each of two skids, five, 55-gal drum of cleaning agent would be consumed per year at a cost of about $ 6,600. The assumption was also made that the cleaning solution is diluted by 50 percent prior to discharge into the town sewer. Based on the total volume of cleaning agent and this dilution factor, approximately 15,500 gallons would disposable discharged per year which at a sewer charge of $ 1.00 per 1000 gal amounts to a disposal cost of only $15.50/year. Thus, the CIP costs are almost entirely due to the cost of the chemical cleaning agent. Given that no chemical agents are needed for backwashing of a dual-media filter, the need for CIP in MF greatly increases the cost of backwash water disposal over that of conventional filtration.

**Distribution of Operating Costs**

The distribution of operating costs among different cost categories was also compared for the old conventional facility and new MF facility. The annual budget and payroll were obtained from the Town of Carthage to make this comparison. From the daily operations data collected by plant personnel, chemical and backwash costs were estimated over the study period for both the new and old facilities. Power costs were obtained from the monthly power bill while CIP costs were estimated from preparation of one 55-gal drum of chemical agent and four CIPs (two on each skid). All other costs were estimated to be administrative or other costs incurred by each facility. The pie charts presented in Figures 32 and 33 depict the percent contribution of each cost component towards the total budget for the conventional and MF facility, respectively.

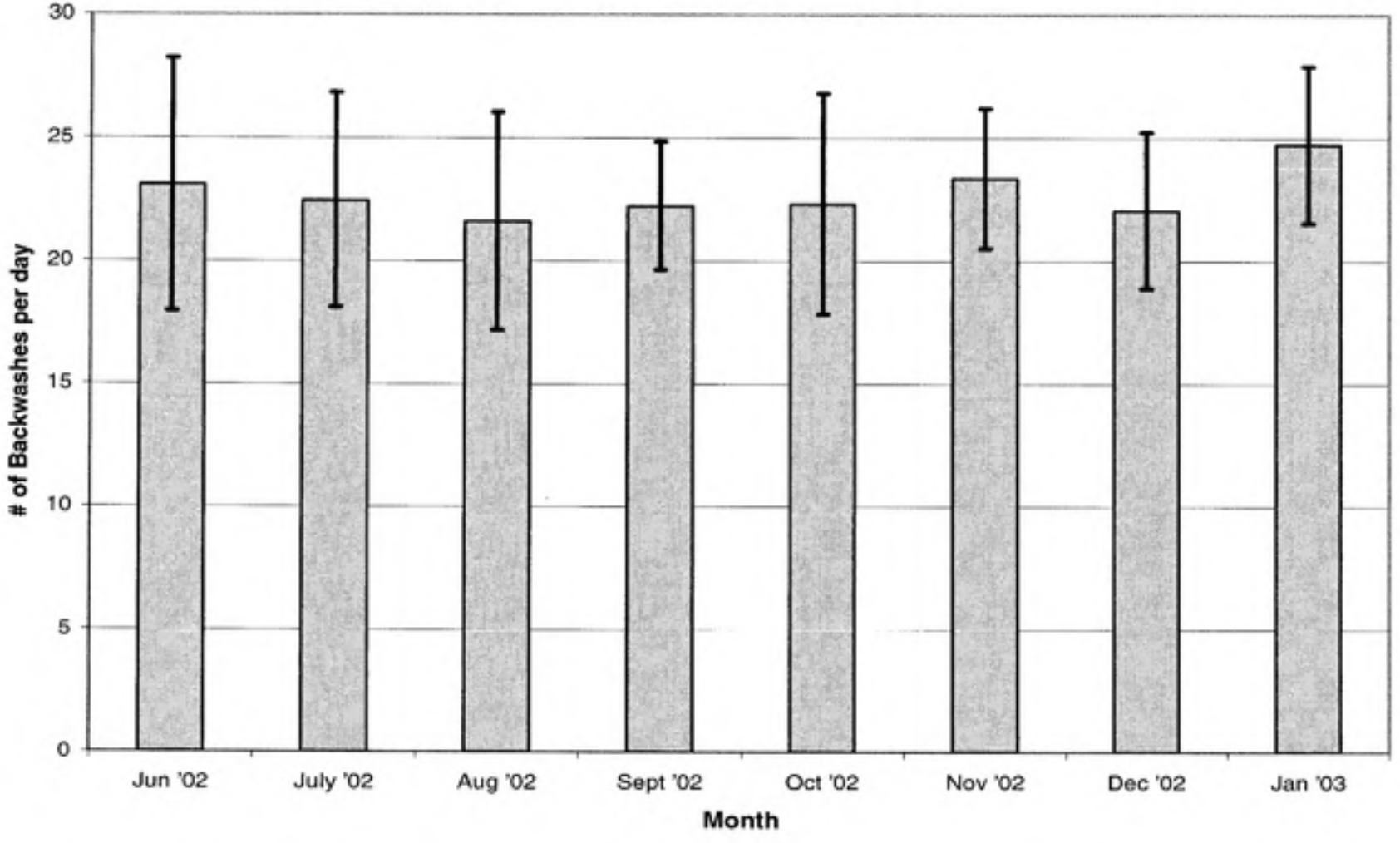



Figure 31. Monthly average of daily MF backwashes

Labor costs were again assumed to be the same for both facilities. While labor costs are nearly 40 percent of the budget for MF, more efficient plant operation with the SCADA system is anticipated to reduce labor costs. Administrative/Other costs remain practically the same (33.4 percent for MF vs. 35.3 percent for conventional filtration). Power and residuals disposal costs together account for a larger portion of the budget than before, while chemical costs decreased by 6.4 percent. The addition of CIP costs was a very small percentage (3.4 percent) of the total cost.

**Other Costs**

Other costs that were calculated were for membrane replacement, capital recovery, and labor. The membrane replacement costs are listed in Table 9. Each of the cost elements will be discussed separately.

Membrane replacement costs were calculated from an estimate cost of the membrane modules as given by the project engineer responsible for the Carthage facility (Bishop, 2003); a membrane life of 5 years was assumed. Costs were annualized over a 20-year lifetime at interest rates of 6, 4 and 2 percent and were then normalized to a cost per 1,000 gal at 0.75 mgd. Table 9 shows high, average and low estimates of costs that were derived from a high and low cost of each membrane modules ($ 1,150 vs. $ 650); an average value was calculated from these two estimates. The low cost applies only to purchase of 100 or more modules. In the case of Carthage, each membrane skid contains 48 modules, thus giving 96 in total.

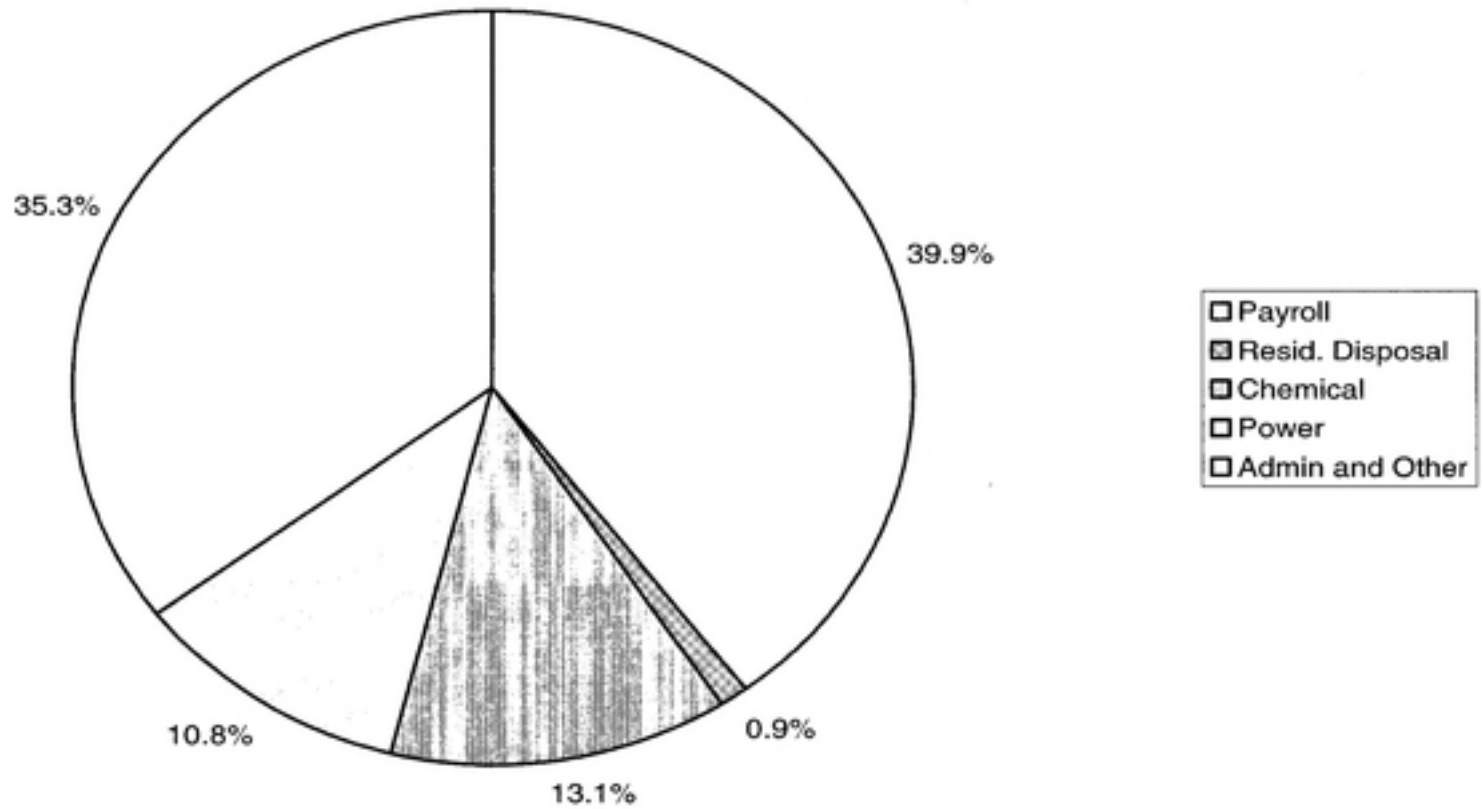



Figure 32. Percent distribution of operating costs of the conventional filtration facility at Carthage during the sample period (October 2001 to May 2002)

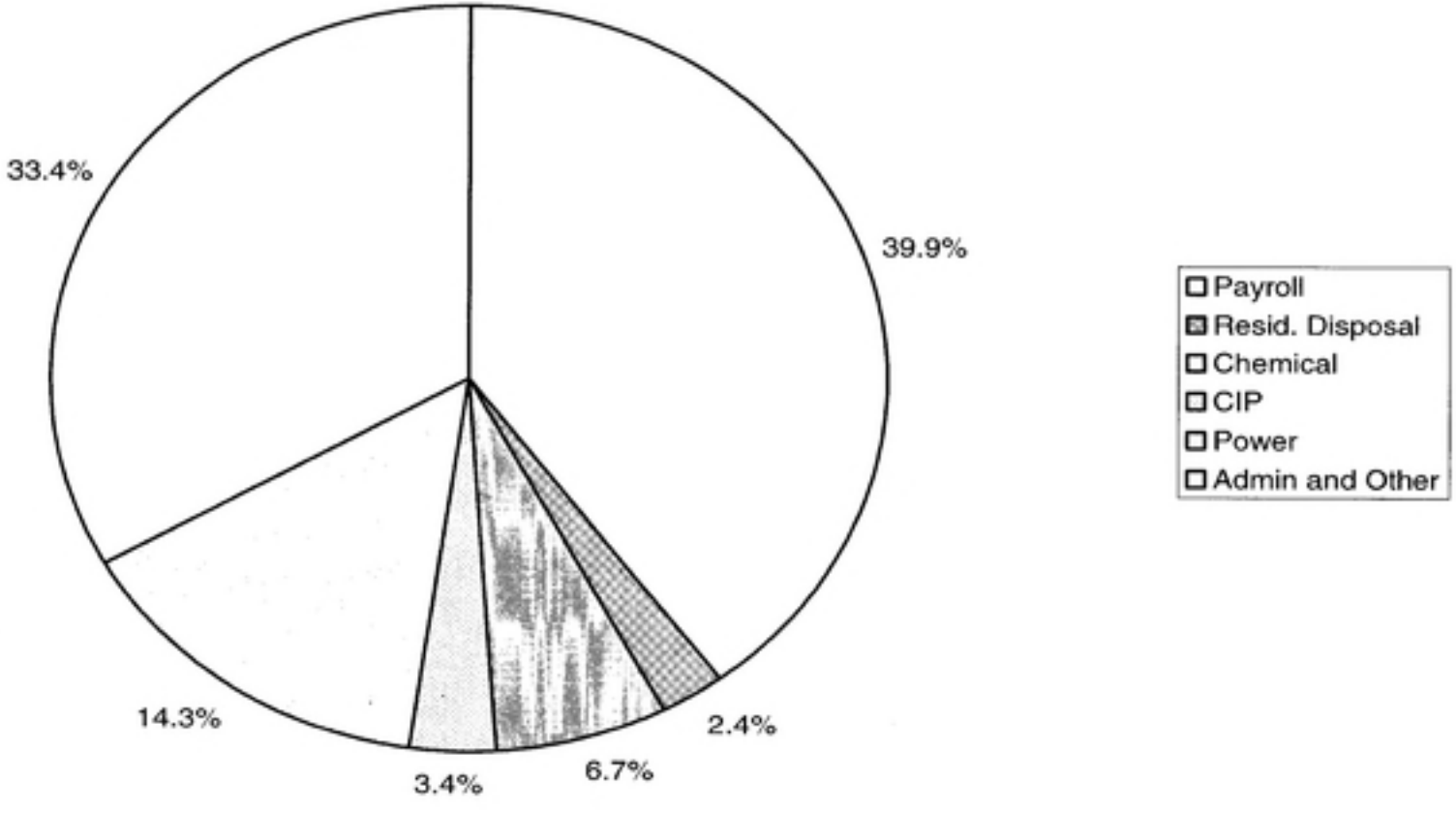



Figure 33. Percent distribution of operating costs of the MF facility at Carthage during the sample period (June 2002 to January 2003

Table 9. Summary of Membrane Replacement Costs

| | Annual Cost | | | Cost/1,000 gal | | |
|---|---|---|---|---|---|---|
| Interest Rate (%) | 6 | 4 | 2 | 6 | 4 | 2 |
| High | $38,508 | $32,502 | $27,206 | $0.141 | $0.119 | $0.099 |
| Low | $21,765 | $18,371 | $15,276 | $0.080 | $0.067 | $0.056 |
| Average | $30,136 | $25,436 | $21,151 | $0.110 | $0.093 | $0.077 |

Membrane replacement costs cited by others were given as a cost per $m^2$ of membrane surface area; these range from $ 70.62/$m^2$ (Dharmappa and Hagare, 1999) to $100/$m^2$ (Sethi and Wiesner, 2000). Given that each module at the Carthage WTP contains approximately 15 $m^2$ and given the average cost per module, which was derived from the high and low estimates provided by project engineer, the membrane cost is $60/$m^2$, or about $10/$m^2$ less than those reported elsewhere. If the high estimate of cost is used, the membrane cost is $76.67/$m^2$, which is close to the value reported by Dharmappa and Hagare (1999).

The membrane replacement cost was compared to an estimated cost to replace the dual media filter. A summary of the dual filter media replacement costs are shown in Table 10. The existing filters in the old conventional treatment plant had an area of 180 square feet with approximately 2.5 feet of anthracite coal and an unknown depth of sand. For the comparison, one foot of silica sand was assumed. Anthracite coal and silica sand were estimated at $10/$ft^3$ and $6/$ft^3$. Both estimates were obtained from The Roberts Filter Group (Duffy and Rogenmuser, 2003). In addition, it was assumed that two filters would need media replacement once every 10 years. The annualized costs were calculated over a 20 year plant life with a 6 percent interest rate. Finally, costs were normalized to a cost per 1,000 gallons produced.

| Table 10. Summary of Dual Media Filter Media Replacement | |
|---|---|
| Annual Cost | Cost per 1000 gallons |
| $1,946 | $0.0071 |

The construction cost for the entire MF facility was $ 2,822,594. In addition to the costs directly associated with the MF facility, the total cost includes demolition of the old WTP, installation of raw water supply line from the reservoir to the WTP, and construction of a new CT tank. As such, the capital cost would be higher than for an MF facility that was retrofit into an existing WTP. This entire amount has already been paid by a combination of a grant from the State of North Carolina for $2,000,530 and a cash payment by the Town of Carthage. However, other potential facilities may not be able to obtain a state grant. Thus, a capital recovery factor was used to provide a more realistic way to estimate capital costs. The amortized capital cost component of the total costs shown in Table 11, therefore, was obtained by using interest rates of 6, 4, and 2 percent over a plant life 20 years. These capital costs were also prorated per 1,000 gal of water produced in the table. A plant capacity was estimated to make this calculation. WTP facilities, as in Carthage, are typically constructed with excess capacity for the future. The current average flow rate at Carthage is 0.4 mgd, and the new MF facility is rated at 1 mgd. A capacity of 0.75 mgd was chosen, therefore, for a mid-life estimate.

The actual cost of the MF facility was also compared to that of a conventional WTP of the same capacity. A power cost function model for a conventional WTP, which was developed at UNC (Kirsch, 2003):

$$Capital(\$) = 257 \times \left(6222/4406\right) \times Q^{0.67} \qquad (5)$$

where Q is the flow rate in gal/day and the reference year for capital cost is 2001. The model coefficients were obtained by Kirsch through a calibration using costs of WTP

facilities in North Carolina. Based on a design flow rate of 1.0 mgd, the estimated capital cost for a conventional facility was $ 3,800,307, which is $ 977,713 more costly than the MF facility installed at Carthage. Therefore, MF is a fiscally competitive alternative to conventional filtration as reported by other authors (Owen et al., 1995; Wiesner and Chellam, 1999; Pickering and Wiesner, 1993). In addition, the capital cost for MF is approximately 74 percent of the conventional facility cost, which is close to the 90 percent reported by Dharmappa and Hagare (1999).

A comparison of annual capital recovery costs for conventional and MF facilities was made using capital recover factors of 6, 4, and 2 percent over a plant life of 20 years. The resulting capital recovery values are shown in Table 11. On this basis, the capital cost of MF is seen to be significantly less than for conventional treatment.

Table 11. Summary of Capital Recovery Costs ($) for a 1-mgd Facility Operating at 0.75-mgd

| | Conventional Facility | | MF Facility | |
|---|---|---|---|---|
| Interest Rate | Annual Cost | Cost/1,000 gal | Annual Cost | Cost/1,000 gal |
| 0.06 | 331,387 | 1.211 | 246,130 | 0.899 |
| 0.04 | 279,703 | 1.022 | 207,743 | 0.759 |
| 0.02 | 232,579 | 0.850 | 172,743 | 0.631 |

Labor costs used in Table 12 were obtained by using the annual payroll at the Carthage WTP. The payroll has not changed from conventional filtration to MF. Thus, the labor costs were estimated to be the same although this raises the question of the time distribution of labor and whether the same amount of labor is still required for the new facility.

Table 12. Summary of Labor Costs ($) for a 1-mgd Facility Operating at 0.75-mgd

| Annual Cost | Cost per 1000 gallons |
|---|---|
| 77,352 | 0.283 |

Interviews with plant personnel were performed to determine the differences in duties between operating the conventional and MF facility. Daily duties at the new facility include: checking the MF system for leaks and loose fittings; checking the SCADA for alarms; and conducting laboratory analyses for process control. The most time consuming duties are membrane replacement, CIPs, sonic tests to measure membrane integrity, and checking of fittings and valves. Keeping the very old and worn equipment operating was the most time consuming duty before the conversion to MF. The maintenance consisted of replacing belts, drives and electrical equipment as well as lubrication. The initial set up and operation of the new MF facility was considered a major difficulty over the past six months. However, the plant duties should eventually become more process than maintenance related. The SCADA system appears cause difficulties for the operators. While the SCADA system has been helpful, it was difficult for the operators to learn how it operates. SCADA offers the ability for the personnel to troubleshoot problems easily because of the online monitoring (alarms). It is speculated that the majority of operator difficulties related to SCADA is due to an inadequate knowledge of computers in general.

**Hypothetical Example: Total Costs of Conventional and MF Treatment**

While the Carthage plant data showed that the conversion to MF has caused the operating costs to increase significantly, the analysis does not account for a comparison of capital costs between conventional and MF treatment. The inclusion of capital costs can only be done for a hypothetical example assuming that Carthage would have considered the choice between new conventional plant and an MF plant. The total costs include amortized capital costs and operating costs per 1,000 gal of water produced. The

total costs and the breakdown into component costs are given for conventional filtration and MF in Figure 34. Conventional filtration is slightly cheaper expensive than MF ($ 1.970 vs. 1.978 per 1,000 gal). Capital recovery is the largest percentage of total costs for both facilities (61.4 percent for conventional vs. 45.5 percent for MF). However, the capital recovery cost for MF has been inflated because the capital costs at Carthage included the cost for demolition of the old WTP, a new raw water supply line and a new CT tank.

The labor costs were assumed to be the same for both facilities as noted above. These costs can also be expressed as a percent of the total cost (14.3 percent for conventional vs. 14.3 percent for MF). The percent contribution of labor for the MF treatment is very close to 20 percent as reported by Dharmappa and Hagare (1999). It is anticipated that labor costs for MF will decrease in time due to the implementation of the SCADA system and its ability to monitor and troubleshoot problems more rapidly than at present Figure 34 also shows that the percent contributions to total costs for MF treatment are different than for conventional treatment, mainly due to lower capital costs recovery, higher power costs (17.4 vs. 10.5 percent), lower chemical costs (8.3 vs. 12.5 percent) and higher backwash disposal costs (2.9 vs. 0.9 percent). Dharmappa and Hagare (1999) gave a power cost of 31 percent while Gere et al. (1997) gave 28.6 percent.

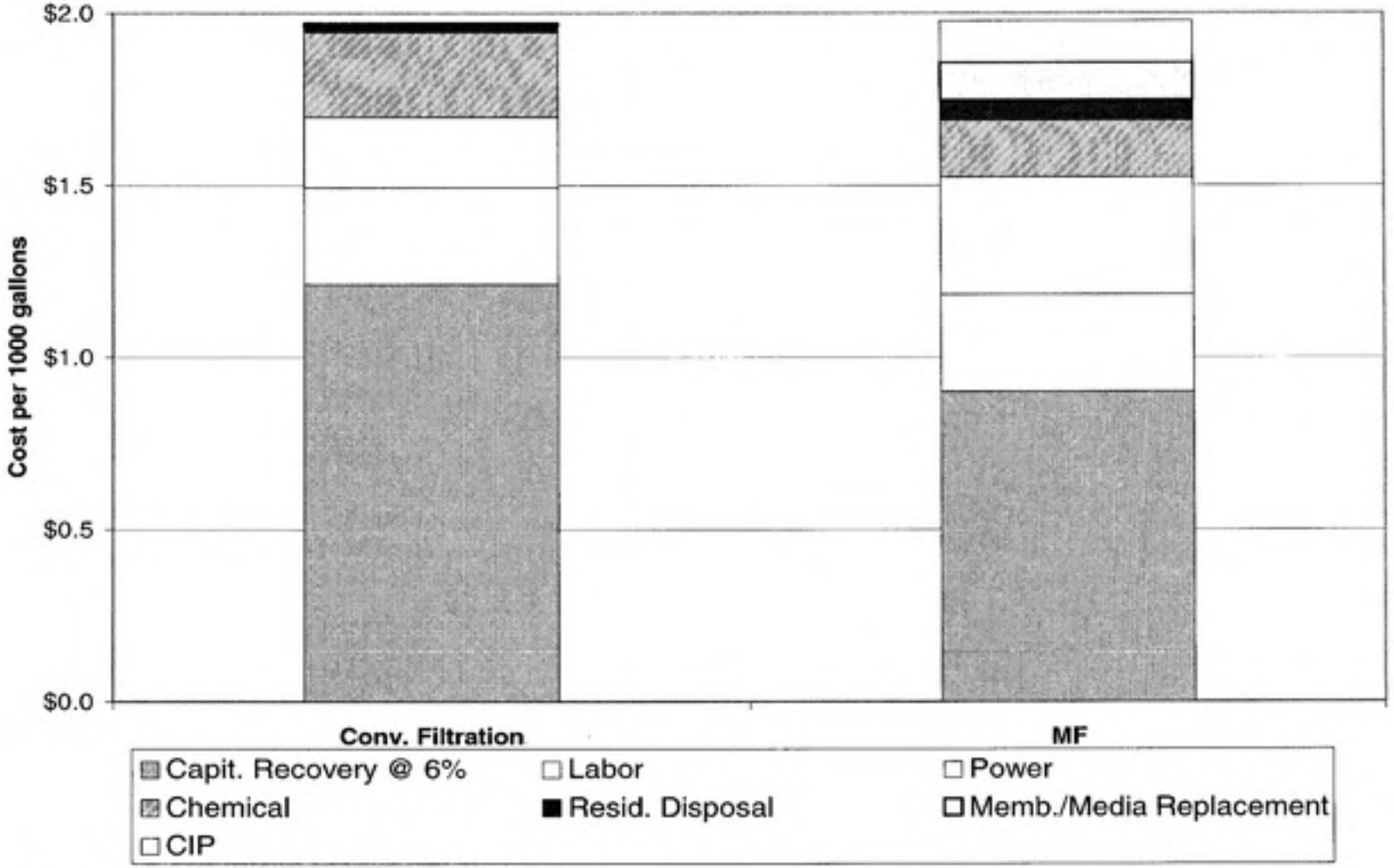



Figure 34. Comparison of the total costs for 1-mgd conventional and a 0.75-mgd MF treatment facility

If capital recovery was not included in the total costs shown in Figure 34, then the power cost would increase to 31.8 percent of the total costs for MF, which is much closer to those reported by others. The 33-percent reduction in chemical costs for MF treatment agrees with the 30-percent reduction reported by Gere et al. (1997). Residuals disposal consumes over three times more of the overall cost for MF than for conventional filtration. Backwash costs are dominated by the CIP costs. Membrane replacement costs (based on $ 900 per module) are obviously a cost that is not included in conventional filtration but is represented by media replacement. The cost to replace the membranes is about 15 times the cost to replace the filter media per 1,000 gallons ($0.110 vs. $0.0071). Membrane replacement cost represents 5.6 percent of the total cost for MF whereas the dual media replacement only represents 0.4 percent of the total cost for conventional treatment.

## CONCLUSIONS

Implementation of MF in place of conventional filtration for water treatment in Carthage improved the water quality when measured by turbidity and particle counts. Finished water turbidity and particle counts from MF have been significantly lower that that produced by conventional filtration. The average reductions in turbidity and particle counts were 95.4 and 98 percent, respectively, in contrast to 84.9 and 71 percent, respectively for conventional filtration. In terms of absolute values, the average turbidity following MF was 0.046 ntu compared to 0.83 ntu for the old conventional WTP. The removal of Fe was about 96 percent by MF although this was based on only two monthly measurements in the permeate.

While MF should theoretically provide a nearly absolute barrier to particles with diameters greater than the nominal pore size of the membrane (0.2 µm), the average particle count as measured by counters that only measured those particles greater than 2 µm was 36.7 particles/mL. Particles greater than 2 µm in the permeate could be explained by failure of mechanical seals or individual hollow fibers. Membrane integrity tests are performed every three hours by plant personnel to insure that these failures are minimal. Alternative explanations include the presence of an artifact in the measurement technique and the presence of pores larger than the nominal pore size. Micro-bubbles that are formed during the high pressure, air backwash cycles could be released into the permeate and be counted in the particle counter (Taylor et al., 1999; Kothari and Taylor,

1998). Although, each hollow-fiber membrane is rated at a nominal pore size of 0.2 µm, a study by Sadr-Ghayeni et al. (1999) found that pores could be twice the nominal size.

Total coliforms and HPC were measured in the MF permeate as well. No coliforms were detected over the entire study period. Prior to July 2002, atypical colonies were noted in the total coliforms test although these were not confirmed as members of the coliform group. These organisms were not found again after the CIP in July 2002. One possible explanation for their initial presence was contamination on the permeate side of the membrane which was later removed by the CIP. This contamination could have occurred because the permeate lines were not disinfected before the membrane modules were placed into service. HPC in the permeate generally decreased after CIPs which suggests that regrowth had occurred in either the permeate piping or sample lines. A follow-up study concluded that regrowth had not occurred in the sample line but this did not rule out the possibility of growth on the permeate side of the membrane surface.

Turbidity and particle counts in the DS were shown to be lower after the conversion to MF. Nonetheless, both parameters were found to increase somewhat at sampling stations most distant from the WTP. In addition, both parameters were much higher at the station that was located in the small, isolated section of cast iron pipe than any other station in the DS. In fact, the turbidity values were far above the proposed MCL of 0.5 ntu and also above the current MCL of 1 ntu. This was caused by excessive corrosion that released particles from the pipe surface as evident also from the very high total Fe levels, which routinely exceeded 0.4 mg/L and reached over 1 mg/L.

The discontinuance of coagulation caused the TOC to increase significantly in the DS. MF is not expected to be as effective at removing TOC as conventional treatment that includes chemical coagulation. Only the particulate form of TOC will be removed by MF and this generally accounts for a very small fraction of the total. The failure to remove TOC has three possible serious consequences. First, more TOC will cause more chlorine demand thus making it more difficult to carry a residual into the DS. Second, more TOC coupled with loss of residual will lead to more bacterial regrowth. Third, more TOC provides for more formation of DBPs within the WTP when chlorine is added to meet the CT rule. A separate study of DBPs was conducted during this study period and will be reported separately (DiGiano et al. 2003). The DBP data show violations of MCLs for both TTHM and $HAA_5$ both before and after the conversion to MF.

HPC data showed that bacterial regrowth occurred within the DS when disinfectant residual could not be maintained. Regrowth was evident both before and after the conversion to MF, especially at stations in the DS that have long residence times. The regrowth problem was particularly acute in this DS where HPC values routinely exceeded 10,000 cfu/mL at several sampling stations. In fact, values of over 100,000 cfu/mL were found on a few occasions, especially at a station that is far from the WTP and behaves like a dead-end. Even after accounting for the fact the higher HPC are usually found when using the more sensitive R2A agar than the Standard Plate Count agar as is used for state reporting purposes, these HPC values are still considered very high. In addition, very high HPC were found in a small section of the DS that remains with cast iron piping. Although this site is very close to the WTP (1.2 miles), the total

chlorine residuals (both combined and free depending upon operations) were very low because of the reduction reaction between corrosion products and chlorine.

The summer of 2002 was marked by an extreme drought that could have negatively affected water quality. One of the most important consequences were extremely high chlorine demands that made it impossible to carry residual through the 1 MG clear well. One theory had been that the chlorine demand was caused by cleaning of the clear well just prior to start up of the MF, but this was not proven. Another theory could be an increase in reduced material from the poor quality waters at the lower depths of the reservoir from which water was being withdrawn. The difficulty in maintaining a chlorine residual was compounded by problems with delivery of ammonia to the WTP and with metering pumps such that chloramination was abandoned for several months after the MF startup.

Operating costs over the 7-month period following conversion to MF were shown to be higher for MF ($ 0.69/1,000 gal) than for the preceding 7-month period of conventional treatment ($ 0.47/1,000 gal). Important contributors to the increase in costs between conventional treatment and MF were costs for power, residuals disposal and CIP chemicals. These increases were offset significantly by a decrease in chemical costs for alum and lime. Labor costs were the same because the same personnel have operated both facilities. Interviews with plant personnel determined that duties have shifted from maintenance (e.g. changing old equipment, equipment lubrication, etc.) to more process control for MF. Labor costs could decrease once full advantage is taken of the SCADA system, which provides the operators with more accessible information about the operation of the system, such that down-town caused by troubleshooting can be

minimized. However, the Town of Carthage would need to make a decision to reduce staffing in order to save labor costs. Whether this is possible remains to be determined.

To compare equally sized facilities, the total costs (capital plus operating) for a 1-mgd conventional filtration facility theoretically operating at 0.75-mgd were compared to the MF facility operating theoretically at 0.75-mgd. The total costs of the MF facility were shown to be slightly more than for a conventional facility ($ 1.978 vs. 1.970 per 1,000 gal). This cost comparison suggests that if a new plant were to be constructed, the higher operating costs of MF (as was shown from the Carthage operating records) would be offset somewhat by its lower capital costs such the total costs for MF and conventional treatment are nearly equal. The MF facility would appear more favorable if the costs for demolition of the old WTP, a new raw water supply line and a new CT tank were not included.

## RECOMMENDATIONS

1. Possible explanations for measurements of particle counts for particles greater than 2 μm and for HPC in the MF permeate should be explored
2. Coagulation ahead of MF is needed to remove TOC so as to reduce chlorine demand, reduce DBPs (although these are not reported here), and reduce the potential for bacterial regrowth in the DS.
3. Long storage in the 1 MG clear well (estimated as more than 2.5 days) causes deterious effects on water quality, foremost by allowing time for chlorine oxidation reactions. Replacement of the 1 MG clear well with a smaller tank would be possible if the WTP were to operate continuously rather than 12 hours per day. Continuous operation of the plant would only be possible if remote control were used to avoid the need for extra plant shifts
4. The loss of both chloramines and chlorine residual in the DS is cause of serious concern with respect to bacterial regrowth; a complete analysis of the water quality and residence time with the aid of water quality and hydraulic model of the DS is needed
5. More detailed study of labor costs to determine if the plant can operate with less staff

6. The cost analysis needs to be repeated to include the possibility that a chemical coagulant would be used ahead of MF to meet future MCLs for DBPs

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# APPENDIX I

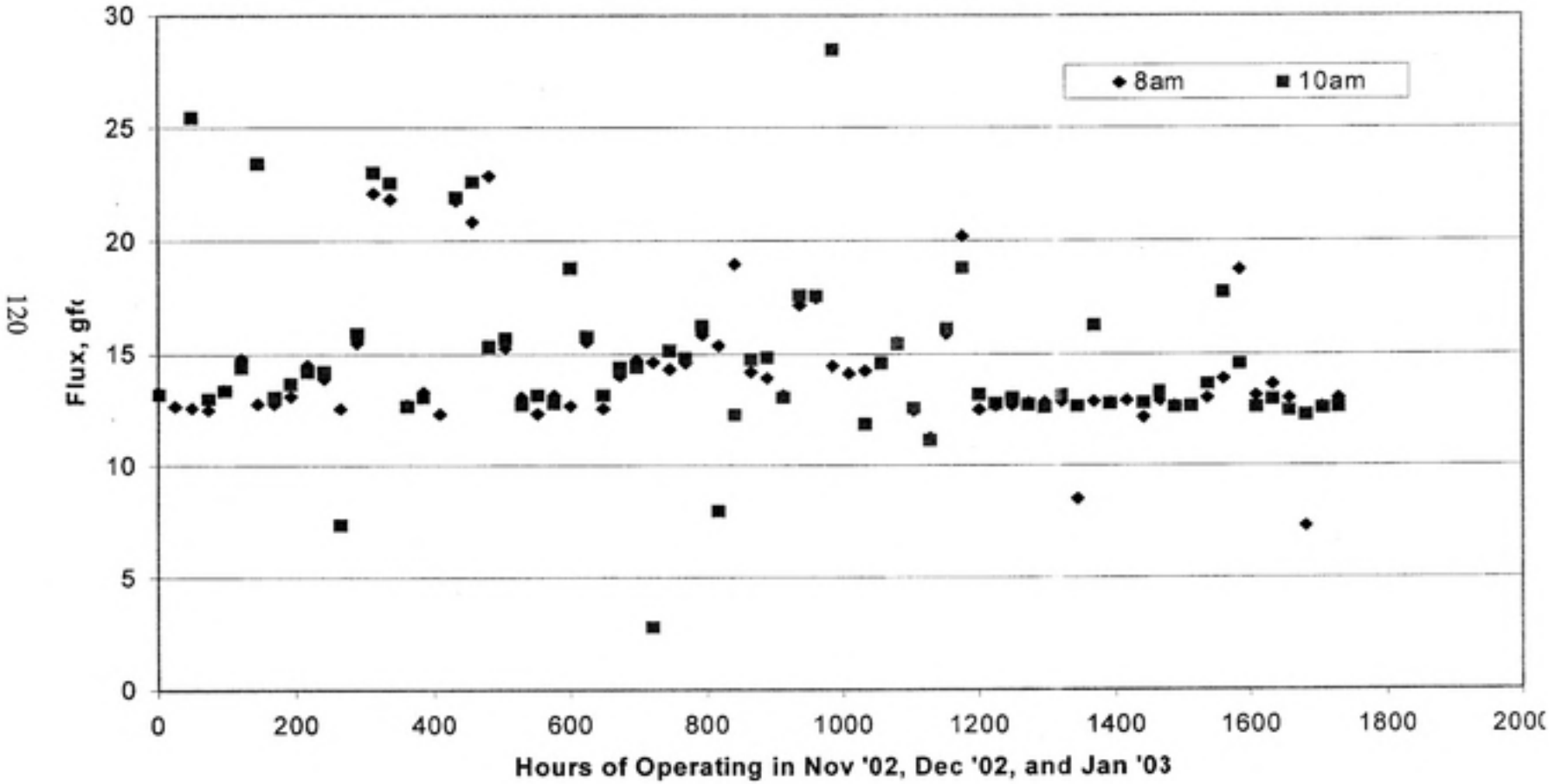



Figure A1. Membrane Flux from November 2002 to January 2003.