Use of Treated Municipal Wastewater as Power Plant Cooling System Makeup Water: Tertiary Treatment versus Expanded Chemical Regimen for Recirculating Water Quality Management

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ABSTRACT

Treated municipal wastewater is a common, widely available alternative source of cooling water for thermoelectric power plants across the U.S. However, the biodegradable organic matter, ammonia-nitrogen, carbonate and phosphates in the treated wastewater pose challenges with respect to enhanced biofouling, corrosion, and scaling, respectively.

The overall objective of this study was to evaluate the benefits and life cycle costs of implementing tertiary treatment of secondary treated municipal wastewater prior to use in recirculating cooling systems. The study comprised bench- and pilot-scale experimental studies with three different tertiary treated municipal wastewaters, and life cycle costing and environmental analyses of various tertiary treatment schemes. Sustainability factors and metrics for reuse of treated wastewater in power plant cooling systems were also evaluated. The three tertiary treated wastewaters studied were: secondary treated municipal wastewater subjected to acid addition for pH control (MWW_pH); secondary treated municipal wastewater subjected to nitrification and sand filtration (MWW_NF); and secondary treated municipal wastewater subjected nitrification, sand filtration, and GAC adsorption (MWW_NFG).

Tertiary treatment was determined to be essential to achieve appropriate corrosion, scaling, and biofouling control for use of secondary treated municipal wastewater in power plant cooling systems. The ability to control scaling, in particular, was found to be significantly enhanced with tertiary treated wastewater compared to secondary treated wastewater. MWW_pH treated water (adjustment to pH 7.8) was effective in reducing scale formation, but increased corrosion and the amount of biocide required to achieve appropriate biofouling control. Corrosion could be adequately controlled with tolytriazole addition (4-5 ppm TTA), however, which was the case for all of the tertiary treated waters. For MWW_NF treated water, the removal of ammonia by nitrification helped to reduce the corrosivity and biocide demand. Also, the lower pH and alkalinity resulting from nitrification reduced the scaling to an acceptable level, without the addition of anti-scalant chemicals. Additional GAC adsorption treatment, MWW_NFG, yielded no net benefit. Removal of organic matter resulted in pitting corrosion in copper and cupronickel alloys. Negligible improvement was observed in scaling control and biofouling control. For all of the tertiary treatments, biofouling control was achievable, and most effectively with pre-formed monochloramine (2-3 ppm) in comparison with NaOCl and ClO_2.

Life cycle cost (LCC) analyses were performed for the tertiary treatment systems studied experimentally and for several other treatment options. A public domain conceptual costing tool (LC^3 model) was developed for this purpose. MWW_SF (lime softening and sand filtration) and MWW_NF were the most cost-effective treatment options among the tertiary treatment alternatives considered because of the higher effluent quality with moderate infrastructure costs and the relatively low doses of conditioning chemicals required.

Life cycle inventory (LCI) analysis along with integration of external costs of emissions with direct costs was performed to evaluate relative emissions to the environment and external costs associated with construction and operation of tertiary treatment alternatives. Integrated LCI and LCC analysis indicated that three-tiered treatment alternatives such as MWW_NSF and MWW_NFG, with regular chemical addition for treatment and conditioning and/or regeneration, tend to increase the impact costs and in turn the overall costs of tertiary treatment. River water supply and MWW_F alternatives with a single step of tertiary treatment were associated with lower impact costs, but the contribution of impact costs to overall annual costs was higher than all other treatment alternatives. MWW_NF and MWW_SF alternatives exhibited moderate external impact costs with moderate infrastructure and chemical conditioner dosing, which makes them (especially MWW_NF) better treatment alternatives from the environmental sustainability perspective since they exhibited minimal contribution to environmental damage from emissions.

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EXECUTIVE SUMMARY

Treated municipal wastewater is a common, widely available alternative source of cooling water for thermoelectric power plants across the U.S. Approximately 81% of power plants planned for construction by the U.S. Energy Information Administration as of 2007 would have sufficient cooling water supply from 1-2 publicly owned treatment works (POTWs) within a 10-mile radius, while 97% of the proposed power plants would be able to meet their cooling water needs from 1-2 POTWs within 25 miles of these plants.

Results from the previous study indicated that it is feasible to use secondary treated municipal wastewater as cooling system makeup. However, the biodegradable organic matter, ammonia-nitrogen, carbonate, and phosphates in the treated wastewater pose challenges with respect to enhanced biofouling, corrosion, and scaling, respectively. In the previous study it was demonstrated that these problems can be controlled to varying extents through chemical management, by deploying different combinations and doses of biocides and corrosion and scale inhibitors. However, chemical management alone would be costly.

The overall objective of the study described herein was to evaluate the benefits and life cycle costs of implementing tertiary treatment of secondary treated municipal wastewater prior to use in recirculating cooling systems versus an expanded chemical regimen for managing the quality of the cooling water when secondary treated municipal wastewater is used as makeup.

Specific objectives of the research were as follows:

(a) Determine the benefits and costs of subjecting secondary treated municipal wastewater to additional treatment to remove NH_3 by nitrification, BOD by GAC adsorption, and particles by filtration prior to use as cooling water makeup.

(b) Determine different chemical treatment regimens required to manage cooling water quality for different levels of tertiary treatment ranging from no tertiary treatment to inclusion of nitrification and treatment for the removal of particles and residual organic matter.

(c) Perform comparative life-cycle cost analyses for different levels of tertiary treatment and the corresponding chemical treatment regimens.

(d) Determine critical economic, technical, and social factors that need to be considered in comparative evaluation of tertiary treatment alternatives for secondary treated municipal wastewater when used as cooling system makeup water.

The study comprised bench- and pilot-scale experimental studies with three different tertiary treated municipal wastewaters, and life cycle costing and environmental analyses of various tertiary treatment schemes. Sustainability factors and metrics for reuse of treated wastewater in power plant cooling systems were also evaluated. The three tertiary treated wastewaters studied were: secondary treated municipal wastewater subjected to acid addition for pH control (MWW_pH); secondary treated municipal wastewater subjected nitrification and sand filtration (MWW_NF); and secondary treated municipal wastewater subjected to nitrification, sand filtration, and GAC adsorption (MWW_NFG). Key findings and conclusions from the experimental studies, life cycle analyses, and sustainability analysis are presented below.

Experimental Studies with MWW_pH

Tertiary treatment of secondary treated municipal wastewater (MWW) by acidification increased the corrosiveness of the cooling water but reduced the scaling potential of MWW. Ammonia present in the MWW_pH water was aggressive towards metal alloys. Addition of 4-5ppm tolyltriazole (TTA) successfully reduced the corrosion rate of copper, cupronickel, and mild steel to acceptable levels according to industrial corrosion control criteria. Scale deposition was reduced with pH control at 7.8 and the addition of 5 ppm polymaleic acid (PMA). With this scaling mitigation method, the formation of calcium carbonate was inhibited and calcium phosphate was the primary form of mineral scale on both unheated and heated surfaces.

Biofouling control was difficult and unpredictable when using MWW_pH as cooling tower makeup water for an extended testing period. Although pH adjustment helped to mitigate the scaling problem, significant input of organic matter into the system still resulted in high variation in biocide residual. It was found that maintaining 3-4 ppm of total chlorine residual was much less effective for biofouling control than maintaining 3-4 ppm of pre-formed monochloramine residual with MWW_pH as makeup water.

Experimental Studies with MWW_NF

Tertiary treatment with nitrification and sand-filtration increased the corrosivity and reduced the scaling potential of secondary treated municipal wastewater. Removal of the corrosion-causing constituent ammonia by nitrification was helpful for corrosion management of the system. TTA successfully inhibited the corrosion of copper, cupronickel, and mild steel to acceptable levels according to industrial corrosion control criteria. Sand filtration removed a significant amount of total solids in the system. As a result there was lower scaling in the system compared to MWW without tertiary treatment. Calcium phosphate was the main precipitate formed when MWW_NF was used. Both bench- and pilot-scale studies showed that scaling was not a significant issue on unheated surfaces and no antiscalant was needed for the scaling control. However, on heated surfaces it was necessary to control the pH in a range around pH 7.8 since comparatively lower pH would retain more soluble calcium and phosphate in the solution and yield crystalline fouling in the high temperature ranges.

Pilot-scale tests with MWW_NF as makeup water indicated that biofouling in the cooling tower could be controlled by maintaining monochloramine residual above 2 ppm in the recirculating water. Use of pre-formed monochloramine was more reliable in controlling biological growth compared to free chlorine and chlorine dioxide. Maintaining 4-5 ppm of total chlorine residual by the addition of sodium hypochlorite could achieve target biofouling control criteria. Also, chlorine dioxide residual between 0.5-1 ppm ClO_2 successfully achieved biofouling control criteria.

Pilot-scale tests with MWW_NF as makeup water in the recirculating cooling system with continuous addition of monochloramine, free chlorine or chlorine dioxide revealed complete absence of Legionella species in the system. This result is significant in terms of public health protection for those that work or live around cooling systems using tertiary-treated municipal wastewater.

Experimental Studies with MWW_NFG

Bench- and pilot-scale studies using secondary treated municipal wastewater with nitrification, filtration, and activated carbon adsorption revealed that MWW_NFG caused pitting corrosion to copper and cupronickel alloys in both bench-scale and pilot-scale studies. Mild steel corrosion rate was maintained at acceptable levels, according to industrial corrosion control criteria, due to scale formation. Use of 2 ppm TTA in the system could not prevent the pitting corrosion of copper and cupronickel alloys in pilot-scale experiments. The presence of pitting corrosion in copper and cupronickel is unacceptable according to industrial corrosion control criteria and thus renders MWW_NFG unsuitable for use in recirculating cooling systems.

Scaling experiments revealed that MWW_NFG showed equally low scaling potential as MWW_NF. The removal of organic matter did not cause significant changes in the scaling characteristics compared to MWW_NF.

Biological growth could be controlled adequately in bench- and pilot-scale tests with MWW_NFG using monochloramine, NaOCl, and chlorine dioxide. The performance of pre-formed monochloramine in pilot-scale cooling systems was more reliable than the other two biocides tested regardless of total organic carbon (TOC) level. Comparison between MWW_NF and MWW_NFG indicated that the TOC removal did not limit biological growth.

Life Cycle Conceptual Cost (LC^3) Model Estimates for Tertiary Treatment Alternatives

Life cycle cost (LCC) analysis was performed for the tertiary treatment processes MWW_pH, MWW_NF, and MWW_NFG, examined in bench- and pilot-scale experiments in this study, and also for MWW with sand filtration (MWW_F), MWW with lime softening and sand filtration (MWW_SF), and MWW with nitrification, lime softening, and sand filtration (MWW_NSF). A public domain conceptual costing tool (LC^3 model) was developed and used to determine LCC for each tertiary treatment process. Estimated costs for the selected tertiary treatment scenarios were compared with established costs of river water and city water supply as alternative sources of cooling water.

MWW with advanced treatment and addition of conditioning chemicals is promising as a power plant cooling system makeup water based on the cost range of \$0.91 - \$1.32 (in 2009\$/kgal), which is between the river water withdrawal and treatment costs of \$0.74 (in 2009\$/kgal) and the average city water costs of \$2.95 (in 2009\$/kgal). By applying the LC^3 model to the case of a MWW effluent with flowrate of 7.75 Mgal/d, the typical make-up water requirement for a 550 MW thermoelectric power plant, it was found that treated water supply and chemical conditioning costs dominated the overall cost.

MWW_SF and MWW_NF were the most cost-effective treatment options among the tertiary treatment alternatives considered in this study because of the higher effluent quality with moderate infrastructure costs and the relatively low doses of conditioning chemicals required. It was also found that reuse of MWW would prevent surface water withdrawal and improve the cost-efficiency in the long term when compared to using river water or city water as makeup water source.

Life Cycle Inventory Emission Estimates for Tertiary Treatment Alternatives

Life cycle inventory analysis along with integration of external costs of emissions with direct costs was performed to evaluate relative emissions to the environment and external costs associated with construction and operation of tertiary treatment alternatives for reusing secondary municipal wastewater in cooling systems. Output inventories of greenhouse gas, sulfur dioxide, nitrogen oxides, and particulate matter (<2.5 micron) emissions to air showed that manufacture of chemicals for tertiary treatment and conditioning, and electric power generation associated with process operation are the main processes that contribute to these emissions. Transportation of chemicals and construction of tertiary treatment infrastructure to obtain higher quality effluent cause the increase in NO_x and $PM_{2.5}$ emissions, respectively.

MWW_pH exhibited relatively high NO_x, $PM_{2.5}$ and BTEX emissions to air, but remaining emissions to air, water and land were relatively moderate. MWW_SF and MWW_NF exhibited moderate emissions, except for greenhouse gas emissions, wherein the former alternative emits more greenhouse gases compared to the latter alternative, mainly due to the addition of lime in MWW_SF. MWW_NF has somewhat higher potential to cause eco-toxicity compared to MWW_SF. The three-tiered treatment alternatives MWW_NSF and MWW_NFG were associated with higher emissions in each category, with MWW_NSF having the highest emissions for all categories except for the BTEX emissions to water and soil/land, which were higher for MWW_NFG. The least emissions were estimated for river water supply (RW), and the reference case MWW, as there is not much infrastructure constructed and mainly addition of chemical conditioners is required. Water supply infrastructure and delivery showed high conventional pollutant emissions which are correlated with the pumping distance.

Relative toxic release inventories (TRI) of air emissions for the various tertiary treatment processes followed the same trend as observed for relative GHG, SO_2, NO_x and $PM_{2.5}$ emissions estimated for the processes. Though MWW_SF and MWW_NF exhibited relatively low and comparative TRI emissions to all compartments, MWW_SF had high TRI releases to air and land compared to MWW_NF, due to addition of lime. Therefore, MWW_NF would be environmentally preferred over MWW_SF.

Accounting for damages caused by air emissions along with the direct economic costs can help balance the external damage and the value added by a particular service, such as tertiary treatment to provide water that can be reused in power plant cooling. From the perspective of total costs incurred from tertiary treatment of MWW for reuse in power plant cooling systems, a treatment alternative can be selected based on cost-effectiveness and overall sustainability.

LCI and LCC analysis integration indicated that three-tiered treatment alternatives such as MWW_NSF and MWW_NFG, with regular chemical addition for treatment and conditioning and/or regeneration, tend to increase the impact costs and in turn the overall costs of tertiary treatment, even though their percent contribution to the total annual costs is moderate to low. RW and MWW_F alternatives with a single step of tertiary treatment were associated with lower impact costs due to a single level of tertiary treatment, but the contribution of impact costs to overall annual costs was higher than for all other treatment alternatives. This result indicates that the RW and MWW_F alternatives are less cost-effective when the impact costs are integrated in the overall analysis. MWW_NF and MWW_SF alternatives with two-tiered tertiary treatment exhibited moderate external impact costs with moderate infrastructure and chemical conditioner dosing, which makes them (especially MWW_NF) better treatment alternatives from the environmental sustainability perspective since they exhibited minimal incremental environmental damage from emissions.

Sustainability Metrics for Reuse of Treated Municipal Wastewater in Cooling Systems

Analysis of the sustainability aspects of wastewater reuse in power plant cooling systems revealed that the sustainability of alternative approaches related to biofouling, scaling, and corrosion control and wastewater delivery can be partially assessed using available sustainability tools and metrics. However, the available tools and metrics are inadequate to address public perception of wastewater reuse, and valuation of freshwater versus wastewater. The primary element that makes treated wastewater reuse in power plant cooling systems sustainable, i.e., reuse of wastewater, is not adequately captured by existing sustainability tools.

Review of literature on behavioral analysis and public perception regarding wastewater reuse indicated the importance of outreach programs and positive communications in changing public perception towards wastewater reuse. The review of community acceptance of wastewater reuse indicates that initial acceptance hinges in large part on the public's awareness of whether or not the reuse applications reflect sustainability principles, improve local water supply, and involve sufficient quality of reclaimed water. Also, the public seeks assurance that the reuse application being considered involves minimal risk of accidental exposure of workers and the public. Opinions about wastewater reuse are different for non-potable and potable uses. Public perception is largely influenced by the potential for human contact; factors such as health and extent of treatment are very important.

Case studies of wastewater reuse in power plant cooling systems and other applications revealed that the public is wary of using wastewater in cooling systems. There are several concerns typically voiced, but potential for human exposure is primary.

Efforts to communicate with the public about wastewater reuse and to build public acceptance should be conducted in the early stages of a project to provide sufficient time to address public concerns and needs. Addressing social challenges involved with wastewater reuse is most effective when information is imparted continuously during the decision making process, and in effective iterations during the multiple stages of the project to develop public confidence and trust.

1.0 Introduction

This study extended a previous DOE-sponsored research project on “Reuse of Treated Internal or External Wastewaters in the Cooling Systems of Coal-based Thermoelectric Power Plants” (Vidic and Dzombak, 2009). The primary objective of that work was to assess the potential of three different impaired waters, namely, secondary treated municipal wastewater, passively treated coal mine drainage, and ash pond effluent, for use as cooling water in thermoelectric power generation. The characteristics and relative availability of these waters were evaluated in the context of reuse in power plant cooling systems. In addition, the study included evaluation of the effects of these waters on corrosion, scaling, and biofouling in cooling systems, and the chemical treatments required to control these effects.

Evaluation of availability of the three impaired waters (Vidic and Dzombak, 2009) revealed that treated municipal wastewater is a common and widespread alternative cooling water source in the U.S. Analysis revealed that 81% of power plants planned for construction by the U.S. Energy Information Administration as of 2007 would have sufficient cooling water supply from 1-2 publicly owned treatment works (POTWs) within a 10-mile radius, while 97% of the proposed power plants would be able to meet their cooling water needs from 1-2 POTWs within 25 miles of these plants (Vidic and Dzombak, 2009; Li et al., 2011). Thus, municipal wastewater is an impaired water source likely to be locally available in sufficient and reliable quantities for many power plants.

Results from the previous study (Vidic and Dzombak, 2009) indicated that it is feasible to use secondary treated municipal wastewater as cooling system makeup. However, the biodegradable organic matter, ammonia-nitrogen, carbonate, and phosphate in the treated wastewater pose challenges with respect to enhanced biofouling, corrosion, and scaling. In the previous study it was demonstrated that these problems can be controlled to varying extents through chemical management, by deploying different combinations and doses of biocides and corrosion and scale inhibitors. However, chemical management alone would be costly.

The overall objective of the study described herein was to evaluate the benefits and costs of implementing tertiary treatment of secondary treated municipal wastewater prior to use in recirculating cooling systems versus an expanded chemical regimen for managing the quality of the cooling water when secondary treated municipal wastewater is used as makeup.

1.1. Availability of Secondary Treated Municipal Wastewater for Cooling Needs

One of the most widely available non-traditional sources of cooling water is treated municipal wastewater. The 11.4 trillion gallons of wastewater collected and treated annually by the public utilities in the U.S. (USEPA, 2003) represent a significant potential source for cooling water needs in thermoelectric power plants.

In the previous related study (Vidic and Dzombak, 2009), an analysis was conducted to assess the extent to which treated municipal wastewater resource is available in sufficient quantities and with adequate geographic distribution to satisfy future needs for cooling water in electricity generation. Data for POTWs across the U.S. were extracted from the Clean Watersheds Needs Survey (CWNS) conducted by the U.S. Environmental Protection Agency (USEPA, 2003). The distribution of 17,864 POTWs identified in the CWNS and included in the analysis is shown in Figure 1.1. The distribution of existing and future power plants among the various different North American Electric Reliability Corporation (NERC) regions was also assesses. Figure 1.2 shows the locations of 110 power plants projected for future development (USEIA, 2007).

Both datasets were imported into a geographic information system (GIS) to calculate available wastewater flowrate within a given distance from each proposed power plant location. Wastewater availability was compared to cooling water demand that was calculated using an average of 1200 gallons per MWh of power generation (NETL, 2006).

About 81% of proposed power plants could meet their cooling water needs considering the POTWs within a 10 mile radius, and 97% of proposed power plants could satisfy their cooling water needs from POTWs within a 25 mile radius. In addition, the analysis indicated that only 1.15 POTWs within a 10 mile radius from the proposed power plants are needed to satisfy total cooling water needs of each plant. If the coverage is extended to 25 miles, each proposed power plant has an average of 18.4 POTWs within that region and only 1.10 POTWs are needed to satisfy its cooling water needs. The fact that a fairly low number of POTWs (i.e., close to one) can meet the cooling water needs of the proposed power plants suggests that the cost of transporting wastewater can be kept at a minimum (i.e., only one or two pipes may be needed to transport the cooling water to the power plant). The analysis thus suggests that using reclaimed water for cooling purposes can be both economical and reliable and can facilitate the development of thermoelectric power plants in the regions where other water sources are not readily available. Of course, in many areas treated municipal wastewater is already in use for maintenance of stream flows, irrigation, and other purposes and may not be available for other uses.

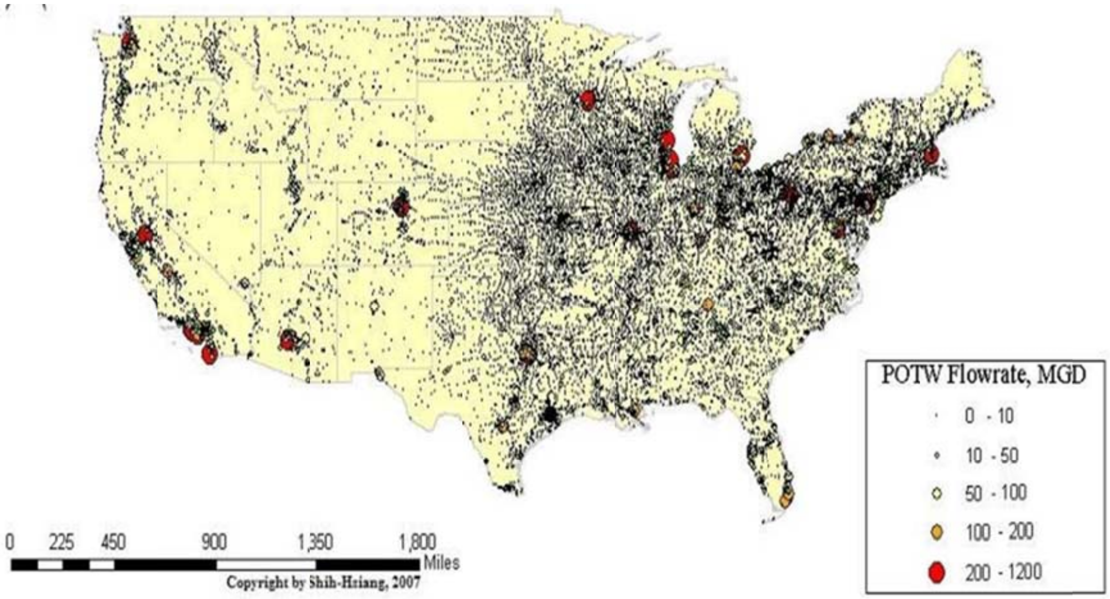

Figure 1.1 Locations of 17,864 POTWs in the U.S., from the USEPA Clean Watersheds Needs Survey database (USEPA, 2003).

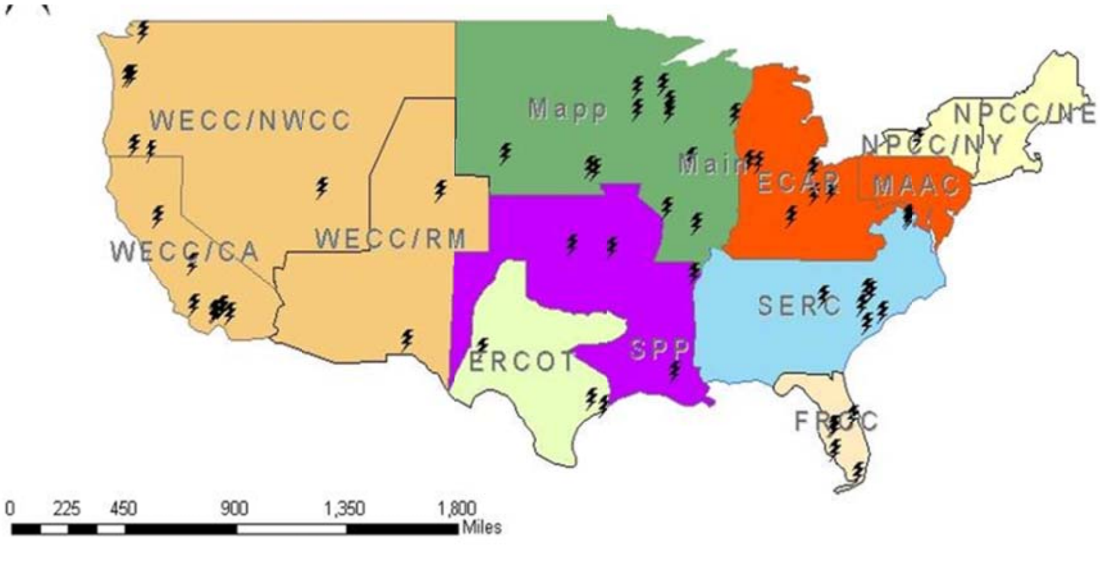

Figure 1.2 Locations of 110 proposed power plants (USEIA, 2007) in the continental U.S.

1.2 Current Experience with the Use of Treated Municipal Wastewater for Cooling

The previous study identified approximately 50 power plants across the U.S. that are already using treated municipal wastewater to meet their cooling water needs (Vidic and Dzombak, 2009). In Burbank, California, about 5 mgd of municipal secondary effluent has been utilized for cooling water makeup in the city's power generating plant since 1967. The reclaimed water is treated with the addition of chlorine, acid, and corrosion inhibitors. The City of Las Vegas and Clark County Sanitation District supply 90 mgd of secondary treated wastewater to meet approximately 35 percent of the water demand in power generating stations operated by the Nevada Power Company. Additional treatment of the secondary treated wastewater is performed, consisting of 2-stage lime softening, filtration, and chlorination prior to use as cooling tower makeup. A reclaimed water reservoir provides backup for the water supply. The Arizona Public Service 1,270-MW Palo Verde nuclear power plant is located 55 miles from Phoenix, Arizona, and uses almost all of the reclaimed water from the City of Phoenix and nearby communities at an average rate of 68 mgd.

Most of the thermoelectric power plants utilizing secondary treated municipal wastewater either provide significant additional treatment (e.g., clarification, filtration, advanced nitrogen and phosphorus removal) prior to use in their cooling system or use it with significant dilution, i.e., only a fraction of their total cooling water need in the plant is satisfied with secondary effluent. Such practice is dictated by concerns related to water quality problems that may occur if as-received secondary treated effluent is used to satisfy 100% of cooling water needs in the plant. It appears that these practices may be overly conservative. The study described herein examined the technical performance and costs of employing different levels of tertiary treatment.

1.3 Study Objectives

The overall objective was to evaluate the benefits and lifecycle economic and environmental costs of different levels of tertiary treatment of secondary treated municipal wastewater prior to use in recirculating cooling systems

Specific objectives of the research were as follows:

(a) Determine the benefits and costs of subjecting secondary treated municipal wastewater to additional treatment to remove NH_3 by nitrification, BOD by GAC adsorption, and particles by filtration prior to use as cooling water makeup.

(b) Determine different chemical treatment regimens required to manage cooling water quality for different levels of tertiary treatment ranging from no tertiary treatment to inclusion of nitrification and treatment for the removal of particles and residual organic matter.

(c) Perform comparative life-cycle cost analyses for different levels of tertiary treatment and the corresponding chemical treatment regimens.

(d) Determine critical economic, technical, and social factors that need to be considered in comparative evaluation of tertiary treatment alternatives for secondary treated municipal wastewater when used as cooling tower makeup water.

1.4 Organization of the Report

The report contains eight major chapters following the Introduction, and six appendices. Chapter 2 presents the details of the equipment and methods employed in the bench- and pilot-scale experiments used to study corrosion, scaling and biofouling and their control with use of tertiary treated municipal wastewater. Chapter 3 presents the results of experiments with secondary treated municipal wastewater subjected to tertiary treatment by pH adjustment. Chapter 4 presents the results of experiments with secondary treated municipal wastewater subjected to tertiary treatment by nitrification and filtration. Chapter 5 presents the results of experiments with secondary treated municipal wastewater subjected to tertiary treatment by nitrification, filtration, and carbon adsorption. In Chapter 6, the results of lifecycle cost analyses are presented for the various tertiary treatment options examined. Chapter 7 presents the results of analysis of combined economic and environmental costs of the tertiary treatment options examined. Development and analysis of sustainability metrics for reuse of treated municipal wastewater is presented in Chapter 8. Finally, in Chapter 9 the main findings and conclusions of the study are presented.

1.5 References

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2.0 Bench-scale and Pilot-scale Experimental Systems and Methods

The study employed bench-scale recirculation water systems and pilot-scale cooling towers for testing of various chemical control schemes for corrosion, scaling, and biofouling in systems using different tertiary treated municipal wastewaters. Initial bench-scale experiments included batch-reactor experiments for performance evaluation of different scaling inhibitors and biocides. An experimental setup was prepared for scaling studies on a heated surface. A synthetic wastewater recipe was formulated for different types of tertiary treated municipal wastewaters having constituent concentrations similar to that at four cycles of concentration, a usual concentration of recirculating cooling waters in power plant cooling systems. The synthetic cooling water recipes were used in initial bench-scale experiments for corrosion, scaling, and biofouling analysis in different types of tertiary treated municipal wastewater. Details about the bench-scale and pilot-scale systems are described in this section. General wastewater quality and synthetic recipe formulas derived from the general wastewater qualities are described briefly. Also the chemical treatment methods and analytical methods used in the study are outlined in this section.

2.1 Bench-scale Systems

Bench-scale systems used in the study included both batch reactor systems and recirculating water systems. The batch reactor systems were used for stability and performance analysis of different scaling inhibitors and biocides. Bench-scale recirculating water systems were used to analyze corrosion, scaling, and biological fouling while using different tertiary treated municipal wastewater as makeup water in the cooling systems. Results obtained from bench-scale recirculating water system experiments formed the basis for selecting optimum chemical treatment scenarios for pilot-scale experiments.

2.1.1 Bench-scale batch reactor configuration

A 1.4 L beaker was used as batch reactor in this study. The beaker was covered with plastic foil to minimize water loss due to evaporation. The contents in the beaker were completely mixed with a magnetic stirrer and the bulk temperature was controlled at target value using the heating plate.

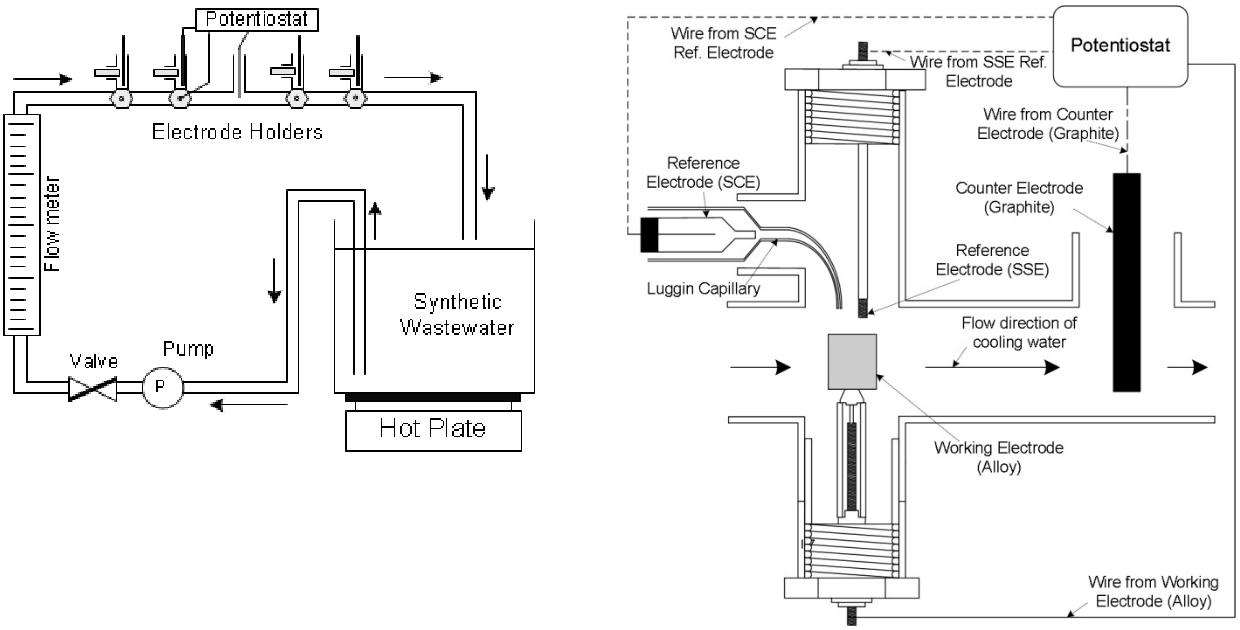
2.1.2 Bench-scale recirculation water system configuration

A bench scale recirculation water system was used to expose metal and metal alloy samples to temperature, flow velocity, and water quality similar to those in recirculating cooling water systems. The bench-scale recirculating water system consisted of a centrifugal pump, a water bath on hotplate (to control the water temperature) and a 1.91 cm (0.75 in) nominal diameter PVC pipe circulation system with a pipe rack to hold different electrodes. Design of the pipe rack holder was modified to accommodate two different reference electrodes (Figure 2.1.1a). The pipe rack system included four tee-sections into which alloy specimen holders were mounted. Each tee-section was connected to another tee section, which accomodated two different reference electrodes (Figure 2.1.1b) for side by side polarization resistance measurement. Temperature of the recirculating water was maintained at 40°C in the bench-scale system to simulate the temperature of cooling water in the pilot-scale recirculating cooling water systems (Hsieh et al., 2010). Flow rate through the system was maintained at 11.4 L/min (3 gpm) to achieve flow velocity of 0.66 m/s (2.18 ft/s) and Reynolds number of 1.9x10^4 .

Both Weight Loss Method (WLM) and Polarization Resistance Method (PRM) of corrosion monitoring were carried out in the bench-scale recirculation system. The reference electrodes were placed close (within 2mm) to the working electrode: the SCE was placed in a Luggin capillary and the SSE was located at one of the tee section ends (Figure 2.1.1b). The pipe rack holder also accommodated a graphite counter electrode and a metal specimen holder (made of an end-cap plug containing a threaded stainless steel rod with an alloy specimen attached at the end). The metal alloy specimen, reference electrode (either SCE of SSE), and counter electrode were connected to a potentiostat for polarization resistance (R_P) measurements (Figure 2.1.1b). After the desired immersion period, the metal and metal alloy specimens were removed for weight loss analysis. Semicontinuous R_P measurements were carried out on the metal alloy coupon samples throughout the immersion period in the bench-scale recirculation systems.

Similar bench-scale recirculation systems were constructed and used for scaling and bio-fouling studies with more sampling ports incorporated in the pipe rack system. Stainless steel circular discs (5.61 cm^2 in area) (Figure 2.1.2b) were inserted through sampling ports into the recirculating water system to provide collecting surfaces for scaling or biofouling.

A mass gain method, similar to the mass loss method for corrosion, was used to as a straight forward means to record the scale forming quantities at different water chemistries and scaling control conditions. Scaling kinetics of different treated municipal wastewaters were studied at varying cycles of concentration (CoC) in the bench-scale recirculating system. Water temperature was fixed at 40°C and the flow rate was 11.4 L/min (3 GPM). The system was open to air so that the alkalinity may approach equilibrium with atmospheric CO_2, a condition similar to actual cooling tower operation.



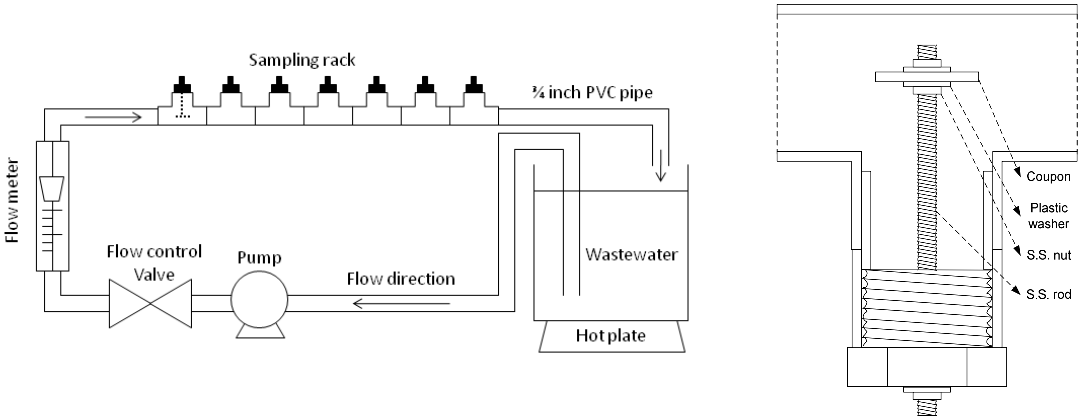
(a) (b)

Figure 2.1.1 Schematic diagrams of (a) bench-scale recirculating system, (b) metal alloy specimen holder and ports for counter electrode and reference electrodes in the bench scale recirculating system.

The scale samples collected on the test discs were air-dried and inspected using Scanning Electron Microscopy (SEM, FEI/Philips XL30), and the elemental composition was determined by energy dispersive X-ray spectroscopy (EDS, EDAX/AMETEK). Surface images were obtained by microphotography (Fisher Micromaster optical microscope coupled with a Sony digital camera) and SEM. These studies helped to identify connections between scaling kinetics and scale characteristics. Insights drawn from the connections facilitate the selection of most effective scaling control methods. For example, identification of the mineral deposits by SEM/EDS would instruct the selection of the appropriate antiscaling chemicals to inhibit formation of the specific minerals identified.

For biofilm growth studies similar circular stainless steel disks were used in a bench-scale recirculating system (Figure 2.1.2a). The total duration of each bench-scale experiment was 24 hours with intermittent biocide dosing to maintain the desired residual. The circular stainless steel coupons were first sterilized by autoclaving and then suspended in the recirculating flow (Figure 2.1.2b) to monitor biofilm growth in the system. Collection of sessile biological growth from the coupons was performed according to the ASTM E 1427 Method

(ASTM, 2000). After withdrawing the coupons aseptically from the coupon rack, visible water residual was carefully removed from the coupon surface (Bradshaw et. al., 1996) and the coupon was then immersed in a 50 mL phosphate buffered saline solution. The solution was sonicated for 5 minutes to dislodge accumulated biological growth from the coupon and then vortexed for 30 seconds to make it homogeneous (Prosser et. al., 1987). The most probable number of sessile heterotrophic bacteria was measured by plating serial dilutions of this solution, and the result was converted to CFU per cm^2 of the coupon area to obtain the sessile heterotrophic bacterial count.In addition, planktonic heterotrophic bacteria, total chlorine and monochloramine residuals in the system were also monitored throughout the experiments.



(a) (b)

Figure 2.1.2 Schematic diagram of (a) bench scale circulating system, and (b) circular coupon with coupon holder for scale and biofilm sampling.

2.1.3 Experimental setup for evaluating scaling on heated surface

A primary concern in managing water quality in the recirculating cooling water system of a thermoelectric power plant is the control of heat exchanger (condenser) fouling. Fouling is generally defined as an unwanted deposition of suspended, dissolved, or chemically generated materials in the process fluids onto the heat transfer surfaces (Yang et al., 2002). Even a thin film of scale can degrade the heat transfer efficiency significantly because of its low thermal conductivity. The efficiency of heat transfer and degree of cooling achieved influences the overall efficiency of a power plant.

Normally, fouling is classified into the following categories: particulate, crystallization, corrosion, biofouling, and chemical reaction (Pahlavanzadeh et al., 2007). The focus of the studies in this project is mainly on crystalline deposits caused by the growth of mineral salt crystals on the surfaces, because the fouling from deposition and attachment of precipitated particles is usually the predominant mechanism for the formation of hard and tenacious fouling in heat exchangers (Sheikholeslami, 1999).

Previous study (Li et al., 2010) has shown that calcium carbonate and calcium phosphate are the main mineral scales formed on non-heated surface when MWW was used as makeup for recirculating cooling systems operated at CoC 4-6. Both calcium carbonate and calcium phosphate have inverse temperature solubility, which means that the solubility of these minerals decreases with an increase in temperature. High skin temperature on the heat transfer surfaces can enhance the scaling propensity and reduces heat transfer efficient. Thus, it is necessary to study fouling mechanisms and mitigation methods to protect heat exchanger surfaces and allow the use of wastewater in cooling systems of thermoelectric power plants.

In this project, we modified the design of the bench-scale recirculating systems to incorporate a heated cartridge, which will be used to simulate the heat exchanger tubing surface in mineral scaling investigation. The design principle for the new system is taken from ASTM Standard Test Method for Determination of Corrosion and Fouling Tendency of Cooling Water under Heat Transfer Conditions (ASTM D 4778-05).

The schematic diagram of a bench-scale recirculating system design for the heat exchanger fouling study is shown in Figure 2.1.3. As seen there, a cartridge electric heater, with built-in type J thermocouple, is inserted vertically into the water recirculating system to simulate the heat transfer situation. The power applied to the heater is regulated by a transformer to control the heat flux into the recirculating water. The flow rate of the recirculating water is adjusted by a centrifugal pump and measured continuously using an in-line flow meter. In order to control the bulk water temperature in the desired range, a cold loop composed of a 6 ft of copper coil is immersed into the bulk solution tank and cold tap water is directed through the copper coil in a once-through pattern to dissipate the excess heat.

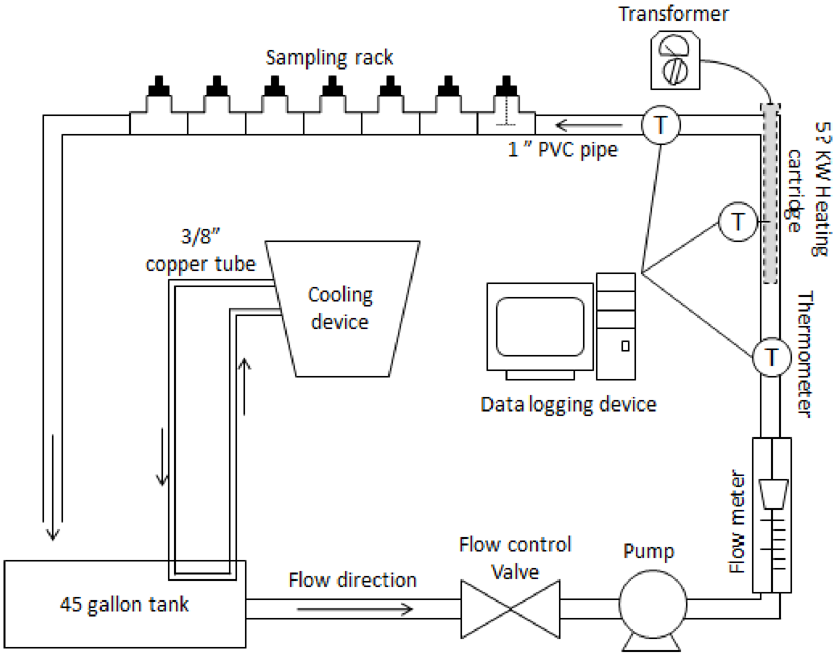


Figure 2.1.3 Schematic diagram of a bench-scale recirculating system for fouling study

For a heated surface experimental fouling system, the more closely the system simulates conditions of the heat exchanger, the more relevant is the conclusion that can be obtained from such a system. According to the available literature and recommendations from industry practitioners, the operating conditions of the experimental system should be as follows:

- Heat flux: 5000-15,000 Btu/hr˙ft^{2} (15673.98 to 47021.94 W/m$_{2}$) (GE Power & Water, 2009).
- Flow velocity: 2-8 ft/s (0.6 to 2.4 m/s) (Nalco Company, 2009).
- Bulk water temperature: 90-110^{o}F (32.2 to 43.3^{o}c) (Aull, 2011)
- Surface temperature of the heater 120-140^{o}F (48.9-60 ^{o}c) (Aull, 2011)

In order to obtain the fouling resistance from this experimental arrangement, the following parameters in the test section of the experimental system (as shown in Figure 2.1.4) are recorded continuously by the data logger system (shown in Figure 2.1.5):

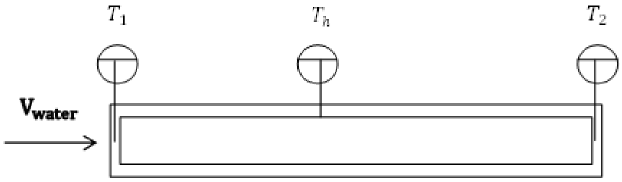
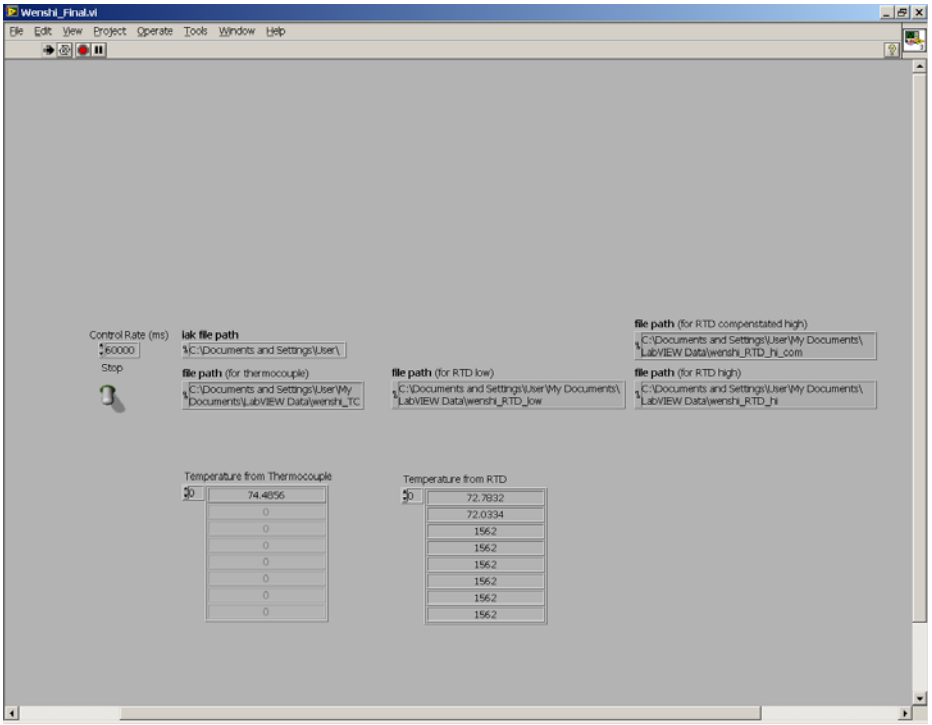


Figure 2.1.4 Schematic diagram of the test section with the heater in the water recirculating system for fouling study (The test section is vertical in the system)



Control Rate (ms) 60000

iak file path C:\Documents and Settings\User\

file path (for thermocouple) C:\Documents and Settings\User\My Documents\LabVIEW Data\wenshi_TC

file path (for RTD low) C:\Documents and Settings\User\My Documents\LabVIEW Data\wenshi_RTD_low

file path (for RTD compenstated high) C:\Documents and Settings\User\My Documents\LabVIEW Data\wenshi_RTD_hi_com

file path (for RTD high) C:\Documents and Settings\User\My Documents\LabVIEW Data\wenshi_RTD_hi

Temperature from Thermocouple	Temperature from RTD
74.4856	72.7832
0	72.0334
0	1562
0	1562
0	1562
0	1562
0	1562
0	1562

Figure 2.1.5 The interface of the data logger system

- The temperature of water at the inlet to the annular section T_1
- The temperature of water at the outlet from the annular section T_2
- The skin temperature of the immersion heater Th

Then the fouling resistance can be calculated using the bulk water temperature in the test section, T_b, determined based on the inlet water temperature (T_1) and the outlet water temperature (T_2):

$$T_b = \frac{1}{2}(T_1 + T_2) \quad (2.1.1)$$

The heat flux, $\dot{q}$, can be calculates as:

$$q = \frac{UI}{A} = \frac{U^2/R}{A} \qquad (2.1.2)$$

Where, U is the voltage applied to the immersion heater,

I is the electrical current through the immersion heater,

R is the electrical resistance of the immersion heater,

A is the heated surface area.

The overall heat transfer coefficient, K, is then calculated as:

$$K = \frac{q}{T_s - T_b} \qquad (2.1.3)$$

Where T_s is the measured surface temperature of the cartridge heater

With the mineral scales built up on the heated surface, the overall heat transfer coefficient would decrease mainly due to the low thermal conductivity of the scale. The the fouling resistance, R_f, is determined as:

$$R_f = \frac{1}{K_{(t)}} - \frac{1}{K_{(0)}} \qquad (2.1.4)$$

where, $K_{(t)}$ is the overall heat transfer coefficient at time t

$K_{(0)}$ is the overall heat transfer coefficient at time 0

In addition to monitoring on the fouling resistance, stainless steel coupon discs specimen are also inserted into the recirculating water to track the mineral scale formation on the non-heated surface.

2.2 Pilot-scale Systems

The use of batch and bench-scale recirculating systems helps to understand the fundamental aspects of the corrosion, scaling, and biofouling processes and relative effectiveness of different inhibition mechanisms in a well-controlled laboratory environment. However, batch reactor lacks the ability to simulate the hydrodynamic forces commonly encountered in full-scale cooling systems while bench-scale recirculating system lacks continuous replenishment with nutrients and minerals coming with the makeup water. In addition, laboratory tests are unable to simulate temperature variation due to intensive air-to-water contact along with stripping effects. Therefore, a pilot-scale cooling system was utilized to verify the findings of laboratory-scale studies under more realistic process conditions.

2.2.1 Pilot-scale cooling tower configuration

To examine the effectiveness of corrosion, scaling, and biofouling mitigation strategies under the conditions similar to those in full-scale cooling systems, pilot-scale cooling towers were constructed and deployed in the field. Each pilot-scale system capable of generating 27,000 kJ/h heat load and maintaining recirculating water flow at 3 GPM with a Reynolds number of 1.92×10^4 was designed to study these critical processes under conditions that are similar to full-scale systems (Figure 2.2.1).

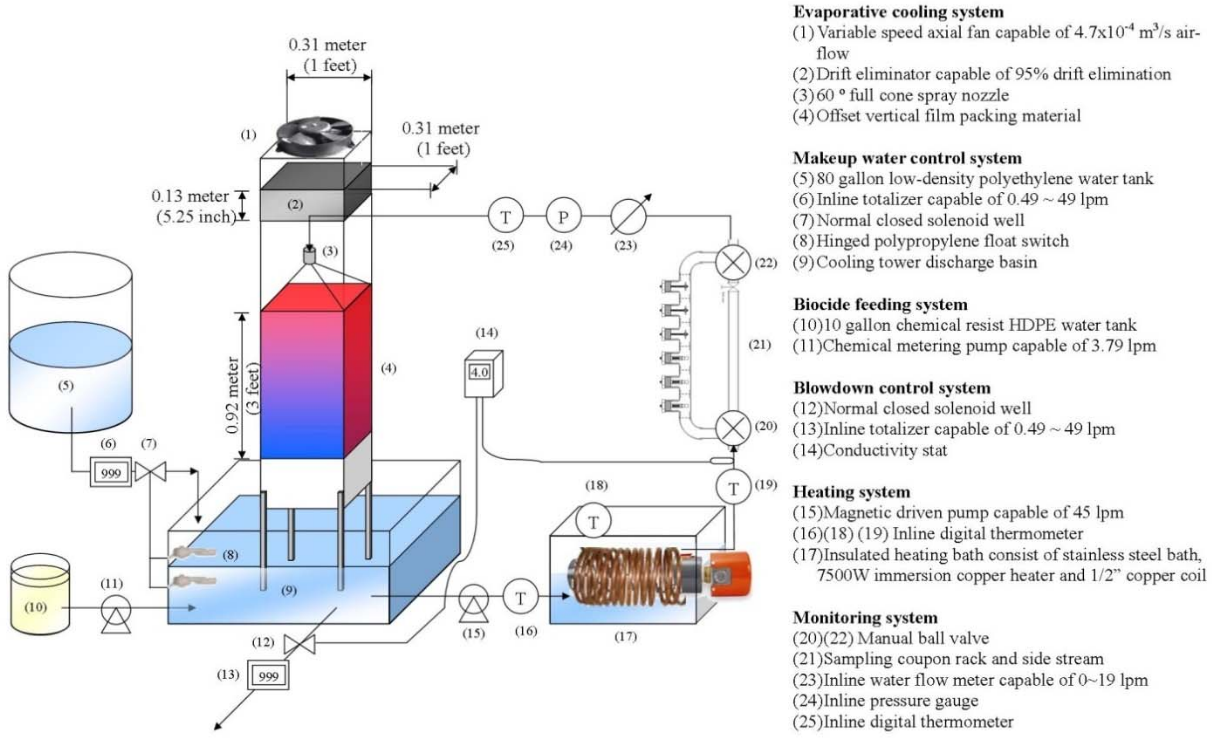


Figure 2.2.1 Schematic of the pilot-scale cooling tower.

The pilot-scale cooling tower was equipped with automatic makeup water control system, automatic blowdown control system, semi-automatic biocide feeding system, and corrosion, scaling, and biofouling monitoring system. Observed operational data revealed that the major operating parameters, including temperature change (6.6 ºC; 10 ºF), cycles of concentration (CoC = 4-6), water flow velocity (0.66 m/s; 2.17 ft/s), and air mass velocity (3660 kg/h·m^2; 1000 CFM), were controlled quite well for several consecutive experiments (up to 2 months). The performance of the pilot-scale cooling towers using treated municipal wastewaters was shown to be suitable to study critical processes (corrosion, scaling, biofouling) and evaluate cooling water management strategies for make-up waters of complex quality. Details and principle mechanisms of the pilot scale cooling system arere described elsewhere (Chien et al., 2012).

2.2.2 Pilot-scale operation

The field tests with pilot-scale cooling towers were conducted at Franklin Township Municipal Sanitary Authority (FTMSA) in Murrysville, PA, using the three pilot-scale cooling towers developed in the previous related study (Vidic et al., 2009). These pilot-scale cooling towers were operated in the field in 2010 and 2011 for 28-day and 60-day test periods, plus 3 days to reach CoC 4.

Tests with nitrified filtered tertiary-treated municipal wastewater (MWW_NF) began in late May 2010 and were conducted for two months. After that, the pilot-scale systems were used in a two-month long tests with secondary treated municipal wastewater (MWW) and secondary treated municipal wastewater with pH adjustment (MWW_pH). The MWW_NF was collected directly from sand filter effluent channels. The MWW was collected from the channel between secondary clarifier and nitrification tower and was further treated with 0.5 mM sulfuric acid to achieve target pH (MWW_pH) prior to addition to pilot-scale cooling systems. The first three test conditions evaluated in the pilot-scale system from May to July 2010 were: (1) MWW_NF with addition of pre-formed monochloramine (MCA) as biocide and without scaling and corrosion control; (2) MWW_NF with addition of pre-formed MCA as biocide, low dosage of tolytriazole (TTA) as corrosion inhibitor, and polymaleic acid as scaling inhibitor; and (3) MWW_NF with addition of pre-formed MCA as biocide, high dosage of TTA as corrosion inhibitor, and polymaleic acid (PMA) as scaling inhibitor.

The test conditions evaluated in the pilot-scale system from August to October 2010 were: (1) MWW with addition of pre-formed MCA as biocide, TTA as corrosion inhibitor, and PMA as scaling inhibitor; (2) MWW_pH with addition of pre-formed MCA as biocide, TTA as corrosion inhibitor, and PMA as scaling inhibitor; and (3) MWW_pH with addition of diluted NaOCl as biocide, TTA as corrosion inhibitor, and PMA as scaling inhibitor.

The tests with different tertiary effluents were initiated in May 2011 at FTMSA. Secondary-treated wastewater that had been subsequently treated by nitrification and sand filtration was subjected to GAC treatment in the field. An 8” diameter PVC pipe filled with F400 granular activated carbon (Calgon Carbon, Pittsburgh, PA) was assembled to reduce the organic matter concentration in the wastewater. Details of the GAC adsorption column design and filtration performance can be found in Appendix C The GAC-treated wastewater, designated MWW_NFG, was then used in pilot-scale testing. The test conditions evaluated in the pilot-scale system in Summer 2011 were: (1) MWW_NF with addition of NaOCl as biocide and without corrosion and scaling inhibitors; (2) MWW_NF with addition of chlorine dioxide as biocide and TTA as corrosion inhibitor; and (3) MWW_NFG with addition of pre-formed MCA as biocide and TTA as corrosion inhibitor.

Characteristics of the waters tested and recirculating water quality are provided in Chapter 2.3. During the pilot-scale tests, detail information on the operation of the towers was recorded, including temperature of water at specific locations, air flowrate in the cooling towers, conductivity of the recirculating water, makeup water volume, blowdown volume, water flowrate, and ambient conditions (weather, temperature, relative humidity). For each testing period, detail operational and performance data for the three pilot-scaling cooling towers are presented in Appendix B. Cycles of concentration were validated based on conductivity of the recirculating water, blowdown and makeup water volumes, and chloride concentration in the recirculating water.

2.2.3 Pilot-scale experimental matrix

The chemical inhibitor regimen for the pilot-scale tests with the tertiary treated wastewater MWW_NF from May-July 2010 were selected to examine variable doses of the corrosion control agent (TTA). Due to the presence of ammonia and relatively low pH in MWW_NF, corrosion is the major concern when using this water as cooling tower makeup. The chemical inhibitor dosing used in tests with MWW_NF are shown in Table 2.2.1. Cooling Tower A (CTA1) was a control test, with biofouling control only. In the tests with Cooling Tower B (CTB1) and Cooling Tower C (CTC1), TTA concentrations of 2ppm (CTB1) and 4ppm (CTC1) were evaluated. Concentrations of PMA as anti-scaling agent and pre-formed MCA as biofouling control agent were held constant in both systems as shown in the Table 2.2.1.

Table 2.2.1 Chemical inhibitor regimen for initial pilot-scale cooling tower tests with MWW_NF

Chemical	CTA1	CTB1	CTC1	Function	Dosing location
TTA, ppm as dose	0	2	4	Anti-Corrosion	Makeup water tank
PMA, ppm as dose	0	5	5	Anti-scaling	Makeup water tank
MCA, ppm as residual	2~3^a	2~3^a	2~3^a	Biocide	Basin

Note: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); a: residual as monochloramine

The previous related study (Vidic et al., 2009) showed that scaling was the major concern when using secondary treated municipal wastewater as cooling tower makeup. Corrosion problem was impeded by severe scaling formation and biofouling problem was controlled at acceptable level by maintaining monochloramine residual above 3 mg/L. In order to determine a proper scaling mitigation strategy, the second set of pilot-scale tests conducted in

2010 focused on the effectiveness of pH adjustment for scaling control when using secondary treated wastewater as cooling system makeup water, and on a direct comparison of pre-formed monochloramine with sodium hypochlorite for biofouling control. The chemical regimens in three cooling systems are shown in Table 2.2.2. It should be noted that PMA was added to all three pilot-scale cooling systems at the same dose as secondary scaling inhibitor. Cooling tower A (CTA2) was a control system with no pH adjustment and corrosion control using a lower TTA dose. In the Cooling Tower B (CTB2) and Cooling Tower C (CTC2), TTA concentration was maintained at 5 ppm and pH was adjusted at 7.7. The biofouling control agents in CTA2 and CTB2 were pre-formed MCA, while NaOCl was used in CTC2.

Table 2.2.2 Chemical inhibitor regimen for second pilot-scale cooling tower tests with MWW

Chemical	CTA2	CTB2	CTC2	Function	Dosing location
TTA, ppm as dose	2	5	5	Anti-corrosion	Makeup water tank
PMA, ppm as dose	5	5	5	Anti-scaling	Makeup water tank
MCA, ppm as residual	3~4[a]	3~4[a]	-	Biocide	Basin
NaOCl, ppm as residual	-	-	3~4[b]	Biocide	Basin
pH control	NC	7.7	7.7	Anti-scaling	Basin

Notes: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); NaOCl (sodium hypochlorine); NC = no pH adjustment; a: residual as monochloramine; b: residual as total chlorine.

The third set of pilot-scale tests with MWW_NF and MWW-NFG focused on the effectiveness of different biocides when using tertiary effluent, e.g., MWW_NF and MWW_NFG, as cooling tower makeup. The GAC-treated wastewater, designated MWW_NFG, was used with addition of pre-formed monochloramine. PMA was not used in this set of tests since it was proved that MWW_NF has negligible scaling potential. Cooling tower A (CTA3), which used MWW_NF as makeup, was treated with NaOCl without any corrosion and scaling inhibitors to represent a control system for corrosion study. The pilot-scale test with NaOCl as biocide and without the addition of any corrosion control agent was conducted in order to test the field portable electrochemical corrosion measurement device under aggressive corrosion conditions. Cooling tower B (CTB3), which used MWW_NF as makeup, was treated with NaOCl and TTA. Cooling tower C (CTC3), which used MWW_NFG as makeup, was treated with pre-formed monochloramine and TTA. The test conditions for the third set of pilot-scale tests in 2011 are summarized in Table 2.2.3.

Table 2.2.3 Chemical inhibitor regimen for third pilot-scale cooling tower tests with MWW_NF and MWW_NFG

Chemical	Pilot-scale Cooling Towers			Function	Dosing location
	CTA3	CTB3	CTC3		
TTA, ppm as dose	0	2	2	Anti-Corrosion	Makeup water tank
NaOCl, ppm as residual	1~2[a]	0	0	Biocide	Basin
ClO_2, ppm as residual	0	0.5~1[b]	0	Biocide	Basin
MCA, ppm as residual	0	0	2~3[c]	Biocide	Basin

Notes: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); NaOCl (sodium hypochlorine); NC = no pH adjustment; a: residual as free chlorine; b: residual as total chlorine; c: residual as monochloramine.

2.3 Characterization of Secondary and Tertiary Treated Wastewater, and Synthetic Wastewater Preparation

The study used actual treated municipal wastewater from the Franklin Township Municipal Sanitary Authority, FTMSA (Murrysville, PA) facility in different bench and pilot-scale recirculating cooling water system experiments. The general water quality of secondary treated municipal wastewater (MWW), nitrified-filtered MWW (MWW_NF) and nitrified-filtered-granular activated carbon adsorbed MWW (MWW_NFG), given in Table 2.3.1 were obtained by averaging the qualities of water samples collected weekly from FTMSA during the summer of 2010 and 2011 (Choudhury, 2012).

For MWW with pH reduction by acidification, continuous sulfuric acid was added to the pilot-scale system to keep the pH at 7.7. Actual water from FTMSA was concentrated in laboratory by heated evaporation (at 40^{o}C) to reach four cycles of concentration (CoC4) as determined by 75% volume reduction for bench-scale experiments.

The tertiary treated municipal wastewater qualities, which significantly influence corrosion and scaling potential, were replicated in synthetic municipal wastewater samples at four cycles of concentration (CoC4) for bench-scale experiments. CoC4 implies that the water has four times the total dissolved constituents compared to the makeup water.

Table 2.3.1 General water characteristics of treated municipal wastewater collected from Franklin Township Municipal Sanitary Authority (Murrysville, PA). Units are in mg/L except for pH.

Analyses	Secondary treated municipal wastewater (MWW) [a]	MWW with Nitrification and Filtration (MWW_NF) [a]	MWW with Nitrification, Filtration and GAC adsorption (MWW_NFG) [b]
Calcium, Total	33.3	46.7	39.8
Copper, Total	0.06	0.17	0.06
Iron, Total	0.32	0.31	0.09
Magnesium, Total	6.55	11.1	8.44
pH	7.16	6.65	7.94
Ammonia (NH_3-N)	24.1	1.42	0.39
COD	102	39.6	15.5
Chloride (Cl^-)	199	212	162
Nitrate (NO_3-N)	9.62	12.1	11.8
Sulphate (SO_4)	67.0	57.8	59.5
Phosphate (PO_4)	9.98	7.16	8.46
Total Alkalinity	123	25.1	44.2
Total Organic Carbon	26.5	8.94	3.21
Total Dissolved Solids	644	362	439

[a] : sampled during May, 2010 – August, 2010
[b] : sampled during May, 2011 – June, 2011

The two synthetic water recipes used in bench-scale experiments were: (a) synthetic secondary treated municipal wastewater (SynMWW), and (b) synthetic secondary treated municipal wastewater with nitrification-filtration (SynMWW_NF). Since no organic matter was considered in the recipe water used in bench-scale experiments, there was no separate synthetic recipe prepared for MWW_NFG. The SynMWW at CoC4 (pH 8.8) and the SynMWW_NF at CoC4 (pH 7.2) were prepared using the recipe given in Table 2.3.2.

Concentrations of different constituents in these recipes were obtained by averaging water quality data of previous similar pilot-scale cooling water system experiments performed using treated municipal wastewaters from FTMSA facility (Vidic et al., 2009). For bench-scale experiments with acidified synthetic secondary treated wastewater (SynMWW_pH), intermittent sulfuric acid was added in the recirculating bench-scale system cooling water to keep the pH of the water at 7.7.

Table 2.3.2 Recipe of synthetic secondary treated municipal wastewater (SynMWW) and synthetic secondary treated municipal wastewater with nitrification-filtration (SynMWW_NF) at four cycles of concentration (CoC4). Units are in mM except for pH.

Analyses	SynMWW (CoC4)	SynMWW_NF (CoC4)
pH	8.8	7.2
K^+	0.48	0.48
Na^+	13.0	8.60
Mg^{2+}	1.60	1.60
Ca^{2+}	4.00	4.00
Cl^-	11.2	11.2
CO_3^{2-}	6.00	0.40
SO_4^{2-}	3.50	3.50
NO^{3-}	0.00	1.20
PO_4^{3-}	0.48	0.48

Note: For MWW with pH control, the synthetic recipe of MWW was modified to keep the pH value fixed at 7.7 in the bench scale system. Also 50mg/L NH3-N was added separately to access effect of ammonia on corrosion with MWW_pH.

2.4 Chemical Treatment Methods

As mentioned earlier, the overall goal of this study was to investigate the feasibility of using tertiary treated municipal wastewater as makeup in recirculating cooling systems of thermoelectric power plants. Chemical treatment is commonly used in cooling systems to mitigate corrosion, scaling, and biofouling problems. In this study, Tolyltriazole (TTA) was selected as corrosion inhibitor; Polymaleic acid (PMA) was selected as scaling inhibitor; and oxidizing biocides, e.g., sodium hypochlorite (NaOCl), monochloramine (MCA), and chlorine dioxide (ClO_2), were selected as biofouling control agents. Detail information and fundamental mechanism of the inhibitors and biocides are described below.

2.4.1 Corrosion inhibitors

The use of corrosion inhibitors is the most widely employed approach to control corrosion in recirculating cooling water system (Frayne, 1999). Corrosion inhibitors usually form barrier layers on the surface of a metal and thus decrease corrosion rate. Barrier forming inhibitors are categorized into three types: adsorbed layer formers, oxidizing inhibitors (passivators), and conversion layer formers (Dean et al., 1981). Briefly, adsorbed layer formers function by adsorbing to the metal surface; oxidizing inhibitors function by shifting the metal's electrochemical potential to a region where the metal oxide or hydroxide is stable (passivating); and conversion layer formers function by forming a low solubility deposition on the metal surface (Dean et al., 1981).

Inhibitor selected for testing in this study was TTA from the National Colloid Company (Steubenville, OH). TTA is an inhibitor specifically tailored for copper alloys. Its NH group can adsorb onto the metal surface thus forming a barrier layer (Hollander and May, 1985). Copper corrosion inhibition by TTA can be reduced by oxidizing chemicals, such as free chlorine, monochloramine, etc. (Breske, 1983; Harrison and Kennedy, 1986), which is often maintained in cooling systems to prevent biological growth or biofouling. Contrary to hypochlorite, monochloramine has lower oxidizing power and is considered to have lower impact on TTA. However, studies of the influence of MCA on the effectiveness of TTA for inhibiting copper and other metal alloy corrosion are limited.

2.4.2 Scaling inhibitors

Scaling inhibitors (antiscalants) are widely used to prevent mineral scaling in recirculating cooling water systems (Frayne, 1999). Scaling inhibitors usually interact with the mineral nuclei to disrupt crystallization process and decrease the size of the precipitating colloids, making them less prone to sedimentation and surface deposition. Some antiscaling polymers adsorb onto surfaces, acting as a barrier to prevent mineral deposition. Another mechanism of scaling inhibition is through the formation of metal complexes with polymeric antiscalants to increase the operational solubility of precipitating metals, primarily Ca and Mg.

Numerous polymeric antiscalants with varied structural features and effectiveness in different waters are available. PMA was selected in this study as a principal antiscalant based on the review of the cooling water treatment literature (USDOE, 2007; Metcalf & Eddy, 2007; EPRI, 2008) and consultation with experts in cooling water design and operation (Scandolari 3/12/2008; Beardwood 3/17/2009; Christophersen 12/19/2007). PMA is believed to be effective as both dispersant and crystal distorter, particularly for Ca precipitates.

2.4.3 Biocides

The most common practice of dealing with the biological growth is to use a chemical disinfectant. Previous studies have shown that oxidizing biocides can serve as excellent biological growth control agent in treating municipal wastewater prior to reuse (USEPA, 1999; Tchobanoglous et al., 2003). Application of biocides to inactivate or eliminate microorganisms may be used on a continuous basis or intermittently, depending on the severity of the problem and cost. Grant and Bott (2005) suggest that only dosing the system with sufficient amount of biocide can succeed in biofouling control; otherwise, the addition of biocide might contribute to an increase in biofilm formation. Oxidizing biocides, such as free chlorine, monochloramine, and chlorine dioxide, have shown their effectiveness in water treatment processes as drinking water disinfectants (EPA, 1999). These three biocides are also effective biofouling control agents in treating cooling towers using freshwater (Kim et al., 2002; Ludensky, 2005).

Chlorination is often used to inhibit biological growth in cooling systems (Frayne, 1999). However, in order to maintain a certain level of free residual chlorine, a large amount of chlorine would be required to satisfy the biocide demand of organic matter and ammonia. This is especially true in the case where secondary treated municipal wastewater is used as cooling system makeup. When free chlorine is added at levels below those required for breakpoint chlorination, it will react with ammonia to form chloramines. Among inorganic chloramines, monochloramine has the highest disinfecting power and is commonly used as secondary disinfectant in drinking treatment (Wolfe et. al., 1984).

Monochloramine is regarded as a weaker disinfectant than free chlorine (hypochlorite) and higher levels may be required when it is used as primary biocide (Wolfe et. al., 1984; Morris, 1967). Turetgen (2004) observed that a 1 mg/L of monochloramine was significantly more effective than a 1.5 mg/L dose of free chlorine against cooling tower biofilm within 30 minutes contact time when using potable water as makeup water in both full scale and model cooling systems. Rao et al. (1998) demonstrated that initial monochloramine and free chlorine doses of 1, 2, and 3 mg/L showed similar biocidal activity in controlling biofilm growth formed on piping surfaces in a once-through cooling system. Monochloramine has been tested as alternative to hypochlorite for biological growth control in wastewater treatment (Aieta et al., 1980; Havelaat and Nieuwstad, 1985).

Chlorine dioxide was introduced as drinking water disinfectant because of low THM production (EPA, 1999). It has also been demonstrated to be effective against Legionella in drinking water distribution systems in Europe (Hood et al. 1999; Hamilton et al. 1996). Chlorine dioxide was reported to be more effective in treating Legionella pneumophila than chlorine and was less affected by pH and temperature (Botzenhart et al., 1993). Walker et al. (1995) demonstrated that a 10-minute continuous dose of 50-80 ppm of ClO_2 followed by 3-5 mg/L residual can ultimately remove biofilm and eradicate Legionella in a hospital hot water system.

2.5 Analytical Methods

2.5.1 Water quality analysis

Water quality parameters measured in the study include pH, conductivity, total solids (TS), total suspended solids (TSS), total dissolved solids (TDS), alkalinity, total organic carbon (TOC), chemical oxygen demand (COD), ammonia (NH_3) concentration, anions (chloride (Cl^-), Nitrate (NO_3^-), Sulfate (SO_4^{2-}), Phosphate (PO_4^{3-}) concentration, and cations (calcium (Ca), magnesium (Mg), Iron (Fe) and Copper (Cu)) concentration. The methods and instruments used in each analysis are summarized in Table 2.5.1.

Table 2.5.1 Analytical methods used in the study

Parameters	Methods	Instruments
pH	pH meter	Fisher Science Education pH meters[a]
		Oakton* pH 11 Standard Portable Meter[b]
Conductivity	Conductivity meter	Fisher Scientific* accumet* AP75 Portable Waterproof Conductivity/TDS meter
Total solids (TS)	Standard Methods for the Examination of Water and Wastewater Section 2540	-
Total Suspended Solids (TSS)		
Total dissolved solids (TDS)		
Alkalinity	Standard Methods for the Examination of Water and Wastewater Section 2320	Filtered water was titrated with standard sulfuric acid to end-point pH at 4.5
Total Organic Carbon (TOC)	Total Organic Carbon (TOC) analyzer	Total organic carbon analyzer (O.I. Analytical, TX)
Chemical Oxygen Demand (COD)	Hach® Method 8000 (USEPA-approved method)	Hach COD reactor Model 45600 COD Reagent and Digestion Vials Hach® DR/850 Portable Colorimeter
Ammonia (NH_3) Concentration		
Chloride (Cl^-)	Standard Methods for the Examination of Water and Wastewater Section 4410B	DIONEX Ion Chromatography System (DIONEX, Sunnyvale, CA)
Nitrate (NO_3^-)		
Sulfate (SO_4^{2-})		
Phosphate (PO_4^{3-})	Standard Methods for the Examination of Water and Wastewater Section 4500 (molybdate/ascorbic acid method)	Milton Roy Spectronic 20D Spectrophotometer
Calcium (Ca)	Flame Atomic Absorption Spectrometry	Perkin Elmer 1100B Atomic Absorption Spectrometer
Magnesium (Mg)		
Iron (Fe)		
Copper (Cu)		

Notes: [a] For bench-scale studies; [b] For Pilot-scale studies

2.5.2 Solids characterization

For the solids collected in bench-scale studies with synthetic wastewater, the crystalline characteristics of the solids were analyzed by x-ray diffraction (XRD, PW 1830, Philips Analytical Inc., Natick, MA) with Cu Kα radiation. Both manual matching of the peak positions and a computer-aided search for the compounds were performed for the diffraction patterns obtained for each specimen. The morphology of the solids was inspected using Scanning Electron Microscopy (SEM, Philips XL30, FEI Co., Hillsboro, OR) and the elemental compositions of selected samples were determined by Energy Dispersive X-ray Spectroscopy (EDS, EDAX Inc., Mahwah, NJ). Prior to the SEM/EDS analyses, samples were sputtered with Pd.

Complicated water chemistry in pilot-scale studies with real wastewater made it impossible to form well-developed solid crystals and only SEM/EDS analysis was conducted on these samples to identify key components of solids formed in these studies. Prior to analysis, the solids were first dried at 104°C for 3.5 hours and then burned at 500°C for 3.5 hours to remove volatile organic components of these deposits.

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3.0 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: pH Adjustment (MWW_pH)

Increasing scarcity of freshwater supply for use in power plant recirculating cooling systems has led to the search for alternative impaired water sources. Secondary treated municipal wastewater (MWW) is a widely available alternative cooling water source in terms of quantity and geographic proximity to most of the existing and future power plants (Chien et al., 2008). MWW contains elevated concentrations of organic matter, hardness, orthophosphate, ammonia, and total dissolved solids relative to freshwater (Williams, 1982; Weinberger, 1966). The main challenge when secondary treated municipal wastewater is reused in recirculating cooling water systems is more complicated control of corrosion, scaling, and biological fouling due to the lower quality of the water. Thus use of MWW in power plant cooling systems requires additional treatment and/or inhibitor dosing attention to mitigate corrosion, scaling and biological fouling problems that can rise due to lower water quality (EPRI, 2003).

Previous study showed that use of MWW resulted in low corrosion of mild steel, copper and cupronickel alloys (Hsieh et al., 2010). The low corrosiveness of MWW was due to its high scaling potential, which caused formation of a protective scaling layer on the metal surfaces (Hsieh et al., 2010; Uhlig and Revie, 1985). However, scaling on metal and metal alloy surfaces reduces the heat transfer efficiency of the cooling system.

In this study, pH of the secondary treated municipal wastewater (MWW) was reduced by continuous acid addition. Tertiary treatment applied in the form of pH reduction reduces the scaling potential of the MWW (Keister, 2001). As a result an improved water quality was available with less scaling problems in the cooling systems. The feasibility of controlling corrosion, scaling, and biofouling when using acidified secondary treated municipal wastewater (MWW_pH) in cooling water systems was investigated through laboratory and pilot-scale experiments. Bench-scale recirculating systems and three pilot-scale cooling towers were employed for testing of various chemical control schemes for corrosion, scaling, and biofouling in systems using secondary treated municipal wastewater. The testing was conducted with conditions of temperature, flow velocity, and water constituent concentration similar to those in a recirculating cooling water system. Synthetic acidified secondary treated municipal wastewater recipe (SynMWW_pH) was simulated to have similar constituent concentration of actual MWW_pH used as cooling water in pilot-scale systems. The effectiveness of chemical treatment strategies in inhibiting corrosion, scaling, and biomass growth was studied through exposure and monitoring specially designed coupons in extended duration tests.

3.1 Corrosion Control for MWW_pH Used for Cooling

In this study corrosion of different metals and metal alloys was evaluated while acidified secondary treated municipal wastewater (MWW_pH) was used as makeup water in power plant cooling systems. Tertiary treatment applied in the form acidification lowered the pH of the secondary treated municipal wastewater. As a result the scaling problem in cooling systems, associated with the use of secondary treated municipal wastewater (MWW), was alleviated. However, reduction in the scaling problem caused a new scenario for corrosion management as the metals and metal alloys surfaces became exposed to the flowing cooling water. The overall goal of this study was to investigate the feasibility of using acidified secondary treated municipal wastewater (MWW_pH) in cooling water systems by metal alloy corrosion rate monitoring and comparison with general corrosion criteria (Hsieh et al., 2010).

The use of corrosion inhibitors is the most widely employed approach to control corrosion in recirculating cooling water system (Frayne, 1999). Corrosion inhibitors usually form barrier layers on the surface of a metal and thus decrease corrosion rate. Barrier forming inhibitors are categorized into three types: adsorbed layer formers, oxidizing inhibitors (passivators), and conversion layer formers (Dean et al., 1981). Briefly, adsorbed layer formers function by adsorbing to the metal surface; oxidizing inhibitors function by shifting the metal's electrochemical potential to a region where the metal oxide or hydroxide is stable (passivating); and conversion layer formers function by forming a low solubility deposition on the metal surface (Dean et al., 1981). Among the commonly used corrosion inhibitors, tolyltriazole was selected to be tested in this research based on review of previous studies (Hsieh et al., 2010; McCoy, 1974; Frayne 1999; Jones, 1996; Harston, 2004).

Tolyltriazole (TTA) is an inhibitor specifically for copper alloys. Its NH group can adsorb onto the metal surface thus forming a barrier layer (Hollander and May, 1985). Copper corrosion inhibition by TTA can be reduced by free chlorine residual (Breske, 1983; Lu et al., 1994; Harrison and Kennedy, 1986), which is often maintained in cooling systems to prevent microbial growth and biofouling. On the other hand, chloramine has lower oxidizing power but studies of its influence on copper corrosion inhibition by TTA are limited. Previous studies (Hsieh et al., 2010) also showed that TTA was able to lower the corrosion rate of mild steel immersed in MWW.

Specifically, the objectives of this study were to 1) design an experimental matrix and perform laboratory experiments to evaluate metal alloy corrosion immersed in SynMWW_pH, 2) assess corrosion of metal alloys immersed in MWW_pH in a bench-scale recirculation water system with identified corrosion control program, and 3) apply the optimal corrosion control program identified from bench-scale experiments to pilot-scale cooling systems operated with acidified secondary treated municipal wastewater (MWW_pH) in the field.

3.1.1 Testing protocols

3.1.1.1 Metal alloy pre-exposure and post-exposure treatment

The metal and metal alloys chosen for this study were mild steel (UNS G10180), copper (UNS C10100), and cupronickel (UNS C70600) as they are suitable for cooling water systems using municipal wastewater as makeup water (Hsieh et al., 2010; Herro and Port, 1993). The specimens were cylinder-shaped with a nominal diameter of 0.375 in., nominal length of 0.5 in. and were obtained from Metal Samples Co. (Munford, AL). Before immersing the metal and metal alloy samples in recirculating cooling water (both in bench-scale and pilot-scale systems), they were wet polished with SiC paper to a 600 grit surface finish, dried, weighed to 0.1 mg, degreased with acetone and rinsed in distilled water.

During exposure of the metal alloy specimens in the bench-scale experiments, the polarization resistance (R_P) of the specimens was semi-continuously monitored and specimens were withdrawn after approximately 1, 3, and 5 days of exposure to measure the weight loss. After withdrawal, the specimens were cleaned following the ASTM G1 and then reweighed to 0.1 mg to determine the weight loss (ASTM G1-03, 2005).

In the pilot scale cooling system tests, the metal and metal alloy specimens were immersed for periods of 7, 14, 28, and 56 days. The weight loss method was used to determine the average corrosion rate during the exposure period. After withdrawal, similar post-exposure treatments were performed on the metals and metal alloys as was done in the bench-scale experiments.

3.1.1.2 Corrosion experiment matrix for the bench-scale experiments with MWW_pH

Synthetic secondary treated municipal wastewater recipe was used in the bench-scale recirculation system for corrosion analysis of mild steel, copper and cupronickel alloys. pH of the synthetic secondary treated municipal wastewater was lowered to pH 7.7 using intermittent sulfuric acid addition in the system. Average 5-day corrosion rates of different metal alloys were determined. In the laboratory experiment with acidified synthetic secondary treated municipal wastewater tolyltriazole (TTA) was used as corrosion inhibitor. Instantaneous corrosion rate profiles for different metal alloys were produced from intermittent R_P measurement data (Choudhury et al., 2012)

Four different experiments were carried out in the bench-scale recirculation water system with synthetic secondary treated municipal wastewater. The matrix of experiments conducted is shown in Table 3.1.1. The matrix was designed to 1) study the influence of pH reduction on mild steel, copper, and cupronickel corrosion, and 2) evaluate the effectiveness of TTA in corrosion inhibition of mild steel, copper and cupronickel alloys immersed in acidified synthetic secondary treated municipal wastewater.

Actual secondary treated municipal wastewater collected from the Franklin Township Municipal and Sanitary Authority (FTMSA) facility was concentrated in the laboratory by heated evaporation (at 40°C) to reach four cycles of concentration (CoC4) as determined by 75% volume reduction for bench-scale experiments. The temperature (40°C), used for heated evaporation in the lab, simulates the temperature of recirculating cooling water in the pilot-scale cooling systems (Hsieh et al, 2010). At CoC4 the secondary treated municipal wastewater was used in the bench-scale recirculation water system for corrosion analysis of mild steel, copper and cupronickel alloys. pH of the actual water was controlled to 7.7 with intermittent sulfuric acid addition.

Two experiments were carried out in the bench-scale recirculation system using acidified secondary treated municipal wastewater (with and without addition of 5 ppm TTA) for corrosion analysis of mild steel, copper, and cupronickel alloys. The samples were immersed for 5 days for average corrosion rate analysis. Intermittent R_P measurements were performed throughout the immersion period of the metal alloy samples for producing an instantaneous corrosion rate profile for the metal alloys.

Table 3.1.1 Experimental matrix for bench-scale corrosion analysis of metal and metal alloy samples in contact with synthetic secondary treated municipal wastewater. Flow rate 0.189 L/s (3gpm), temperature 40 ± 1 ^{o}C.

Synthetic secondary treated municipal wastewater composition tested	Tolyltriazole (TTA) concentration used, mg/L	Objective of experiments with different synthetic wastewater compositions	
		Effectiveness of inhibitors	**Influence of pH control**
CoC4 (no pH)	0		X
CoC4 [a]	0		X
CoC4_NH [a]	0	X	
CoC4_NH_TTA5 [a]	5	X	

Note: CoC4: Four cycles of concentration. TTA5: Tolyltriazole 5 ppm. nopH: no additional pH control was employed, pH of secondary treated municipal wastewater was kept at 8.8. NH: ammonia added (50 mg/L as N).
a pH controlled to 7.7 by intermittent dosing of sulfuric acid.

3.1.1.3 Pilot-scale corrosion experiment with MWW_pH

The experimental matrix for the pilot-scale testing was based on the results of the laboratory experiments with respect to corrosion, scaling, and biofouling studies (scaling and biofouling control studies are not covered in this section). TTA was chosen as corrosion inhibitor, and PMA (a scaling inhibitor identified to be effective in scaling control through the laboratory experiments in another study) as a scaling inhibitor. Monochloramine and free chlorine both were used as biofouling control agents. Three different pilot-scale cooling towers were operated for 56 days after reaching CoC 4. One of the cooling towers (CTA) was operated using secondary treated municipal wastewater at CoC 4, while the remaining two cooling towers (CTB and CTC) were operated using acidified secondary treated municipal wastewater at CoC 4. Intermittent sulphuric acid dosing was done directly in the cooling tower basin water to control the pH to 7.7. Table 3.1.2 shows the pilot-scale experimental matrix with chemical treatment strategies.

In the pilot-scale testing, corrosion inhibitor TTA was monitored with high performance liquid chromatography (HPLC 1100 Series, Agilent Technologies) (Choudhury et al., 2012). Other water chemistry parameters (PMA, monochloramine, free chlorine, anions, alkalinity, pH and conductivity) and cooling tower operational parameters (water flow rate, air flow velocity, temperature, makeup water flow rate, and blowdown water flow rate) were also monitored. The inhibitor TTA was added once per day to the makeup water tank. Concentration of TTA dosed in the makeup water was one-fourth of the target concentration of TTA in the cooling tower system.

TTA concentration in the pilot scale cooling tower basin reached to desired target concentration as the recirculating cooling water reached CoC 4.

Table 3.1.2 Experimental matrix for pilot-scale cooling tower experiments showing different chemical treatment strategies for corrosion, scaling, and bio-fouling management. Flow rate 0.189 L/s (3gpm).

Cooling Tower Designation	pH	Corrosion Inhibitor	Scaling Inhibitor	Bio-fouling Control
Cooling Tower A (CTA)	8.4±0.3	TTA (2 ppm)	PMA (5 ppm)	MCA (3~4 ppm residual)
Cooling Tower B (CTB) [a]	7.5±0.5	TTA (5 ppm)	PMA (5 ppm)	MCA (3~4 ppm residual)
Cooling Tower C (CTC) [a]	7.7±0.5	TTA (5 ppm)	PMA (5 ppm)	FC (3~4 ppm as TC residual)

Note: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); FC (Free chlorine); TC (Total chlorine). [a] pH controlled by intermittent dosing of sulfuric acid into the cooling tower basin.

3.1.2 Bench-scale studies

3.1.2.1 Bench-scale recirculation system experiments with SynMWW_pH: corrosion analysis

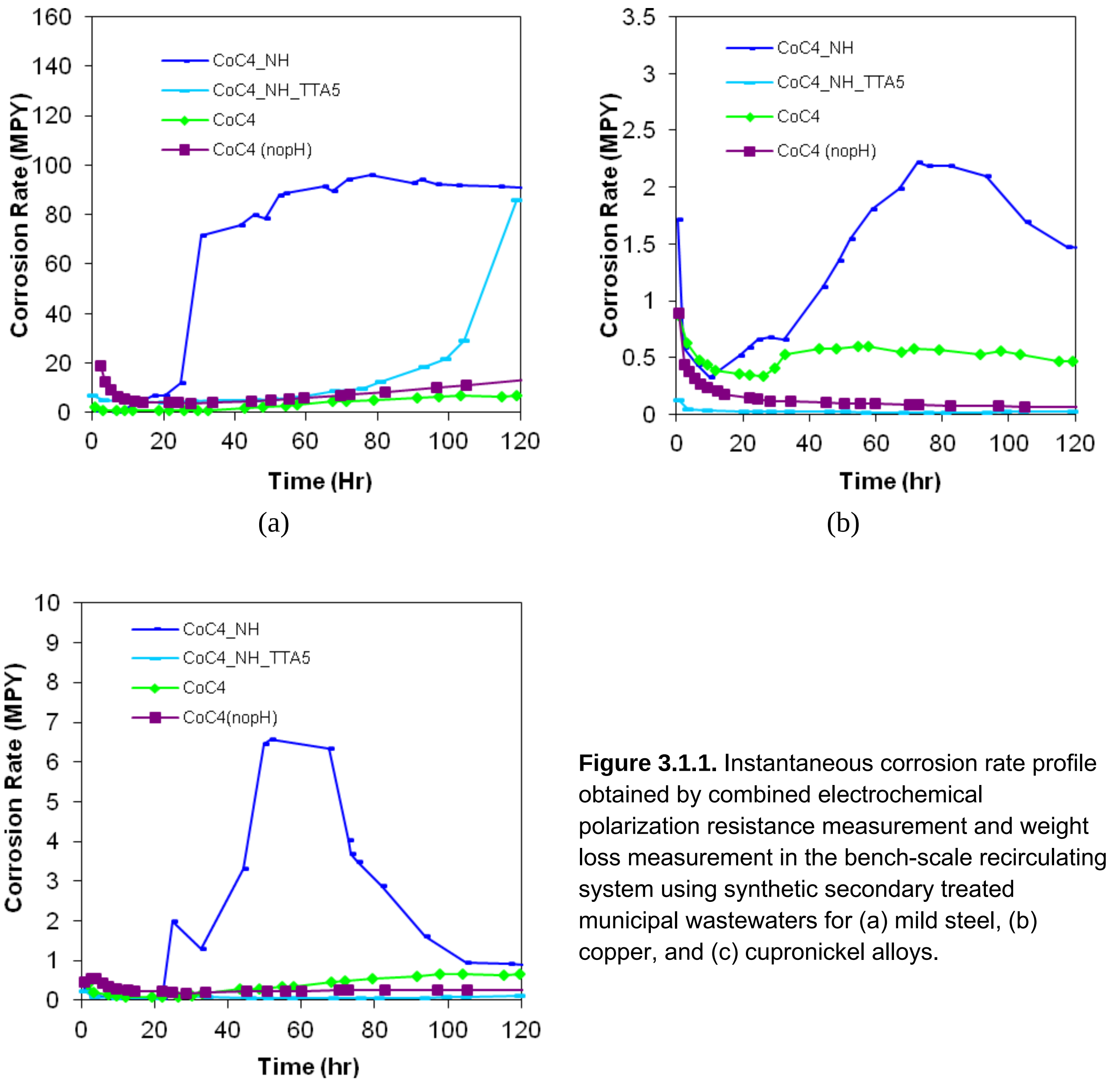
The average 5-day corrosion rates from the four bench-scale recirculation system experiments are given in table 3.1.3. Corrosion inhibition efficiency values are also given with respect to the control experiment using pH controlled synthetic secondary treated municipal wastewater with ammonia. From the results given in Table 3.1.3 it is observed that pH reduction increased the corrosivity of the secondary treated municipal wastewater. Presence of ammonia at 50 ppm as N exacerbated the corrosion of mild steel, copper and cupronickel alloys. Use of 5 ppm TTA reduced copper and cupronickel corrosion rates significantly. Mild steel corrosion rates were also reduced by about 65% in the presence of 5 ppm TTA.

Table 3.1.3 Average corrosion rates of metal and metal alloy samples, and corresponding corrosion inhibition efficiencies in the bench-scale recirculating cooling water experiments. The metal and metal alloys were exposed to synthetic cooling water for five days.

Synthetic secondary treated municipal wastewater composition tested	Tolyltriazole (TTA) concentration used, mg/L	Average Corrosion Rate (MPY) (corrosion inhibition efficiency, %)		
		Mild Steel	**Copper**	**Cupronickel**
CoC4 (no pH)	0	2.09 (97.8%)	0.108 (90.9%)	0.208 (93.5%)
CoC4 [a]	0	2.35 (97.6%)	0.382 (67.7%)	0.310 (90.2%)
CoC4_NH [a]	0	96.9 (--)	1.18 (--)	3.17 (--)
CoC4_NH_TTA5 [a]	5	34.3 (64.6%)	0.064 (94.6%)	0.075 (97.6%)

Note: CoC4: Four cycles of concentration. TTA5: Tolyltriazole 5mg/L. nopH: no additional pH control was employed, pH of secondary treated municipal wastewater was kept at 8.8. NH: ammonia added (50 mg/L as N).
[a] pH controlled to 7.7 by intermittent dosing of sulfuric acid.

The instantaneous corrosion rate (ICR) profiles for mild steel, copper, and cupronickel samples in the bench-scale recirculation system experiments are shown in Figure 3.1.1. The ICR profiles indicated that presence of ammonia exacerbated corrosion of all the metal alloys. TTA was able to inhibit corrosion of mild steel, copper and cupronickel in the presence of ammonia.



(a)

(b)

(c)

Figure 3.1.1. Instantaneous corrosion rate profile obtained by combined electrochemical polarization resistance measurement and weight loss measurement in the bench-scale recirculating system using synthetic secondary treated municipal wastewaters for (a) mild steel, (b) copper, and (c) cupronickel alloys.

3.1.2.2 Bench-scale recirculation system experiments with MWW_pH: corrosion analysis

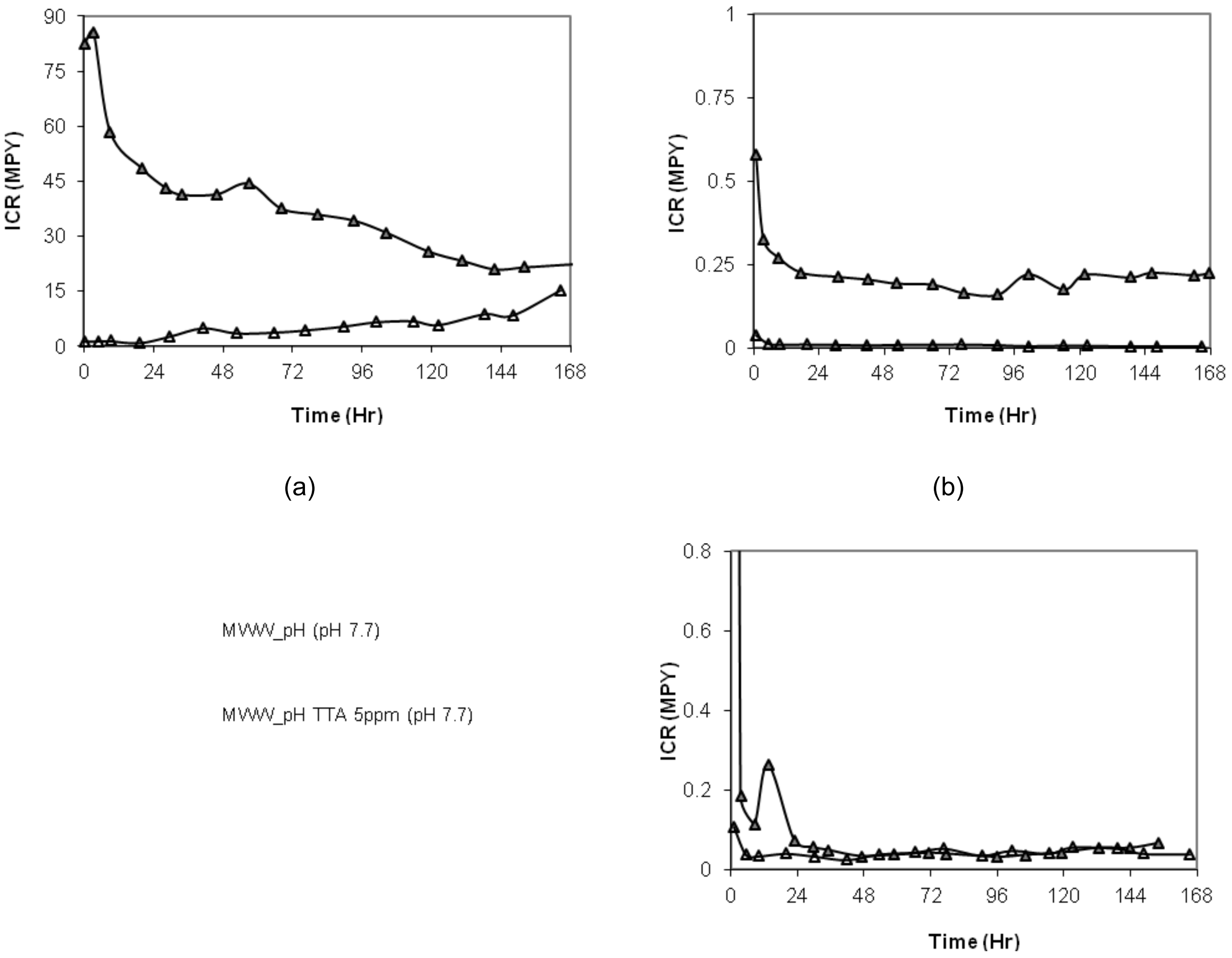
MWW_pH water collected from FTMSA facility was used in bench-scale recirculation system experiments at CoC 4 with and without the addition of corrosion inhibitor TTA. Table 3.1.4 shows the average 5-day corrosion rates of metals and metal alloys immersed in MWW_pH. Results from the table indicated that TTA successfully inhibited the corrosion of copper and cupronickel alloys to excellent levels according to the cooling tower system corrosion criteria. Mild steel corrosion rate was significantly inhibited and lowered to acceptable limits using TTA as corrosion inhibitor.

Table 3.1.4 Average 5-day corrosion rates of mild steel, copper and cupronickel samples exposed to MWW_pH at 4CoC, collected from Franklin Township Municipal Sanitary Authority (FTMSA) wastewater treatment facility, in the bench-scale recirculating system.

Type of actual tertiary treated wastewater	Average 5-day corrosion rate, MPY (Corrosion category [b])		
	Mild Steel	**Copper**	**Cupronickel**
MWW_pH (Control)	33.9 (Unacceptable)	0.295 (Good)	0.285 (Good)
MWW_pH (TTA 5mg/L)	5.79 (Poor)	0.011[a] (Excellent)	0.065 [a] (Excellent)

Note: [a] weight loss determined from electrochemical polarization resistance measurements (Choudhury, 2012).
[b] b Hsieh et al., 2010.
MPY: milli-inch per year. Control: no corrosion inhibitor was dosed in the system. TTA: Tolyltriazole.

Figure 3.1.2 shows the instantaneous corrosion rate measurement profiles for mild steel, copper and cupronickel alloy samples measured for an interval of 7 days in the bench-scale recirculating system with MWW_pH. The pH of each type of recirculating water along with respective TTA dosing is also indicated in Figure 3.1.2.



(a)

(b)

(c)

Figure 3.1.2 Instantaneous corrosion rate profiles for (a) mild steel, (b) copper, and (c) cupronickel alloy samples immersed in bench-scale recirculating cooling system using MWW_pH.

3.1.3 Pilot-scale studies

Pilot-scale experiments were carried out with three cooling towers (CTA, CTB and CTC) using secondary treated municipal waste water. In these experiments CTA, CTB and CTC were initially dosed with 2ppm TTA. However pH control to 7.7 in CTB and CTC enhanced the corrosion rate of metal alloys in these towers. So the TTA dosing to CTB and CTC was increased to 5ppm after day 10 of the MWW run.

Table 3.1.5 shows the average corrosion rates of metal and metal alloy specimens immersed in secondary-treated wastewaters at CoC 4 in the three pilot-scale cooling towers for different time intervals. It can be seen from Table 3.1.5 that satisfactory corrosion control was achieved with copper and cupronickel in the three cooling towers. However, pH control to 7.7 required use of additional TTA for corrosion control of alloys in the system. Hence pH control required a different corrosion control strategy.

When pH control was not done, as in CTA, use of 2ppm TTA was sufficient to control the corrosion rate of copper and cupronickel. Based on the 56-day average corrosion rate, both copper and cupronickel had average corrosion rate less than 0.2 MPY. This indicates excellent copper corrosion control in CTA.

pH control to 7.7 in CTB and CTC required a higher dose of TTA (5ppm) to control the average corrosion rate for copper and cupronickel. Table 3.1.5 shows the 56-day average corrosion rate of the copper and cupronickel in CTB and CTC. Except for the cupronickel alloy in CTB the rest of the specimens displayed an average 56-day corrosion rate lower than 0.2 MPY. However the 56-day samples were initially dosed with 2ppm TTA for the first 10 days. In Table 3.1.5 only the 28-day samples were dosed with 5ppm TTA in CTB and CTC for the entire immersion period, and the average 28-day corrosion rates are less than 0.2 MPY. Hence use of 2ppm TTA with secondary treated wastewater and 5ppm TTA with pH (7.7) controlled secondary treated wastewater were able to control the corrosion rate of copper and cupronickel in the cooling systems to acceptable levels.

Mild steel corrosion control with the use of TTA in systems with secondary treated municipal wastewater was fair. The 56-day average corrosion rate in CTA (4.99 MPY) and CTC (3.75 MPY) were within the fair corrosion control range. CTB showed a 56-day average corrosion rate of 8.98 MPY, indicating a poor corrosion control. The 7-day average corrosion rate shows that the systems with lower TTA dosing (2ppm) exhibited severe corrosion to mild steel in pH controlled cooling towers CTB (19.94 MPY) and CTC (12.84 MPY). From the average corrosion rate data it can be said that TTA moderately controls the corrosion rate of mild steel.

Table 3.1.5 Average corrosion rates of metal alloys contacted with secondary-treated wastewater (with and without pH control) at CoC 4 in pilot-scale cooling systems for different time intervals and their category according to general corrosion criteria[b] for cooling systems.

Cooling Tower	Immersion Days	Average Corrosion Rates of Metal Alloys (MPY) (Corrosion category)		
		Mild Steel	**Cupronickel**	**Copper**
CTA [a]	7	14.43 (Unacceptable)	0.04 (Excellent)	0.34 (Good)
	14	8.76 (Poor)	0.27 (Good)	0.17 (Excellent)
	28	5.27 (Poor)	0.03 (Excellent)	0.05 (Excellent)
	56	4.99 (Fair)	0.05 (Excellent)	0.03 (Excellent)
CTB [a]	7	19.94 (Unacceptable)	0.04 (Excellent)	0.34 (Good)
	14	12.75 (Unacceptable)	0.57 (Fair)	0.25 (Good)
	28	6.63 (Poor)	0.08 (Excellent)	0.02 (Excellent)
	56	8.98 (Poor)	0.21 (Good)	0.06 (Excellent)
CTC [a]	7	12.84 (Unacceptable)	0.04 (Excellent)	0.26 (Good)
	14	10.65 (Unacceptable)	0.07 (Excellent)	0.21 (Good)
	28	2.24 (Good)	0.06 (Excellent)	0.07 (Excellent)
	56	3.75 (Fair)	0.10 (Excellent)	0.03 (Excellent)

[a] CTA : Cooling Tower A with 2 ppm TTA, 5 ppm PMA and 3-4 ppm MCA; CTB: Cooling Tower B with 5 ppm TTA, 5 ppm PMA, 3-4 ppm MCA and pH control to 7.7; CTC: Cooling Tower C with 5 ppm TTA, 5 ppm PMA, 3-4 ppm (as total chlorine) FC and pH control to 7.7.

[b] Hsieh et al., 2010.

Figure 3.1.3 shows the concentration profiles of TTA in CTA, CTB and CTC. The average free TTA concentration in CTA was 1.82±0.44 ppm, which was close to the desired 2ppm target TTA dose. The average TTA concentrations in CTB and CTC for the entire field experiment run were 2.03±1.23 ppm and 1.61±0.66 ppm respectively. Both CTB and CTC were dosed with 2ppm TTA in the first 10 days of the testing. This resulted in a depletion of free TTA in CTB on day 10 of the experiment and caused severe corrosion in the system. The excess dose of TTA (5 ppm) was necessary to keep residual free TTA in solution for CTB and CTC. The average TTA concentrations in the CTB and CTC excluding the first 10 days were 2.52±1.23 ppm and 1.96±0.75 ppm respectively.

The TTA concentrations in CTB and CTC were always less than the target 5ppm concentration. The high corrosivity of the pH controlled secondary treated wastewater enhanced the corrosion rate and required more TTA in solution for the repair of the corrosion protection film. The presence of cupric ion in solution, due to enhanced corrosivity of the water, likely contributed to the reduction in the free TTA concentration by formation of copper-TTA complex.

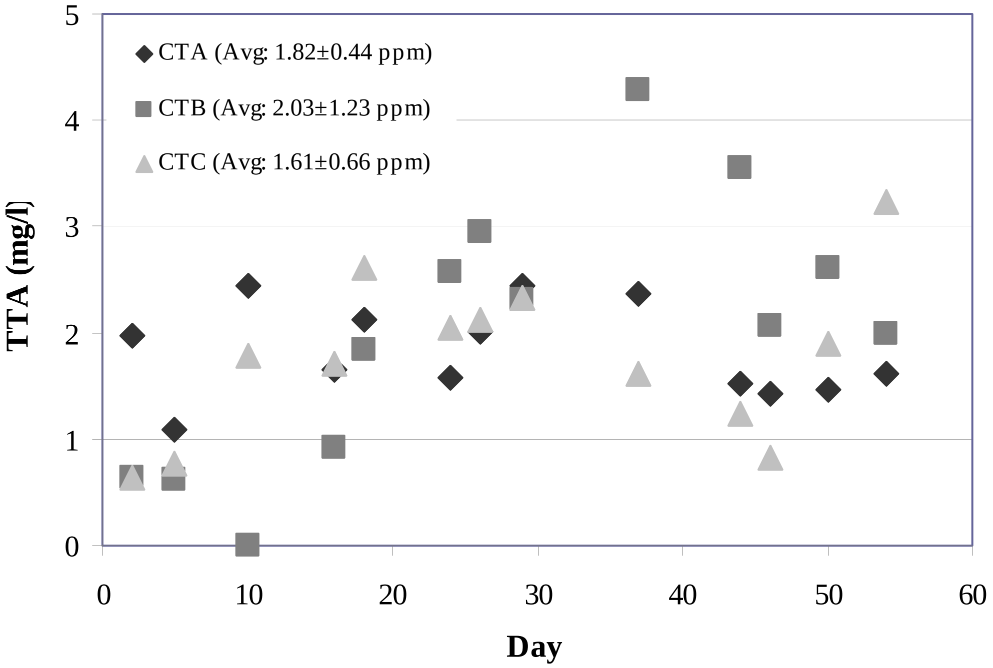


Figure 3.1.3 Concentration profile of TTA in CTA, CTB, and CTC (measured twice each week) in the pilot scale cooling tower experiments using Secondary treated municipal wastewater (MWW) and pH controlled secondary treated municipal wastewater (MWW_pH) from FTMSA.

The pH variation (Figure 3.1.4) in the cooling water was not very much in the pilot-scale experiments which resulted in consistent average corrosion rates across different time intervals as shown in Table 3.1.4. The 7-day, 14-day, 28-day and 56-day immersion periods all had similar pH conditions. Hence it was expected to see higher average corrosion rates for shorter (7-day, 14-day) immersion periods than longer (28-day, 56-day) immersion periods. The mild steel and copper corrosion rates in the three cooling towers confirmed this expected result. The cupronickel corrosion rates in the three cooling towers were very low in most cases and the variation in the average corrosion rates was not significant.

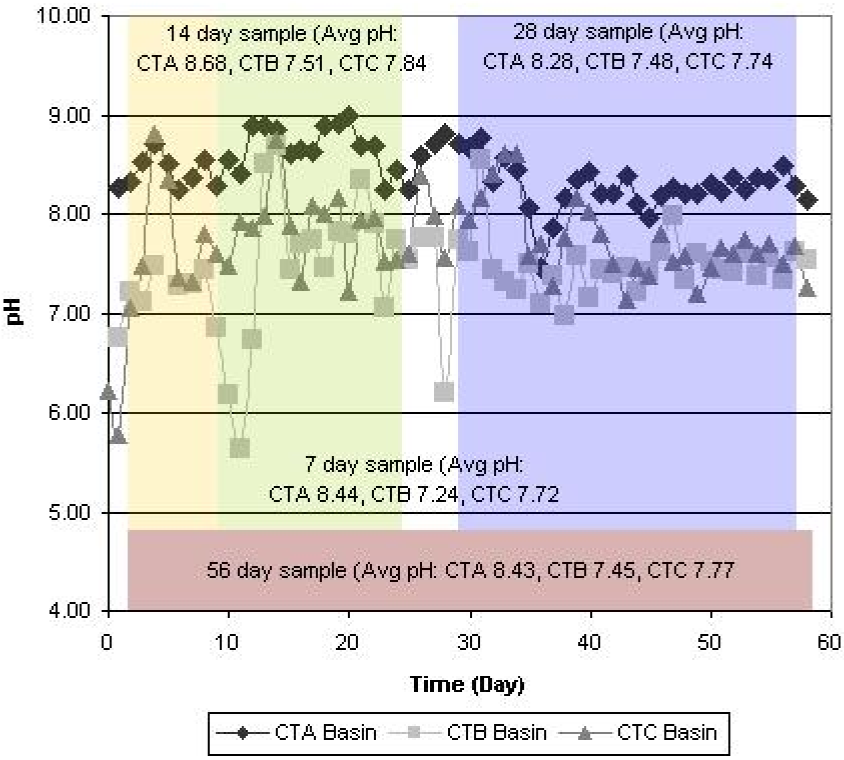


Figure 3.1.4 pH profile in CTA, CTB and CTC in the pilot scale cooling tower experiments using secondary treated municipal waste water from FTMSA. Average pH values in shaded regions, indicating the immersion period of different metal alloys in cooling waters, are given for all three cooling towers.

3.2 Scaling Control for MWW_pH Used for Cooling

Mineral scaling is a critical issue with MWW as make-up water for recirculating cooling systems (Vidic et al., 2009). Among common antiscalants, phosphorous-based chemicals are not appropriate due to their significant loss through precipitation reactions with calcium (Vidic et al., 2009). Polymaleic acid (PMA), as a model of carboxylic polymers, was shown to be effective in bench-scale tests. However, its effectiveness was compromised in extended pilot-scale cooling tower tests (Li et al., 2011). It was also shown that the major mineral scales formed in recirculating cooling systems using secondary-treated MWW as make-up water are calcium carbonate and to a lesser extent calcium phosphate (Li et al., 2011).

Precipitation of inorganic carbonates and phosphates is highly dependent on the solution pH and it is reasonable to assume that lowering the solution pH should lower the formation of these solids by lowering the solution alkalinity and thus lowering the driving force for scale formation. However, concerns like enhanced corrosion rates would arise at lower pH (Troup and Richardson, 1978; Hsieh et al., 2010). Although the impact of pH reduction on scaling control is well known, proper acid dosing and the combined effects of acid addition and other chemical treatment programs, such as antiscalants, corrosion inhibitors and biocides, in cooling systems using MWW_pH have not been investigated before.

In this study, batch tests and bench-scale recirculating system studies were conducted to determine the desired pH range and the synergism between pH adjustment and antiscalant addition on scaling mitigation on both unheated and heated surfaces in recirculating cooling systems when using MWW_pH as make-up water. Pilot-scale cooling tower tests were then conducted to evaluate the applicability of selected scaling mitigation strategies in the field and to test their compatibility with corrosion and biofouling control methods in the integrated chemical regimen for MWW_pH reuse in recirculating cooling systems.

3.2.1 Testing protocols

Typical operation of a recirculating cooling system will concentrate the feed water as much as 4-6 times due to evaporative losses in the system. A synthetic wastewater (Table 3.2.1) was prepared to represent secondary-treated MWW at 4 cycles of concentration (CoC 4) in terms of its mineral content for detailed investigation in batch and bench-scale recirculating system tests (Li et al., 2011). It must be noted that ammonia is not considered in the recipe since most of it would be stripped out during recirculation in the cooling tower (Hsieh et al., 2010; Rebhum and Engel, 1988).

Table 3.2.1 Chemical compositions of the synthetic secondary-treated municipal wastewater (MWW) at 4 cycles of concentration (CoC 4) for batch and bench-scale recirculating system tests

Cation	Concentration (mM)	Anion	Concentration (mM)
Ca2+	7.60	SO42-	2.84
Mg2+	7.16	HCO3-	13.44
Na^+	26.88	Cl^-	37.35
K^+	0.70	PO_4^{3-}	0.21

In the batch tests, precipitation behavior of MWW_ pH at CoC 4 was first studied in a 1.4 L beaker covered with plastic foil to minimize water loss due to evaporation. The contents in the beaker were completely mixed with a magnetic stirrer and the bulk temperature was controlled at 40°C (typical bulk temperature of power plant recirculating cooling water) using the heating plate. The experiments were always initiated by adding all the salts listed in Table 3.2.1 except for $CaCl_2$. For the tests intended to study the impact of pH on precipitation behavior of MWW_ pH at CoC 4, the solution pH was then lowered to 6.8 with 0.5 M hydrochloric acid (previous tests have shown that there will be no immediate bulk precipitation at pH 6.8) followed by the addition of $CaCl_2$. pH of the synthetic solution was then adjusted and maintained at a desired pH with 0.5 M hydrochloric acid or 0.5 M sodium hydroxide. The effectiveness of Polymaleic acid (PMA, Kroff Chemical Company, Pittsburgh, PA) as a model antiscalant on the precipitation was tested by adding it to the solution before the addition of any salts. Precipitation behavior in the beaker was monitored by withdrawing 5 mL of the solution at predetermined intervals. The water sample was filtered through a 0.45 μm membrane and immediately acidified to pH< 2 with concentrated HNO_3. Calcium and phosphate concentrations were determined by atomic absorption spectroscopy and the molybdate/ascorbic acid method (American Water Works Association, 2005), respectively. Mg concentration was not monitored because previous study (Li et al., 2011) has shown that Mg precipitation was negligible. Precipitated solids were collected, washed with DI water, and air-dried for subsequent petrographic and chemical characterization.

A bench-scale recirculating system (described in Section 2.1.2) was used to investigate the deposition behavior of MWW_ pH under different scaling mitigation strategies. Synthetic treated MWW was used for reproducible solution chemistry. Mineral mass deposited on stainless steel (SS) discs (5.61 cm^2) was monitored to track the scaling process. Water temperature and SS surface temperature were 40°C (105°F) while flow velocity was 0.6 m/s, to reflect actual conditions of industrial cooling systems. In a typical test, the recirculating water was exposed to air so that the alkalinity may approach equilibrium with CO_2 (g), as is the case with actual cooling system operation. Before use, the SS specimens were cleaned by ultrasonic wash for 5 min in an acetone/ethanol solution (1:1 v/v ratio), rinsed with DI water and air-dried in a laminar flow hood. At predetermined time intervals, the SS specimens were taken out of the recirculating water through the sampling ports. The water remaining on the disc surface was carefully removed by paper tissue without disturbing the solid deposited on the surface. The discs were then air-dried for at least 48 hours and the mass of each disc was measured using an analytical balance (Mettler AE163, detection limit 0.01 mg). Final weighing was performed only after a constant mass was achieved (mass measurement variance < 0.05 mg/h). Three measurements were taken for each specimen and the average value was reported as the mineral mass on the disc. After weighing, the morphology of the scale samples was inspected by SEM methods. After each experiment, the recirculating system was cleaned with HCl solution (pH 2-3) for about 1 h, followed by two DI water rinses, with 0.5 hour of water recirculation each time.

Besides the tests on the unheated surfaces with SS discs, mineral scaling on the heated surfaces was also conducted in the bench-scale fouling study system. Fouling resistance was recorded according to the temperature measurement by the datalogger system. When the fouling resistance reached equilibrium, the cartridge heater was removed and the deposits were analyzed using XRD and SEM/EDX.

Field tests with pilot-scale cooling towers were conducted at the Franklin Township Municipal Sanitary Authority (FTMSA, Murrysville, PA). All three towers were operated at CoC 4-6, using a flow velocity of 0.6 m/s. The temperature of water entering the tower was about 40°C (105°F) and leaving the tower was about 35°C (95°F). The schematic diagram of the pH control system shown in Figure 3.2.1 used dilute sulfuric acid fed by gravity through solenoid valve that opened when the measured pH in the tower sump was higher than the target value set in the pH controller. Similar to the bench-scale recirculating system tests, stainless steel discs were used to provide collecting surfaces for scaling/deposition and were withdrawn at predetermined time intervals to monitor the scaling process. One important difference between the scaling studies with synthetic treated MWW in the laboratory and real treated MWW in the field is the biomass growth on SS discs used in the field tests. Therefore, the SS discs were first placed at 104°C for 3.5 hours to dry and then at 500°C for 3.5 hours to remove volatile organic component of the deposit. The inorganic deposits on selected SS disc specimens were analyzed by SEM/EDS to obtain their elemental composition. In parallel with the solids analysis, important information about the chemistry of the makeup and recirculating water was recorded throughout the field tests.

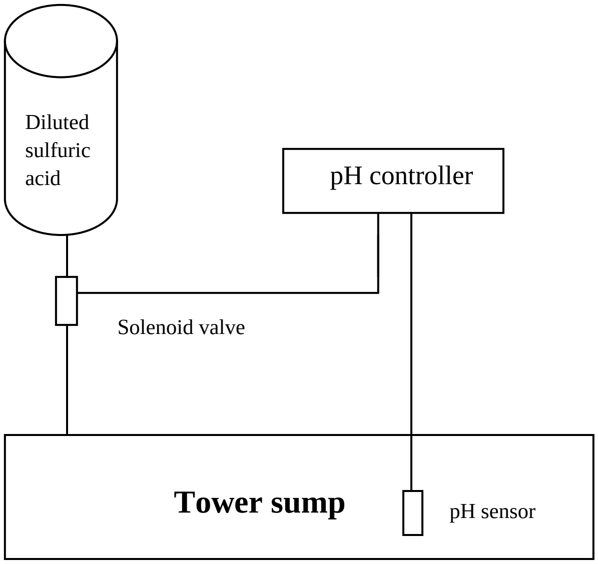


Figure 3.2.1 Schematic diagram of pH control system in pilot-scale cooling tower tests with MWW_ pH as recirculating water

3.2.2 Bench-scale batch tests

Batch tests with synthetic MWW_ pH (CoC 4) at pH 8.4, 8.2, 7.8, 7.4, and 7.0 (MWW_pH) were conducted at 40°C and the residual Ca and phosphate concentrations are plotted as a function of time on Figure 3.2.2. As can be seen in Figure 3.2.2 and, Ca and phosphate concentrations in the synthetic secondary-treated MWW were sensitive to pH in the range of 8.4 to 7.8. Typically, lower pH resulted in the increase in solution concentration. Phosphate precipitation was essentially complete in less than 10 minutes when pH was above 7.8 and lowering pH to 7.4 extended the reaction time to about 1 hour. Soluble Ca and phosphate concentrations at pH 7.0 remained almost unchanged throughout the test, suggesting that maintaining the pH of synthetic MWW_ pH at 7.0 would prevent precipitation of calcium carbonate and calcium phosphate for at least 3 hours.

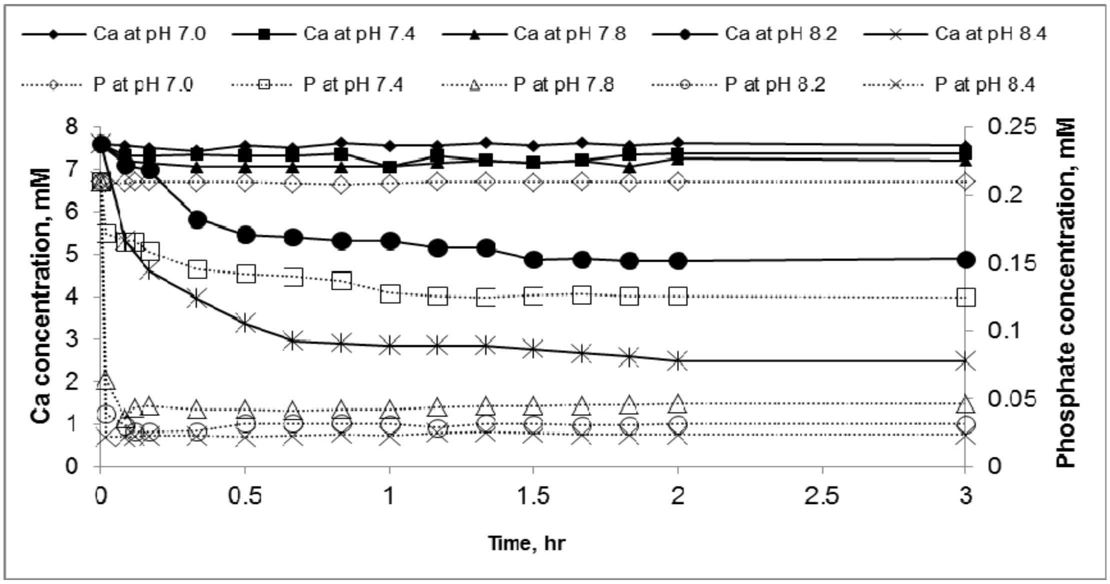


Figure 3.2.2 Residual Ca and phosphate concentrations in batch tests with synthetic secondary-treated MWW (CoC 4) as a function of time at typical cooling water temperature (40°C) when pH was maintained at 8.4, 8.2, 7.8, 7.4, and 7.0.

XRD analysis (Figure 3.2.3) showed that magnesian calcite was the main crystalline components of the precipitates from the synthetic secondary-treated MWW at pH 8.4, 8.2, and 7.8. Magnesian calcite is generally formed by the coprecipitation of $MgCO_3$ and $CaCO_3$ and contains small but variable amounts (less than 5%) of $MgCO_3$ in solid solution (Thorstenson and Plummer, 1977) .Comparison of the XRD patterns at these pH conditions revealed that the peaks of magnesian calcite decreased with pH reduction. Maintaining the pH at 7.4 could totally inhibit the formation of magnesian calcite as evidence by the absence of its characteristic peaks in XRD spectra. Morphology of amorphous calcium phosphate was observed in SEM images (Chesters, 2009) and EDS analysis verified that the precipitates were mainly composed of Ca and P with a small amount of Mg and C incorporated (Figure 3.2.4).

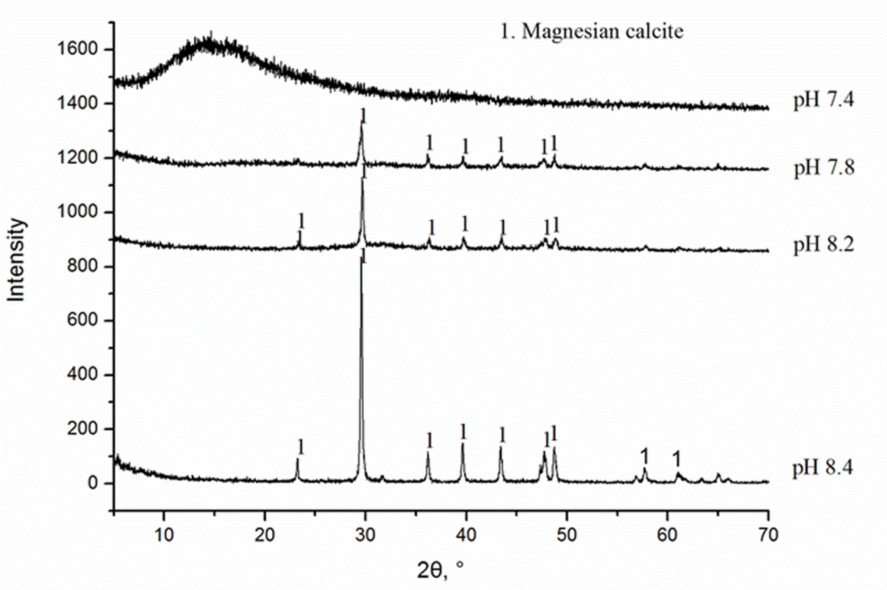


Figure 3.2.3 XRD pattern of the precipitates from batch tests with synthetic secondary-treated MWW (CoC 4, 40°C) when pH was maintained at pH 8.4, 8.2, 7.8, and 7.4

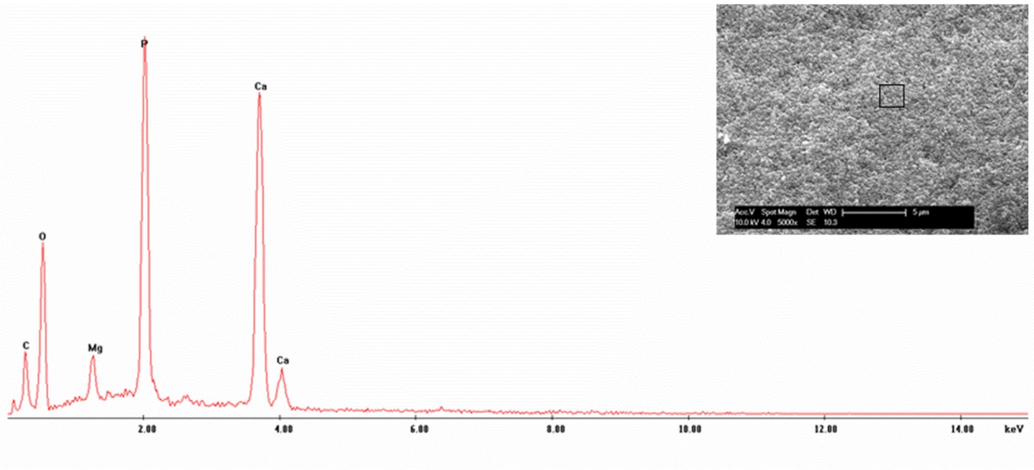
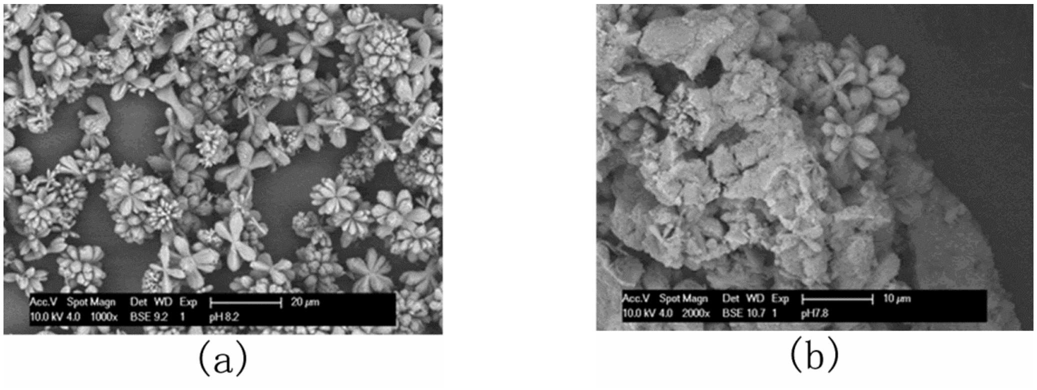


Figure 3.2.4 SEM image and the elemental composition of the precipitates from batch tests with synthetic secondary-treated MWW (CoC 4, 40°C) when pH was maintained at pH 7.4. EDS scan was performed on the area outlined by the square box on the SEM image

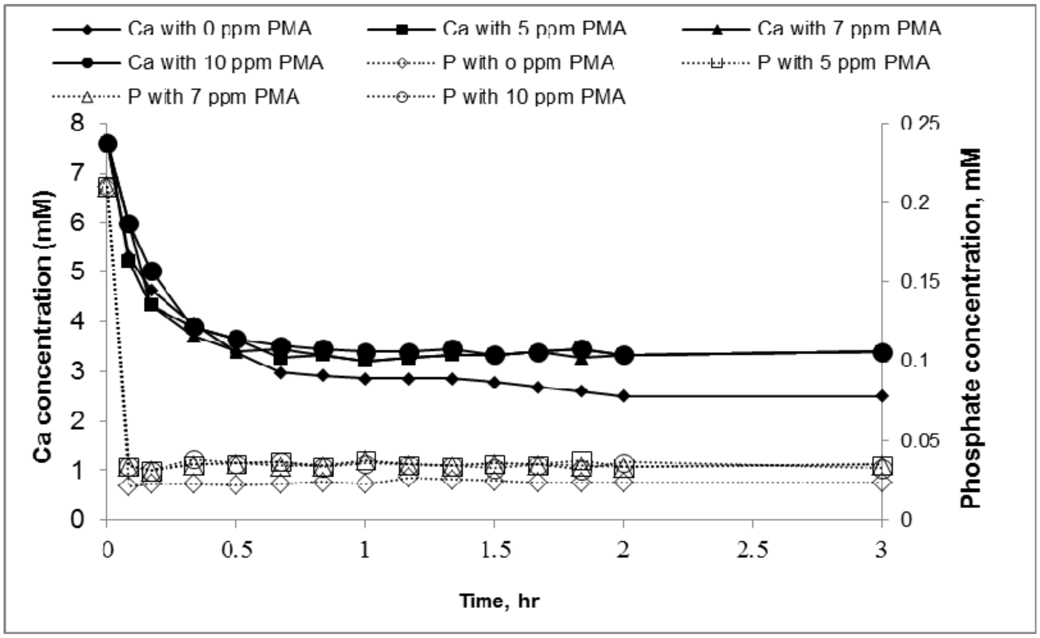
Morphology of precipitates formed in the batch tests is shown in the SEM images of Figure 3.2.5. Only "broccoli-like" minerals were identified in the precipitates formed from the synthetic secondary-treated MWW (CoC 4) at pH 8.2. Normally, three polymorphs of $CaCO_3$ could form in the absence of biological activity: calcite with rhombohedral shape (Wu et al., 2010), aragonite with needle-like shape (Hardikarl and Matijevic, 2001), and spherical vaterite (Hou and Feng, 2006). However, the morphology of the precipitates formed in these tests could not be recognized as either of these polymorphs. The incorporation of Mg ions into the lattice of calcite would poison the side faces of calcite and allow the crystals to grow in the c-axis direction, producing elongated morphology rather than equant crystals with rhombohedral shape (Folk, 1974; Davis et al., 2000). It is thus concluded that broccoli-like minerals in Figure 3.2.5(a) were clusters of single elongated calcite crystals (Swietlik et al., 2011). When pH was lowered to 7.8, two predominant morphologies were observed in the SEM images (Figure 3.2.5 (b)): amorphous minerals and "broccoli-like" minerals.



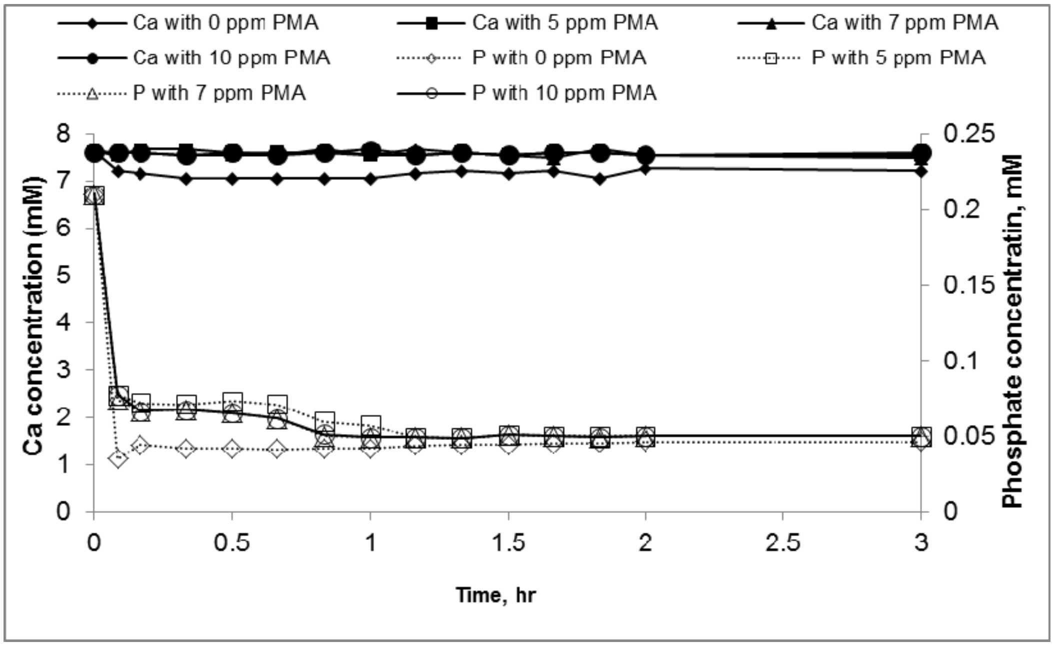
(a) (b)

Figure 3.2.5 SEM images of precipitates from batch tests with synthetic secondary-treated MWW (CoC 4, 40°C) when pH was maintained at different value: (a) pH 8.2; (b) pH 7.8

Batch tests were also conducted to evaluate the combined effects of PMA addition and pH control on mineral precipitation. PMA doses of 5 ppm, 7 ppm, and 10 ppm were added to synthetic MWW_pH (CoC 4) when the pH was maintained at 8.4 or 7.8. Residual Ca and phosphate concentrations were monitored as a function of time as shown in Figure 3.2.6. At pH 8.4, both calcium and phosphate concentrations were slightly elevated by the addition of PMA, revealing that PMA had minimal impact on precipitation propensity of minerals and associated increase in the solubility of the scaling species (Figure 3.2.6(a)). Fairly limited increase in solubility was also observed at pH 7.8 (Figure 3.2.6 (b)). In addition, amorphous calcium phosphate dominated the solids characterization spectra collected at pH 7.8 as evidence by the absence of distinct XRD peaks (data not shown) and SEM analysis in Figure 3.2.7 (Andritsos et al., 2002). Adsorption of the PMA molecules at the active sites on the surface of pre-critical nuclei would prevent the growth of the crystals beyond the pre-critical size and formation of stable magnesian calcite (Meldrum and Hyde, 2001). Although the residual phosphate concentration measurement showed that the precipitation of calcium phosphate was still significant at pH 7.8 (Figure 3.2.6 (b)), the addition of the PMA extended the reaction time from less than 5 min to about 60 min. It should be noted that increasing the PMA dosage at both pH conditions did not result in better precipitation inhibition. Such behavior is likely due to the PMA adsorption saturation on the active sites of the crystals formed under these conditions once 5 ppm PMA was added to the system.



(a)



(b)

Figure 3.2.6 Residual Ca and phosphate concentrations in batch tests with synthetic secondary-treated MWW (CoC 4) as a function of time at typical cooling water temperature (40°C) when 0, 5, 7, and 10 ppm PMA was dosed at different pH: (a) 8.4; (b) 7.8.

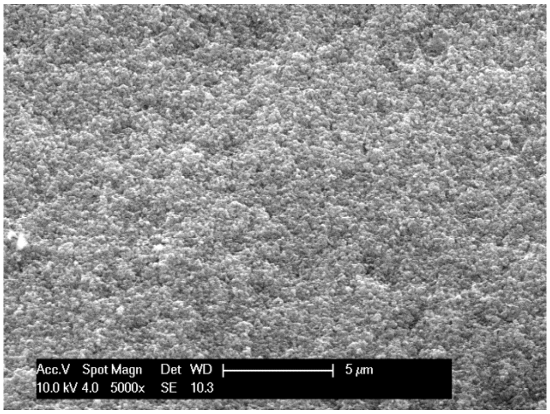


Figure 3.2.7 SEM images of precipitates from batch tests with synthetic secondary-treated MWW (CoC 4, 40°C) when pH was maintained at 7.8 with the addition of 5 ppm PMA

3.2.3 Bench-scale recirculating system tests

A series of experiments in the bench-scale recirculating system was conducted to test the effectiveness of scaling control strategies on mineral deposition from flowing synthetic MWW. The scaling control strategies were designed based on the results from the batch tests. Scaling behavior of the synthetic MWW_ pH under different scaling control strategies is shown in Figure 3.2.8.

Mass gain on the disc specimen exposed to recirculating synthetic MWW_pH at pH 8.4 developed quickly and reached 1.6 mg in 24 hours. Although total mineral deposits accumulated after 48 hours were nearly the same with or without the addition of PMA, the addition of PMA significantly delayed the scaling process. The PMA can function as colloid dispersant through electrostatic and/or steric interactions to keep mineral particles dispersed in aqueous suspension and render them less prone to deposition (Eriksson et al., 2007; Li et al., 2011). Lowering the pH to 7.8 reduced the deposition of mineral scales by about 50%. Under these conditions, the dosage of 5 ppm PMA not only reduced the scaling rate but also decreased the final mass gain on the stainless steel disc from 0.89 mg to 0.43 mg. pH adjustment to 7.0 was the most effective method for scaling control as evidenced by minimal scale accumulation in 72 hours.

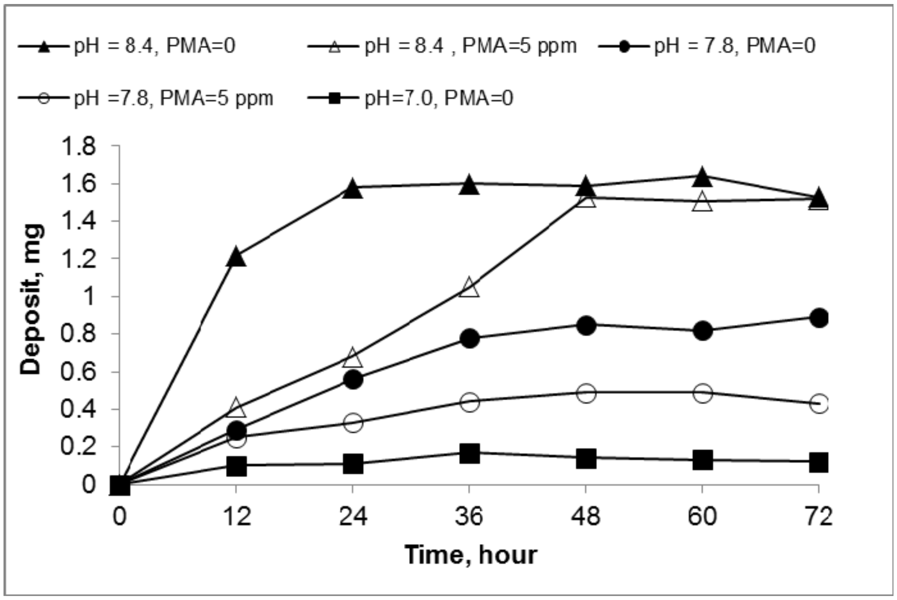


Figure 3.2.8 Scaling behavior of synthetic MWW_pH under different scaling control strategies in bench-scale recirculating system tests

Figure 3.2.9 depicts morphologies of mineral deposits on stainless disc specimens collected from bench-scale recirculating system tests. At pH 8.4 (Figure 3.2.9 (a)), the morphology of magnesian calcite dominated on the disc specimen (Swietlik et al., 2011). However, the minerals did not grow in clusters as in batch tests but formed separately and thus appeared as needle-like structures. Addition of 5 ppm PMA to the system operated at pH 8.4 (Figure 3.2.9(b)) made it difficult to identify well-developed magnesian calcite thereby, revealing the role of PMA as crystal distorter in the deposition inhibition. Adjusting the pH to 7.8 (Figure 3.2.9(c)) decreased the percentage of magnesian calcite and amorphous deposits appeared on SEM images. Only amorphous solids (Figure 3.2.9(d)) were identified on the collector surface when pH was controlled around 7.8 and 5 ppm PMA was added to recirculating water.

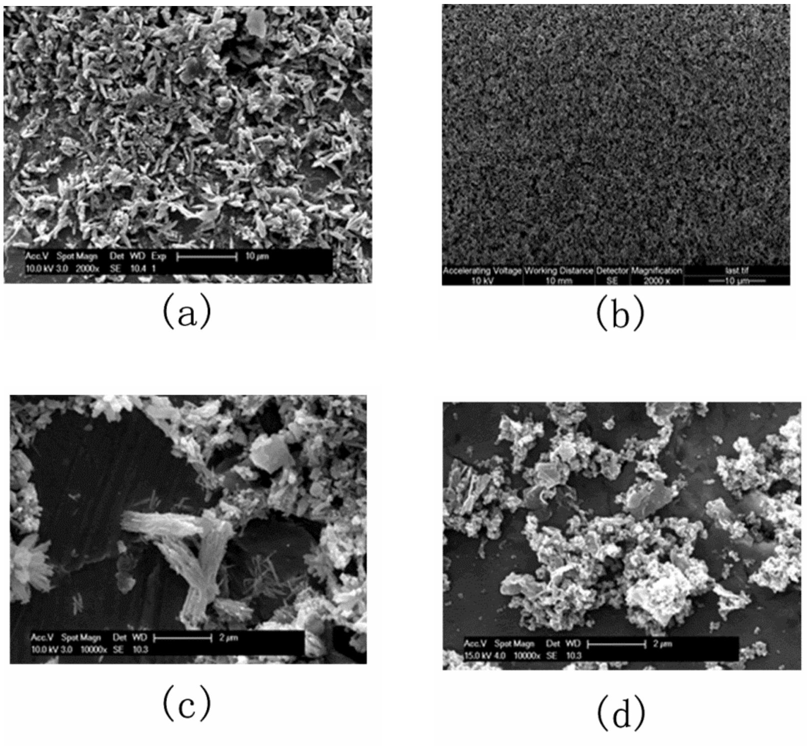


Figure 3.2.9 SEM images of the mineral deposits collected on the stainless disc specimens from bench-scale recirculating system tests under different scaling mitigation strategies: (a) pH control at 8.4; (b) pH control at 8.4 plus 5 ppm PMA; (c) pH control at 7.8; (d) pH control at 7.8 plus 5 ppm PMA

3.2.4 Pilot-scale cooling tower experiments

In the pilot-scale cooling tower tests with MWW_pH as the make-up water source, recirculating water in all three towers was dosed with 5 ppm PMA as scaling inhibitor. In addition, pH in Towers B and C was initially adjusted to pH 7.0 by adding 0.1 M H_2SO_4 because batch and bench-scale recirculating system tests revealed that pH control at 7.0 could totally inhibit the precipitation and deposition of mineral scales from the synthetic MWW at CoC 4. However, serious corrosion of the copper coil employed in the heating section occurred as evidenced by the presence of green copper oxide retained on the 0.45 μm filter paper after filtering the recirculating water in Tower B on day 9 (Figure 3.2.10). Copper concentrations in the recirculating water in Towers B and C on days 2, 4, and 9 were all above 2 mg/L and reached as high as 48 mg/L in Tower B on day 9. It is well known that ammonia is a critical component in secondary-treated MWW that can attack copper and form soluble complexes (Strmčnik et al., 2009). Significant reduction in ammonia concentration in all pilot-scale cooling tower was always observed due to air stripping during the wastewater recirculation (Hsieh et al., 2010; Rebhum and Engel, 1988). However, the efficiency of ammonia stripping was reduced at lower pH and high concentration of ammonium ions thus contributed to significant corrosion rates and considerable Cu concentration in the recirculating water. Consequently, target pH in Towers B and C was elevated to 7.8 and 0.05 M H_2SO_4 was used to control the pH in the recirculating water starting on day 12. Monitoring of Cu concentration during the following test period (0.8-2.0 mg/L in Tower B and 0.8- 1.4 mg/L in Tower C) revealed that the corrosion of the copper coil was reduced with the new pH control procedure.

Figure 3.2.10 “Green deposits” retained on 0.45um filter paper after filtering the recirculating water in Tower B on day 9 in pilot scale tests with MWW_pH at pH 7.0

Figure 3.2.11 depicts the time course of inorganic scale deposition in the three cooling systems during the tests with MWW_ pH. The total mass of the inorganic deposits on day 58 in Tower A was as high as 11.57 mg, while the mass gains in Towers B and C were only 0.61 and 1.80 mg, respectively. It is clear that just adding 5 ppm PMA failed to mitigate scaling in Tower A. Significant reduction in the inorganic deposition was observed with pH adjustment in Towers B and C. Furthermore, use of monochloramine as biocide in Tower B resulted in even lower scale accumulation as compared to Tower C where free chlorine was used for biofouling control. This finding is in agreement with the previous related study which revealed enhanced PMA oxidation with free chlorine when compared to monochloramine (Li et al., 2011).

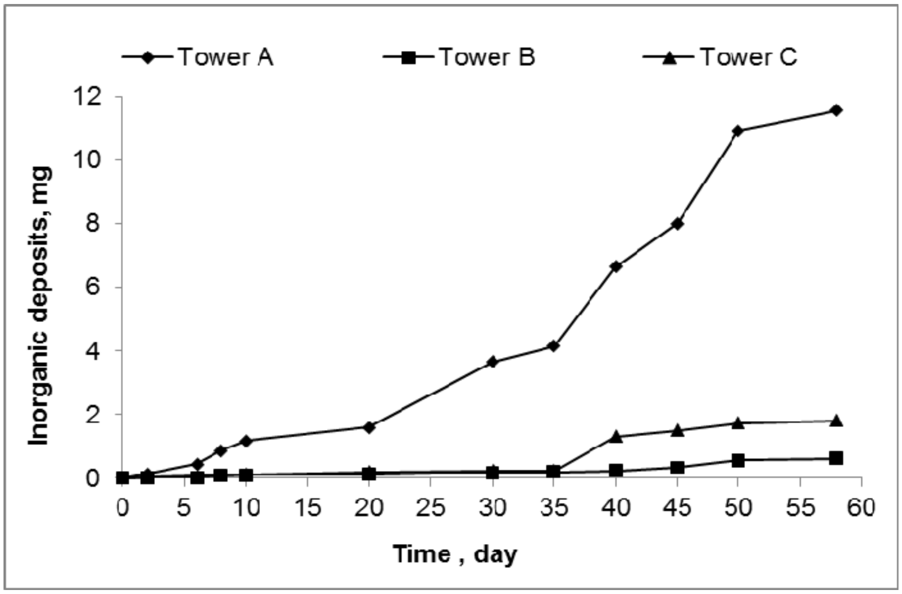


Figure 3.2.11 Inorganic deposit mass measurements in the pilot-scale cooling tower tests with MWW_pH

Water samples from the recirculating loop of each cooling tower operated at steady state were analyzed for key constituents. Previous study with MWW revealed that calcium carbonate was the dominant mineral scale on the sampling discs with a small amount of magnesium and calcium phosphate (Li et al., 2011). Langelier Saturation Index (LSI), Ryznar Stability Index (RSI), and Puckorius Scaling Index (PSI) which are widely used to estimate the scaling potential of calcium carbonate in cooling towers, was calculated based on the quality of the recirculating water (EPRI, 2003; Sheikholeslami, 2004) for all three towers operated with MWW_pH. LSI, RSI, and PSI calculations in Table 3.2.2 revealed that the recirculating water in Tower A had significant calcium carbonate precipitation potential while there was no scaling propensity in Tower B on most of the sampling days. The scaling tendency in Tower C was a bit more erratic and the LSI value was close to zero with positive values on several sampling days.

Comparison of orthophosphate concentrations in the make-up water and recirculating water for all three towers operated with MWW_pH is shown in Figure 3.2.12. In Tower A, more than 80% of orthophosphate precipitated from the solution. On the other hand, orthophosphate concentration in Towers B and C was always 2.0-3.5 times that of the make-up water. Water quality analysis suggests that calcium phosphate is the primary mineral scale when pH of the recirculating water is adjusted at 7.8 and 5 ppm PMA is added. SEM/EDS results shown in Figure 3.2.13 confirmed this conclusion.

Table 3.2.2 LSI, RSI, and PSI of the recirculating water in Towers A, B and C in pilot-scale tests with MWW_pH

Day	LSI[a]			RSI[b]			PSI[c]		
	CTA	CTB	CTC	CTA	CTB	CTC	CTA	CTB	CTC
2	0.80	-0.64	-2.55	6.72	8.48	10.87	7.29	8.85	10.37
9	1.17	-1.48	0.25	5.96	9.80	7.31	6.14	10.29	7.68
16	1.46	-0.04	-0.36	5.72	7.77	7.80	6.28	8.19	7.09
23	0.82	-0.64	0.29	6.61	8.34	7.37	7.07	8.14	7.93
30	1.61	-0.12	0.59	5.46	7.85	6.91	5.93	8.21	7.49
37	0.41	-0.36	0.05	7.04	8.08	7.59	7.30	8.27	7.91
44	0.95	-0.58	-0.70	6.21	8.38	8.53	6.22	8.36	8.58
58	1.05	0.22	0.41	6.05	7.10	6.86	6.13	6.74	6.55

[a] If LSI is negative, there is no potential to form scale and the water will dissolve CaCO$_3$; if LSI is positive, scale can form and CaCO$_3$ precipitation may occur; if LSI is close to zero, the water is neutral with respect to scale formation;

[b] If RSI < 6, the scaling tendency increases as the index decreases; if RSI > 7, calcium carbonate scale will not form; if RSI > 8, mild steel corrosion may be of concern;

[c] If PSI < 6, the scaling tendency increases as the index decreases; if PSI > 6, calcium carbonate scale will not form.

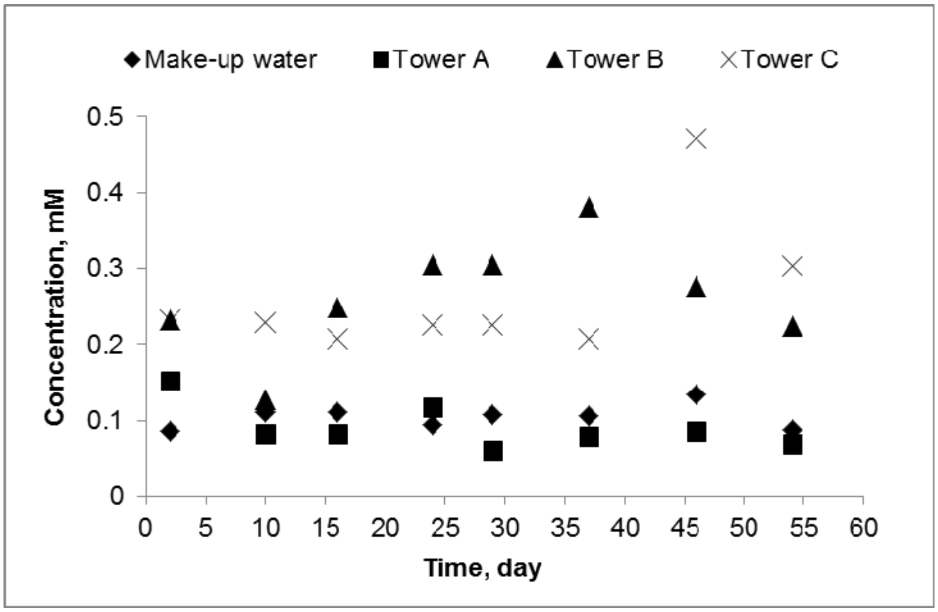


Figure 3.2.12 Orthophosphate concentration in the make-up water and recirculating water in the pilot-scale cooling tower tests with MWW_pH

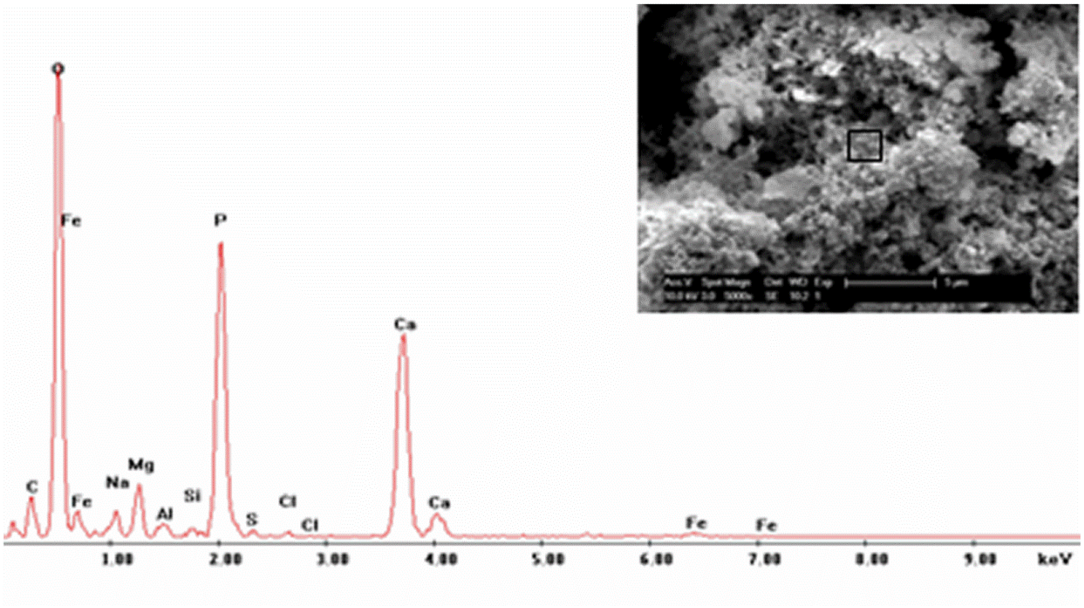


Figure 3.2.13 SEM images and the elemental composition of the solid deposits collected on stainless steel discs immersed in pilot-scale cooling towers operated at CoC 4-6: Day-50 sample from Tower B using MWW_ pH. EDS scan was performed on the area outlined by the square box on the SEM image

Another parameter indicating the development of scales in the recirculating cooling tower system is the flow rate of the recirculating water since the accumulation of mineral scales would increase the head loss and even plug the conveying pipes and copper heating coils. The flow rates in all the three towers were measured daily as shown in Figure 3.2.14. As can be seen in this Figure, the water flow rate in Tower A with no pH adjustment dropped to 2.2 GPM on day 20. An acid wash (5 gallon 10% HCl solution for 20 min) was used to clean the copper coil of the tower A and recover the flow rate to 2.9 GPM on day 22. However, the flow rate decreased again to 2.3 GPM on day 33. The copper coil in Tower A was washed again by 5 gallons of 10% HCl for 20 min on day 41 and the flow rate was recovered to 2.7 GPM. On the last day of tower operation, the flow rate in Tower A was still reduced to 2.4 GPM. The above phenomenon indicated that the recirculating water in Tower A had significant scaling potential and 5 ppm PMA was not effective in the scaling control. The flow rate in Tower C was generally above 2.4 GPM during the course of the test and was around 2.60 GPM during the last 8 days of tower operation. The flow rate in Tower B, was above 2.70 GPM during the entire test. It must be noted that no acid wash was required in Towers B and C during the whole run. According to the flow rate recorded daily in all the towers, Tower A with no pH control had the most significant scaling problem while the least serious fouling has occurred in Tower C. These conclusions are identical to those from the mass gain data and the calculations of scaling indices.

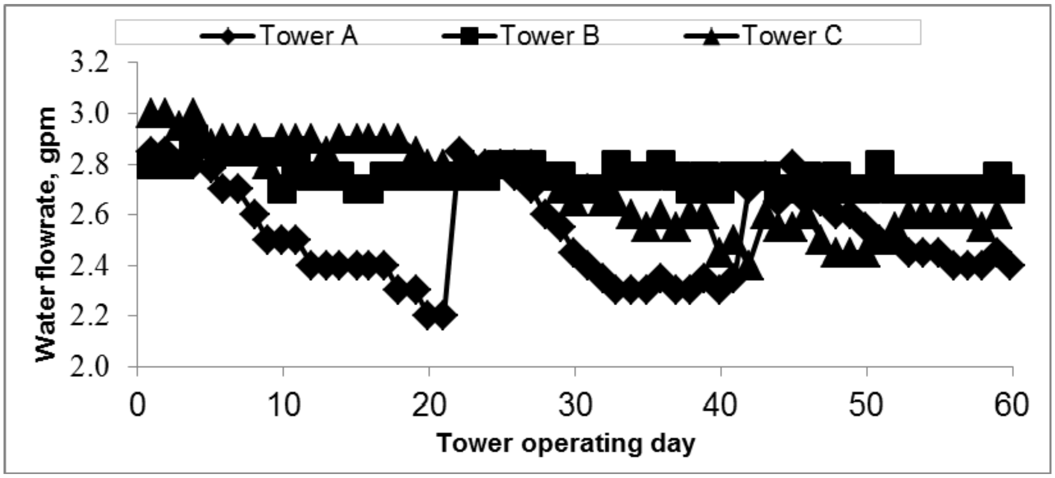


Figure 3.2.14 Water flow rates measured in the pilot-scale cooling tower tests with MWW_ pH as recirculating water. Target flow rate is 3 GPM for the system.

3.2.5 Mineral scaling on heated surface

Synthetic MWW simulating the secondary-treated effluent from Franklin Township Municipal Sanitary Authority (FTMSA, Murrysville, PA) was used in this study. The flow velocity in the test section was 4.74 ft/s, resulting in Reynolds number as high as 24,039. Figure 3.2.15 shows rapid development of fouling resistance with synthetic MWW at CoC 4 in the bench-scale experimental system. The fouling curve on Figure 3.2.15, can be divided into 3 stages: rapid growth period (0-2 hr), slow growth period (3-7 hr), and equilibrium period (7-16 hr). Figure 3.2.16 and 3.2.17 indicate that the rapid growth period is closely related with the reduction of total alkalinity (Figure 3.2.16) and Ca concentration (Figure 3.2.17). Although the total alkalinity and Ca concentration were almost constant during the slow growth period, the suspended solids concentration (Figure 3.2.19) decreased continuously, suggesting the attachment of suspended particles formed in the bulk solution was responsible for the development of fouling resistance on the heater during the second stage. During the equilibrium period, the fouling resistance fluctuated in a “sawtooth” fashion. XRD analysis (Figure 3.2.20) of the deposits showed that magnesian calcite (calcium carbonate with a small amount of magnesium incorporated in the solid phase) was the primary crystalline components. Chemical compositions further identified by EDX (Figure 3.2.21) indicated that phosphates also accounted for 6.1 wt.% of the deposits, corresponding to the sharp decrease of the soluble phosphate in the recirculating water as shown in Figure 3.2.18. It could be concluded that magnesian calcite and calcium phosphate were the main components of the mineral scale deposited on the heater during the fouling study. The reason for failure to identify the existence of calcium phosphate with XRD methods might be due to the fact that the amount of calcium phosphate was below the detection limit or the amorphous characteristics of the mineral.

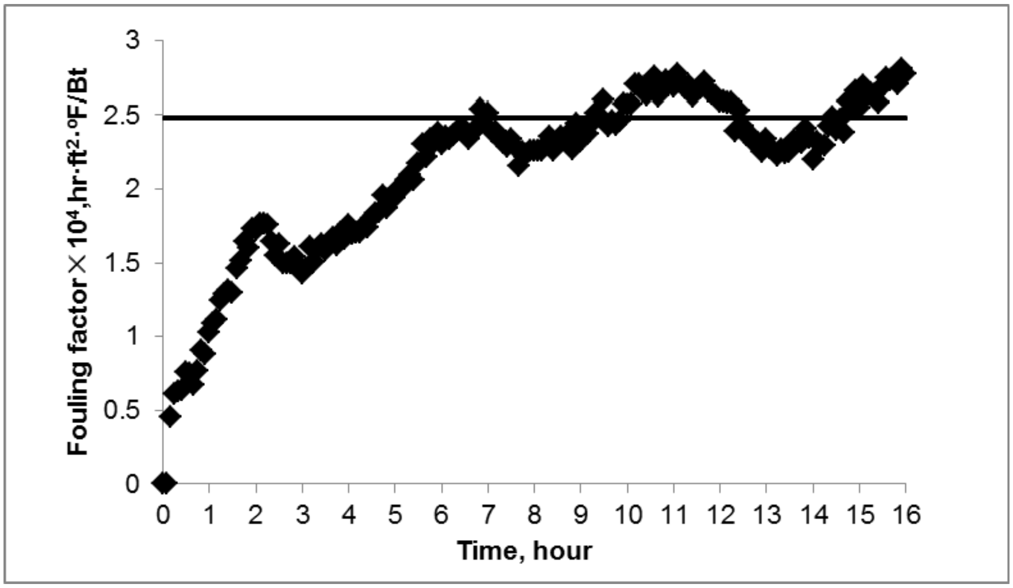


Figure 3.2.15 Fouling curve of the MWW at CoC 4

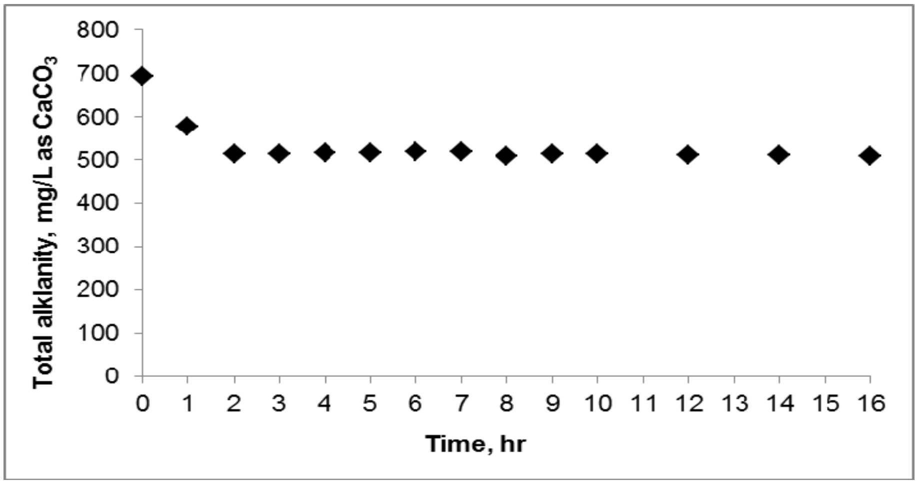


Figure 3.2.16 Total alkalinity of the recirculating water during the fouling study with MWW

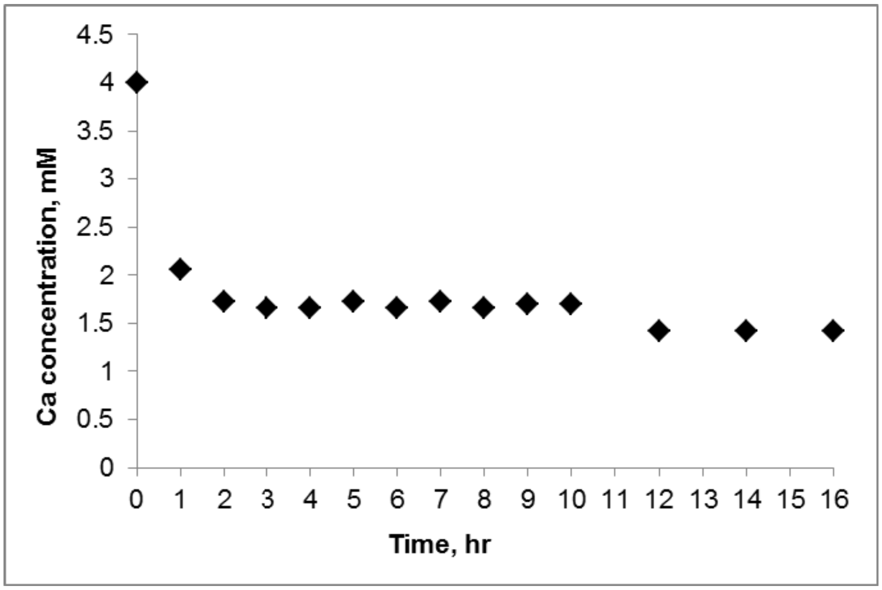


Figure 3.2.17 Calcium concentration in the recirculating water during the fouling study with MWW

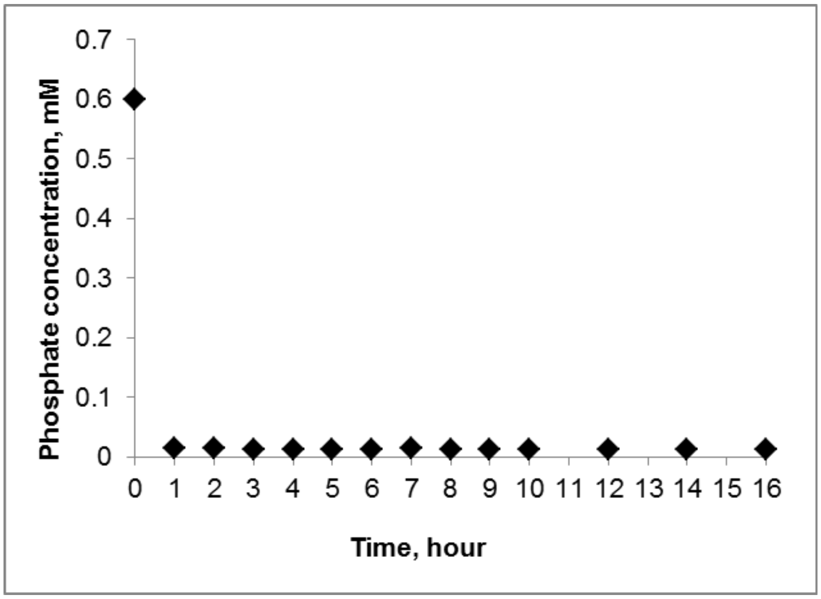


Figure 3.2.18 Phosphate concentration in the recirculating water during the fouling study with MWW

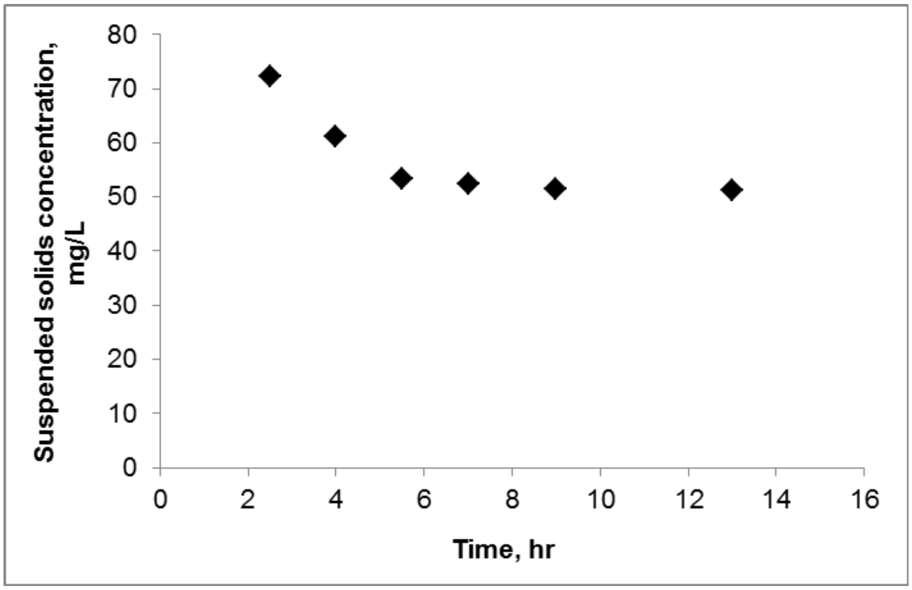


Figure 3.2.19 Suspended solids concentration in the recirculating water during the fouling study with MWW

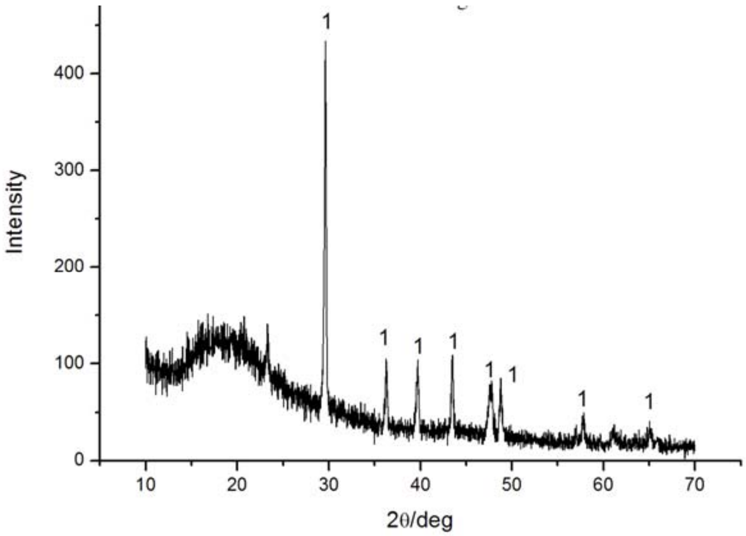


Figure 3.2.20 XRD pattern of the deposits formed on the heater during the fouling test with MWW. “1” denotes magnesian calcite.

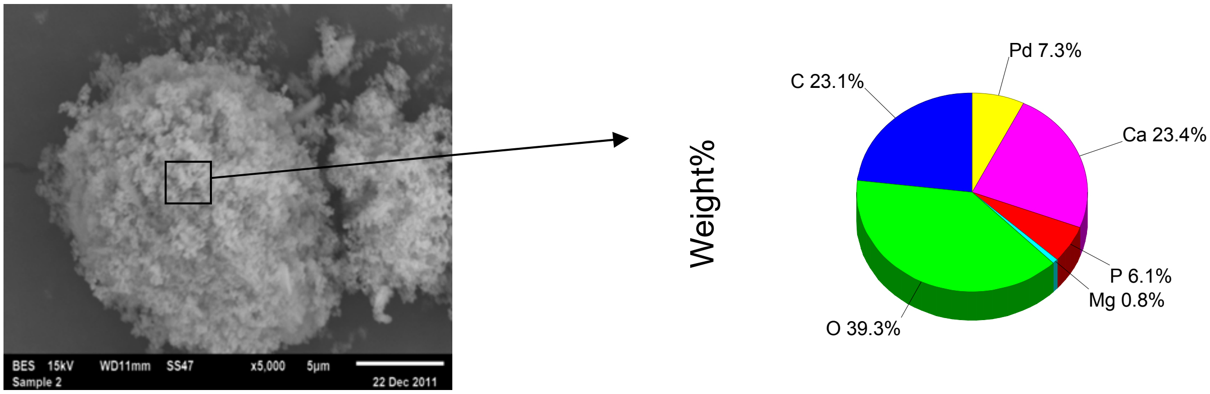


Figure 3.2.21 Morphology and elemental compositions of the deposits formed on the heater during the fouling test with MWW

Overall, the mechanism of the fouling development in the rapid growth period was crystalline fouling and particulate fouling while only particulate fouling contributed to the fouling in the slow growth stage since the concentration of scale-forming ions already reached equilibrium state during the second phase. The difference in fouling mechanisms is one possible explanation for the difference in fouling rate between the rapid growth stage and the slow growth stage. Finally, the attachment and detachment of suspended particles in the bulk came to equilibrium because of the equilibrium between the attachment force and shear force caused by the flowing water in the test section.

For MWW_pH, the addition of acid is to reduce the scaling propensity by consuming the alkalinity and controlling the pH at comparatively low level. According to the pilot-scale cooling tower tests with real MWW, the total alkalinity of MWW_pH was generally 74.25-104.36 mg/L as $CaCO_3$, which was in the same level with MWW_NF at CoC 4. Thus, the fouling tests with MWW_pH at pH 7.8 were conducted on the MWW_NF at CoC 4 and bulk pH was controlled around 7.8 all through the tests. The fouling curve was shown in Figure 3.2.22. As shown in Figure 3.2.22, negligible fouling was identified. That is to say, insignificant mineral scaling was formed on the heat transfer surface with MWW_pH at pH 7.8 under relevant test conditions. The discussion on the test results will be presented in Section 4.2.4.

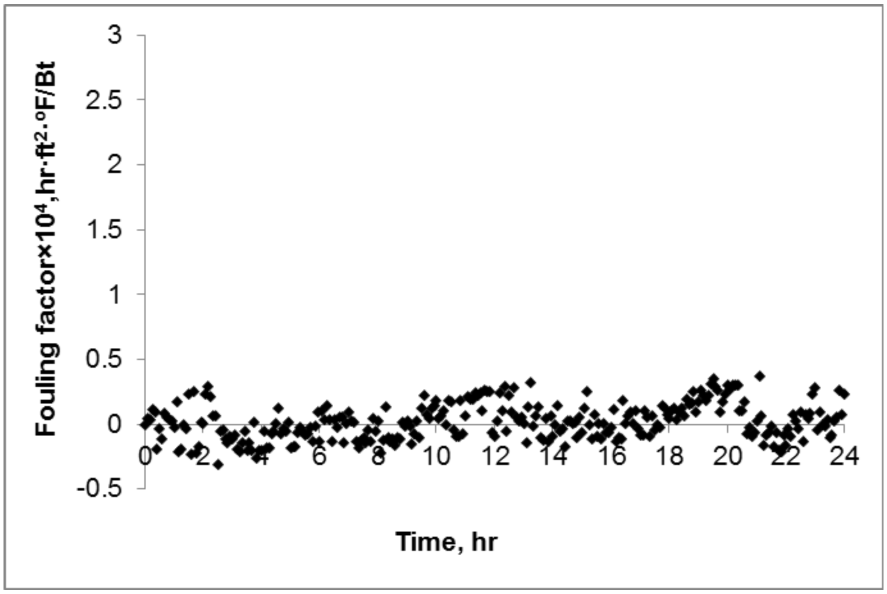


Figure 3.2.22 Fouling curve of the MWW_NF at CoC 4 with bulk pH controlled at 7.8 (equivalent to MWW_pH at pH 7.8)

3.3 Biofouling Control for MWW_pH

3.3.1 Testing protocols

Study of MWW_pH for cooling water makeup was only conducted in pilot-scale tests because of almost identical water quality as MWW. The two objectives of pilot-scale studies for biofouling control were to: 1) evaluate the impact of pH adjustment on chloramination in cooling systems and 2) compare the effectiveness of chloramination and chlorination to control biological growth in a cooling tower at pH 7.7. First objective was achieved by comparing results of cooling tower test CTA2 and cooling tower test CTB2. Second objective was achieved by comparing results of cooling tower test CTB2 and cooling tower test CTC2. Target monochloramine residual concentration for MWW_pH test CTA2 and CTB2 was 3 to 4 ppm as Cl_2; target total chlorine residual for MWW_pH test with NaOCl in CTC2 was 3 to 4 ppm as Cl_2. After the regular testing periods, systems with severe biofouling problem were treated with glutaraldehyde (GA) by dosing 100 and 1000 mg/L of GA into the system. Efficacy of GA in treating sessile biological growth was determined based on planktonic HPCs derived from the recirculating waters.

The pilot-scale cooling towers were operated in the field during a 60-day testing period. Throughout the pilot-scale tests, both planktonic and sessile heterotrophic bacteria were monitored along with water quality analysis. Sessile bacteria were enumerated by immersing circular stainless steel coupons in circulating cooling water. Coupons were taken from the sampling rack on Day 7, 10, 28, and 56.

Biocides used in both laboratory and field tests were prepared on-site prior to each experiment. Free chlorine stock solution was prepared by diluting a 5% commercial sodium hypochlorite (NaOCl) solution (Ricca Chemical). Monochloramine was pre-formed by mixing 4 to 1 ratio of sodium hypochlorite and ammonia at pH 9. The biocide stock solution was 1000 ppm as Cl_2 in both cases and biocide was continuously added to the recirculating system though a solenoid-driven diaphragm liquid metering pump. Residuals and HPCs were monitored following the analytical methods described in Chapter 2.5.

3.3.2 Pilot-scale studies

3.3.2.1 Effectiveness of monochloramine in MWW at pH 7.7

Pre-formed monochloramine addition and pH control at 7.7 were both applied in CTB2 and the results are shown in Figure 3.3.1. In CTB2, monochloramine residual was maintained at 2.80 ± 3.48 ppm as Cl_2 and total chlorine residual was maintained at 4.24 ± 5.02 ppm for 56 days. It was observed that monochloramine residual averaged 59% of total chlorine residual. Extremely high biocide residual was observed on Days 5, 29 and 31 because of the failure to add appropriate amounts of makeup water. Addtional mechanical failure of liquid metering pump was also observed in CTB2 and caused a significant variation of monochloramine residual throughout the experiment. During the first 10 days, the pH control was not stable and the increase in copper ion concentration in the recirculating water was observed due to excessive copper corrosion. Copper toxicity led to the low HPC for both planktonic and sessile bacteria during this period.

It was observed that monochloramine residual decreased to zero on Day 34. Subsequently, planktonic heterotrophic bacteria grew out of control. Between Day 43 and Day 46, the same problem of biocide stock solution was observed in CTB2 and caused the planktonic HPC to reach 10^7 CFU/mL. For the rest of the experiment, monochloramine residual was not properly maintained and both planktonic HPC and sessile HPC were beyond the control criteria on last day (Day 56).

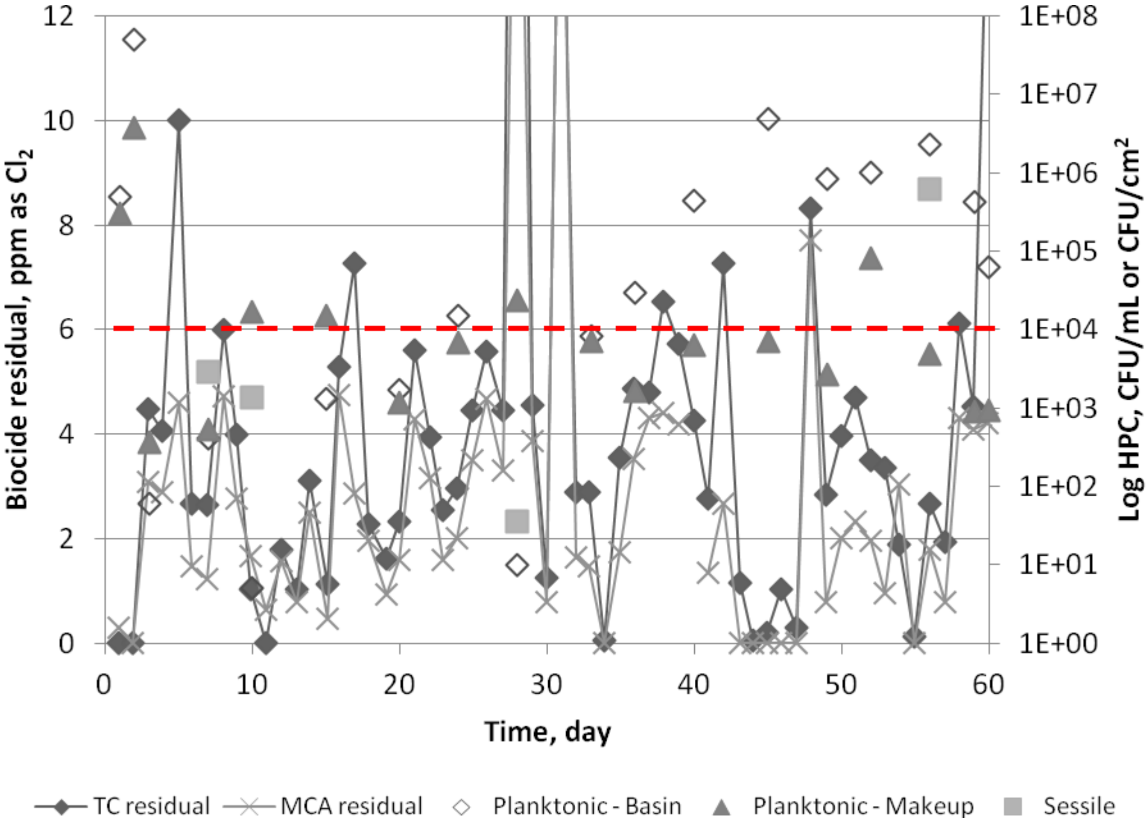


Figure 3.3.1 Monochloramine residual, total chlorine residual, planktonic HPC in recirculating water, planktonic HPC in makeup water, and sessile HPC in CTB2. Horizontal dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2.

CTB2 was operated for additional 7 days to examine the effectiveness of shock dose of GA and the results of an extended experiment with GA addition are shown in Figure 3.3.2. A shock dose of 100 ppm GA was added to the recirculating system on Days 59 and 63. With the addition of GA, the planktonic HPC decreased by 1 log and then gradually grew back to 10^6 CFU/mL within 3 days. However, result of sessile HPC collected inside the recirculating pipe on Day 66 showed a decrease in biofilm formation. It can be assumed that additional shock doses of GA may be required to suppress the planktonic heterotrophic bacteria growth in recirculating water after biofilm formation is placed under control.

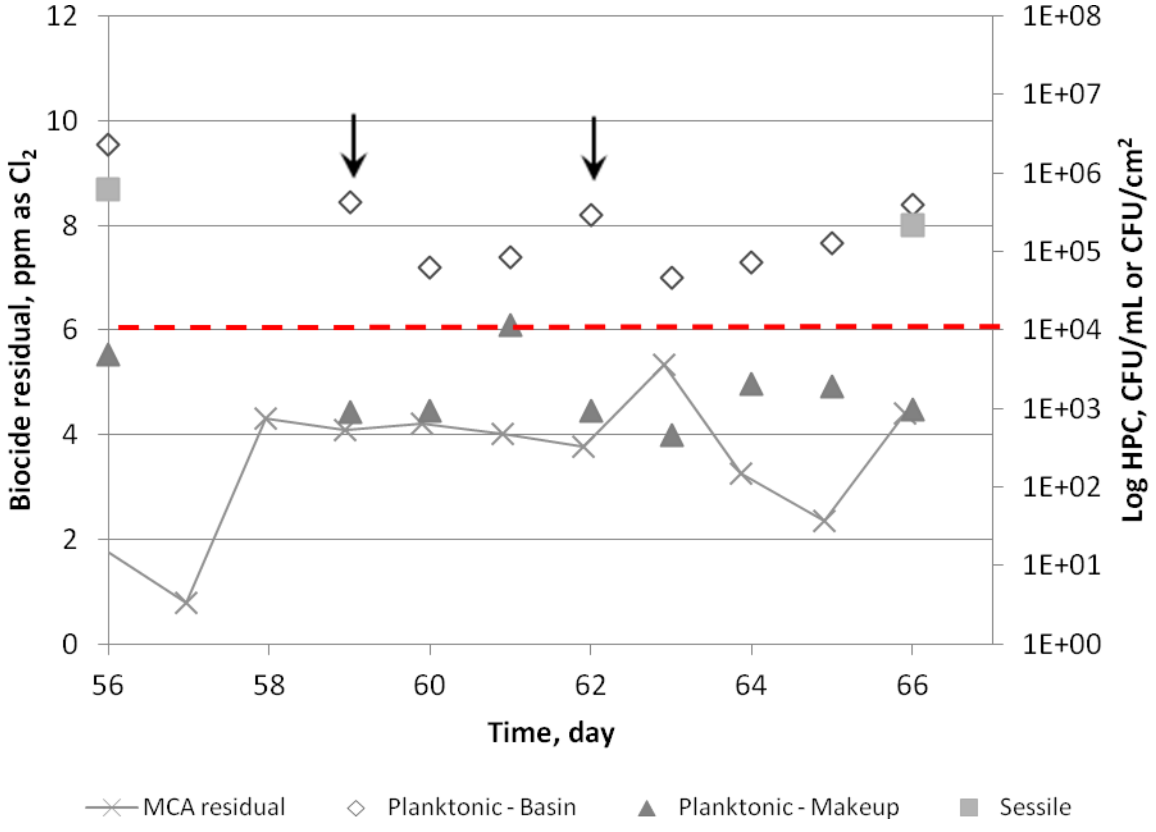


Figure 3.3.2 Monochloramine residual, total chlorine residual, planktonic HPC in recirculating water, planktonic HPC in makeup water, and sessile HPC in Monochloramine residual, total chlorine residual, planktonic HPC in recirculating water, planktonic HPC in makeup water, and sessile HPC in CTB2 during the extended experiment. Horizontal dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2. Arrows indicate the dates when 100 ppm of glutaraldehyde was added to the recirculating system.

3.3.3.2 Effectiveness of sodium hypochlorite in MWW at pH 7.7

Total chlorine residual, planktonic HPC in recirculating water and makeup water, and sessile HPC in CTC2 test are shown in Figure 3.3.3. Monochloramine residual averaged 0.81 ± 0.75 ppm as Cl_2 and total chlorine residual averaged 4.13 ± 1.95 ppm during the 56-day test. As mentioned in the previous section, the pH control was not very reliable and caused an increase in copper ion concentration in the recirculating water during the first 10 days. High copper toxicity to microorganisms resulted in low HPC for both planktonic and sessile bacteria in CTC2 during this period. The copper ion concentration decreased after Day 10 and planktonic heterotrophic bacteria growth entered exponential growth period. In the following 20 days, planktonic HPC increased to $10^{5.4}$. Although the total chlorine residual was mostly maintained above 4 ppm, monochloramine residual did not reach 2 ppm as Cl_2 during this period. Thereafter, neither planktonic HPC nor sessile HPC were controlled well in this system. The decrease in water flowrate from 3.0 to 2.45 GPM in the recirculating system also reflected the severe biofouling problems in CTC2.

Glutaraldehyde was used to address excessive biological growth in CTC2. A shock dose of 1,000 ppm of GA was added to the recirculating water on Day 52. Planktonic HPC decreased to 10^3 CFU/mL in 24 hours; however, the planktonic HPC recovered in 4 days because there was no detectable monochloramine residual in the recirculating water.

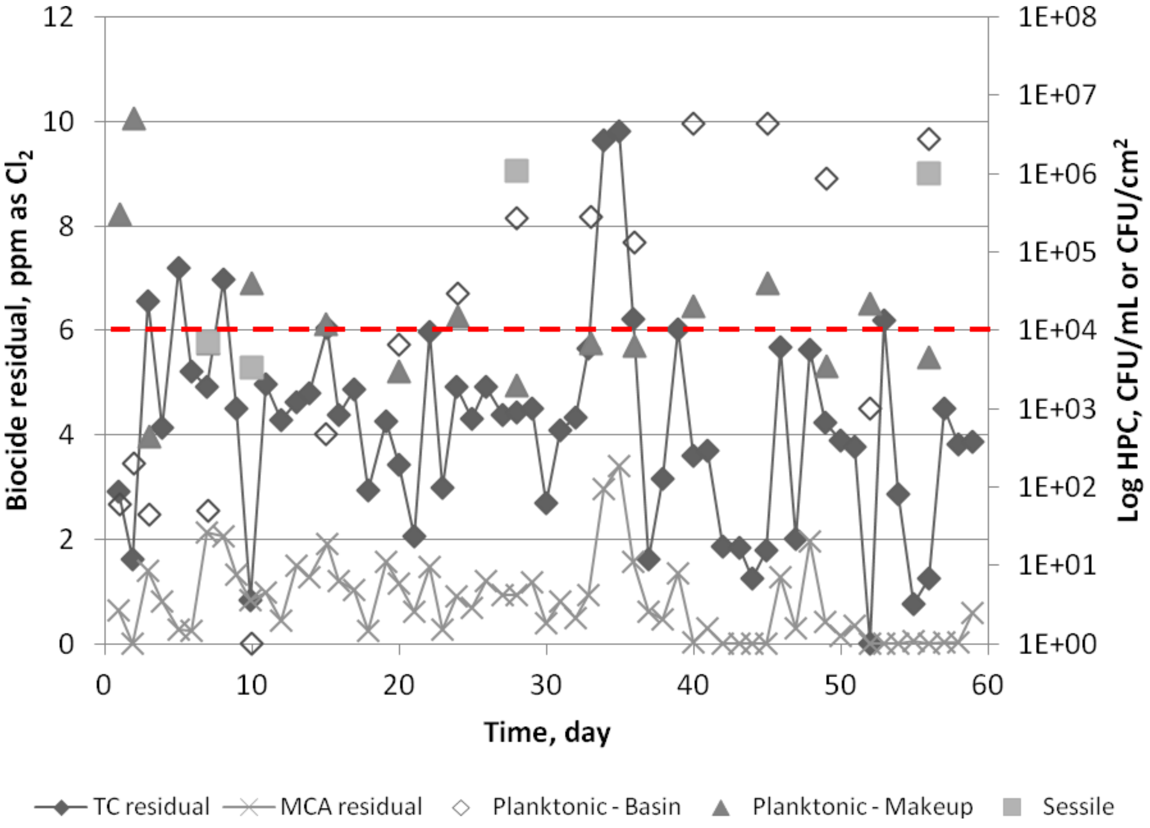


Figure 3.3.3 Monochloramine residual, total chlorine residual, planktonic HPC in recirculating water, planktonic HPC in makeup water, and sessile HPC in CTC2. Horizontal dashed line indicates the biofouling control criteria, 10^4 CFU/ml or CFU/cm^2.

3.3.2.3 Biocide usage

Total biocides usage in CTB2 and CTC2 is shown in Table 3.3.1. A normalized biocide consumption rate was calculated based on the amount of biocide added to the system, the biocide concentration in stock solution, and daily makeup water rate. The daily makeup water rates for CTB2 and CTC2 were 49.88 and 51.50 gal/day, respectively. Both tests had similar normalized biocide consumption rate about $4\text{x}10^{-5}$ lb/gal·day. However, it is clear that adding 3.6 gallons of 1000 ppm pre-formed monochloramine solution to CTB2 or 3.77 gallons of 100 ppm sodium hypochlorite to CTC2 could not control biofouling in pilot scale cooling systems using secondary effluent with pH adjustment as cooling tower makeup for a period of 2 months.

Table 3.3.1 Biocide residual, dosing rate, and normalized biocide consumption in MWW_pH tests

Unit	Total Chlorine residual (mg/L)	MCA residual (mg/L)	Biocide dosing (gal/day)	Normalized biocide consumption (lb/gal·day)[a]
CTB2	4.25±5.02	2.80±3.48	3.61±1.68	$4.2\text{x}10^{-5}$
CTC2	4.13±1.95	0.81±0.75	3.77±1.20	$4.3\text{x}10^{-5}$

Note: a. Normalized biocide usage = $\frac{V_{Bio} \times C_{stock}}{R_{MK}}$,

where V_{Bio} = Volume of daily used biocide, gallon
C_{stock} = Monochloramine stock solution concentration, mg/L
R_{MK} = Daily makeup water rate, gallon/day

3.4 Summary and Conclusions

The bench-scale and pilot-scale studies using pH controlled secondary treated municipal wastewater (MWW_pH) indicated an increase in the corrosiveness of the cooling water after acidification. The abatement of the scaling problem, after pH control, resulted in the exposure of metal alloys to the flowing cooling water. At low pH the secondary treated water containing some ammonia was aggressive towards mild steel, copper, and cupronickel alloys. Ammonia present in the secondary treated municipal wastewater was identified as a significant corrosion-influencing element in the bench-scale experiments. Lower pH resulted in higher concentration of hydrogen ion (H^+) in the cooling water. The increased hydrogen ion reacted with the electrons at the cathodic site and enhanced metal corrosion. Addition of 5 ppm TTA successfully reduced the corrosion rate of copper and cupronickel alloys within excellent levels according to the industrial corrosion control criteria. Mild steel corrosion control was also significantly reduced after TTA addition, and it was lowered just within acceptable limits according to industrial corrosion control criteria.

Based on the results from batch tests, magnesium calcite and amorphous calcium phosphate are the mineral scales formed when pH of the synthetic secondary-treated MWW is controlled at 8.4 and 7.8. Less pH resulted in less precipitation. Scales formed when pH of the synthetic secondary-treated MWW is controlled at 8.4 and 7.8. Bench-scale recirculating tests showed that the addition of 5 ppm PMA delayed the scale deposition process but had minimal impact on the final total mineral deposits at pH 8.4, while the combination of pH control at 7.8 and the addition of 5 ppm PMA not only reduced the scaling rate but also decreased the final mass gain on the sampling specimens. Addition of PMA at pH 7.8 inhibited the formation of magnesian calcite and slowed the kinetics of calcium phosphate precipitation. The pilot-scale cooling tower tests were consistent with conclusion from bench-scale studies, i.e., pH adjustment to 7.8 was needed besides the addition of 5 ppm PMA to reduce the scaling rate significantly. With this scaling mitigation method, the formation of calcium carbonate was inhibited and calcium phosphate was the primary form of mineral scale on both unheated and heated surfaces.

Results obtained in this study are in good agreement with previous findings observed in short term pilot-scale tests (Vidic et al., 2009). However, biofouling control was more difficult and unpredictable when using secondary effluent with or without pH adjustment as cooling tower makeup for an extended testing period (i.e., up to 60 days). Although pH adjustment helped to mitigate the scaling problem, significant input of organic matter into the system still resulted in high variation in biocide residual and low active monochloramine concentration. Pilot-scale tests demonstrated that the biofilm formation can get out of control once monochloramine residual decreases below 3~4 ppm as Cl_2.

Application of free chlorine at a doe bellow that required for breakpoint chlorination at pH 7.7 resulted in total chlorine residual above 4 mg/L but was unable to generate sufficient monochloramine residual to control biological growth. It can be concluded that maintaining 3~4 ppm of total chlorine residual is much less effective for biofouling control than maintaining 3~4 ppm of pre-formed monochloramine residual when MWW is used as makeup water. The main reason for such behavior is that monochloramine was not the dominant disinfectant among the combined chloramine species. In addition, appropriate biocide dosing rate is one of the key requirements to achieve reliable biofouling control in cooling systems using acidified secondary treated municipal wastewater.

Comparing biocide dosing rate and biocide residual in CTA2 and CTB2, it can be concluded that pH control at 7.7 lead to a slight increase in active monochloramine residual in cooling water but had limited impact on biological growth control. Additional tests with glutaraldehyde showed that a single shock dose of 1000 ppm can reduce planktonic bacteria in the recirculating water for about 24 hours. However, planktonic bacterial population will recover in the absence of adequate primary biocide residual. Sessile biological growth was not affected by a one-time shock dose of glutaraldehyde. Several consecutive 100-ppm doses of glutaraldehyde in CTA2 and CTB2 can effectively reduce both planktonic and sessile bacterial growth.

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4.0 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: Nitrification and Filtration (MWW_NF)

Increasing scarcity of freshwater supply for use in power plant recirculating cooling systems has led to the search for alternative impaired water sources. Secondary treated municipal wastewater (MWW) is a widely available alternative cooling water source in terms of quantity and geographic proximity to most of the existing and future power plants (Chien et al., 2008). MWW contains elevated concentrations of organic matter, hardness, orthophosphate, ammonia, and total dissolved solids relative to freshwater (Williams, 1982; Weinberger, 1966). The main challenge when secondary treated municipal wastewater is reused in recirculating cooling water systems is more complicated control of corrosion, scaling, and biological fouling due to the lower quality of the water. Thus use of MWW in power plant cooling systems requires additional treatment and/or inhibitor dosing attention to mitigate corrosion, scaling and biological fouling problems that can rise due to lower water quality (EPRI, 2003).

Previous study showed that use of MWW resulted in low corrosion of mild steel, copper and cupronickel alloys (Hsieh et al., 2010). The low corrosiveness of MWW was due to its high scaling potential, which caused formation of a protective scaling layer on the metal surfaces (Hsieh et al., 2010; Uhlig and Revie, 1985). However, scaling on metal and metal alloy surfaces reduces the heat transfer efficiency of the cooling system.

In this study, the feasibility of controlling corrosion, scaling, and biofouling when using tertiary treated (nitrified-filtered) municipal wastewater (MWW_NF) in cooling water systems was investigated through laboratory and pilot-scale experiments. Nitrification with subsequent filtration will result in ammonia removal, lower pH and lower total solids in the water (Metcalf and Eddy, 2003). Ammonia causes corrosion of metals by forming soluble complexes with copper (Strmcnik et. al., 2009) and iron (Uhlig and Revie, 1985). Hence removal of ammonia will reduce the corrosiveness of MWW. As a result, MWW with additional treatment by nitrification and filtration may require fewer chemicals for corrosion and scaling management. Bench-scale recirculating systems and three pilot-scale cooling towers were employed for testing of various chemical control schemes for corrosion, scaling, and biofouling in systems using secondary treated municipal wastewater. The testing was conducted with conditions of temperature, flow velocity, and water constituent concentration similar to those in a recirculating cooling water system. Synthetic tertiary treated (nitrified-filtered) municipal wastewater recipe (SynMWW_NF) was simulated to have similar constituent concentration of actual MWW_NF used as cooling water in pilot-scale systems. The effectiveness of chemical treatment strategies in inhibiting corrosion, scaling, and biomass growth was studied through exposure and monitoring specially designed coupons in extended duration tests.

4.1 Corrosion Control for MWW_NF Used for Cooling

In this study corrosion of different metals and metal alloys was evaluated while tertiary treated (nitrified-filtered) municipal wastewater (MWW_NF) was used as makeup water in power plant cooling systems. . Nitrification with subsequent filtration will result in ammonia removal, lower pH and lower total solids in the water (Metcalf and Eddy, 2003). Ammonia causes corrosion of metals by forming soluble complexes with copper (Strmcnik et. al., 2009) and iron (Uhlig and Revie, 1985). Hence removal of ammonia will reduce the corrosiveness of MWW. As a result, MWW with additional treatment by nitrification and filtration may require fewer chemicals for corrosion and scaling management. So nitrification with subsequent filtration results in a new scenario for corrosion management of metals and metal alloys exposed to the flowing cooling water. The overall goal of this study was to investigate the feasibility of using tertiary treated (nitrified-filtered) municipal wastewater (MWW_NF) in cooling water systems by metal alloy corrosion rate monitoring and comparison with general corrosion criteria (Hsieh et al., 2010).

The use of corrosion inhibitors is the most widely employed approach to control corrosion in recirculating cooling water system (Frayne, 1999). Corrosion inhibitors usually form barrier layers on the surface of a metal and thus decrease corrosion rate. Barrier forming inhibitors are categorized into three types: adsorbed layer formers, oxidizing inhibitors (passivators), and conversion layer formers (Dean et al., 1981). Briefly, adsorbed layer formers function by adsorbing to the metal surface; oxidizing inhibitors function by shifting the metal's electrochemical potential to a region where the metal oxide or hydroxide is stable (passivating); and conversion layer formers function by forming a low solubility deposition on the metal surface (Dean et al., 1981). Among the commonly used corrosion inhibitors, polyphosphates, and tolyltriazole were selected for testing in this research based on review of previous studies (Hsieh et al., 2010; McCoy, 1974; Frayne 1999; Jones, 1996; Harston, 2004).

Polyphosphates are identified as cathodic corrosion inhibitors (Saha and Kurmaih, 1986). Polyphosphate compounds in water gradually hydrolyze to orthophosphate (Frayne, 1999), which is an anodic inhibitor (Saha and Kurmaih, 1986). However, phosphate scaling potential increases when the aqueous solution has high hardness (Goldstein et al., 1981). Generally, the feasibility of employing phosphorous based inhibitors in cooling tower systems using impaired waters with high hardness and alkalinity is not very promising (Hsieh et al., 2010). Polyphosphate based inhibitors tend to precipitate in such systems which escalates the scaling problems.

Tolyltriazole (TTA) is an inhibitor specifically for copper alloys. Its NH group can adsorb onto the metal surface thus forming a barrier layer (Hollander and May, 1985). Copper corrosion inhibition by TTA can be reduced by free chlorine residual (Breske, 1983; Lu et al., 1994; Harrison and Kennedy, 1986), which is often maintained in cooling systems to prevent microbial growth and biofouling. On the other hand, chloramine has lower oxidizing power but studies of its influence on copper corrosion inhibition by TTA are limited. Previous studies (Hsieh et al., 2010) also showed that TTA was able to lower the corrosion rate of mild steel immersed in MWW.

Specifically, the objectives of this study were to 1) design an experimental matrix and perform laboratory experiments to evaluate metal alloy corrosion immersed in SynMWW_NF, 2) assess corrosion of metal alloys immersed in MWW_NF in a bench-scale recirculation water system with identified corrosion control program, and 3) apply the optimal corrosion control program identified from bench-scale experiments to pilot-scale cooling systems operated with tertiary treated (nitrified-filtered) municipal wastewater (MWW_NF) in the field.

4.1.1 Testing protocols

4.1.1.1 Metal alloy pre-exposure and post-exposure treatment

The metal and metal alloys chosen for this study were mild steel (UNS G10180), copper (UNS C10100), and cupronickel (UNS C70600) as they are suitable for cooling water systems using municipal wastewater as makeup water (Hsieh et al., 2010; Herro and Port, 1993). The specimens were cylinder-shaped with a nominal diameter of 0.375 in., nominal length of 0.5 in. and were obtained from Metal Samples Co. (Munford, AL). Before immersing the metal and metal alloy samples in recirculating cooling water (both in bench-scale and pilot-scale systems), they were wet polished with SiC paper to a 600 grit surface finish, dried, weighed to 0.1 mg, degreased with acetone and rinsed in distilled water.

During exposure of the metal alloy specimens in the bench-scale experiments, the polarization resistance (R_P) of the specimens was semi-continuously monitored and specimens were withdrawn after approximately 1, 3, and 5 days of exposure to measure the weight loss. After withdrawal, the specimens were cleaned following the ASTM G1 and then reweighed to 0.1 mg to determine the weight loss (ASTM G1-03, 2005).

In the pilot scale cooling system tests, the metal and metal alloy specimens were immersed for periods of 7, 14, 28, and 56 days. The weight loss method was used to determine the average corrosion rate during the exposure period. After withdrawal, similar post-exposure treatments were performed on the metals and metal alloys as was done in the bench-scale experiments.

4.1.1.2 Corrosion experiment matrix for the bench-scale experiments with MWW_NF

Synthetic tertiary treated (nitrification-filtration) municipal wastewater recipe was used in the bench-scale recirculation system for corrosion analysis of mild steel, copper and cupronickel alloys. Average 5-day corrosion rates of different metal alloys were determined. In the laboratory experiment with synthetic tertiary treated (nitrified-filtered) municipal wastewater tolyltriazole (TTA) and sodium hexametaphosphate (SHMP) were used as corrosion inhibitor. Instantaneous corrosion rate profiles for different metal alloys were produced from intermittent R_P measurement data (Choudhury et al., 2012)

Six different experiments were carried out in the bench-scale recirculation water system with synthetic tertiary treated (nitrified-filtered) municipal wastewater. The matrix of experiments conducted is shown in Table 4.1.1. The matrix was designed to 1) Evaluate the effectiveness of TTA and SHMP in corrosion inhibition of mild steel, copper and cupronickel alloys immersed in synthetic tertiary treated municipal wastewater, 2) Study the effect of individual ingredient in the corrosion inhibitors mixture, and 3) Assess the effectiveness of the corrosion inhibitors in the presence of oxidizing agent monochloramine.

Actual nitrified-filtered tertiary treated municipal wastewater collected from Franklin Township Municipal and Sanitary Authority (FTMSA) facility was concentrated in the laboratory by heated evaporation (at 40^{o}C) to reach four cycles of concentration (CoC4) as determined by 75% volume reduction for bench-scale experiments. The temperature (40^{o}C), used for heated evaporation in the lab, simulates the temperature of recirculating cooling water in the pilot-scale cooling systems (Hsieh et al, 2010). At CoC4 the tertiary treated municipal wastewater was used in the bench-scale recirculation water system for corrosion analysis of mild steel, copper and cupronickel alloys.

Two experiments were carried out in the bench-scale recirculation system using actual tertiary treated municipal wastewater (with and without addition of 4 ppm TTA) for corrosion analysis of mild steel, copper, and cupronickel alloys. The samples were immersed for 5 days for average corrosion rate analysis. Intermittent R_P measurements were performed throughout the immersion period of the metal alloy samples for producing an instantaneous corrosion rate profile for the metal alloys.

Table 4.1.1 Experimental matrix for bench-scale corrosion analysis of metal and metal alloy samples in contact with synthetic tertiary treated municipal wastewater. Flow rate 0.189 L/s (3gpm), temperature 40 ± 1 °C.

Synthetic secondary treated municipal wastewater composition tested	Tolyltriazole (TTA) concentration used, mg/L	Sodium Hexametaphosphate (SHMP) concentration used, mg/L	Objective of experiments with different synthetic wastewater compositions		
			Effectiveness of inhibitors	Effect of each ingredient in Mix	Effectiveness in the presence of MCA
CoC4	0	0	X		
CoC4_TTA2	2	0	X	X	X
CoC4_SHMP10	0	10	X	X	
CoC4_Mix	2	10	X	X	
CoC4_TTA2_MCA3	2	0	X		X
CoC4_TTA4_MCA3	4	0	X		X

Note: CoC4: Four cycles of concentration. TTA2, TTA4: Tolyltriazole 2ppm, 4 ppm respectively. SHMP10: Sodium Hexapemtaphosphate 10ppm. Mix: TTA 2ppm, SHMP 10ppm. MCA3: Monochloramine (intermittently maintain 3ppm as total chlorine).

4.1.1.3 Pilot-scale corrosion experiment with MWW_NF

The experimental matrix for the pilot-scale testing was based on the results of the laboratory experiments with respect to corrosion, scaling, and biofouling studies (scaling and biofouling control studies are not covered in this section). TTA was chosen as corrosion inhibitor, and PMA (a scaling inhibitor identified to be effective in scaling control through the laboratory experiments in another study) as a scaling inhibitor. Monochloramine was used as biofouling control agent. Three different pilot-scale cooling towers were operated for 56 days after reaching CoC 4. All three cooling towers were operated using tertiary treated (nitrified-filtered) municipal wastewater. Monochloramine was dosed as a biocide at concentrations of 2-3 ppm in all cooling towers. One of the cooling towers (CTA) was operated without addition of any corrosion and scaling inhibitors. The other two cooling towers were operated with addition of corrosion inhibitor TTA and scaling inhibitor PMA. Table 4.1.2 shows the pilot-scale experimental matrix with chemical treatment strategies.

In the pilot-scale testing, corrosion inhibitor TTA was monitored with high performance liquid chromatography (HPLC 1100 Series, Agilent Technologies) (Choudhury et al., 2012). Other water chemistry parameters (PMA, monochloramine, free chlorine, anions, alkalinity, pH and conductivity) and cooling tower operational parameters (water flow rate, air flow velocity, temperature, makeup water flow rate, and blowdown water flow rate) were also monitored. The inhibitor TTA was added once per day to the makeup water tank. Concentration of TTA dosed in the makeup water was one-fourth of the target concentration of TTA in the cooling tower system. TTA concentration in the pilot scale cooling tower basin reached to desired target concentration as the recirculating cooling water reached CoC 4.

Table 4.1.2 Experimental matrix for pilot-scale cooling tower experiments showing different chemical treatment strategies for corrosion, scaling, and bio-fouling management. Flow rate 0.189 L/s (3gpm).

Cooling Tower Designation	pH	Corrosion Inhibitor	Scaling Inhibitor	Bio-fouling Control
Cooling Tower A (CTA)	7.8±0.6	TTA (0 ppm)	PMA (0 ppm)	MCA (2~3 ppm residual as TC)
Cooling Tower B (CTB)	7.8±0.6	TTA (2 ppm)	PMA (5 ppm)	MCA (2~3 ppm residual as TC)
Cooling Tower C (CTC)	7.8±0.6	TTA (4 ppm)	PMA (5 ppm)	MCA (2~3 ppm residual as TC)

Note: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); TC (Total chlorine).

4.1.2 Bench-scale studies

4.1.2.1 Bench-scale recirculation system experiments with SynMWW_NF: corrosion analysis

The average 5-day corrosion rates from the four bench-scale recirculation system experiments are given in table 4.1.3. Corrosion inhibition efficiency values are also given with respect to the control experiment using synthetic tertiary treated (nitrified-filtered) municipal wastewater. From the results given in Table 4.1.3 it is observed that addition of 2ppm TTA imparted higher corrosion inhibition for mild steel than 10ppm SHMP. It was observed that most of the SHMP precipitated out of the solution in the system (Choudhury, 2012). For system with high calcium, precipitation of solid calcium phosphate will reduce the availability of polyphosphate in solution for effective metal-inhibitor film formation (Saha and Kurmiah, 1986). Hence SHMP was found less effective than TTA, as a corrosion inhibitor, in synthetic municipal wastewater. For copper and cupronickel alloys, immersed in synthetic tertiary treated (nitrified-filtered) municipal wastewater, addition of 2ppm TTA imparted corrosion inhibition efficiencies of more than 90%. Formation of TTA protection film on copper and cupronickel surface reduced the corrosion rates of these metal alloys in solutions (Hollander and May, 1985). Presence of SHMP inhibited corrosion of cupronickel, but SHMP was less effective than TTA. Average copper corrosion rate increased with the use of SHMP in SynMWW_NF.

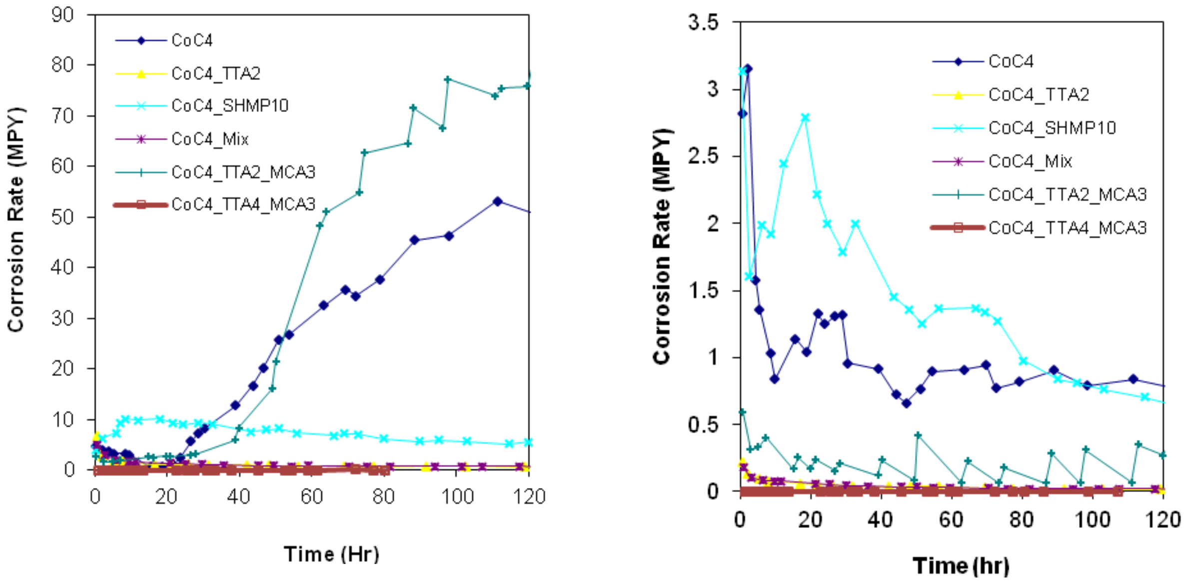
Intermittent dosing of monochloramine as a biocide was done to keep the residual concentration at 3 mg/L as total chlorine. The presence of monochloramine exacerbated the corrosion rate of all the metal and metal alloys tested in the system. TTA dosing of 2 mg/L inhibited copper and cupronickel corrosion, but failed to inhibit mild steel corrosion. Use of 4mg/L TTA reduced the corrosion rate of all three metal and metal alloys in the system.

Table 4.1.3 Average corrosion rates of metal and metal alloy samples, and corresponding corrosion inhibition efficiencies in the bench-scale recirculating cooling water experiments. The metal and metal alloys were exposed to synthetic cooling water for five days.

Synthetic secondary treated municipal wastewater composition tested	Tolyltriazole (TTA) concentration used, mg/L	Sodium Hexametaphosphate (SHMP) concentration used, mg/L	Average Corrosion Rate (MPY) (corrosion inhibition efficiency, %)		
			Mild Steel	Copper	Cupronickel
CoC4	0	0	20.5 (--)	0.766 (--)	3.80 (--)
CoC4_TTA2	2	0	0.655 (96.8%)	0.045 (94.1%)	0.094 (97.5%)
CoC4_SHMP10	0	10	5.68 (72.3%)	0.897 (-17.1%)	0.244 (93.6%)
CoC4_Mix	2	10	0.863 (95.8%)	0.038 (95.1%)	0.086 (97.7%)
CoC4_TTA2_MCA3	2	0	41.0 (-99.6%)	0.206 (73.1%)	0.398 (89.5%)
CoC4_TTA4_MCA3	4	0	7.29 (64.5%)	0.133 (82.6%)	0.094 (97.5%)

Note: CoC4: Four cycles of concentration. TTA2, TTA4: Tolyltriazole 2ppm, 4 ppm respectively. SHMP10: Sodium Hexapemtaphosphate 10ppm. Mix: TTA 2ppm, SHMP 10ppm. MCA3: Monochloramine (intermittently maintain 3ppm as total chlorine).

The instantaneous corrosion rate (ICR) profiles for mild steel, copper, and cupronickel samples in the bench-scale recirculation system experiments are shown in Figure 4.1.1. TTA can be successfully used to inhibit the corrosion of copper and cupronickel alloys. SHMP was less effective than TTA for mild steel corrosion inhibition.



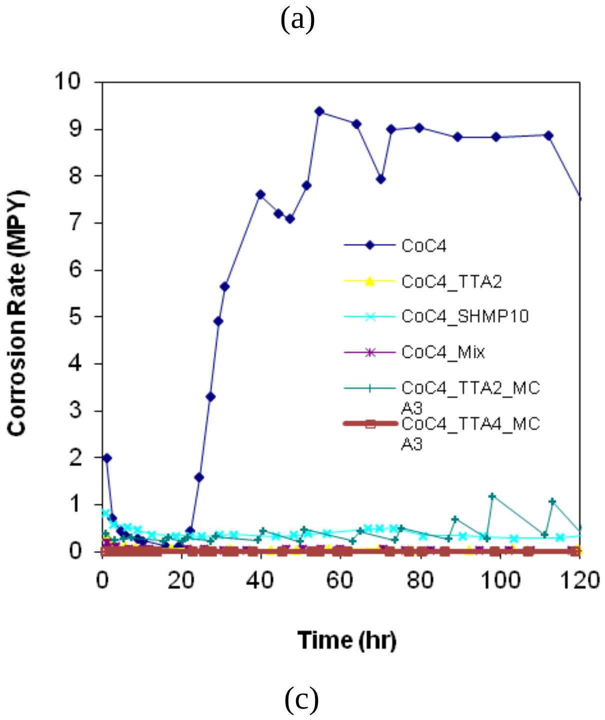


Figure 4.1.1. Instantaneous corrosion rate profile obtained by combined electrochemical polarization resistance measurement and weight loss measurement in the bench-scale recirculating system using synthetic tertiary treated (nitrified-filtered) municipal wastewaters for (a) mild steel, (b) copper, and (c) cupronickel alloys.

4.1.2.2 Bench-scale recirculation system experiments with MWW_NF: corrosion analysis

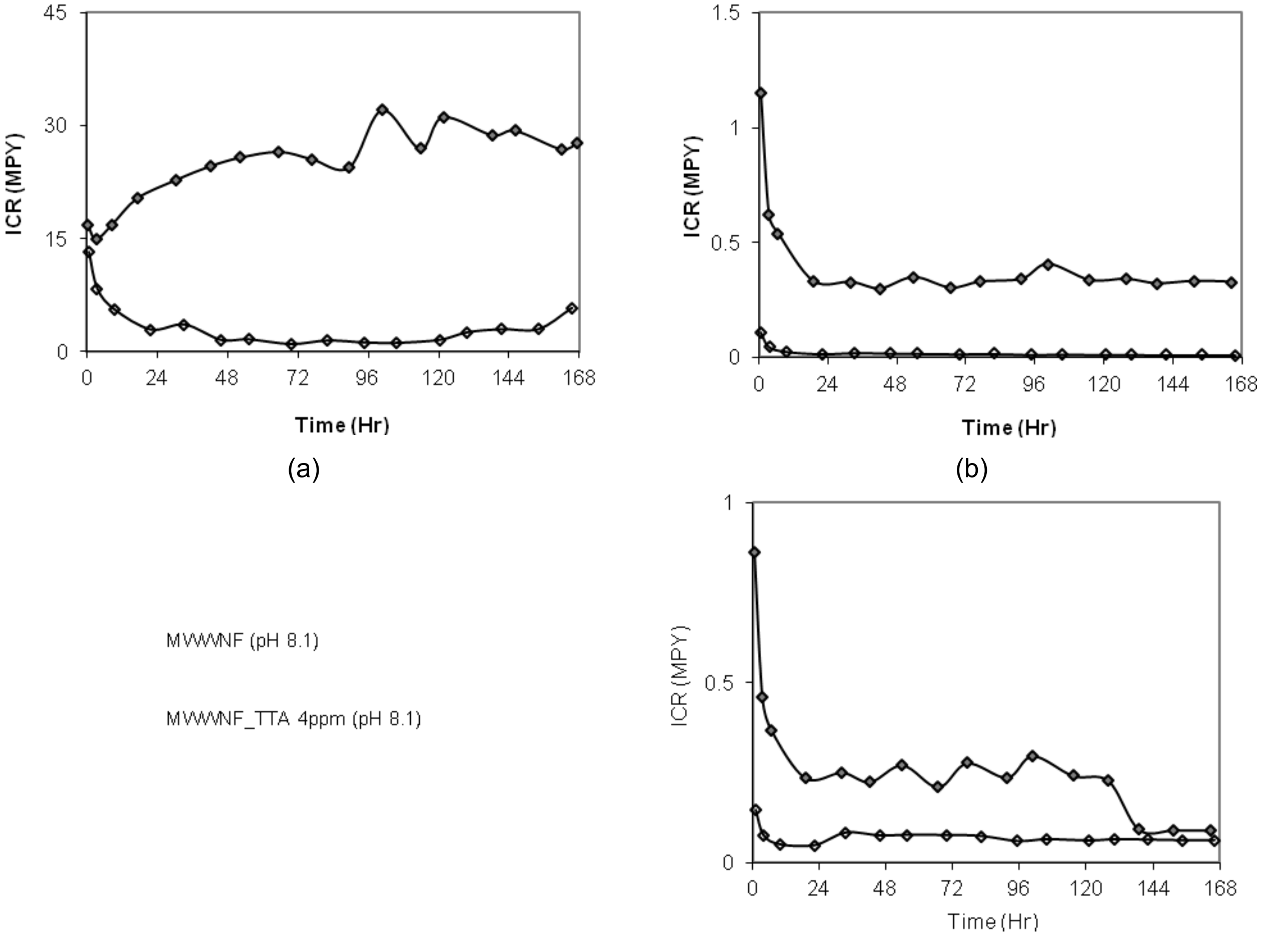
MWW_NF water collected from FTMSA facility was used in bench-scale recirculation system experiments at CoC 4 with and without the addition of corrosion inhibitor TTA. Table 4.1.4 shows the average 5-day corrosion rates of metals and metal alloys immersed in MWW_NF. Results from the table indicated that TTA successfully inhibited the corrosion of copper and cupronickel alloys to excellent levels according to the cooling tower system corrosion criteria. Mild steel corrosion rate was significantly inhibited and lowered to acceptable limits using TTA as corrosion inhibitor.

Table 4.1.4 Average 5-day corrosion rates of mild steel, copper and cupronickel samples exposed to MWW_NF at 4CoC, collected from Franklin Township Municipal Sanitary Authority (FTMSA) wastewater treatment facility, in the bench-scale recirculating system.

Type of actual tertiary treated wastewater	Average 5-day corrosion rate, MPY (Corrosion category [b])		
	Mild Steel	Copper	Cupronickel
MWW_NF (Control)	25.9 (Unacceptable)	0.381 (Good)	0.229 (Good)
MWW_NF (TTA 4mg/L)	7.42 (Poor)	0.022[a] (Excellent)	0.076 [a] (Excellent)

Note: [a] weight loss determined from electrochemical polarization resistance measurements (Choudhury, 2012).
[b] b Hsieh et al., 2010.
MPY: milli-inch per year. Control: no corrosion inhibitor was dosed in the system. TTA: Tolyltriazole.

Figure 4.1.2 shows the instantaneous corrosion rate measurement profiles for mild steel, copper and cupronickel alloy samples measured for an interval of 7 days in the bench-scale recirculating system with MWW_NF. The pH of each type of recirculating water along with respective TTA dosing is also indicated in Figure 4.1.2.



(a)

(b)

(c)

Figure 4.1.2 Instantaneous corrosion rate profiles for (a) mild steel, (b) copper, and (c) cupronickel alloy samples immersed in bench-scale recirculating cooling system using MWW_NF.

4.1.3 Pilot-scale studies

Pilot-scale experiments were carried out with three cooling towers (CTA, CTB and CTC) using tertiary treated (nitrified-filtered) municipal waste water. CTA was used as a control with no addition of corrosion and scaling inhibitors. CTB, and CTC were dosed with 2, and 4 ppm TTA, respectively. The average residual free TTA concentrations in CTB and CTC were 1.78±0.32 ppm and 3.40±0.53 ppm, respectively. All the cooling towers were dosed with 2-3ppm monochloramine for biofouling control.

Table 4.1.5 shows the average corrosion rates of metal and metal alloy specimens immersed in tertiary treated (nitrified-filtered) municipal wastewaters at CoC 4 in the three pilot-scale cooling towers for different time intervals. It can be seen from Table 4.1.5 that satisfactory corrosion control was achieved with copper and cupronickel in the three cooling towers.

Table 4.1.5 Average corrosion rates of metal alloys contacted with tertiary treated (nitrified-filtered) municipal wastewater at CoC 4 in pilot-scale cooling systems for different time intervals and their category according to general corrosion criteria[b] for cooling systems.

Cooling Tower	Immersion Days	Average Corrosion Rates of Metal Alloys (MPY) (Corrosion category)		
		Mild Steel	Cupronickel	Copper
CTA [a]	7	7.67 (Poor)	0.27 (Good)	0.36 (Good)
	14	2.96 (Good)	0.11 (Excellent)	0.11 (Excellent)
	28	13.43 (Unacceptable)	0.17 (Excellent)	0.24 (Good)
	56	15.24 (Unacceptable)	0.23 (Good)	0.28 (Good)
CTB [a]	7	2.73 (Good)	0.05 (Excellent)	0.04 (Excellent)
	14	1.29 (Good)	0.07 (Excellent)	0.04 (Excellent)
	28	9.41 (Poor)	0.09 (Excellent)	0.20 (Excellent)
	56	7.76 (Poor)	0.13 (Excellent)	0.03 (Excellent)
CTC [a]	7	1.27 (Good)	0.04 (Excellent)	0.04 (Excellent)
	14	1.67 (Good)	0.01 (Excellent)	0.01 (Excellent)
	28	7.45 (Poor)	0.13 (Excellent)	0.18 (Excellent)
	56	6.38 (Poor)	0.07 (Excellent)	0.08 (Excellent)

[a] CTA : Cooling Tower A with 0 ppm TTA, 0 ppm PMA and 2-3 ppm MCA; CTB: Cooling Tower B with 2 ppm TTA, 5 ppm PMA, 2-3 ppm MCA; CTC: Cooling Tower C with 4 ppm TTA, 5 ppm PMA, 2-3 ppm MCA.
[b] Hsieh et al., 2010.

The corrosion control of copper and cupronickel alloys were excellent based on the 56-day average corrosion rate in relation to the industrial corrosion criteria (Hsieh et al, 2010). The average corrosion rates for shorter duration (7, 14, 28 days) were also less than 0.2 MPY indicating excellent corrosion control while using TTA. The use of 2ppm TTA in CTB and 4ppm TTA in CTC gave almost similar corrosion control. Additional use of TTA was not advantageous with tertiary treated wastewater. The presence of residual free TTA in solution was sufficient to control the copper and cupronickel corrosion. With use of the tertiary treated wastewater in the control tower CTA, the 56-day average corrosion rates for copper and cupronickel were less than 0.2 MPY, indicating an excellent corrosion rate for these materials.

Mild Steel corrosion control was not satisfactory with the use of tertiary treated municipal wastewater in the cooling tower CTA, which had no TTA added. CTA gave a 56-day average corrosion rate of 15.24 MPY for mild steel, which according to the corrosion criteria (Hsieh et al., 2010) is unacceptable. With addition of TTA the corrosion rate was reduced, but still was poor (between 5 to10 MPY) based on the 56 days average corrosion rate. So it can be said that TTA was able to reduce the mild steel corrosion rate, but its efficacy was not as prominent as with copper and cupronickel.

It was observed that in spite of dosing both cooling towers (CTB and CTC) with 2ppm and 4ppm TTA respectively, the residual free TTA concentration in the system was lower than the target TTA concentration. The TTA concentration profile shown in Figure 4.1.3 indicates this fact. The average TTA concentration of CTB was 1.78±0.32 ppm and that of CTC was 3.40±0.53 ppm. A major reason for reduction of free TTA concentration is hypothesized to be the complexation of TTA with copper corrosion products formed on the surface of copper metal, which makes less free TTA available for effective corrosion control.

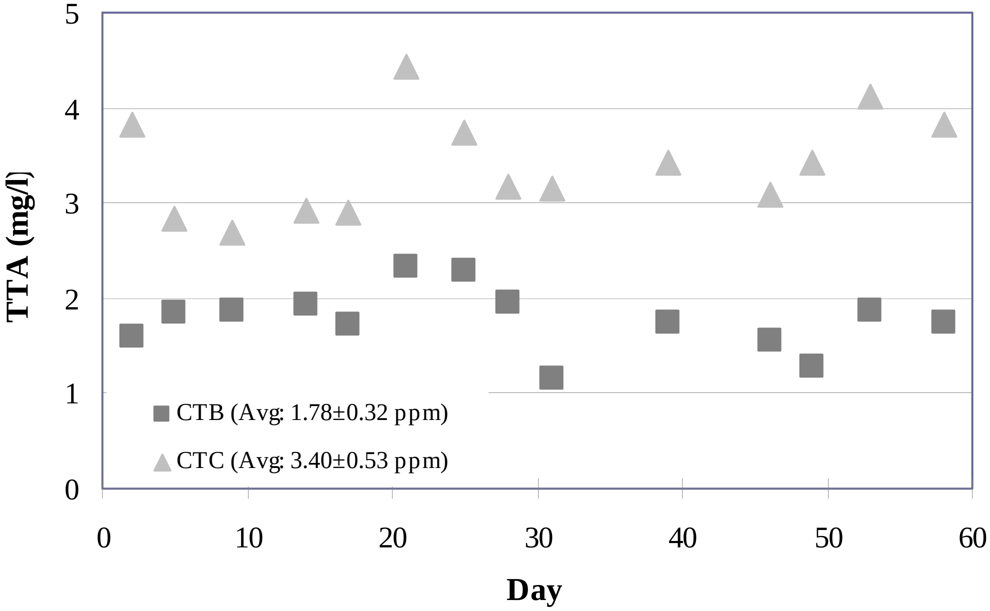


Figure 4.1.3 Concentration profile of TTA in CTB, and CTC (measured twice each week) in the pilot scale cooling tower experiments using Tertiary treated (nitrified-filtered) municipal wastewater (MWW_NF) from FTMSA.

The corrosivity of the cooling water can be correlated with the pH. The results in Table 4.1.5 show that the 7-day and 14-day mild steel samples displayed lower average corrosion rate than the 28-day and 56-day samples. Usually the 7-day or 14-day samples would have displayed higher average corrosion rates. Because the initial corrosion rates were higher and with passivation taking place on the metal surface at later times the corrosion rate gradually gets to a constant lower value.

The lower corrosion rates in the 7-day and 14-day samples can be linked with the average pH during the sample immersion period. Figure 4.1.4 shows the pH profile during the MWW_NF run. The shaded regions in the figure show the 7-day, 14-day, 28-day, and 56-day immersion periods for different metals/metal alloys; the figure also indicates the average pH in these time periods. From the pH profile it can be seen that the 7-day and 14-day samples were at higher pH than the 28-day sample. Hence the lower average corrosion rates in at the earlier times can be linked to this pH variation.

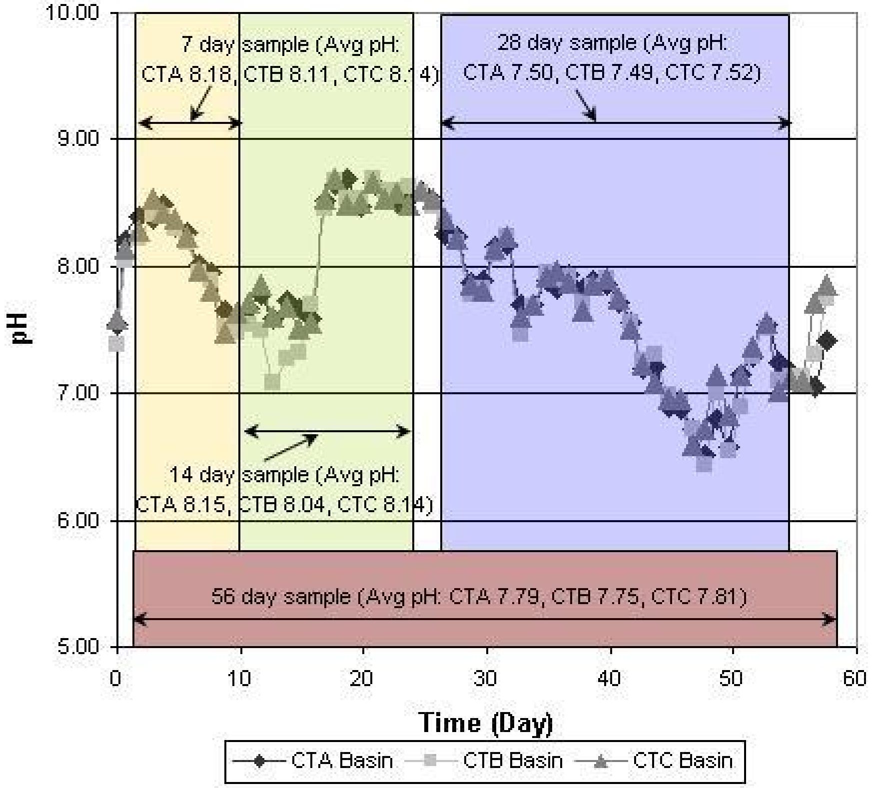


Figure 4.1.4 pH profile in CTA, CTB and CTC in the pilot scale cooling tower experiments using tertiary treated (nitrified-filtered) municipal wastewater from FTMSA. Average pH values in shaded regions, indicating the immersion period of different metal alloys in cooling waters, are given for all three cooling towers.

4.2 Scaling Control for MWW_NF Used for Cooling

It has been reported that the ammonia concentration in the secondary-treated MWW can be as high as 10 to 30 mg-N/L (Selby et al., 1996). Biological nitrification processes for ammonia removal have been adopted by many wastewater treatment plants faced with more stringent requirement (CSUC, 2009). From the prospective of mineral scaling control, tertiary treatment of the MWW by nitrification would depress the pH and alkalinity and thus reduce the scaling propensity in recirculating cooling systems (Metcalf & Eddy, 2003). There are currently no studies that offer guidance on the scaling behavior of MWW_NF when used as make-up water in recirculating cooling systems. In addition, there is no information about suitable scaling control methods that are demonstrated under relevant process conditions.

4.2.1 Testing Protocols

Similar to approach described in Section 3.2, batch tests and bench-scale recirculating system tests were first conducted to determine the effective scaling control strategy, which is then tested in extended pilot-scale cooling tower tests.

Tertiary-treated MWW_NF water (secondary treatment followed by nitrification and sand filtration) from Franklin Township Sanitary Authority, Murrysville, PA was used as a model for the synthetic MWW_NF in the batch and bench-scale recirculating system tests. The chemical composition of the synthetic MWW_NF at CoC 4 is shown in Table 4.2.1.

Table 4.2.1 Chemical compositions of the synthetic MWW_NF at 4 cycles of concentration (CoC 4) for batch and bench-scale recirculating system tests

Cation	Concentration (mM)	Anion	Concentration (mM)
Ca^{2+}	4.00	**SO_4^{2-}**	3.50
Mg^{2+}	1.60	**HCO_3^-**	1.60
Na^+	9.80	**Cl^-**	11.20
K^+	0.48	**PO_4^{3-}**	0.48
		NO_3^- (as N)	1.20

Precipitation behavior of MWW_ NF at CoC 4 was studied in a 1.4 L beaker covered with plastic foil to minimize water loss due to evaporation. The contents in the beaker were completely mixed with a magnetic stirrer and the bulk temperature was controlled at 40°C (typical bulk temperature of power plant recirculating cooling water) using the heating plate. Different from the tests with MWW_pH, pH of the MWW_NF was not controlled and was monitored during each test. Small aqueous sample is withdrawn from the reactor, filtered through 0.45 μm membrane and acidified to pH< 2 prior to analysis for soluble Ca and phosphate. At the end of the test, the bulk precipitates were collected from the reactor, washed with DI water, and air-dried for XRD and SEM/EDX analysis.

Mineral deposition behavior of the MWW_NF was also studied in bench-scale recirculating cooling system. Mass gain method was still employed to track the mineral scale formation in both processes.

Field tests with three pilot-scale cooling towers were conducted at the Franklin Township Municipal Sanitary Authority (FTMSA, Murrysville, PA). Two if the towers received 5 ppm PMA while the third tower was operated as a control tower for scaling study with no antiscalant dosed. The operating conditions of the cooling towers are shown in Section 2.2.3. Mass gain method with stainless steel discs (5.61 cm^2) was used to collect mineral scale samples. In order to exclude the influence of organic materials on the mass gain, the air-dried coupons collected from all three towers were dried at 104^{o}C for 3.5 hours and subsequently combusted at 500^{o}C for 3.5 hours in a muffle furnace. Then the portion remained, considered as the inorganic mineral scales, were analyzed by SEM/EDS to obtain their elemental composition. In parallel with the solids analysis, important information about the chemistry of the makeup and recirculating water was recorded throughout the field tests.

Besides the tests on the unheated surfaces with SS discs, mineral scaling on the heat transfer surfaces was also conducted in the bench-scale fouling study system. Fouling resistances were recorded according to the temperature measurement by the data logger system. In addition, critical water quality parameters including Ca concentration, total alkalinity, suspended solids concentration, and phosphate concentration in the bulk solution were monitored continuously. When the fouling resistance reached equilibrium, the cartridge heater was removed and the deposits were analyzed with XRD and SEM/EDX methods.

4.2.2 Bench-scale studies

No visual precipitate was observed during the first 2 hours of the batch test with synthetic MWW_NF at CoC 4 without any pH adjustment. However, as the solution equilibrated with atmosphere and pH increased from 6.8 to 7.6, it turned turbid after about 2 hours. Residual Ca and phosphate concentrations also decreased with the development of turbidity (Figure 4.2.1) and reached steady state after 14 hours. Solids characterization revealed that the precipitates were mainly in the form of amorphous calcium phosphate. These results suggest that when MWW_NF is used as make-up water in recirculating cooling systems operated at CoC 4, no magnesian calcite would be expected and calcium phosphate would be the primary mineral scale.

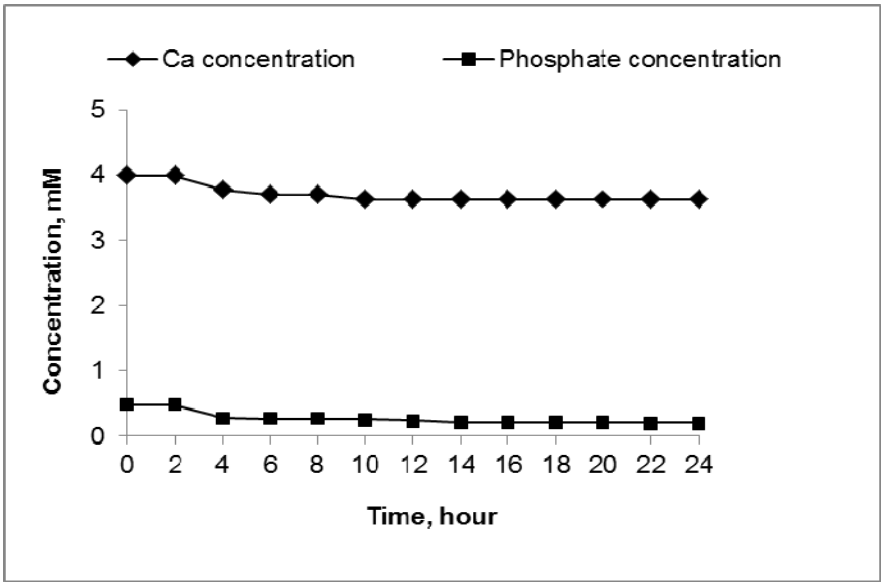


Figure 4.2.1 Residual Ca and phosphate concentrations in batch tests with synthetic MWW_NF (CoC 4) as a function of time at typical cooling water temperature (40°C)

4.2.3 Bench-scale recirculating system tests

Bench-scale recirculating system tests with MWW_NF (Figure 4.2.2) revealed negligible deposits on the disc specimens during 72 hours, indicating that MWW_NF exhibited little potential for mineral deposition. Such behavior is related in part to the lower pH and alkalinity in the MWW_NF. It is thus difficult to assess the effectiveness of PMA in scaling control for MWW_NF because very small amounts of deposits were formed even without PMA.

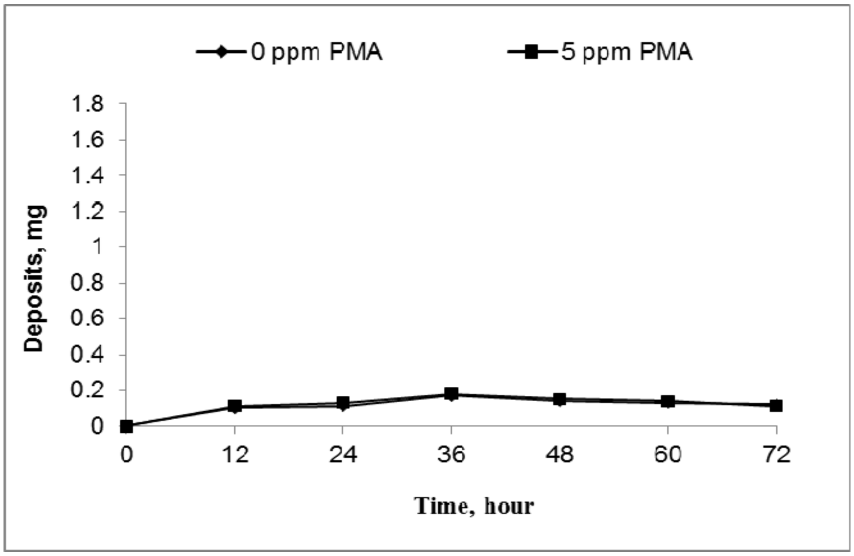


Figure 4.2.2 Scaling behavior of synthetic MWW_NF with and without the addition of 5 ppm PMA in bench-scale recirculating system tests

4.2.4 Pilot-scale studies

Figure 4.2.3 depicts accumulated mineral scale solids deposited on stainless steel disc specimens in the three cooling towers during the tests with MWW_ NF. The inorganic mass gain on the specimens in all three towers was small (0.05-0.67 mg) even after 50 days of immersion in the system. Samples from Tower A showed the least amount of deposits when compared with Towers B and C despite the fact that both Towers B and C received 5 ppm PMA to mitigate the scaling. Because of the small amount of inorganic deposits (less than 0.7 mg) accumulated on the stainless steel coupons, it is difficult to assess the effectiveness of the PMA as antiscalant. The coupons were almost clean even after immersion for 10 days (shown in Figure 4.2.3). LSI, RSI, and PSI values for all three towers on selected days in Table 4.2.2 were negative, suggesting that recirculating water was below saturation with respect to calcium carbonate.

(a) **(b)** **(c)**

Figure 4.2.3 Coupons collected on Day 10 in the pilot-scale cooling tower tests with MWW_NF as make-up water: (a) Tower A; (b) Tower B; (c) Tower C

Table 4.2.2 LSI, RSI, and PSI of the recirculating water in Towers A, B and C on days 36, 40, 44, 49 and 54

	LSI[a]			RSI[b]			PSI[c]		
	CTA	CTB	CTC	CTA	CTB	CTC	CTA	CTB	CTC
Day 36	-1.75	-1.58	-1.56	11.3	11.04	11.08	12.07	11.67	11.84
Day 40	-1.99	-1.80	-1.80	11.84	11.50	11.50	12.67	12.20	12.23
Day 44	-2.78	-2.59	-2.85	12.77	12.48	12.78	13.18	12.90	13.01
Day 49	-3.06	-2.61	-2.72	12.91	12.22	12.59	12.80	12.24	12.81
Day 54	-2.72	-2.66	-2.71	12.68	12.41	12.44	13.12	12.58	12.55

[a] If LSI is negative, no potential to scale, the water will dissolve $CaCO_3$; if LSI is positive, scale can form and $CaCO_3$ precipitation may occur; If LSI is close to zero, the water is on the borderline of the scale formation;

[b] If RSI < 6, the scale tendency increases as the index decreases; If RSI > 7, the calcium carbonate formation does not lead to a protective corrosion inhibitor film; If RSI > 8, mild steel corrosion becomes an increasing problem;

[c] As for PSI, a value below 6 means scale whole a value above 6 means no scale.

The pilot-scale cooling tower tests with MWW_NF were all operated at CoC 4-6 but the orthophosphate concentration in the recirculating water was generally 2~5 times that of the make-up water (Figure 4.2.4). These results indicated that precipitation of calcium phosphate occurred during the test period and is likely the major form of the limited mineral scales collected on sampling coupons. It must be noted that the precipitation of calcium phosphate was still not as severe as in the tests with secondary-treated MWW where 90% of the phosphate precipitated (Li et al., 2011). SEM/EDS analyses were performed on the deposits collected from Tower B after 50 days of operation with MWW_NF at CoC 4-6 (Figure 4.2.5). The EDS spectra showed that calcium phosphate was the predominant mineral scale, which is consistent with bench-scale results and water quality analysis. The carbon peak on Figure 4.2.5 could be due to substitution of carbonate for the phosphate in the deposits (Ferguson and McCarty, 1971; Suchanek et al., 2004).

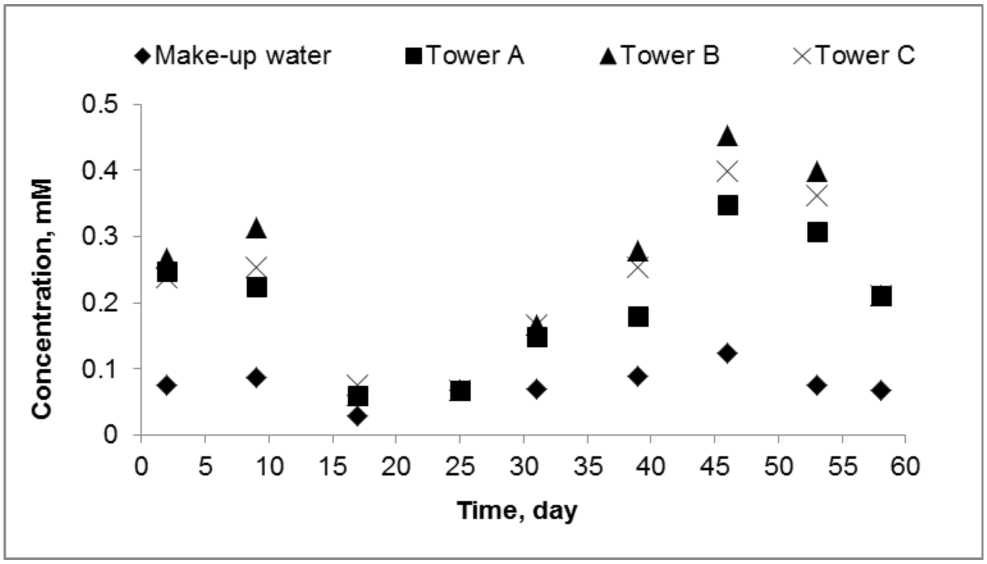


Figure 4.2.4 Orthophosphate concentration in the make-up water and recirculating water in the pilot-scale cooling tower tests with MWW_NF

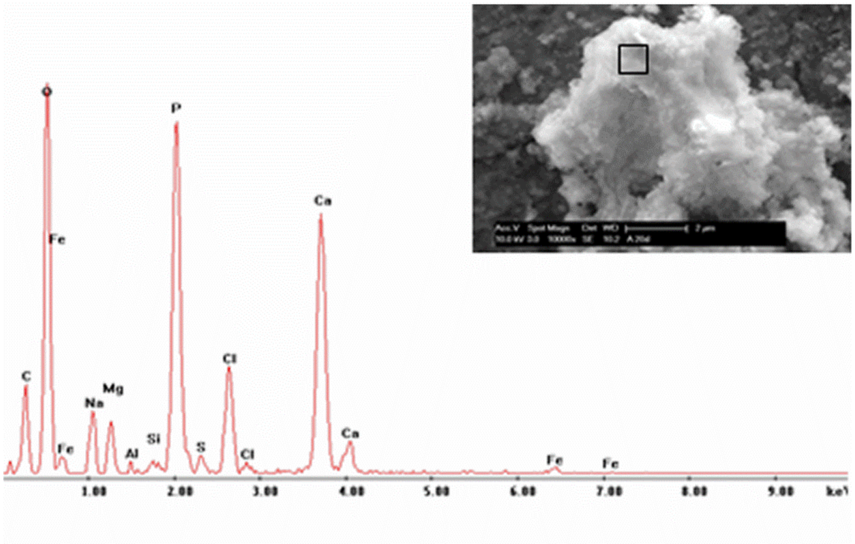


Figure 4.2.5 SEM images and the elemental composition of the solid deposits collected on stainless steel discs immersed in pilot-scale cooling towers operated at CoC 4-6: Day-50 sample from Tower B using MWW_ NF. EDS scan was performed on the area outlined by the square box on the SEM image

As described in Section 3.2, flow rate is another parameter indicating the scaling problem. Different from the results with MWW shown in Section 3.2, the flow rates were always above 2.70 GPM during the entire 60 days of operation, indicating that no significant scaling or biofouling has occurred in the three towers. In the last 30 days, the flow rate in Tower C was around 2.9-3.0 GPM while the flow rates in Tower A and B were generally in the range of 2.8-2.95 GPM. This trend was generally consistent with the mass gain data in which Tower C has shown the least mineral deposition while the mass gain in Tower A and B were a slightly higher.

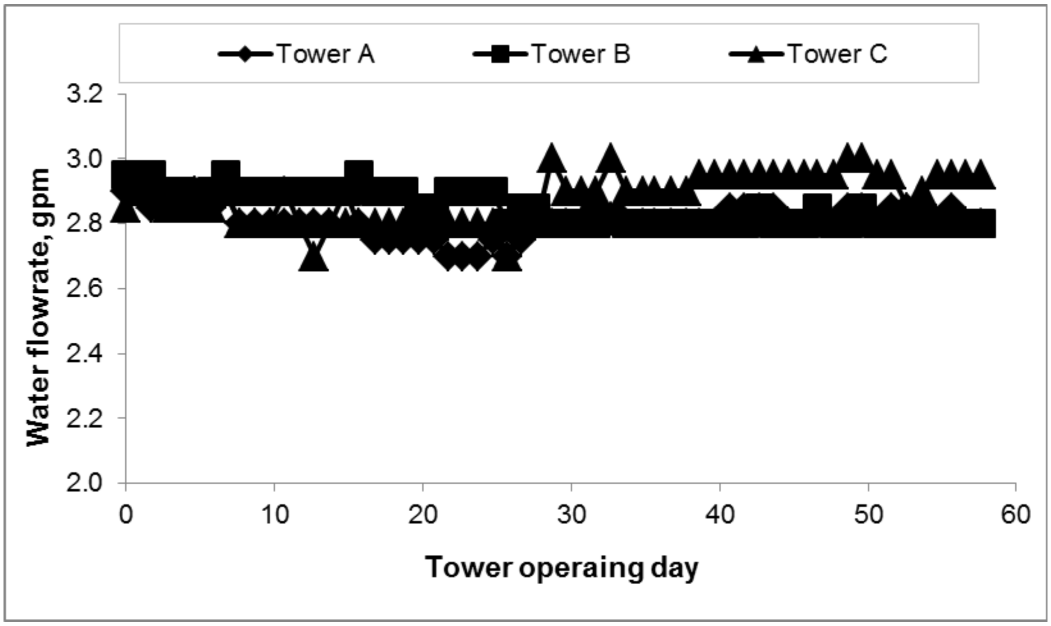


Figure 4.2.6 Water flow rates measured in the pilot-scale cooling tower tests with MWW_ NF as recirculating water. Target flow rate is 3 GPM for the system

4.2.5 Mineral scaling on heated surface

Synthetic MWW_NF simulating the tertiary-treated effluent from Franklin Township Municipal Sanitary Authority (FTMSA, Murrysville, PA) was used in this study. The flow velocity in the test section is 4.74 ft/s, resulting in Reynolds number as high as 24,039. Figure 4.2.7 shows significant development of fouling resistance with synthetic MWW_NF (CoC 4) without pH control in the bench-scale experimental system. Similar to tests with MWW, rapid growth period (0.5-3.0 h), slow growth period (3.0-5.0 h), and equilibrium period (5.0-24.0 h) have also been identified in the tests with MWW_NF (CoC 4). However, an induction period with negative fouling resistance (0-0.5 h) was observed in the experiment with MWW_NF (CoC 4). Similar induction period has also been reported in other studies (Budair, M.O. et al., 1998; Förster, M. et al., 1999). During the induction period when the fouling resistance appears to be small, the building up of the fouling layer causes the roughening of the heat transfer surface resulting in the increase in the local heat transfer coefficient and negative fouling resistance. As the fouling layer thickness increases, its thermal resistance increases due to the lower thermal conductivity of the fouling material. Total alkalinity, calcium concentration, phosphate concentration, and suspended solids concentration are shown in Figure 4.2.8-4.2.11, respectively. Hydroxyapatite ($Ca_5(PO_4)_3(OH)$) was the only crystalline material identified by XRD analysis in the deposits formed on the heater (Figure 4.2.12). Chemical composition of the deposit was further analyzed by EDS and the results are shown in Figure 4.2.13. It can be observed form Figure 4.2.13 that the Ca:P ratio was 1.54, which is lower than the ratio of 1.67 that is expected for hydroxyapatite. The results indicated that amorphous calcium phosphate ($Ca_3(PO_4)_2.nH_2O$) with Ca/P ratio of 1.5 may also exist in the deposits, which was verified by the high base line in the XRD spectra as shown in Figure 4.2.12. Amorphous calcium phosphate was the precursor for the formation of hydroxyapatite as the most stable form of calcium phosphate (Alvarezet al., 2004). However, the transformation of amorphous calcium phosphate to hydroxyapatite is slow, especially with the coexistence of Mg and carbonate (Cao and Harris, 2007). By combining the fouling curve and water quality analysis, rapid growth period is related with the sharp reduction of phosphate concentration in the bulk solution, indicating mixed mechanisms of crystalline fouling and particulate fouling. While phosphate concentration was almost constant in the slow growth period, the development of fouling resistance was accompanied with the reduction in suspended solids concentration, suggesting the particulate fouling would be predominant during this phase of the test.

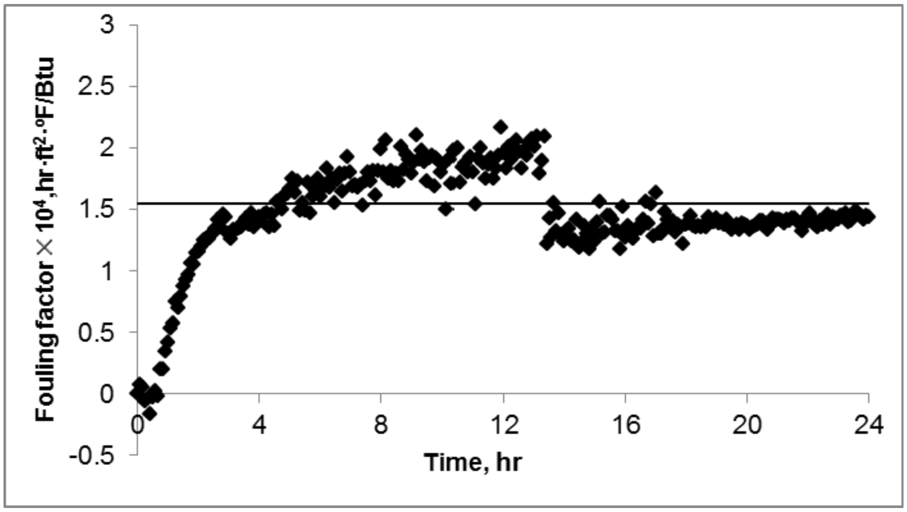


Figure 4.2.7 Fouling curve of the MWW_NF at CoC 4. The horizontal line indicates the approximate steady-state fouling factor achieved.

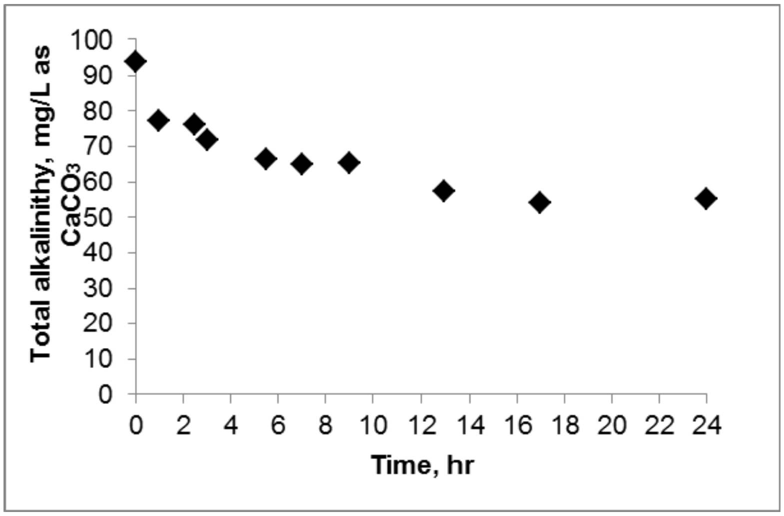


Figure 4.2.8 Total alkalinity of the recirculating water during the fouling study with MWW_NF

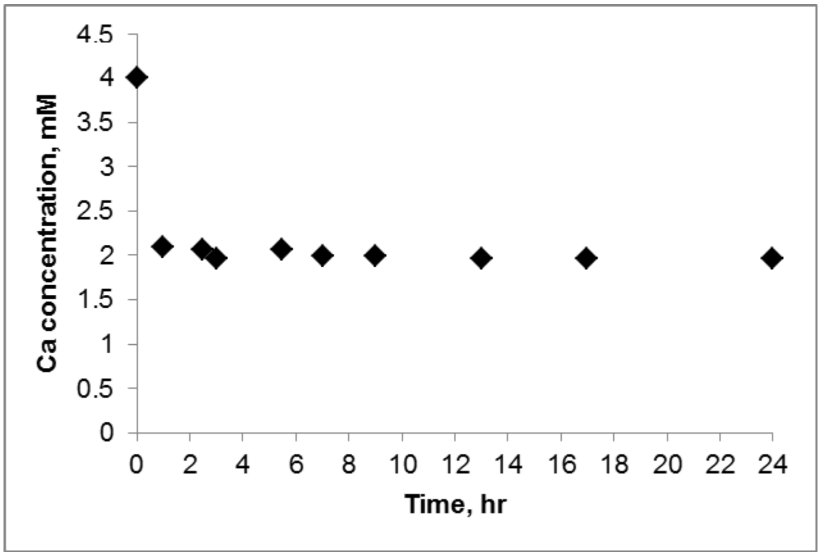


Figure 4.2.9 Calcium concentration in the recirculating water during the fouling study with MWW_NF

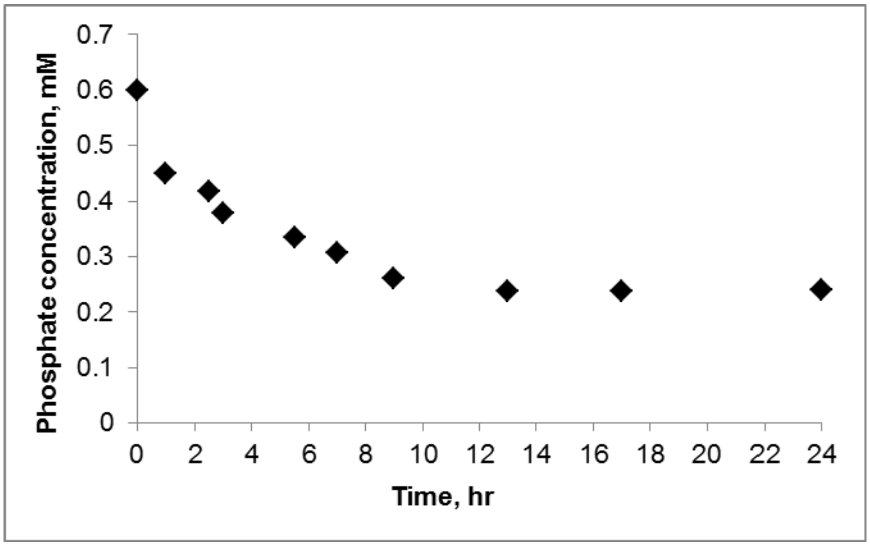


Figure 4.2.10 Phosphate concentration in the recirculating water during the fouling study with MWW_NF

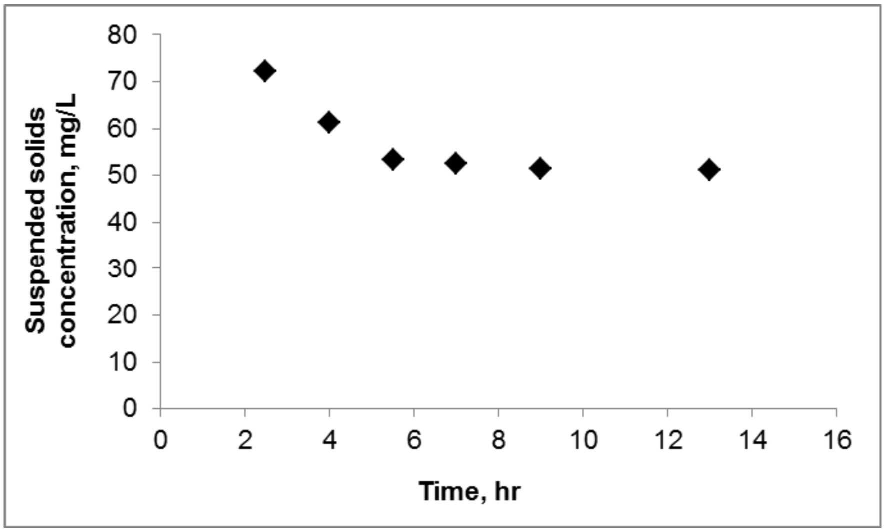


Figure 4.2.11 Suspended solids concentration in the recirculating water during the fouling study with MWW_NF

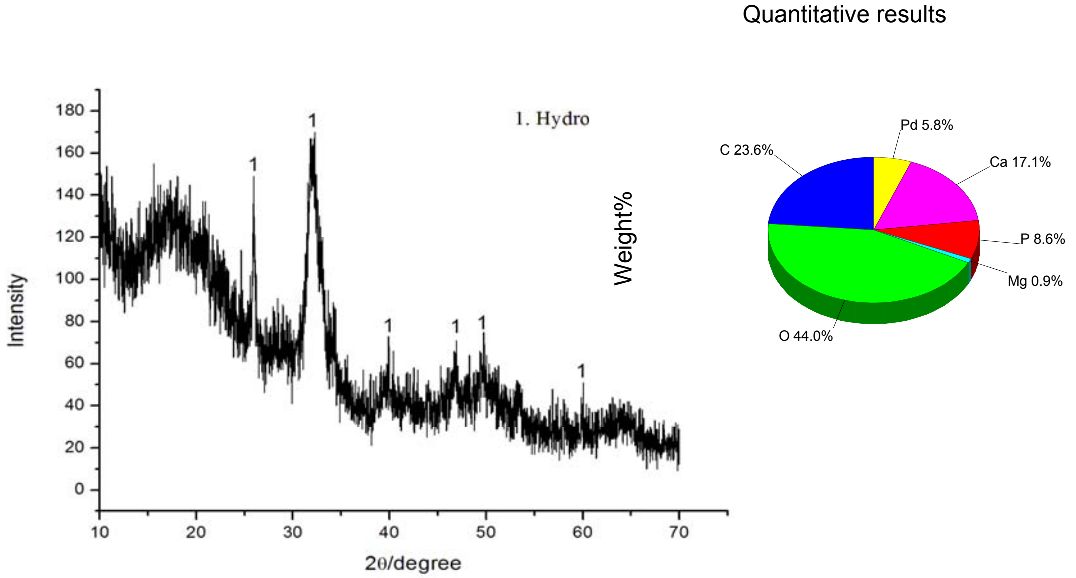


Figure 4.2.12 XRD pattern of the deposits formed on the heater in the fouling study with MWW_NF

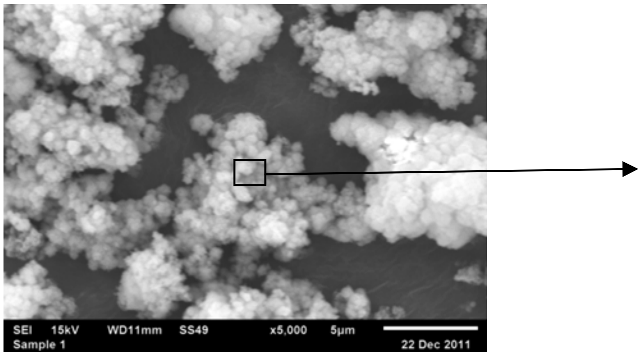


Figure 4.2.13 Morphology and elemental compositions of the deposits formed on the heater during the fouling test with MWW_NF

It must be noted that the fouling study with MWW_NF above was conducted without pH control. pH of the synthetic MWW_NF during the test period was recorded as shown in Figure 4.2.14. It can be observed from Figure 4.2.14 that pH in the first 5 hours was at comparatively lower level (7.2-7.4) and then increased slowly to about 7.8 at the end of the test. Comparing these data to the fouling curve on Figure 4.2.7 revealed that the first 5 hours was the period when fouling occurred. In other words, the fouling curve was mainly formed at pH 7.2-7.4, which is not the typical bulk pH when real MWW_NF was used as make-up water in a recirculating cooling water system of the thermoelectric power plants. According to our previous pilot-scale cooling tower tests with MWW_NF, average pH of the recirculating water was about 7.8. Thus, to simulate the real situation, it is necessary to conduct the fouling study with MWW_NF at representative pH conditions.

Figure 4.2.14 shows the development of fouling factors with synthetic MWW_NF at CoC 4 when bulk pH was controlled around 7.8. Based on this Figure , it can be concluded that the fouling potential of MWW_NF at pH 7.8 is negligible during the first 24 hours of contact with heated surface. Such behavior is vastly different from the one exhibited in Figure 4.2.7 when pH was not controlled. Ca, phosphate, total alkalinity, and total suspended solids in recirculating water throughout the test are depicted in Figure 4.2.15, 4.2.16, 4.2.17, and 4.2.18, respectively. The finding that these water quality parameters stabilized in less than 1 hour indicates that chemical precipitation was essentially complete shortly after the initiation of the test. In other words, most of the scale-forming components were consumed by bulk precipitation, thereby eliminating the possibility of crystalline fouling. Once the particles precipitated in the bulk solution, there was no possibility for their attachment to the immersion heater since the heater with smooth surface is placed vertically in the experimental system. Thus, it is understandable that no particle attached to the heater surface to form resistance for the heat transfer. However, it does not mean that particulate fouling never occurred in the test system. When the crystalline fouling on the heated surface created micro-roughness on the heater, it is still possible for bulk precipitates to attach to the heated surface by mechanical straining, as in the case of slow rapid growth period in Figure 4.2.7.

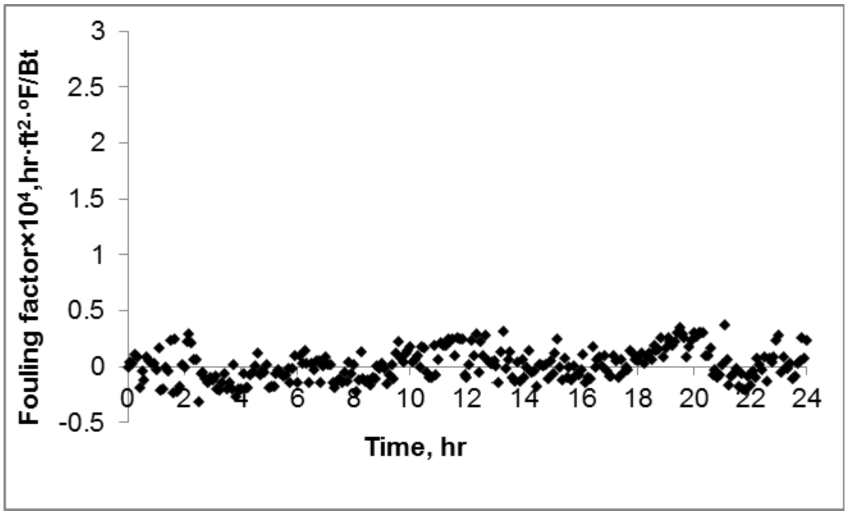


Figure 4.2.14 Fouling curve of the MWW_NF at CoC 4 with bulk pH controlled at 7.8

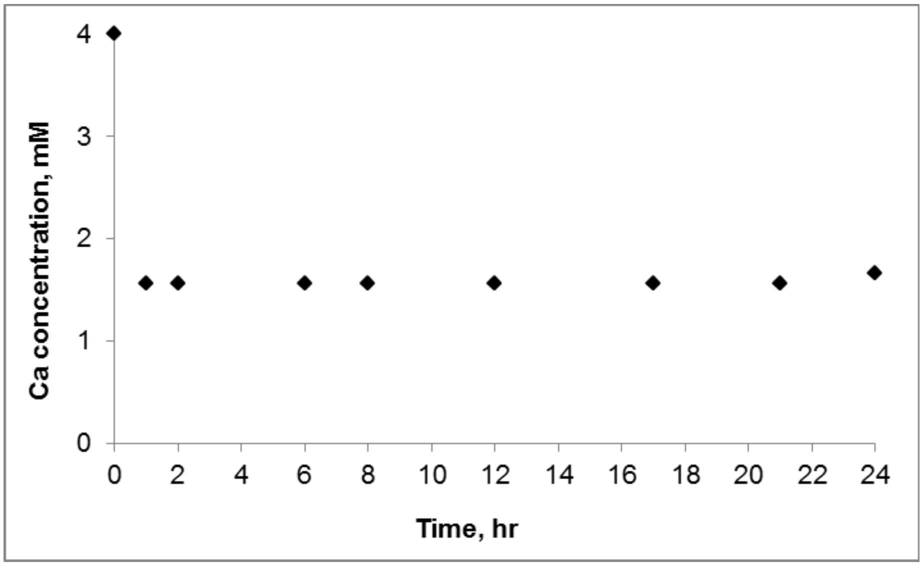


Figure 4.2.15 Calcium concentration in the recirculating water during the fouling study with MWW_NF when pH was controlled at 7.8

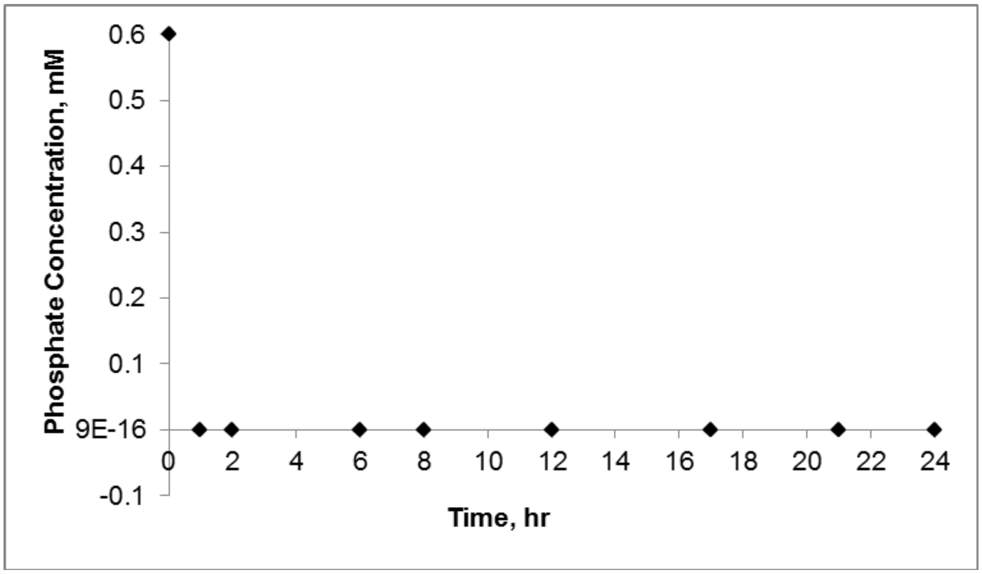


Figure 4.2.16 Phosphate concentration in the recirculating water during the fouling study with MWW_NF when pH was controlled at 7.8

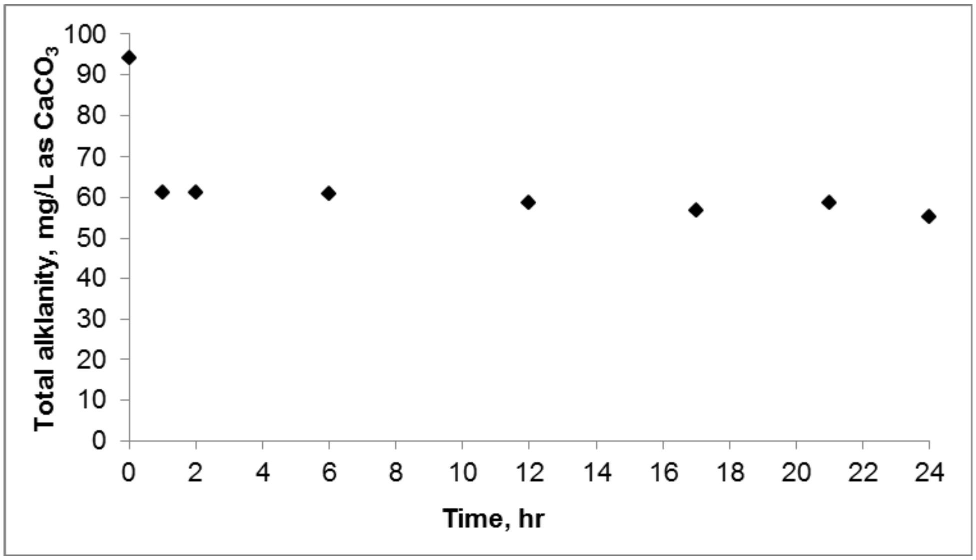


Figure 4.2.17 Total alkalinity in the recirculating water during the fouling study with MWW_NF when pH was controlled at 7.8

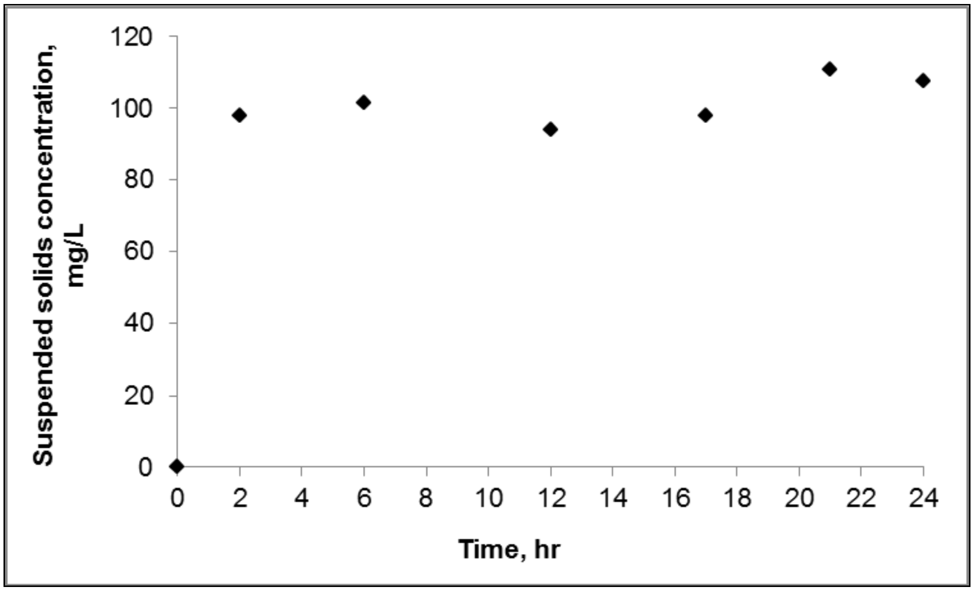


Figure 4.2.18 Total suspended solids concentration in the recirculating water during the fouling study with MWW_NF when pH was controlled at 7.8

In the case of crystalline fouling, the scale-forming ions diffuse to the heated surface due to the concentration gradient where they react to form tenacious scale layer and cause fouling resistance. To verify this hypothesis, another fouling run with MWW_NF at CoC 4 was conducted at pH 6.8-7.2 and the results are shown in Figure 4.2.19. It is clear form this Figure that MWW_NF can exhibit a significant fouling potential when the solution pH is adjusted to lower levels. Corresponding Ca and phosphate concentrations in the bulk solution are depicted on Figure 4.2.20 and 4.2.21, respectively. Slow decline of both Ca and phosphate at lower pH supports the hypothesis of crystalline fouling on the heated surface under these conditions.

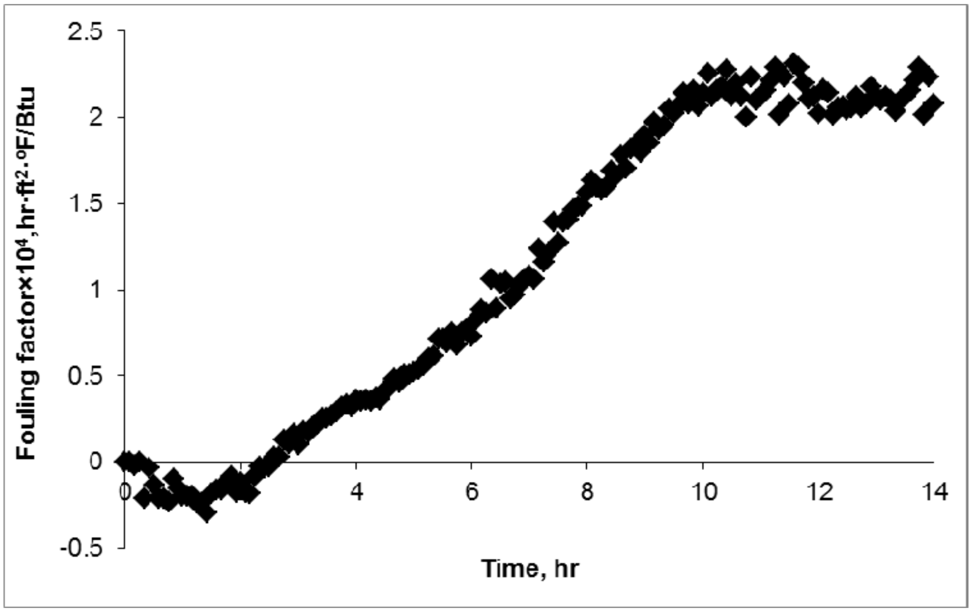


Figure 4.2.19 Fouling curve of the MWW_NF at CoC 4 with bulk pH controlled at 6.8-7.2

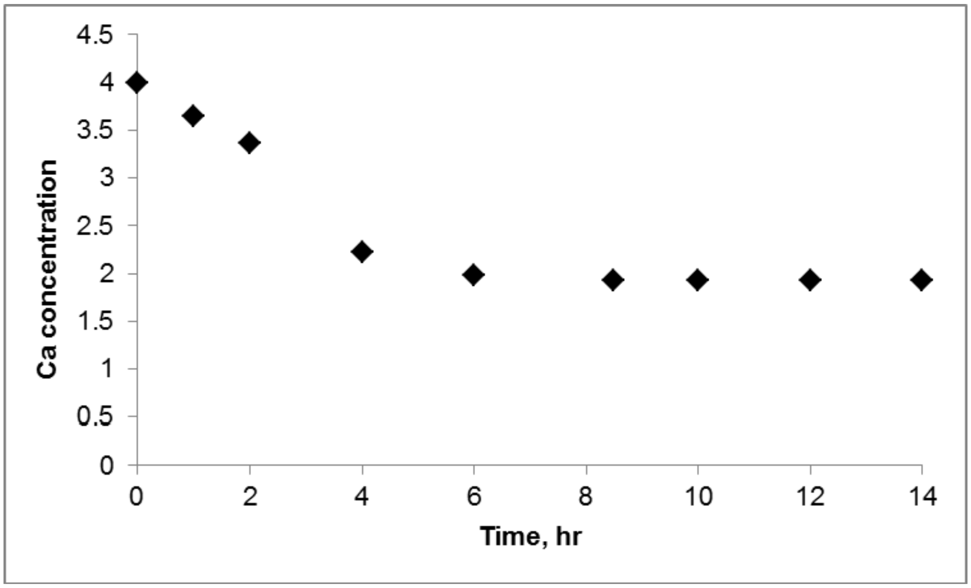


Figure 4.2.20 Calcium concentration in the recirculating water during the fouling study with MWW_NF when pH was controlled at 6.8-7.2

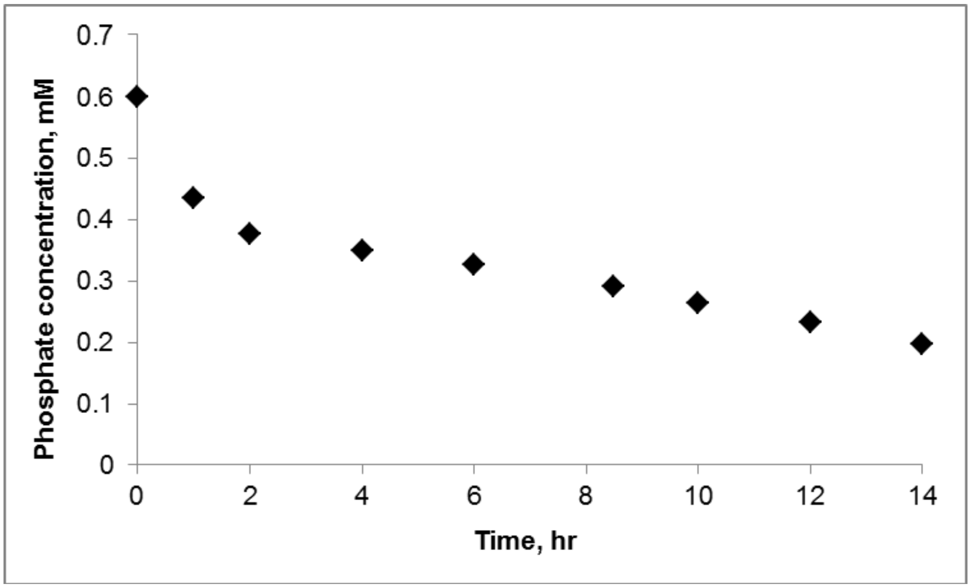


Figure 4.2.21 Phosphate concentration in the recirculating water during the fouling study with MWW_NF when pH was controlled at 6.8-7.2

Scaling rate indicated by the mass gain on stainless steel disc coupons in the fouling run with MWW_NF at pH 7.8 is shown in Figure 4.2.22. These results also indicate the mineral scaling propensity on the heated surface when the heated rod is placed horizontally.

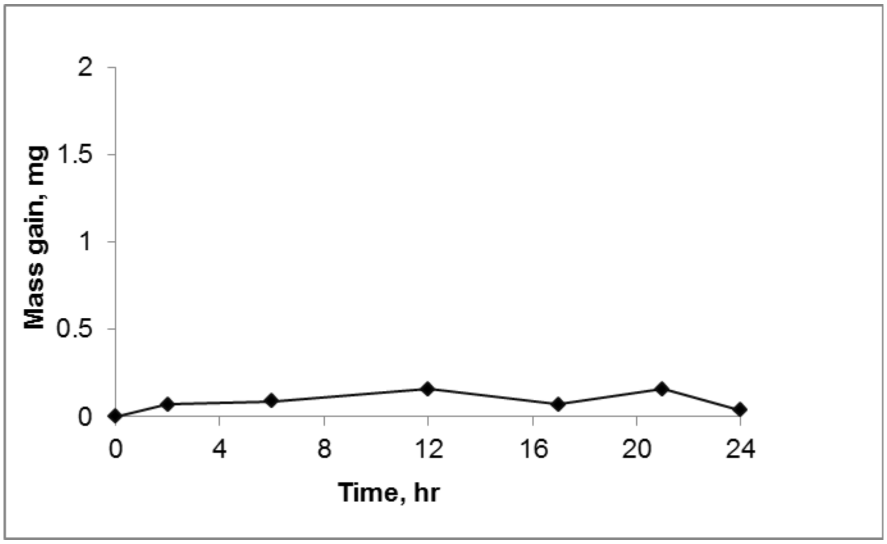


Figure 4.2.22 Mass gain on stainless steel disc coupons during the fouling study with MWW_NF when pH was controlled at 7.8

The scaling potential revealed by the mass gain method was similar to that indicated by the fouling curves where more surface deposits were observed at pH 7.8. As discussed above, when pH was maintained around 7.8, the scale-forming components were consumed by bulk precipitation and the only possibility for fouling development is through the attachment of the precipitated particles. Whether a particle actually touches the pipe wall is governed by short-range forces as well as geometry and surface properties of the pipe material. However, it is clear that an entrained particle must first negotiate the carrier fluid before the short-range forces can exert any influence (Drew, 1988). A study of the forces on a particle moving in a shearing fluid near the pipe wall would help to understand the fate of precipitated particle once they approach the pipe wall.

For small particles near the wall, the following forces appear as shown in Figure 4.2.23:

- Gravity force, F_G
- Buoyancy force, F_B
- Drag force of the flow, F_D
- Lift force, F_L

Among these forces, the drag force and the lift force are caused by the flowing water. The lift force accounts for the inertial forces on the particle due to the presence of the wall and exists in the direction normal to the wall when there is an external force on the sphere in the flow direction (Drew, 1988).

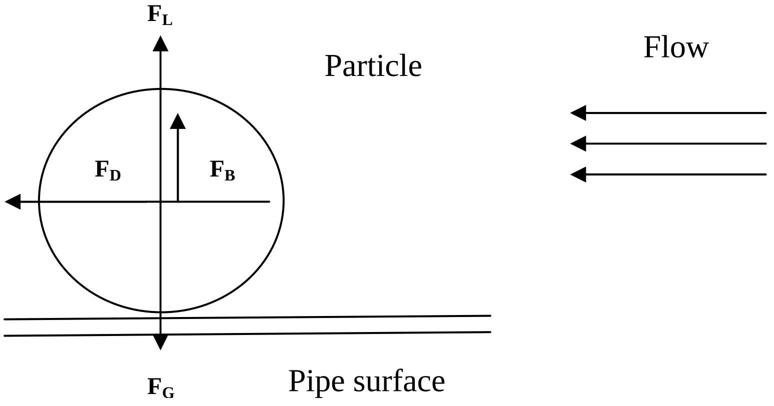


Figure 4.2.23 Forces acting on a single particle in a flowing stream

These forces can be calculated using the equations listed below.

Gravity force:

$$F_G = \rho_s g \times \frac{4}{3} \times \pi \times \left(\frac{1}{2} \times d_p\right)^3 = \frac{1}{6} \pi \rho_p g d_p^3 \qquad (4.2.1)$$

Where, ρ_s is the density of the particle,

d_p is the diameter of the particle

Buoyancy force:

$$F_B = \frac{1}{6}\pi\rho_L g d_p^3 \tag{4.2.2}$$

Where, ρ_L is the density of the water

Drag force (Altmann and Ripperger, 1997):

$$F_D = 3.16 \cdot \pi \cdot \tau_w \cdot d_p^2 \tag{4.2.3}$$

Where, τ_w is the shear stress,

Lift force (Drew, 1988):

$$F_L = 0.761 \cdot \frac{\tau_w^{1.5} \cdot d_p^3 \cdot \rho_L^{0.5}}{\mu} \tag{4.2.4}$$

Shear stress:

$$\tau_w = \frac{f\rho v^2}{8} \tag{4.2.5}$$

Where, f is the Dary friction factor

v is the mean flow velocity

The total force normal to the pipe wall:

$$F_{vertical} = \frac{1}{6}\pi(\rho_S - \rho_L)g d_p^3 - 0.761 \cdot \frac{\tau_w^{1.5} \cdot d_p^3 \cdot \rho_L^{0.5}}{\mu} \tag{4.2.6}$$

If $F_{vertical}$ is positive, then the particles will attach to the pipe surface. If $F_{vertical}$ is negative, there is no possibility for the particles to transport to the pipe surface. Qualitatively, for a certain particle, whether the particle moves toward or away from the pipe surface depends on the hydrodynamic conditions in the flowing stream. High flow rates could result in high shear stress (τ_w) and smaller $F_{vertical}$. Thus, high flow rates will be beneficial for the particle fouling mitigation.

Force calculation for various particle sizes (0.1~50 μm) under the test conditions used in this study (flow rate at 5 GPM, pipe diameter at 1 inch, and Reynolds number at 24,039) are summarized in Figure 4.2.24. The total vertical force for particle sizes included in this analysis was negative, thereby suggesting that these particles will not be able to attach to the pipe surface. Thus, the hydrodynamic conditions tested in the fouling study were favorable for the particle fouling prevention. However, if the flow rate of recirculating water is lowered to 3.5

GPM, the total vertical force on particles in the range of 0.1 - 50 µm was positive (Figure 4.2.25), which suggests that these particles may cause particulate fouling in the pipes.

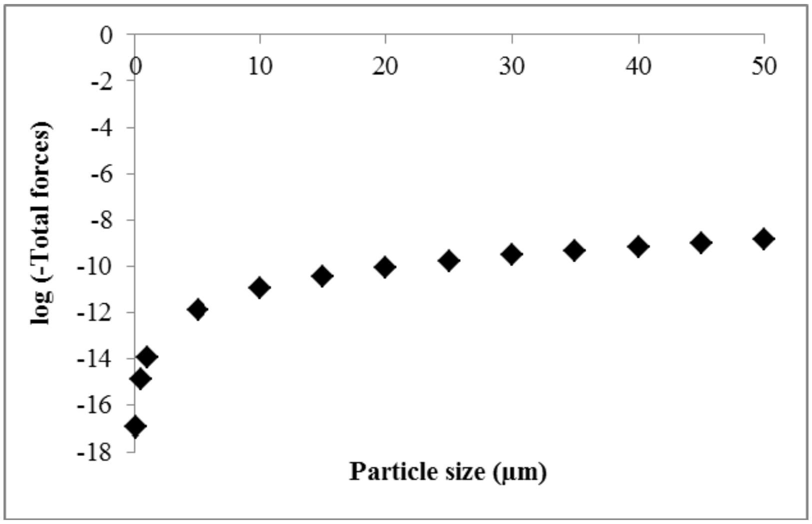


Figure 4.2.24 Total forces on different particle sizes under typical test conditions (flow rate at 5 GPM in 1 inch diameter pipe)

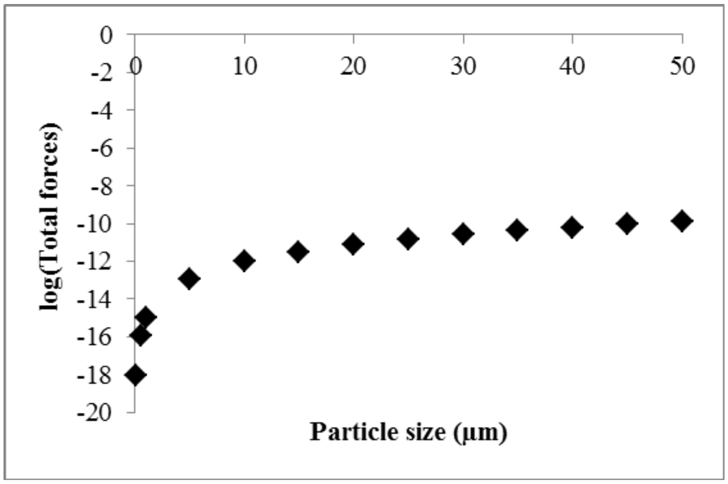


Figure 4.2.25 Total forces on different particle sizes (at 3.5 GPM flow rate through 1 inch diameter pipe)

4.3 Biofouling Control for MWW_NF Used for Cooling

4.3.1 Testing protocols

The objectives of this set of tests were to 1) compare the effectiveness of sodium hypochlorite (NaOCl), chlorine dioxide (ClO_2), and monochloramine (MCA) for biological growth control; and 2) determine the effectiveness of monochloramine for biofouling control in cooling towers using tertiary treated municipal wastewaters (MWW_NF) as makeup during a long-term operation. Differences among these three biocides were evaluated in batch reactor, bench-scale recirculating system, and pilot-scale cooling system.

Batch experiment is designed to evaluate the biological growth potential and biocide performance under static conditions. A 250-mL Erlenmeyer flask used as a batch reactor was covered with aluminum foil to minimize evaporative water loss and photo-degradation of biocide. Water temperature was controlled at 40 ºC to simulate heated recirculating water in the cooling system. Four different biocide dosages (0.5, 1, 2, 4 mg/L as Cl_2) were added into flasks 1 to 4 to evaluate the biocidal effectiveness in treated municipal wastewaters. The total length of batch experiment was controlled at 3 hours and biocide residual concentration and pH were measured throughout the experiment. Biological growth potential was determined based on the plate counts of heterotrophic bacteria cultured for planktonic bacteria counts at 30 and 120 minutes following the spread plate count method (Method 9215 C. Spread Plate Method, APHA, 1998). Biocidal effectiveness of each biocide was evaluated based on the biocide demand after 2-hour contact time, decay rate after 2-hour contact time, and its disinfection efficiency.

4.3.1.1 Biocide demand

Biocide residual and pH measurements were used to determine the biocide demand, which is then used to evaluate the biocide consumption under well-controlled laboratory conditions. The biocide demand refers to the biocide dose required to achieve a given biocide residual at a prescribed contact time, pH, and temperature. Chlorine/chloramine demand measurement followed the Standard Method 2350 B. Biocide demand/requirement. Chlorine dioxide demand measurement followed the Standard Method 2350 C. Chlorine Dioxide demand/requirement. The general equation for calculating biocide demand is:

$$\text{Biocide demand} = \text{Sample Dose} - (\text{Sample Residual} - \text{Interference}) - (\text{Blank Dose} - \text{Blank Residual})$$

4.3.1.2 Decay rate of selected biocides in different treated wastewaters

Biocide residuals measured with time are graphically presented to demonstrate the decay with different treated wastewaters at 23 and 40 ºC. When NaOCl is used as biocide, both free chlorine and total chlorine residuals were reported. In the case of MCA and chlorine dioxide, only monochloramine residual and chlorine dioxide residual were reported.

4.3.1.3 Disinfection efficiency of selected biocides

Planktonic heterotrophic bacteria counts in each water sample are normalized to initial values (t = 0 min) to calculate disinfection efficiency (%) of respective biocides. Disinfection efficiencies calculated in this fashion are analyzed as a function of water temperature, initial dosage, and contact time. For MWW and MWW_NF collected at FTMSA, the average initial planktonic heterotrophic plate counts were collected from batch experiments conducted in 2008. For MWW, average initial bacterial count was $10^{6.33} \pm 10^{6.60}$ CFU/mL. Therefore, the inactivation rate above 99.5% must be achieved to reach biofouling control criteria of 10^4 CFU/mL. As for MWW_NF, the average initial bacterial count was slightly lower ($10^{5.29} \pm 10^{5.41}$ CFU/mL) and an inactivation rate above 94.8% is required to reach the same biofouling control criteria.

The effectiveness of biocide in controlling biofouling was further tested in a bench-scale recirculating system designed to simulate temperature, flow velocity and water quality similar to those in a full-scale recirculating cooling system. The total duration of each experiment was 72 hours with continuous biocide dosing to maintain the desired residual. Planktonic HPCs were cultured at 4, 8, 12, 24, 48, and 72 hours. Sessile HPCs were cultured at 12, 24, 48, and 72 hours.

Results from the previous field studies (Vidic et al., 2009) suggested that monochloramine can be used as an effective biocide against planktonic and sessile bacteria growth in cooling tower systems with MWW_NF and MWW during a 30-day test. Preliminary laboratory studies also supported that biological growth can be restrained by adding at least 4 ppm of free chlorine, 2 ppm of ClO_2, or 4 ppm of pre-formed monochloramine for MWW_NF. Therefore, NaOCl, ClO_2, and pre-formed MCA were tested as the primary biocides for biological growth control with MWW_NF as cooling system make up water during extended pilot-scale tests. In the first test with pre-formed MCA and MWW_NF, the monochloramine residual was maintained in the range of 2 to 3 ppm as Cl_2 in all three cooling systems (CTA1, CTB1, and CTC1) for 60 days (Table 2.2.1). In the second test with NaOCl and ClO_2 and MWW_NF, the total chlorine residual was maintained in the range of 3 to 4 ppm as total chlorine in CTA3 and ClO_2 residual was maintained in the range of 0.5 to 1 mg/L as ClO_2 in CTB3 (Table 2.2.3).

4.3.2 Batch studies with MWW_NF

4.3.2.1 Chlorination for biological growth control

Table 4.3.1 and Table 4.3.2 show the biocide demand of raw (CoC 1) secondary treated wastewater followed by nitrification and sand filtration (MWW_NF) at 23 and 40 ºC, respectively. Ammonia concentration measured in the MWW_NF was 3.4 ppm as N for both tests. Tests conducted with MWW_NF at 23 ºC showed the same behavior as tests with MWW, where the formation of chloramines governed free chlorine and total chlorine demand. Free chlorine demand of MWW_NF at 40 ºC appears to be similar to the total chlorine demand. Results shown in Table 4.3.1 and Table 4.3.2 indicate that the initial free chlorine dosage above 4 ppm is required to achieve free chlorine residual in MWW_NF at either 23. When the water temperature increased to 40 °, it is expected to have much higher initial dose to have detectable free chlorine residual after contact time of 2 hours. On the other hand, to maintain 0.5 ppm total chlorine residual, an initial dosage between 2 and 4 ppm is required for MWW_NF at 23ºC. However, if the water temperature is above 40 ºC, even 4 ppm initial free chlorine dose was not able to achieve total chlorine residual above 0.5 ppm as Cl_2.

Table 4.3.1 Biocide demand after 2 hour contact time with secondary treated municipal wastewater followed by nitrification and sand filtration at 23 ºC

Dose, ppm	pH	Free chlorine residual, ppm	Total chlorine residual, ppm	Free chlorine demand, ppm	Total chlorine demand, ppm
0.5	6.31	0.00	0.00	0.41	0.44
1	6.32	0.00	0.00	0.91	0.94
2	6.36	0.00	0.17	1.91	1.77
4	6.39	0.04	1.48	3.87	2.46

Table 4.3.2 Biocide demand after 2 hour contact time with secondary treated municipal wastewater followed by nitrification and sand filtration at 40 ºC

Dose, ppm	pH	Free chlorine residual, ppm	Total chlorine residual, ppm	Free chlorine demand, ppm	Total chlorine demand, ppm
0.5	6.26	0.00	0.00	0.27	0.35
1	6.31	0.00	0.00	0.77	0.85
2	6.32	0.00	0.00	1.77	1.85
4	6.32	0.00	0.22	3.77	3.63

Figure 4.3.1 shows the decay of NaOCl in MWW_NF at 23 and 40 ºC. It is evident that free chlorine was mostly consumed within 10 minutes after addition to MWW_NF. Due to the low organic matter and ammonia in MWW_NF, only a small portion of free chlorine was converted into organic chloramine. With the increase in temperature, the conversion still occurred but the decay rates of both free chlorine and total chlorine increased dramatically.

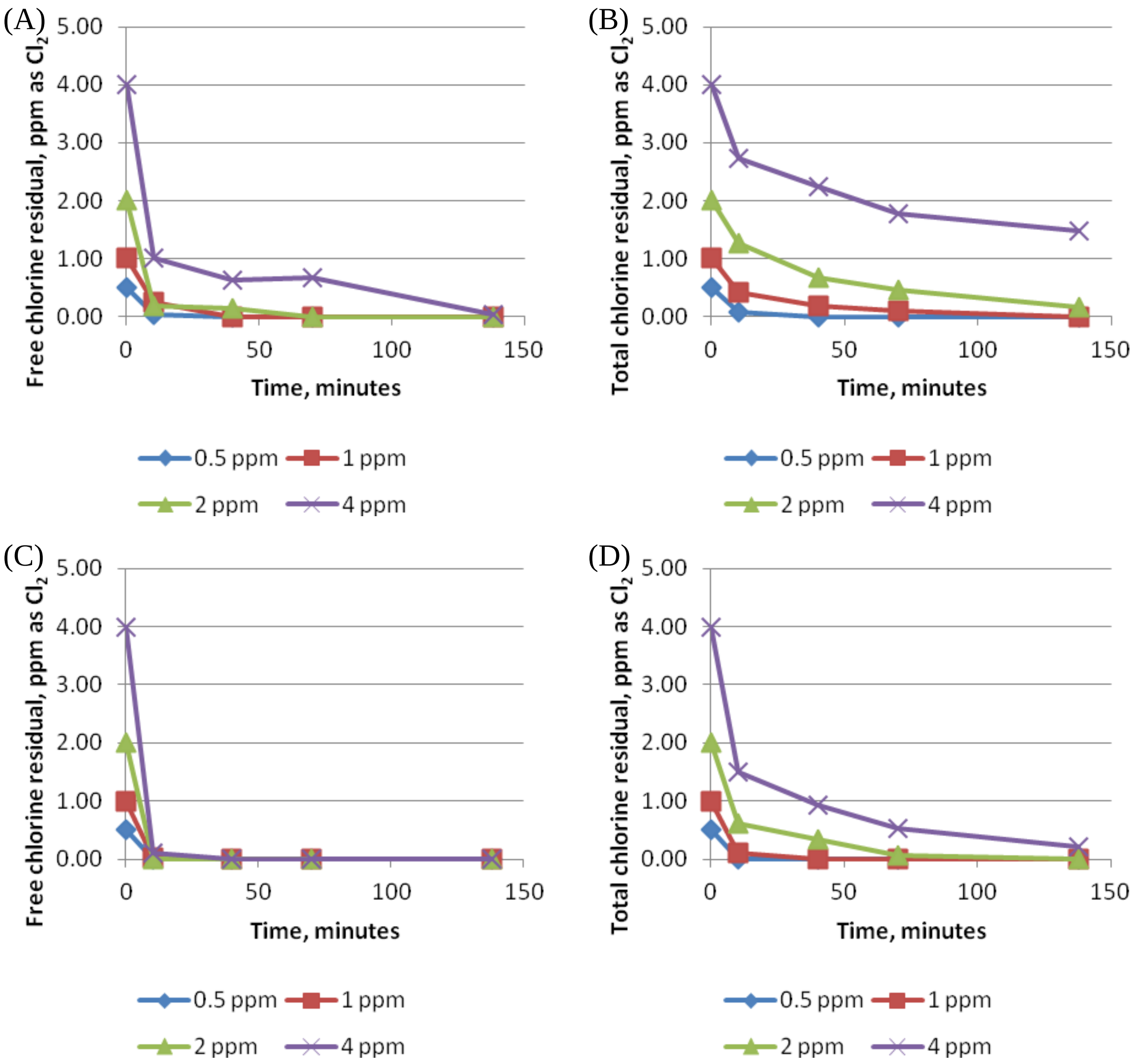


Figure 4.3.1 Decay of NaOCl in secondary treated municipal wastewater plus nitrification and sand filtration. (A) and (B) show the decay trend of free chlorine and total chlorine in MWW_NF at 23 ºC; (C) and (D) show the decay trend of free chlorine and total chlorine in MWW_NF at 40 ºC.

Table 4.3.3 shows the disinfection efficiency of NaOCl in MWW_NF. For MWW_NF at room temperature, initial dosage of 0.5 ppm NaOCl is required to maintain the planktonic bacteria level below 10^4 CFU/mL within a contact time of 2 hours. For MWW_NF at 40 ºC, initial dosage of 0.5 ppm NaOCl is required to maintain the planktonic bacteria level below 10^4 CFU/mL within a contact time of 2 hours.

Table 4.3.3 Results of disinfection efficiency of NaOCl in treating MWW_NF

Dosage, ppm	23 °C_30 mins	23 °C_2 hrs	40 °C_30 mins	40 °C_2 hrs
0.5	96.00%	98.10%	98.12%	99.67%
1	97.65%	99.77%	99.97%	99.99%
2	98.95%	99.91%	99.94%	99.99%
4	99.16%	99.92%	99.99%	99.99%

Despite the low ammonia concentration in MWW_NF, free chlorine was still mostly consumed by organic matter. Low ammonia also contributed to low combined chlorine residual in these samples. As a result, biocide demand in MWW_NF was higher than that in MWW. Although the results indicate that both free and total chlorine residual decreased dramatically within 10 minutes, an initial free chlorine dosage below 1 ppm was effective in controlling biomass growth.

4.3.2.2 Chloramination for biological growth control

Monochloramine demand in MWW_NF shown in Table 4.3.4 indicates that the initial dosage of 2 ppm could achieve a residual above 0.5 ppm as Cl_2 at 23 °C after 2 hours of contact time. Similar conclusion can be made for the test conducted at 40 °C. Figure 4.3.2 that shows the decay of MCA in MWW_NF at 23 and 40 °C indicates that MCA is relatively stable in MWW_NF at both 23 and 40 °C.

Table 4.3.4 Monochloramine demand after 2-hour contact time with secondary treated municipal wastewater after nitrification and sand filtration (MWW_NF) at 23 and 40 °C.

Dose, ppm	At 23 °C			At 40 °C		
	pH	Monochloramine residual, ppm	Monochloramine demand, ppm	pH	Monochloramine residual, ppm	Monochloramine demand, ppm
0.5	7.00	0.11	0.33	8.00	0.09	0.11
1	7.00	0.23	0.71	8.31	0.17	0.54
2	7.00	0.70	1.24	8.80	0.85	0.86
4	7.01	1.86	2.08	9.39	2.46	1.25

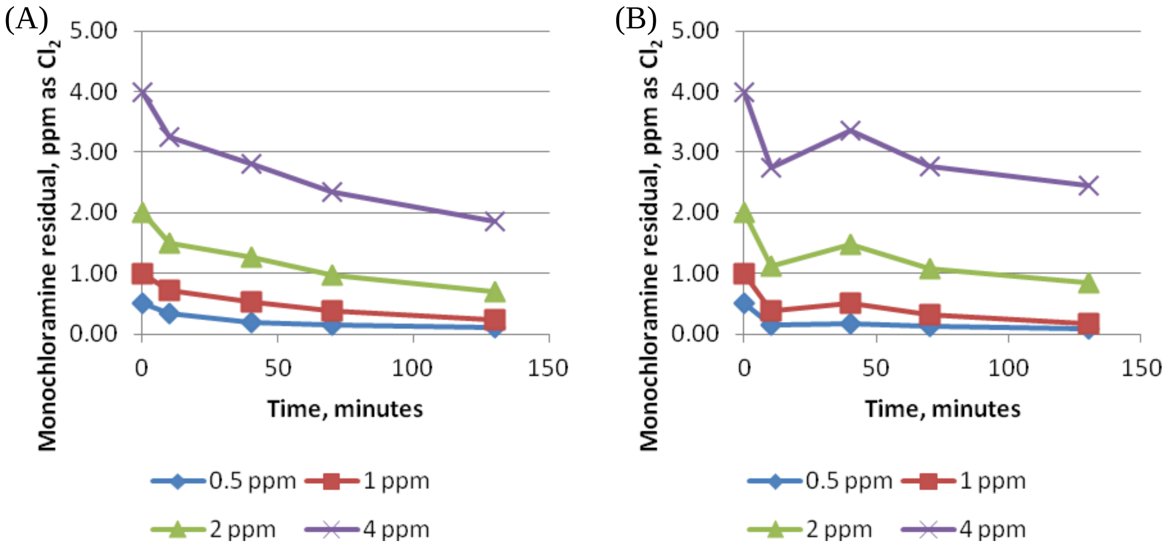


Figure 4.3.2 Decay of MCA in secondary treated municipal wastewater after nitrification and sand filtration. (A) MCA decay in MWW_NF at 23 ºC; (B) MCA decay in MWW_NF at 40 ºC.

Table 4.3.5 shows the disinfection efficiency of MCA in MWW_NF. At room temperature, an initial dosage of 4 ppm MCA is required to keep the planktonic bacteria level below 10^4 CFU/mL with a contact time of 2 hours. For MWW_NF at 40 ºC, an initial dosage of 0.5 ppm MCA is required to maintain the planktonic bacteria level below 10^4 CFU/mL with contact time of 2 hours. Temperature increase greatly improves the HPC inactivated rate in MWW_NF with MCA.

Table 4.3.5 Results of disinfection efficiency of MCA in treating secondary treated municipal wastewater plus nitrification and sand filtration (MWW_NF)

Dosage, ppm	23 ºC_30 mins	23 ºC_2 hrs	40 ºC_30 mins	40 ºC_2 hrs
0.5	50.29%	66.38%	91.98%	99.40%
1	40.72%	82.61%	99.98%	99.98%
2	83.33%	92.14%	99.92%	99.98%
4	93.22%	98.93%	99.71%	99.95%

The data in table above suggest that the required disinfection with MCA residual can be achieved by adding at least 2 ppm of preformed MCA in MWW_NF. Furthermore, the increase in temperature significantly enhanced the disinfection efficiency of MCA in MWW_NF.

4.3.2.3 Chlorine dioxide for biological growth control

The results of chlorine dioxide decay in MWW_NF shown in Table 4.3.6 suggest that even the initial dosage of chlorine dioxide of 4 mg/L could not achieve the residual above 0.5 ppm at 23 or 40 °C after 2 hours of contact time. Figure 4.3.3 shows the decay of ClO_2 in MWW_NF at 23 and 40 ºC. Results are similar to those in MWW. It is shown that ClO_2 residual was consumed within 10 minutes after addition regardless of dosing concentrations and water temperatures. An initial dosage higher than 4 ppm may be required to achieve reasonable residual.

Table 4.3.6 Chlorine dioxide demand after 2 hour contact time with secondary treated municipal wastewater after nitrification and sand filtration (MWW_NF) at 23 ºC and 40 ºC.

Dose, ppm	At 23 ºC			At 40 ºC		
	pH	Chlorine dioxide residual, ppm	Chlorine dioxide demand, ppm	pH	Chlorine dioxide residual, ppm	Chlorine dioxide demand, ppm
0.5	7.31	0.02	0.47	7.64	0.07	0.20
1	7.33	0.00	0.94	7.60	0.14	0.63
2	7.25	0.06	1.88	7.50	0.12	1.65
4	7.10	0.09	3.85	7.31	0.13	3.64

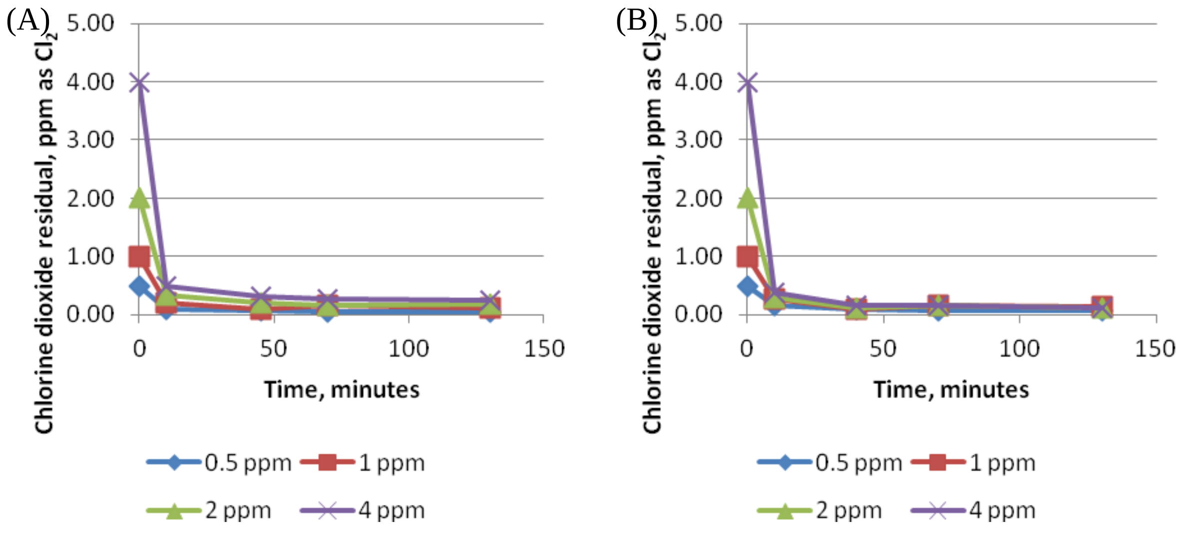


Figure 4.3.3 Decay of ClO_2 in secondary treated municipal wastewater after nitrification and sand filtration (MWW_NF): (A) 23 ºC; (B) 40 ºC.

Disinfection efficiency of ClO_2 in MWW_NF summarized in Table 4.3.7 indicates that the initial chlorine dioxide dosage of 1 ppm is required to reach at least 94.5% inactivation after 2 hours at room temperature. When water temperature increased to 40 °C, the initial ClO_2 below 1 ppm is required to maintain the planktonic bacteria level below 10^4 CFU/mL with contact time of 2 hours (Table 4.3.7). However, none of the initial dosages evaluated in this study could provide sufficient chlorine dioxide residual (0.5-1.0 ppm) in MWW_NF. Higher water temperature enhanced the disinfection efficiency of ClO_2 in MWW_NF.

Table 4.3.7 Results of disinfection efficiency of ClO_2 in treating secondary treated municipal wastewater plus nitrification and filtration.

Dosage, ppm	23 °C_30 mins	23 °C_2 hrs	40 °C_30 mins	40 °C_2 hrs
0.5	68.60%	85.03%	81.12%	50.90%
1	96.13%	97.84%	96.18%	99.19%
2	99.83%	99.88%	99.20%	99.78%
4	99.97%	99.98%	99.82%	99.94%

4.3.3 Bench-scale recirculating studies with MWW_NF

4.3.3.1 Biofouling potential test

Control studies of biofouling potential with CoC 4 MWW_NF were conducted without any biocide addition to validate the biological growth under the impact of hydrodynamic forces. Results of the control test shown in Figure 4.3.4 suggest that the planktonic bacterial population reached exponential growth phase and that both planktonic and sessile bacterial populations exceeded the biofouling control criterion (10^4 CFU/mL or 10^4 CFU/cm^2) within 24 hours.

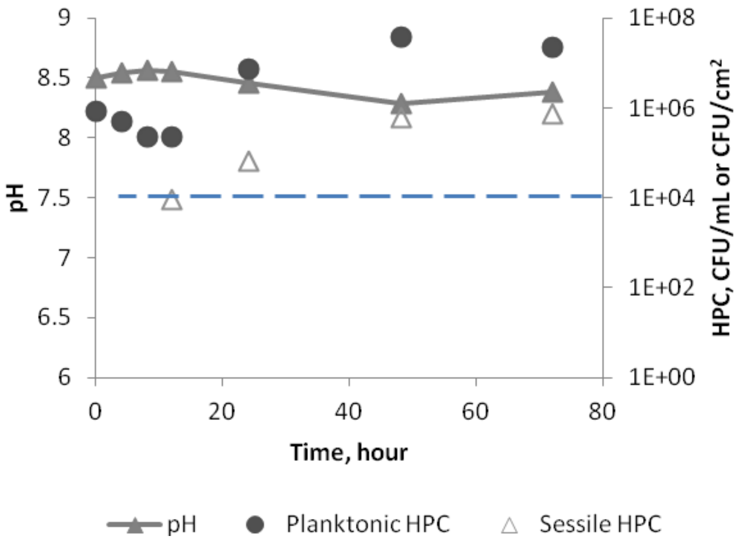


Figure 4.3.4 Planktonic and sessile heterotrophic bacteria growth in CoC 4 MWW_NF in bench scale recirculating system.

4.3.3.1 Chlorination for biological growth control

Planktonic and sessile bacterial activity in the presence of the total chlorine residual between 2-3 ppm as Cl_2 in recirculating system is shown in Figure 4.3.5. The pH in the recirculating system was also measured to validate the major forms of chlorine (combined chlorine) residual (Figure 4.3.6). Since the initial inorganic ammonia was 0.09 ppm as NH_3 in MWW_NF, monochloramine was only detected in the system during the very early stages of the test (i.e., until the breakpoint chlorination was achieved after 8 hours). The total chlorine residual averaged 2.76 ± 0.83 ppm as Cl_2 with free chlorine residual accounting for 1.27 ± 0.75 ppm as Cl_2 after that initial stage. In the rest of the experiment, the free chlorine residual accounted for 44% of measured total chlorine residual. Planktonic heterotrophic bacteria counts decreased below 10^4 CFU/mL after four hours and remained at low levels until the end of the experiment. The analysis of sessile HPC indicated that biofilm formation was reasonably well controlled in the presence of 2~3 ppm of total chlorine residual from periodic NaOCl addition. These results indicate that maintaining 2~3 ppm of total chlorine residual in MWW_NF can control both planktonic and sessile heterotrophic bacteria in the recirculating system. The dosing rate required to maintain 2~3 mg/L of total chlorine residual in CoC 4 MWW_NF was 1.10 ppm per hour.

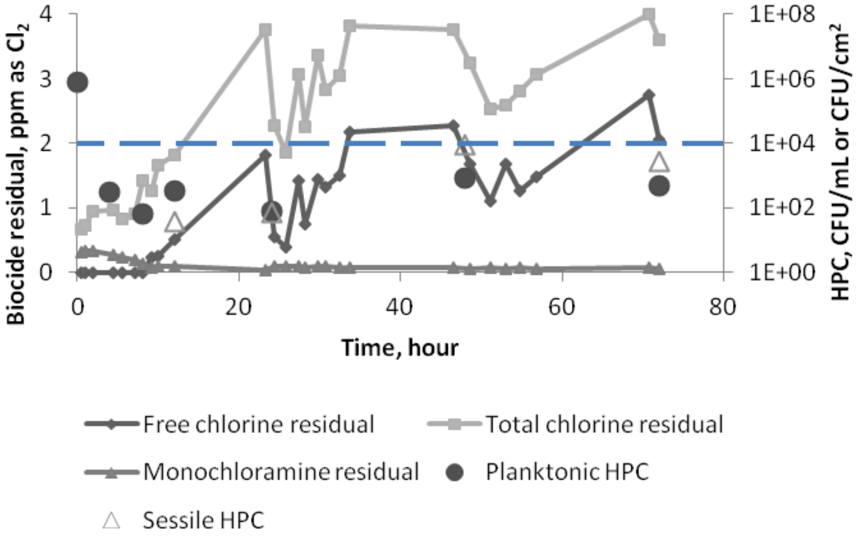


Figure 4.3.5 Total chlorine, free chlorine, and monochloramine residuals and heterotrophic bacteria counts in bench scale recirculating system with CoC 4 MWW_NF treated with sodium hypochlorite

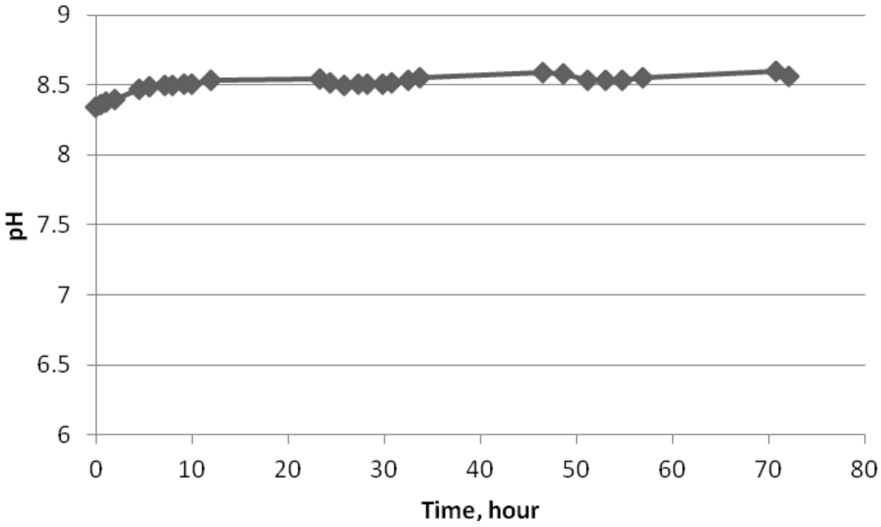


Figure 4.3.6 pH versus time in the bench scale recirculating system with CoC 4 MWW_NF treated with sodium hypochlorite.

4.3.3.2 Chloramination for biological growth control

Planktonic and sessile bacterial activity in the presence of monochloramine residual ranging between 2-3 ppm as Cl_2 in the recirculating system is shown in Figure 4.3.7 while the pH in the recirculating system is depicted on Figure 4.3.8. Monochloramine residual (2.29 ± 0.42 ppm of MCA) dominated the total chlorine residual (2.77 ± 0.50 ppm of total chlorine as Cl_2) and accounted for approximately 83% of the total chlorine residual. Throughout the experiment, both planktonic and sessile bacteria populations were maintained below the control criteria of 10^4 CFU/mL or 10^4 CFU/cm^2, respectively. These results suggest that maintaining 2 ~ 3 ppm of monochloramine residual successfully controlled biological growth with dosing rate of 0.59 ppm per hour in CoC 4 MWW_NF.

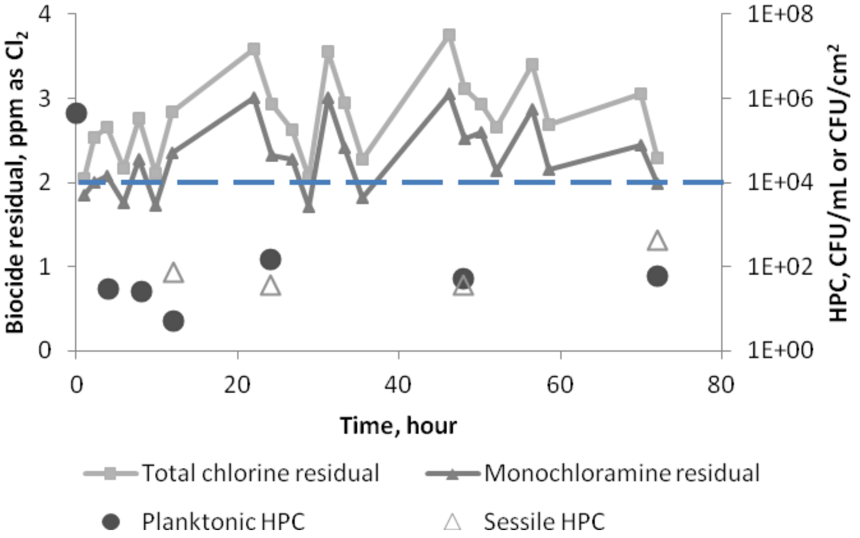


Figure 4.3.7 Total chlorine and monochloramine residuals and heterotrophic bacteria counts in bench scale recirculating system with CoC 4 MWW_NF treated with pre-formed monochloramine

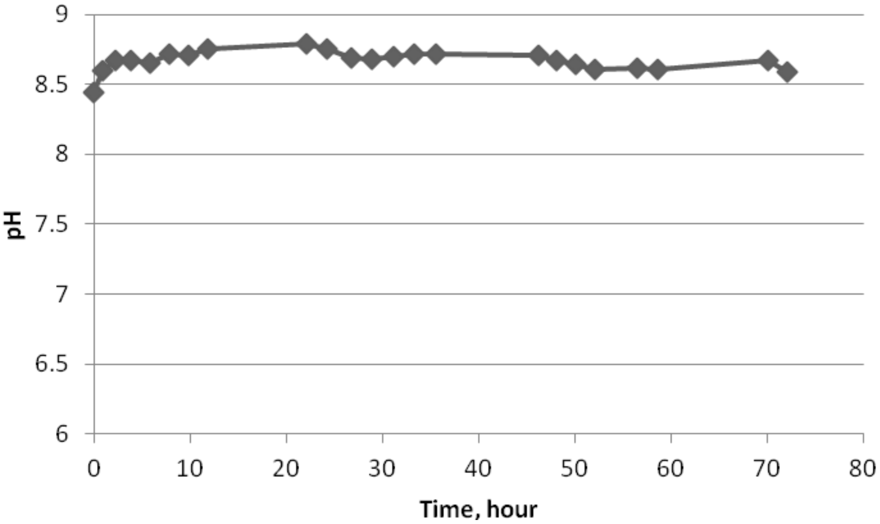


Figure 4.3.8 pH versus time in the bench scale recirculating system with CoC 4 MWW_NF treated with pre-formed monochloramine

4.3.3.3 Chlorine dioxide for biological growth control

Planktonic and sessile bacteria activity in the presence of chlorine dioxide residual between 0.25-0.5 ppm as Cl_2 in recirculating system is shown in Figure 4.3.9, while pH in the recirculating system is depicted on Figure 4.3.10. These results suggest that maintaining chlorine dioxide residual between 0.25-0.5 ppm can control both planktonic and sessile heterotrophic bacteria in the recirculating system. The dosing rate required to maintain this chlorine dioxide residual in CoC 4 MWW_NF was 1.04 ppm per hour.

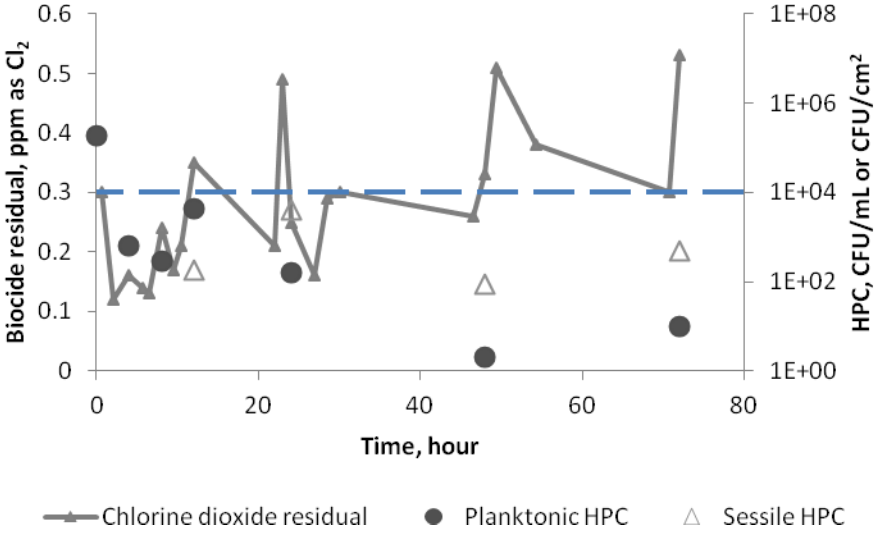


Figure 4.3.9 Chlorine dioxide residuals and heterotrophic bacteria counts in bench scale recirculating system with CoC 4 MWW_NF treated with chlorine dioxide

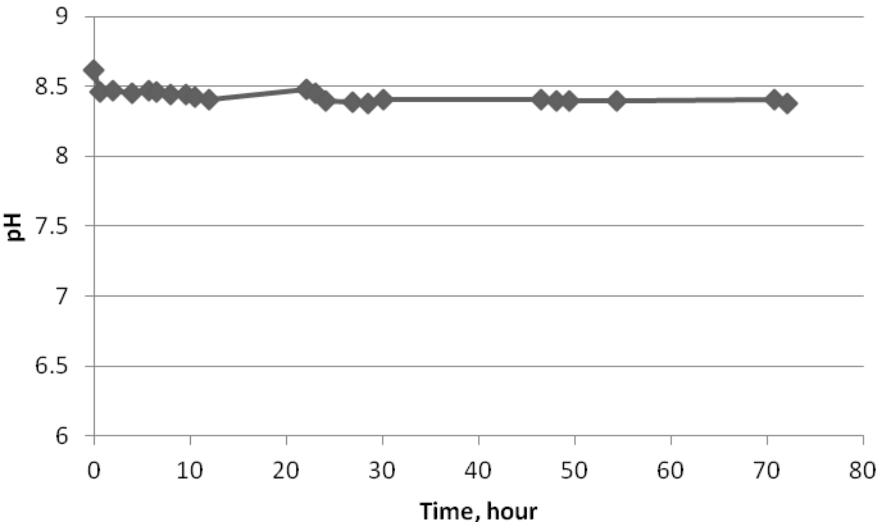


Figure 4.3.10 pH versus time in the bench scale recirculating system with CoC 4 MWW_NF treated with chlorine dioxide

4.3.4 Pilot-scale studies with MWW_NF

The objective of the pilot-scale experiments was to confirm the results observed from bench scale studies. The pilot scale experiment focused on the biocidal effectiveness and optimal dosages of selected biofouling control agents, namely sodium hypochlorite (NaOCl), monochloramine (MCA), and chlorine dioxide (ClO_2), when treating MWW_NF as cooling system makeup water.

Tables 2.3.1 and 2.3.3 show the testing program for biofouling control with these three chlorine-based biocides. Laboratory studies suggested that biological growth in tertiary treated municipal wastewater (MWW_NF) could be controlled by maintaining 1-2 ppm of free chlorine residual, 2-3 ppm of pre-formed monochloramine residual, or 0.25-0.5 ppm of chlorine dioxide residual.

Planktonic heterotrophic bacteria in the recirculating water and in the makeup water were measured every 3-4 days during the field tests. Biocides were stored separately in 10 gallon low density polyethylene tanks fully covered with aluminum foil. Free chlorine stock solution was prepared by diluting 5% sodium hypochlorite to 1,000 mg/L. Monochloramine stock solution was pre-formed by mixing sodium hypochlorite and ammonia (4:1 weight ratio) at pH 9. This solution was then diluted to 1,000 ppm monochloramine as Cl_2 and used in the field tests. Chlorine dioxide stock solution was prepared using a small-scale chlorine dioxide generator (Envirox H1000SRE, Nalco Company, Naperville, IL). Chlorine dioxide concentration in the stock solution varied from 300 and 500 ppm depending on the quality of feed chemicals. All three biocides were continuously injected into the recirculating system using a diaphragm liquid metering pumps.

4.3.4.1 Chlorination for biological growth control

Biofouling control data for CTA3 test are shown in Figure 4.3.11. Free chlorine residual in this test was maintained at 1.99 ± 1.80 ppm as Cl_2, while total chlorine and monochloramine residuals were 5.17 ± 1.94 and 0.09 ± 0.02 mg/L as Cl_2, respectively. It is important to note that free chlorine residual accounted for only 32% of total chlorine residual. Planktonic HPC in both makeup water and recirculating water were below the target criterion, 10^4 CFU/mL throughout the test. However, 10-day and 28-day sessile samples were between 10^4 and 10^5 CFU/cm^2. Although these measurements exceeded the initial target biofilm growth control criterion, they were within the 10^5 CFU/cm^2 limit recently proposed by the Cooling Tower Institute (CTI, 2008). A repeated sessile sample that was collected on the coupon installed in the port which was initially occupied by the coupon taken after three days of contact with recirculating water showed much lower biofilm density.

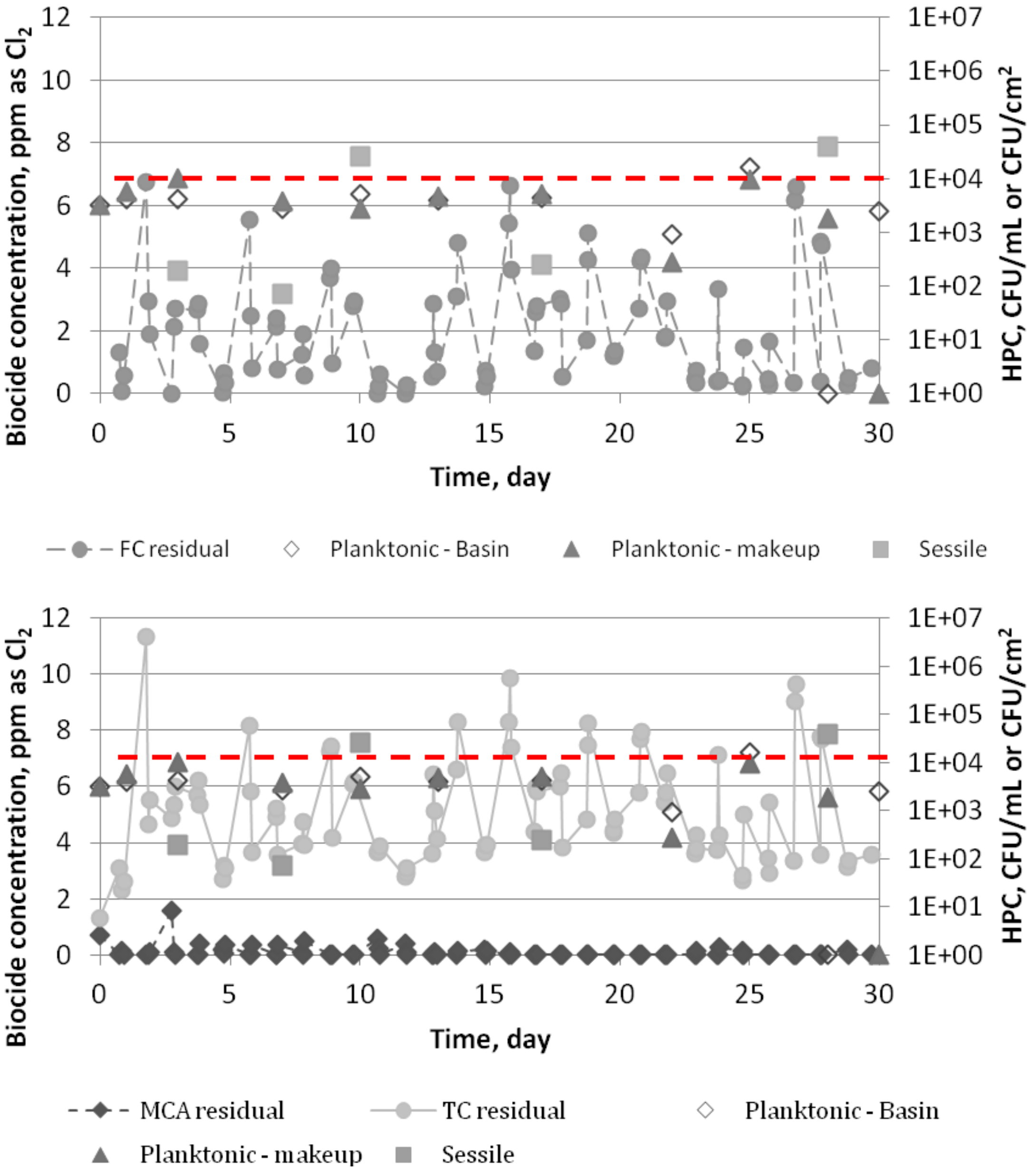


Figure 4.3.11 Biocide residual and HPCs in CTA3 test with free chlorine and MWW_NF. Dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2

4.3.4.2 Chloramination for biological growth control

Target chloramine concentration for pilot-scale tests with MWW_NF was 2 to 3 ppm as Cl_2 in all three towers. Biocide residual analysis during CTA1 test shown in Figure 4.3.12 confirmed that monochloramine residual averaged 2.43 ± 1.18 ppm as Cl_2 during 56 days of the pilot-scale test. During the first 28 days, monochloramine residuals were never below 1 ppm as Cl_2 and the planktonic HPC in the recirculating water was mostly below the target criterion of 10^4 CFU/mL except the sample on Day 15 (Figure 4.X12). However, analysis of sessile HPC samples indicates that the biofilm formation was not well controlled as it steadily increased to above 10^4 CFU/cm^2 after 28 days. In the second half of the experiment, as the monochloramine residual decreased below 2 ppm as Cl_2, planktonic HPC reached 10^5 CFU/mL on Day 30. Although monochloramine residual was maintained above 2 ppm or higher most of the time during the last 28 days of this test, none of the planktonic HPC results during this period were below the biofouling control criteria. The 56-day sessile HPC result also demonstrated that the sessile biological growth formation in CTA1 was not reversed and continued to grow since Day 28 and finally reached 10^6 CFU/cm^2.

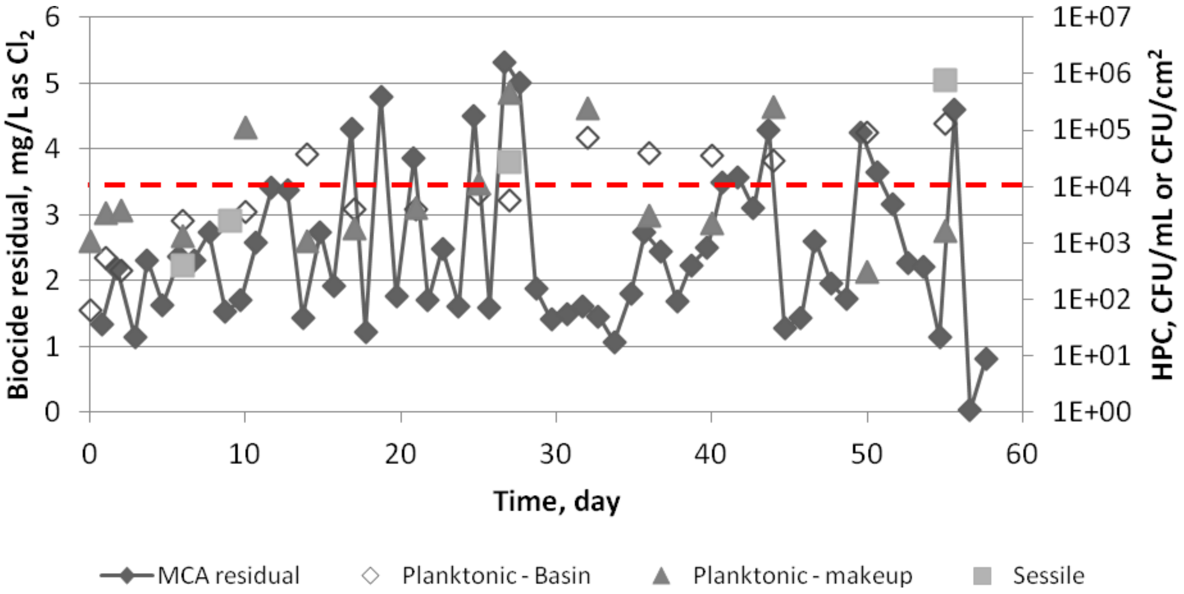


Figure 4.3.12 Biocide residual and HPCs in CTA1 test with monochloramine and MWW_NF. Dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2.

Monochloramine residual and HPC results in CTB1 test are shown in Figure 4.3.13. In this test, monochloramine residual averaged 1.76 ± 1.05 ppm as Cl_2 for 56 days. Due to the mechanical failure of the liquid metering pump, the monochloramine residual did not reach 2 ppm as Cl_2 until Day 4. After maintaining monochloramine the residual above 2 ppm for 2 consecutive days, the biocide residual again decreased to below 0.5 ppm on Day 6. As a result, the planktonic HPC in the recirculating water and first sessile HPC on Day 7 exceeded target criterion of 10^4 CFU/mL and 10^4 CFU/cm^2. After that, none of the planktonic and sessile HPC samples were below the biofouling control criteria until Day 50. It was observed that the planktonic HPC in the makeup water on Day 50 was only 10^3 CFU/mL, which was much lower than the average planktonic HPC in makeup water of 10^5 CFU/mL, resulting in lower bacterial counts. Liquid metering pump was replaced on Day 48 and monochloramine residual reached above 1 ppm and during the last 5 days of the test was above 1.5 ppm as Cl_2. During this period, both planktonic HPC and sessile HPC decreased but were still above the biofouling control criteria.

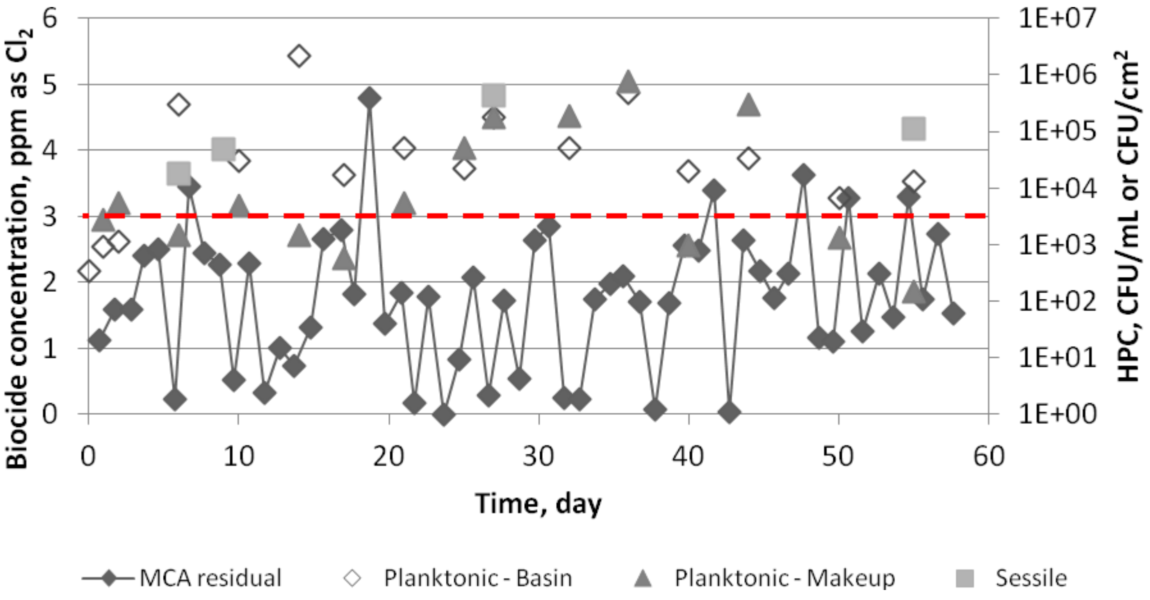


Figure 4.3.13 Biocide residual and HPCs in CTB1 test with monochloramine and MWW_NF. Dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2.

Results for CTC1 are shown in Figure 4.3.14. In CTC1, monochloramine residual was maintained at 2.84 ± 1.15 ppm as Cl_2 for 56 days. Monochloramine residual was well maintained above 2 ppm throughout the 56 days except on Day 23, 46, 50, 56 and 58. Most planktonic HPC results were below the biofouling control criteria. The only two results beyond the control criteria on Day 24 and Day 50 were caused by extremely low monochloramine residual on the sampling day or the day before sampling day. It was observed that all four sessile HPC were below the biofouling control criteria. However, the grow curve of the sessile HPC results demonstrates that the biofilm formation may surpass the biofouling control criteria after 56 days.

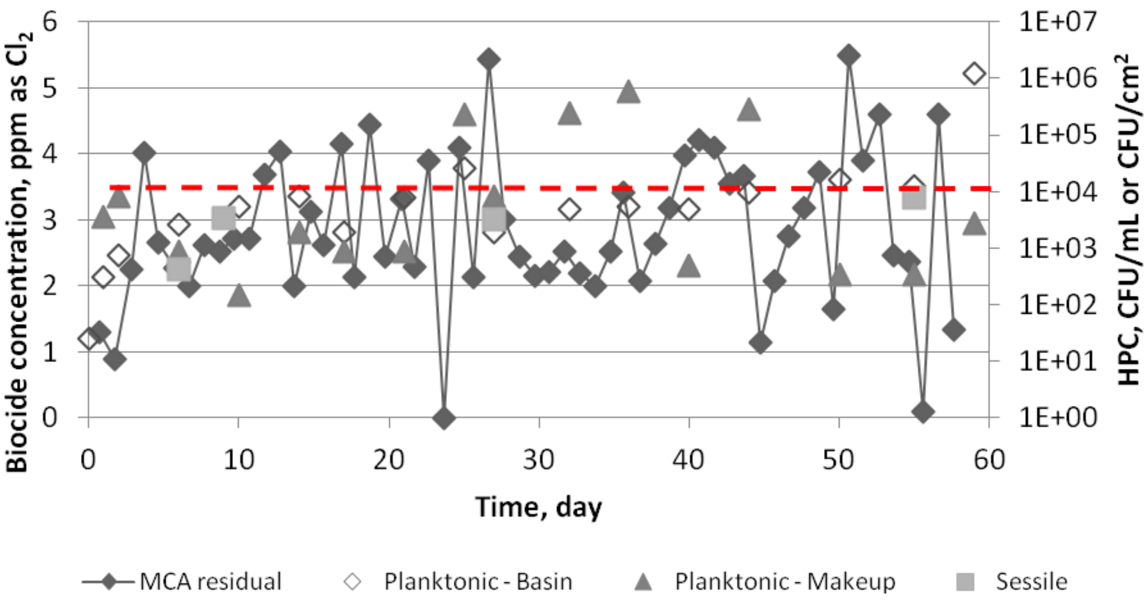


Figure 4.3.14 Biocide residual and HPCs in CTC1 test with monochloramine and MWW_NF. Dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2.

4.3.4.3 Chlorine dioxide for biological growth control

Biofouling control results in CTB3 test are shown in Figure 4.3.15. Chlorine dioxide residual was maintained at 0.41 ± 0.16 ppm as Cl_2 throughout the experiment, which was lower than the target range of 0.5 - 1.0 ppm due to variation in chlorine dioxide concentration in the stock solution. The chlorine dioxide residual reached above 0.5 mg/L after Day 23. Since the makeup water was treated with pre-formed monochloramine, a low concentration of monochloramine residual in the recirculating cooling systems was expected. It was observed that monochloramine residual was 0.42 ± 0.23 mg/L as Cl_2. Although the chlorine dioxide residual was not maintained within the bounds of the target criteria, the planktonic HPC in the recirculating water was consistently below the target criterion of 10^4 CFU/mL. The sessile sample taken on Day 10 was the only sessile sample that exceeded 10^4 CFU/cm^2.

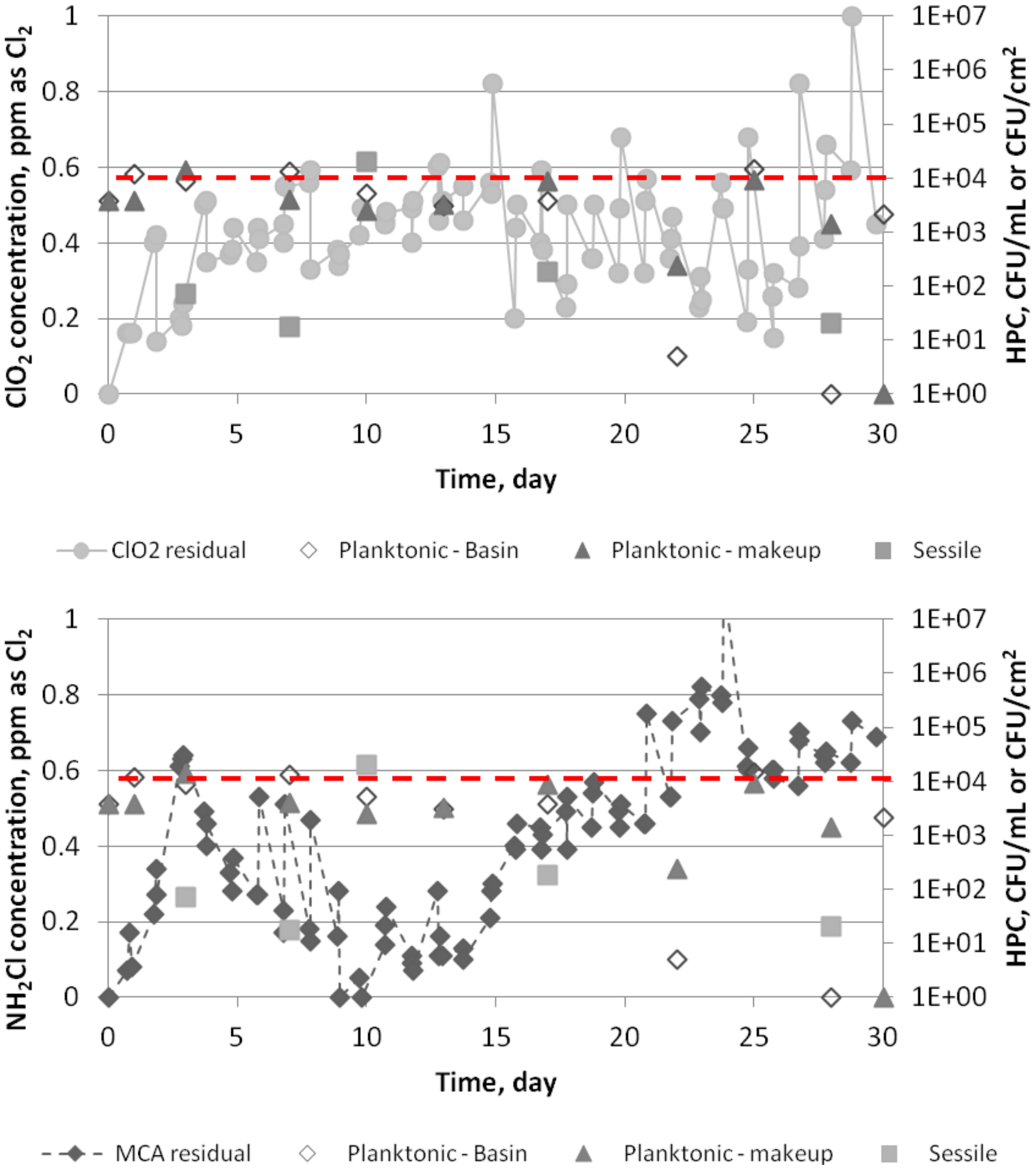


Figure 4.3.15 Biocide residual and HPCs in CTB3 test with chlorine dioxide and MWW_NF. Dashed line indicates the biofouling control criteria, 10^4 CFU/ml or 10^4 CFU/cm^2.

4.3.4.4 Biocide usage in MWW_NF tests

Biocide usage during of MWW_NF tests in 2010 and 2011 are summarized in Table 4.3.8. The daily makeup water rate for CTA1, CTB1, CTC1, CTA3, and CTC3 were 43.31, 50.16, 45.72, 55.30, and 54.11 gal/day, respectively.A total of 3.5 lb of pre-formed MCA was required to achieve the best biofouling control in the pilot-scale cooling systems when using monochloramine to treat cooling towers using MWW_NF as tower makeup water. Although test

CTA1 had similar biocide consumption rate, the higher variation of MCA concentration in stock solution resulted in unacceptable biofouling control in the end of testing period. A biocide consumption rate of 1.4×10^{-5} lb/gal·day was observed when using sodium hypochlorite as biofouling control agent. Although free chlorine residuals were consistently observed (Figure 4. X12), it appeared that under-breakpoint chlorination was unable to biofouling problems.When using chlorine dioxide as biofouling control agent, a biocide consumption rate of 1.3×10^{-5} lb/gal·day was required to achieve desired chlorine dioxide concentrations about 0.5 mg/L in recirculaing cooling water. The outcomes of HPC analyses also indicated that the desired residual was able to control biofouling problems.

Table 4.3.8 Biocide residual, dosing rate, and biocide consumption in MWW_NF tests

Test	Biocide	Total chlorine residual (mg/L)	MCA chlorine residual (mg/L)	Biocide dosing rate (gal/day)	Stock solution concentration (mg/L)	Normalized biocide consumption (lb/gal·day) [a]
CTA1	**MCA**	2.76±1.34	2.43±1.18	2.77±0.99	928±209	3.4×10^{-5}
CTB1	**MCA**	2.11±1.21	1.76±1.05	2.47±1.11	778±273	2.2×10^{-5}
CTC1	**MCA**	3.34±1.29	2.84±1.15	3.05±0.92	909±105	3.5×10^{-5}
CTA3	**NaOCl**	5.17±1.94	0.09±0.02	1.52±0.44	874± 92	1.4×10^{-5}
CTB3	**ClO_2**	0.41±0.16[b]	0.42±0.23	7.15±1.07	163± 61	1.3×10^{-5}

Note: a: calculated using equation shown in Table 3.3.4;; b: chlorine dioxide residual

4.3.4.5 Legionella monitoring in field tests

Water samples from all of the pilot-scale cooling system tests in Summer 2010 and 2011 were sent weekly to the Special Pathogen Laboratory (Pittsburgh, PA) for Legionella detection. At the end of the second run, a small piece of drift eliminator having visible biomass growth from each pilot unit was also analyzed for Legionella species. Water samples were treated with acidification and heating before culturing. Sessile bacteria sample (biofilm on plastic sheet) was first immersed in 10 mL of sterilized phosphate buffer solution and was then sonicated for 5 minutes. Results shown in Table 4.3.9 indicate that Legionella species were not able to grow in the pilot-scale cooling systems fed with MWW_NF and with continuous biocide addition over a period of two months.

Table 4.3.9 Legionella analysis detection in recirculating water samples and on drift eliminators

System	Week								
	1	2	3	4	5	6	7	8	DE
CTA1	-	-	-	-	-	-	-	-	N/A
CTB1	-	-	-	-	-	-	-	-	N/A
CTC1	-	-	-	-	-	-	-	-	N/A
CTA3	-	-	-	-	N/A	N/A	N/A	N/A	-
CTB3	-	-	-	-	N/A	N/A	N/A	N/A	-

Note: "+": Positive; "-": Negative; "U": Unable to identify; "N/A": Not available, "DE": Sessile samples collected on drift eliminator

4.3 Summary and Conclusions

Bench-scale studies carried out for corrosion analysis revealed that secondary treated municipal wastewater with subsequent nitrification and sand-filtration (MWW_NF) increased the corrosivity of the secondary treated municipal wastewater (MWW). Removal of corrosion influencing constituent ammonia by nitrification was helpful for corrosion management of the system. Sand filtration removed a significant amount of total solids in the system. As a result there was lower scaling in the system compared to MWW, and the metal alloys exposed to recirculating cooling water experienced higher corrosion rate when immersed in MWW_NF than in MWW.

Tolyltriazole (TTA) successfully inhibited the corrosion rate of copper and cupronickel alloys in both the bench-scale and pilot-scale studies. TTA also lowered the corrosion rate of mild steel alloys significantly, and was able to control it within acceptable levels according to the industrial corrosion control criteria.

Polyphosphate based corrosion inhibitor sodium hexametaphosphate (SHMP) was found to be less effective than TTA in mild steel corrosion control. Most of the SHMP precipitated out of the solution and contributed to the scaling problem. Precipitation of SHMP in the system lowered the corrosion rate of mild steel and cupronickel alloys. But the reduction in corrosion rate was not significant compared to that using TTA as a corrosion inhibitor.

Bench-scale studies were carried out with 3ppm monochloroamine dosing as a biocide in the recirculating cooling water system. It was found that use of 4ppm TTA was able to lower the corrosion rate of mild steel, copper, and cupronickel alloys. On the other hand, use of 2ppm TTA with monochloramine lowered the corrosion rate of copper, and cupronickel alloys only. Mild steel corrosion rate exacerbated with 2ppm TTA, and 3ppm monochloramine dosing in the bench-scale studies.

Pilot-scale experiments were carried out with a control cooling tower with no addition of corrosion inhibitor TTA, and two other cooling towers with different concentrations (2ppm, and 4 ppm) of TTA dosed for corrosion management. It was observed that even without any corrosion inhibitor, corrosion rates of copper and cupronickel alloys were within acceptable limit (after 28 days, and 56 days) according to the industrial corrosion control criteria. Addition of TTA reduced the corrosion rates of copper and cupronickel alloys to excellent levels. Mild steel corrosion rate was unacceptable in the control cooling tower according to the industrial corrosion control criteria. However, use of 2ppm, and 4ppm TTA reduced the corrosion rate of mild steel significantly (higher reduction achieved with 4ppm TTA dose) and brought it down just within acceptable limits according to the industrial corrosion control criteria.

Based on the results from batch tests, calcium phosphate is the main precipitates formed when MWW_NF was used for the recirculating cooling towers operated at typical operation conditions. Both bench- and pilot-scale studies have showed that scaling is not a significant issue on unheated surfaces and no antiscalant was needed for the scaling control at relevant hydrodynamic conditions. However, on heated surface, it is necessary to control the pH at proper range since comparatively lower pH would retain more soluble calcium and phosphate in the solution and crystalline fouling in the high temperature ranges. This study also showed that the flow velocity should be maintained high enough to prevent the sedimentation of bulk precipitates to both the unheated and heated surfaces.

Laboratory scale batch studies revealed that monochloramine (MCA) could control biological growth below 10^4 CFU/mL in MWW_NF at the initial dosage of 2 mg/L and 2 hours of contact time. Biocidal efficacy of MCA increased dramatically when the water temperature increased from 23ºC to 40 ºC. Under these conditions, free chlorine (NaOCl) and chlorine dioxide could achieve the same results at lower dosages of 0.5 mg/L and 1 mg/L, respectively because of a fairly low organic content in MWW_NF. However, an initial chlorine dioxide dose above 4 ppm is required to achieve detectable ClO_2 residual.

Control tests with MWW_NF at CoC 4 with no biocide addition in a recirculating system indicated that the planktonic bacteria population reached exponential growth phase within 24 hours while the sessile bacterial population easily exceeded the biofouling control criterion of 10^4 CFU/cm^2. Bench-scale studies in a recirculating system with continuous biocide addition demonstrated that all three biocides evaluated in this study can achieve biofouling control criteria for 7 days. Tests in the recirculating system revealed that chlorine dioxide residual of 0.5 ppm could easily achieve biofouling planktonic control criteria (10^4 CFU/mL) with minimal impact of water temperature.

Pilot-scale tests with MWW_NF as makeup water (i.e., CTA1, CTB1, and CTC1 tests) indicate that biofouling in the cooling tower can be controlled by maintaining MCA residual above 2 ppm in the recirculating water for 56 days. However, the growth curve of sessile heterotrophic bacteria indicates that biofilm growth can exceed biofouling control criteria after 56 days. Strict control of biocide dosing rate is the major operating requirement to achieve reliable biofouling control in cooling systems using tertiary treated municipal wastewater. Use of pre-formed monochloramine was more reliable in controlling biological growth compared to free chlorine and chlorine dioxide. Maintaining 4-5 mg/L of total chlorine residual by the addition of sodium hypochlorite cannot achieve target biofouling control criteria. On the other hand, chlorine dioxide residual between 0.5-1 mg/L ClO_2 successfully achieved biofouling control criteria during the extended pilot-scale test.

Pilot-scale tests with MWW_NF as makeup water in recirculating cooling system with continuous addition of monochloramine, free chlorine or chlorine dioxide revealed complete absence of Legionella species in the system. This result is quite significant in terms of public health protection for those that work or live around a recirculating cooling systems using tertiary-treated municipal wastewater.

4.5 References

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5.0 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: Nitrification-Filtration and Carbon Adsorption (MWW_NFG)

Increasing scarcity of freshwater supply for use in power plant recirculating cooling systems has led to the search for alternative impaired water sources. Secondary treated municipal wastewater (MWW) is a widely available alternative cooling water source in terms of quantity and geographic proximity to most of the existing and future power plants (Chien et al., 2008). MWW contains elevated concentrations of organic matter, hardness, orthophosphate, ammonia, and total dissolved solids relative to freshwater (Williams, 1982; Weinberger, 1966). The main challenge when secondary treated municipal wastewater is reused in recirculating cooling water systems is more complicated control of corrosion, scaling, and biological fouling due to the lower quality of the water. Thus use of MWW in power plant cooling systems requires additional treatment and/or inhibitor dosing attention to mitigate corrosion, scaling and biological fouling problems that can rise due to lower water quality (EPRI, 2003).

Previous study showed that use of MWW resulted in low corrosion of mild steel, copper and cupronickel alloys (Hsieh et al., 2010). The low corrosiveness of MWW was due to its high scaling potential, which caused formation of a protective scaling layer on the metal surfaces (Hsieh et al., 2010; Uhlig and Revie, 1985). However, scaling on metal and metal alloy surfaces reduces the heat transfer efficiency of the cooling system.

In this study, the feasibility of controlling corrosion, scaling, and biofouling when using secondary municipal wastewater tertiary treated with nitrification, filtration, and granular activated carbon adsorption (MWW_NFG) in cooling water systems was investigated through laboratory and pilot-scale experiments. Nitrification with subsequent filtration will result in ammonia removal, lower pH and lower total solids in the water (Metcalf and Eddy, 2003). Ammonia causes corrosion of metals by forming soluble complexes with copper (Strmčnik et. al., 2009) and iron (Uhlig and Revie, 1985). Hence removal of ammonia will reduce the corrosiveness of MWW. As a result, MWW with additional treatment by nitrification and filtration may require fewer chemicals for corrosion and scaling management. Granular activated carbon adsorption treatment following nitrification and filtration will reduce the organic material in the wastewater (Freeman and Harris, 1995), and thus lower biocide dose should be required for biological fouling management. Bench-scale recirculating systems and one pilot-scale cooling tower were employed for testing of various chemical control schemes for corrosion, scaling, and biofouling in systems using secondary treated municipal wastewater. The testing was conducted with conditions of temperature, flow velocity similar to those in a recirculating cooling water system. The effectiveness of chemical treatment strategies in inhibiting corrosion, scaling, and biomass growth was studied through exposure and monitoring specially designed coupons in extended duration tests.

5.1 Corrosion Control for MWW_NFG Used for Cooling

In this study corrosion of different metals and metal alloys was evaluated while tertiary treated (nitrified-filtered-granular activated carbon adsorbed) municipal wastewater (MWW_NFG) was used as makeup water in power plant cooling systems. . Nitrification with subsequent filtration will result in ammonia removal, lower pH and lower total solids in the water (Metcalf and Eddy, 2003). Ammonia causes corrosion of metals by forming soluble complexes with copper (Strmčnik et. al., 2009) and iron (Uhlig and Revie, 1985). Hence removal of ammonia will reduce the corrosiveness of MWW. As a result, MWW with additional treatment by nitrification and filtration may require fewer chemicals for corrosion and scaling management. Granular activated carbon adsorption treatment following nitrification and filtration will reduce the organic material in the wastewater (Freeman and Harris, 1995), and thus lower biocide dose should be required for biological fouling management. So use of MWW_NFG as makeup water, results in a new scenario for corrosion management of metals and metal alloys exposed to the flowing cooling water. The overall goal of this study was to investigate the feasibility of using tertiary treated (nitrified-filtered-granular activated carbon adsorbed) municipal wastewater (MWW_NFG) in cooling water systems by metal alloy corrosion rate monitoring and comparison with general corrosion criteria (Hsieh et al., 2010).

The use of corrosion inhibitors is the most widely employed approach to control corrosion in recirculating cooling water system (Frayne, 1999). Corrosion inhibitors usually form barrier layers on the surface of a metal and thus decrease corrosion rate. Barrier forming inhibitors are categorized into three types: adsorbed layer formers, oxidizing inhibitors (passivators), and conversion layer formers (Dean et al., 1981). Briefly, adsorbed layer formers function by adsorbing to the metal surface; oxidizing inhibitors function by shifting the metal's electrochemical potential to a region where the metal oxide or hydroxide is stable (passivating);

and conversion layer formers function by forming a low solubility deposition on the metal surface (Dean et al., 1981). Among the commonly used corrosion inhibitors, tolyltriazole was selected to be tested in this research based on review of previous studies (Hsieh et al., 2010; McCoy, 1974; Frayne 1999; Jones, 1996; Harston, 2004).

Tolyltriazole (TTA) is an inhibitor specifically for copper alloys. Its NH group can adsorb onto the metal surface thus forming a barrier layer (Hollander and May, 1985). Copper corrosion inhibition by TTA can be reduced by free chlorine residual (Breske, 1983; Lu et al., 1994; Harrison and Kennedy, 1986), which is often maintained in cooling systems to prevent microbial growth and biofouling. On the other hand, chloramine has lower oxidizing power but studies of its influence on copper corrosion inhibition by TTA are limited. Previous studies (Hsieh et al., 2010) also showed that TTA was able to lower the corrosion rate of mild steel immersed in MWW.

Specifically, the objectives of this study were to 1) assess corrosion of metal alloys immersed in MWW_NFG in a bench-scale recirculation water system with identified corrosion control program from similar bench-scale experiments using other types of tertiary treated water, and 2) apply the corrosion control program tested in bench-scale experiments to pilot-scale cooling systems operated with tertiary treated (nitrified-filtered-granular activated carbon adsorbed) municipal wastewater (MWW_NFG) in the field.

5.1.1 Testing protocols

5.1.1.1 Metal alloy pre-exposure and post-exposure treatment

The metal and metal alloys chosen for this study were mild steel (UNS G10180), copper (UNS C10100), and cupronickel (UNS C70600) as they are suitable for cooling water systems using municipal wastewater as makeup water (Hsieh et al., 2010; Herro and Port, 1993). The specimens were cylinder-shaped with a nominal diameter of 0.375 in., nominal length of 0.5 in. and were obtained from Metal Samples Co. (Munford, AL). Before immersing the metal and metal alloy samples in recirculating cooling water (both in bench-scale and pilot-scale systems), they were wet polished with SiC paper to a 600 grit surface finish, dried, weighed to 0.1 mg, degreased with acetone and rinsed in distilled water.

During exposure of the metal alloy specimens in the bench-scale experiments, the polarization resistance (R_P) of the specimens was semi-continuously monitored and specimens were withdrawn after approximately 1, 3, and 5 days of exposure to measure the weight loss. After withdrawal, the specimens were cleaned following the ASTM G1 and then reweighed to 0.1 mg to determine the weight loss (ASTM G1-03, 2005).

In the pilot scale cooling system tests, the metal and metal alloy specimens were immersed for periods of 7, 14, 28, and 56 days. The weight loss method was used to determine the average corrosion rate during the exposure period. After withdrawal, similar post-exposure treatments were performed on the metals and metal alloys as was done in the bench-scale experiments.

5.1.1.2 Corrosion experiment matrix for the bench-scale experiments with MWW_NFG

Actual tertiary treated (nitrified-filtered-activated carbon adsorbed) municipal wastewater (MWW_NFG) collected from Franklin Township Municipal and Sanitary Authority (FTMSA) facility was concentrated in the laboratory by heated evaporation (at 40°C) to reach four cycles of concentration (CoC4) as determined by 75% volume reduction for bench-scale experiments. The temperature (40°C), used for heated evaporation in the lab, simulates the temperature of recirculating cooling water in the pilot-scale cooling systems (Hsieh et al, 2010). At CoC4 the tertiary treated municipal wastewater (MWW_NFG) was used in the bench-scale recirculation water system for corrosion analysis of mild steel, copper and cupronickel alloys.

Two experiments were carried out in the bench-scale recirculation system using MWW_NFG (with and without addition of 2 ppm TTA) for corrosion analysis of mild steel, copper, and cupronickel alloys. The samples were immersed for 5 days for average corrosion rate analysis. Intermittent R_P measurements were performed throughout the immersion period of the metal alloy samples for producing an instantaneous corrosion rate profile for the metal alloys.

5.1.1.3 Pilot-scale corrosion experiment with MWW_NFG

The experimental matrix for the pilot-scale testing was based on the results of the laboratory experiments with respect to corrosion, scaling, and biofouling studies (scaling and biofouling control studies are not covered in this section). TTA was chosen as corrosion inhibitor, and PMA (a scaling inhibitor identified to be effective in scaling control through the laboratory experiments in another study) as a scaling inhibitor. Monochloramine was used as biofouling control agents. Only one pilot-scale cooling towers were operated for 28 days after reaching CoC 4. The cooling tower (CTA) was operated using tertiary treated (nitrified-filtered-activated carbon adsorbed) municipal wastewater (MWW_NFG) at CoC 4. Table 5.1.1 shows the pilot-scale experimental matrix with chemical treatment strategies.

Table 5.1.1 Experimental matrix for pilot-scale cooling tower experiment showing different chemical treatment strategies for corrosion, scaling, and bio-fouling management. Flow rate 0.189 L/s (3gpm).

Cooling Tower Designation	pH	Corrosion Inhibitor	Scaling Inhibitor	Bio-fouling Control
Cooling Tower A (CTA)	8.1±0.4	TTA (2 ppm)	PMA (5 ppm)	MCA (2~3 ppm residual)

Note: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine).

In the pilot-scale testing, corrosion inhibitor TTA was monitored with high performance liquid chromatography (HPLC 1100 Series, Agilent Technologies) (Choudhury et al., 2012a). Other water chemistry parameters (PMA, monochloramine, anions, alkalinity, pH and conductivity) and cooling tower operational parameters (water flow rate, air flow velocity, temperature, makeup water flow rate, and blowdown water flow rate) were also monitored. The inhibitor TTA was added once per day to the makeup water tank. Concentration of TTA dosed in the makeup water was one-fourth of the target concentration of TTA in the cooling tower system. TTA concentration in the pilot scale cooling tower basin reached to desired target concentration, after the recirculating cooling water reached CoC 4.

5.1.2 Bench-scale studies

MWW_NFG water collected from FTMSA facility was used in bench-scale recirculation system experiments at CoC 4 with and without the addition of corrosion inhibitor TTA. Table 5.1.2 shows the average 5-day corrosion rates of metals and metal alloys immersed in MWW_NFG. Results from the table indicated that TTA successfully inhibited the corrosion of copper and cupronickel alloys to excellent levels according to the cooling tower system corrosion criteria. Mild steel corrosion rate was significantly inhibited and lowered to acceptable limits using TTA as corrosion inhibitor.

Table 5.1.2 Average 5-day corrosion rates of mild steel, copper and cupronickel samples exposed to MWW_NFG at 4CoC, collected from Franklin Township Municipal Sanitary Authority (FTMSA) wastewater treatment facility, in the bench-scale recirculating system.

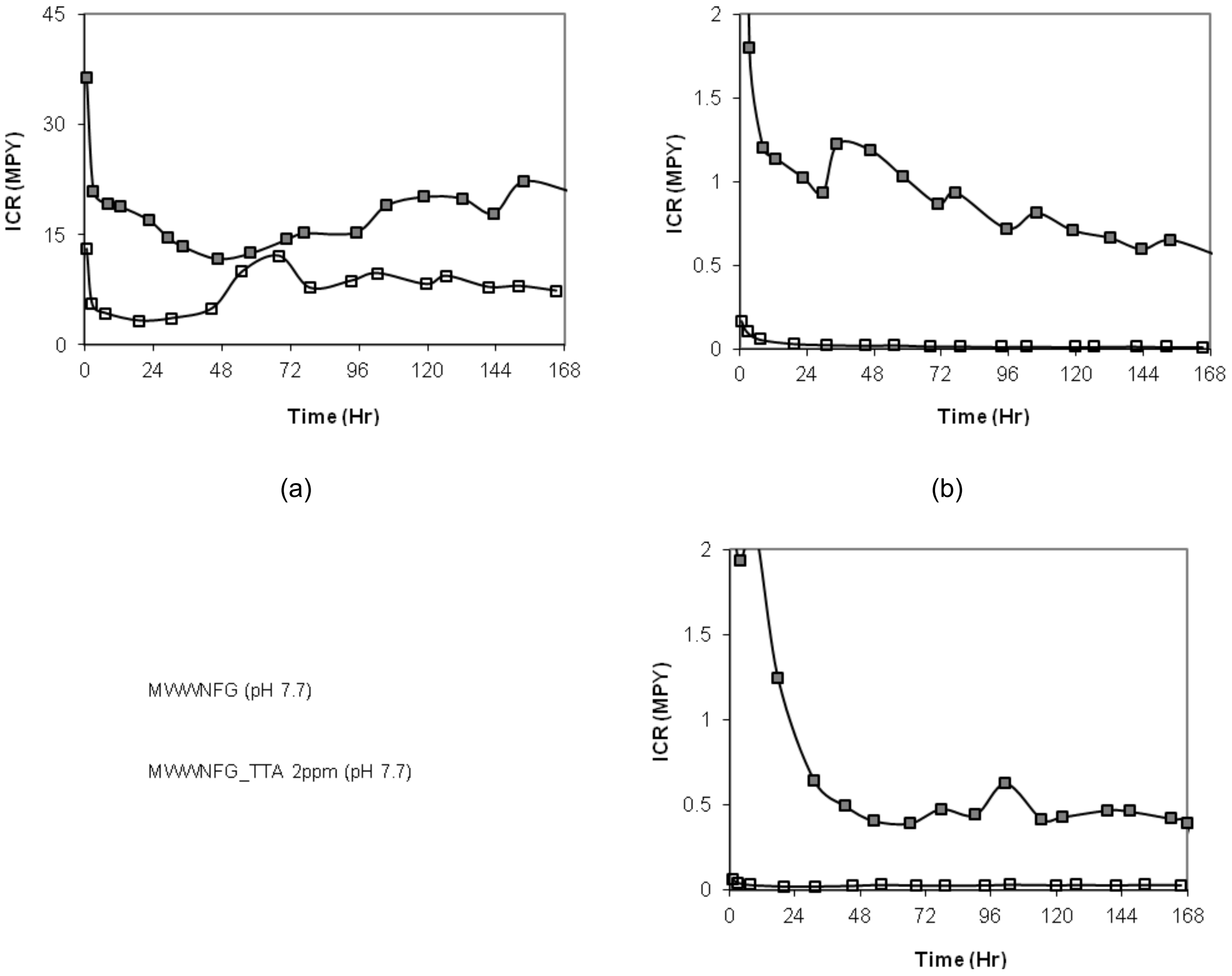
Type of actual tertiary treated wastewater	Average 5-day corrosion rate, MPY (Corrosion category [b])		
	Mild Steel	**Copper**	**Cupronickel**
MWW_NFG (Control)	16.0 (Unacceptable)	1.00 [a] (Poor)	0.671 [a] (Fair)
MWW_NFG (TTA 2mg/L)	7.77 (Poor)	0.070 (Excellent)	0.090 (Excellent)

Note: [a] Pitting corrosion was visually observed on the sample after withdrawal from the system.

[b] b Hsieh et al., 2010.

MPY: milli-inch per year. Control: no corrosion inhibitor was dosed in the system. TTA: Tolyltriazole.

Figure 5.1.1 shows the instantaneous corrosion rate measurement profiles for mild steel, copper and cupronickel alloy samples measured for an interval of 7 days in the bench-scale recirculating system with MWW_pH. The pH of each type of recirculating water along with respective TTA dosing is also indicated in Figure 5.1.1.



(a)

(b)

(c)

Figure 5.1.1 Instantaneous corrosion rate profiles for (a) mild steel, (b) copper, and (c) cupronickel alloy samples immersed in bench-scale recirculating cooling system using MWW_NFG.

5.1.3 Pilot-scale studies

5.1.3.1 Corrosion analysis in pilot-scale cooling tower experiments

Pilot-scale experiments were carried out with one cooling tower (CTA) using secondary municipal wastewater tertiary treated with nitrification, filtration, and activated carbon adsorption (MWW_NFG). In this experiment CTA was dosed with 2ppm TTA. The average residual free TTA concentrations in CTA was 1.74±0.50 ppm

Table 5.1.3 shows the average corrosion rates of metal and metal alloy specimens immersed in MWW_NFG at CoC 4 in the three pilot-scale cooling tower for different time intervals. It can be seen from Table 5.1.3 that corrosion rates of copper and cupronickel alloys were inhibited in the cooling tower. However, pitting corrosion was observed in the copper and cupronickel samples immersed for 28 days. So MWW_NFG was not suitable for copper and cupronickel alloys according to corrosion criteria (Hsieh et al., 2010) with 2ppm TTA dosed for corrosion inhibition.

Mild steel corrosion rate was low in MWW_NFG compared to the other tertiary treated municipal wastewaters used previously. Presence of scaling in the cooling system while using MWW_NFG caused lower corrosion of mild steel.

Table 5.1.3 Average corrosion rates of metal alloys contacted with tertiary treated (nitrified-filtered-activated carbon adsorbed) municipal wastewater (MWW_NFG) at CoC 4 in pilot-scale cooling systems for different time intervals and their category according to general corrosion criteria[b] for cooling systems.

Cooling Tower	Immersion Days	Average Corrosion Rates of Metal Alloys (MPY) (Corrosion category)		
		Mild Steel	**Cupronickel**	**Copper**
CTA [a]	7	3.16 (Fair)	0.28 (Good)	0.23 (Good)
	28	1.33 (Good)	0.26 [c] (Unacceptable)	0.22 [c] (Unacceptable)

[a] CTA : Cooling Tower A with 2 ppm TTA, 5 ppm PMA and 2-3 ppm MCA

[b] Hsieh et al., 2010.

[c] Pitting corrosion was visually observed on the sample after withdrawal from the system.

Figure 5.1.3 shows the concentration profiles of TTA in CTA. The average free TTA concentration in CTA was 1.74±0.50 ppm, which was close to the desired 2 ppm target TTA dose. The TTA concentration in CTA was always less than the target 2 ppm concentration, likely due to the reduction in the free TTA by formation of copper-TTA complex.

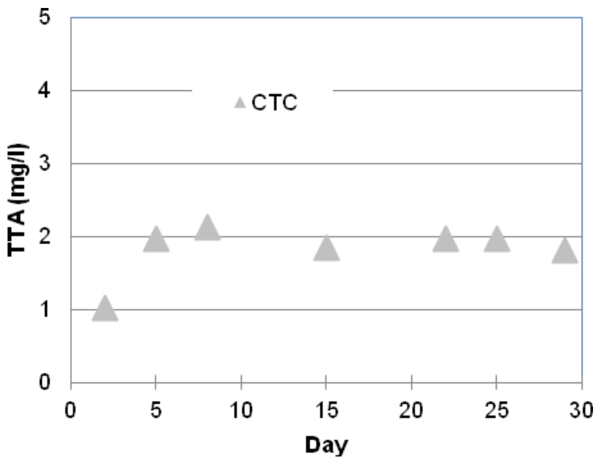


Figure 5.1.3 Concentration profile of TTA in CTA (measured twice each week) in the pilot scale cooling tower experiments using tertiary treated (nitrified-filtered-activated carbon adsorbed) municipal wastewater (MWW_NFG) from FTMSA.

5.2 Scaling Control for MWW_NFG Used for Cooling

Organic matter in cooling system make-up water promotes biofouling and increases biocide requirements to control biofouling. In addition, the organic matter can adsorb on metal surfaces and contribute directly to fouling. One of the unit processes that can be utilized for the removal of residual dissolved organic matter is activated carbon adsorption. The main objective in this part was to identify the effects of TOC removal in a fixed bed granular activate carbon (GAC) adsorber on scaling behavior.

5.2.1 Testing protocols

It was expected that only organic matter in the treated municipal wastewater is influenced significantly by the GAC treatment. Thus, no batch and bench-scale recirculating system tests which focused on synthetic treated municipal wastewater composed of mineral contents only were conducted.

Three pilot-scale cooling towers (Tower A, B, and C) were operated side-by-side at the Franklin Township Municipal Sanitary Authority (FTMSA, Murrysville, PA) in Summer 2011. MWW_NF after GAC treatment is designated as MWW_NFG and was fed into Tower C while the other two towers were used as control towers for scaling study. The operating conditions of the cooling towers are shown in Section 2.2.3. Traditional stainless steel (SS) coupon discs were immersed in the recirculating water and sampled at a predetermined schedule. The air-dried SS coupons were dried at 104°C for 3.5 hours and subsequently combusted at 500°C for 3.5 hours in a muffle furnace. The deposits after the combustion were considered as the inorganic mineral scales. The inorganic deposits on selected SS disc specimens were also analyzed by SEM/EDS to obtain their elemental composition. In parallel with the solids analysis, important information about the chemistry of the makeup and recirculating water was recorded throughout the field tests.

During the field tests, the alkalinity of the MWW_NF was unusually high in the first 15 days resulting in significant mass gain in all three towers (this will be discussed in the following text). In order to obtain representative information, a new test was initiated on day 16 and lasted for 12 days when the alkalinity of makeup water was in the normal range. For the sake of simplicity in discussion, the whole test period was divided into two phases: Phase 1 (day 1 to day 15) and Phase 2 (day 16 to day 28).

5.2.2 Pilot-scale studies

Figure 5.2.1 depicts the time course of inorganic mineral scale deposition on the stainless steel coupon discs in the three cooling towers during the two phases of pilot-scale cooling tests.

As can be seen in Figure 5.2.1, mineral deposition on the stainless steel coupon discs immersed in Towers A, B, and C for the first 15 days (phase 1) was significant, especially in Towers A and C. This result was completely inconsistent with the pilot-scale cooling tower tests with MWW_NF in summer 2010 as shown in Section 4.2. In order to explain these unusual results, SEM/EDS analysis was first conducted to study the elemental compositions of the inorganic deposits collected from Towers A, B, and C, as shown in Figure 5.2.2, 5.2.3, and 5.2.4 respectively. These analysis showed that the inorganic deposits were mainly composed of calcium carbonate and calcium phosphate. In order to provide reasonable explanations for the significant elevation of mass gain in the first phase, it was necessary to assess the reactions leading to the formation of calcium phosphate and calcium carbonate individually.

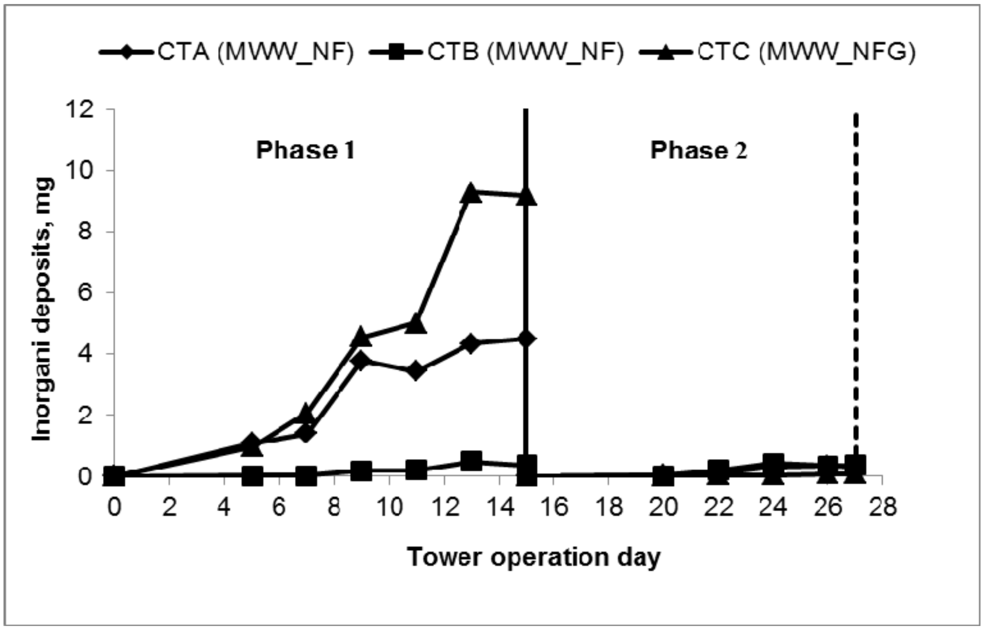


Figure 5.2.1 Inorganic deposits measured in the pilot-scale cooling tower tests with MWW_NFG

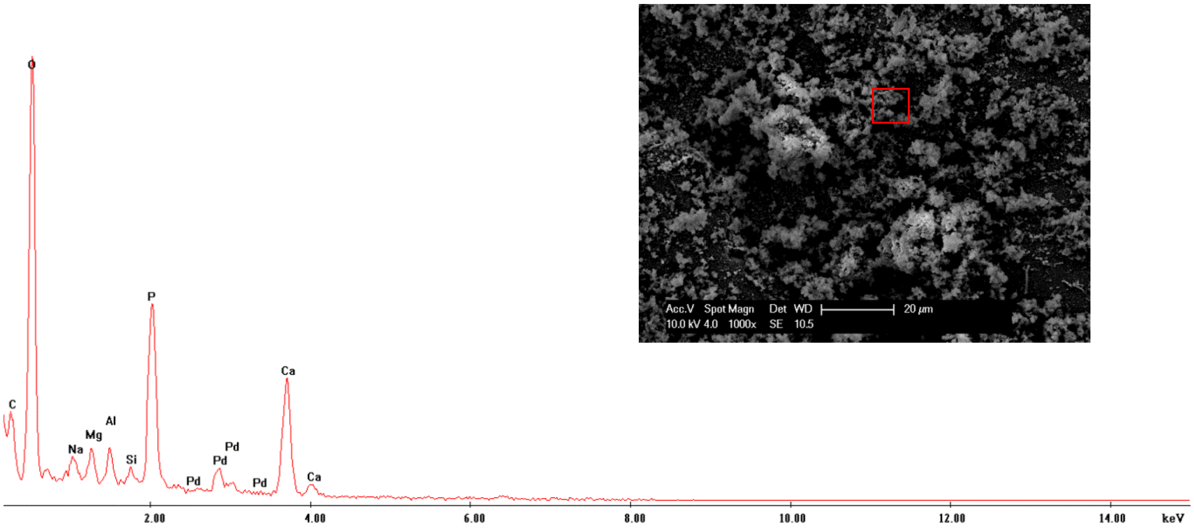


Figure 5.2.2 SEM image and elemental composition of the solid deposits collected on stainless steel discs immersed in pilot-scale cooling towers operated at CoC 4-6: Day-15 sample from Tower A using MWW_ NF. EDS scan was performed on the area outlined by the square box on the SEM image.

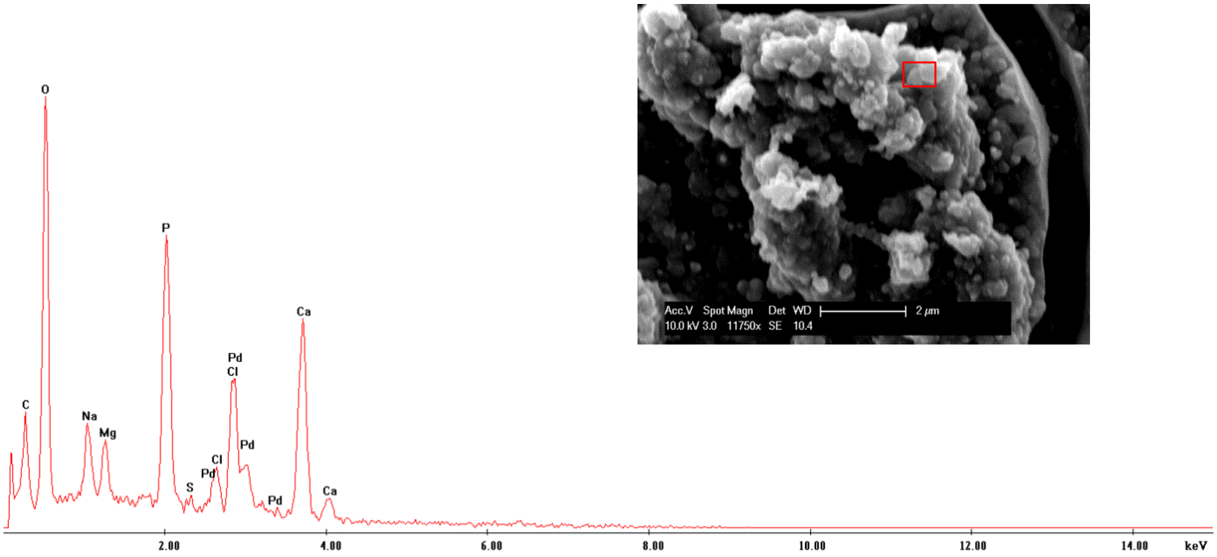


Figure 5.2.3 SEM image and elemental composition of the solid deposits collected on stainless steel discs immersed in pilot-scale cooling towers operated at CoC 4-6: Day-15 sample from Tower B using MWW_ NFG. EDS scan was performed on the area outlined by the square box on the SEM image.

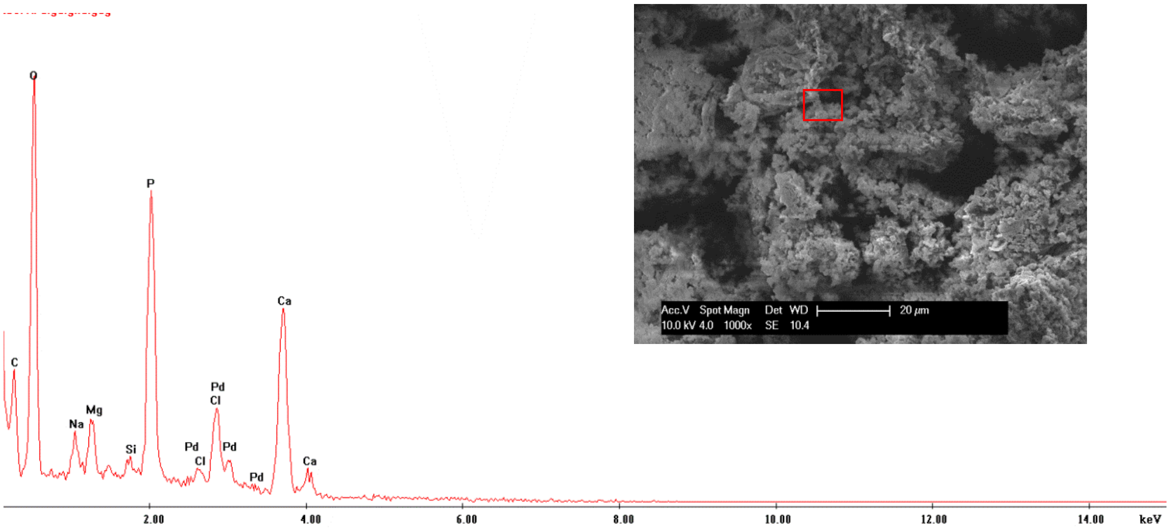


Figure 5.2.4 SEM image and elemental composition of the solid deposits collected on stainless steel discs immersed in pilot-scale cooling towers operated at CoC 4-6: Day-15 sample from Tower C using MWW_ NFG. EDS scan was performed on the area outlined by the square box on the SEM image.

Figure 5.2.5 and 5.2.6 depict phosphate concentration profiles in the make-up water and recirculating water in pilot-scale cooling towers during summer 2010 and phase1 in this section, respectively. These Figures suggest that there was not much difference between the phosphate concentrations in the make-up water during the two tests. However, significant differences in phosphate concentrations were observed in the recirculating water between the two tests despite the fact that all the pilot-scale cooling towers were operated at similar cycles of concentration (CoC 4-6). Phosphate concentrations in the recirculating water in Towers A and C in Phase 1 (day 1 to day 15) were much lower (5-10 ppm as PO_4^{3-}) compared with the data obtained in summer 2010 (>20 ppm as PO_4^{3}), indicating that more phosphate scales formed in these systems. Among the three towers, highest phosphate concentration in the recirculating water was in Tower B, corresponded to the least mineral scale collected in this tower.

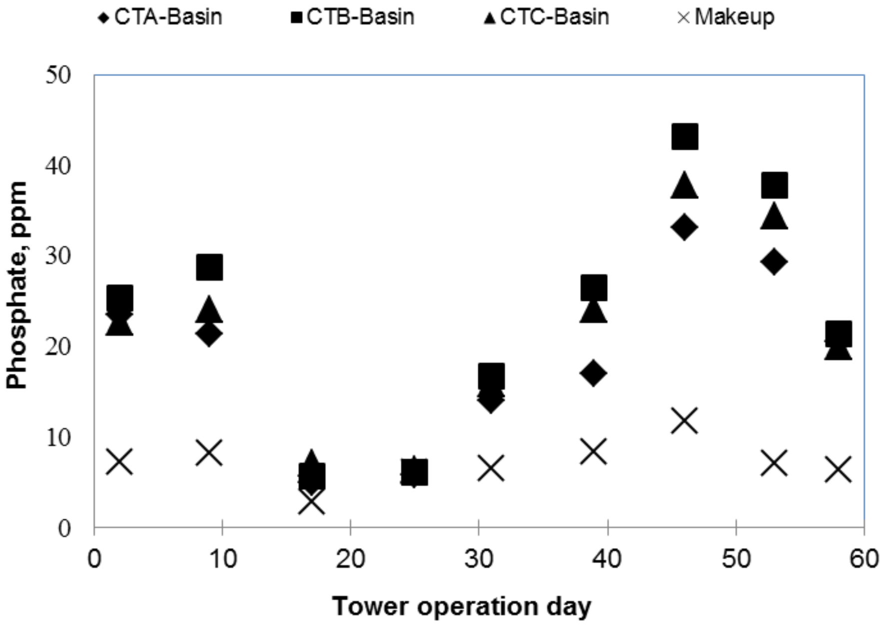


Figure 5.2.5 Phosphate concentrations in pilot-scale cooling tower tests using MWW_NF as make-up water, summer 2010

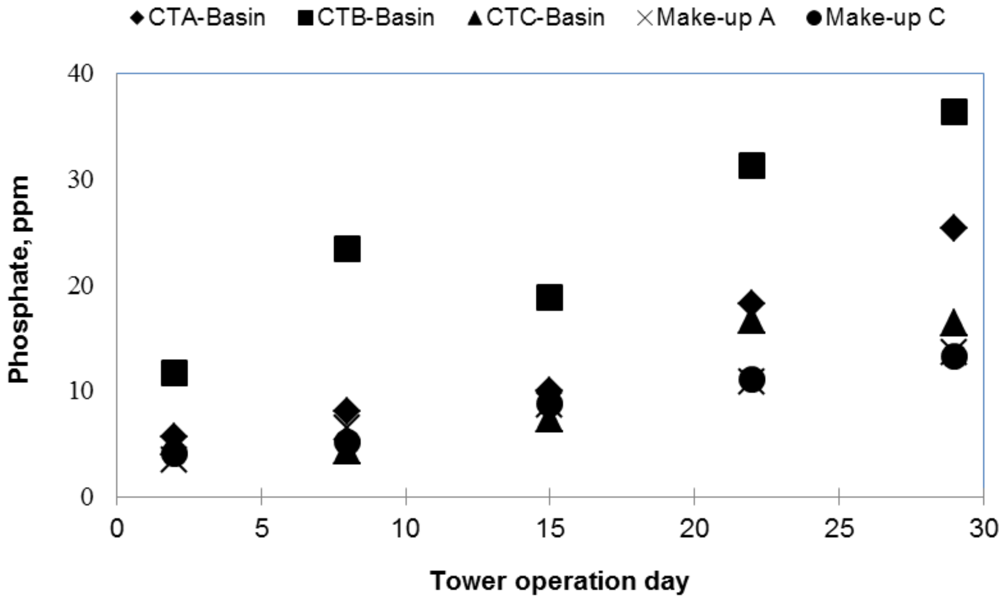


Figure 5.2.6 Phosphate concentrations in pilot-scale cooling tower tests using MWW_NF and MWW_NFG as make-up water, summer 2011

Langelier Saturation Index (LSI) could be used to estimate the potential for calcium carbonate scale formation. LSI in the recirculating water in Towers A, B, and C is shown in Table 5.2.1. For comparison, LSI in the recirculating water in all towers operated during summer 2010 is shown in Table 5.2.2.

Table 5.2.1 Langelier Saturation Index for the recirculating water in Towers A, B, and C during the pilot-scale cooling tower test, summer 2011

	Langelier Saturation Index				
	Day-1	Day-8	Day-15	Day-22	Day-29
Tower A	1.55	0.66	1.56	0.07	-0.46
Tower B	1.25	-1.52	1.68	-0.58	-0.48
Tower C	1.00	0.62	1.07	0.61	0.09

Note: If LSI is negative, there is no potential to form $CaCO_3$ scale and the water will dissolve $CaCO_3$; if LSI is positive, scale can form and $CaCO_3$ precipitation may occur; if LSI is close to zero, the water is neutral with respect to scale formation.

Table 5.2.2 The Langelier Saturation Index of the recirculating water in Towers A, B and C on Day 36, 40, 44, 49, and 54 during summer 2010

	Langelier Saturation Index				
	Day-36	**Day-40**	**Day-44**	**Day-49**	**Day-54**
Tower A	-1.75	-1.99	-2.78	-3.06	-2.72
Tower B	-1.58	-1.80	-2.59	-2.61	-2.66
Tower C	-1.56	-1.80	-2.85	-2.72	-2.71

Note: If LSI is negative, there is no potential to form $CaCO_3$ scale and the water will dissolve $CaCO_3$; if LSI is positive, scale can form and $CaCO_3$ precipitation may occur; if LSI is close to zero, the water is neutral with respect to scale formation.

Nearly all LSI values for the recirculating water in phase 1 of the pilot-scale test in this section were positive, indicating that there was a significant potential for the precipitation of calcium carbonate. On the contrary, Table 5.2.2 clearly shows that there was no driving force for the formation of calcium carbonate scale during the experiments conducted in summer 2010. Positive scaling propensity of calcium carbonate in all three towers operated in summer 2011 arises from high total alkalinity in the recirculating water. A comparison between the total alkalinity of the recirculating water during phase 1 of the test in 2011 (Figure 5.2.7) and in summer 2010 (Figure 5.2.18), clearly illustrates significant differences in total alkalinity between these tests. For example, total alkalinity in the recirculating water in Towers A and C during the first 15 days in summer 2011 was in the range from 92.8- 215.5 mg/L as $CaCO_3$ and 77.4-199.3 mg/L as $CaCO_3$, respectively. On the other hand, total alkalinity in the recirculating water during summer 2010 was generally in the range of 50-70 mg/L as $CaCO_3$. The high alkalinity in the recirculating water was from the unusually high total alkalinity in the make-up water during the first phase of this section (50.83-77.35 mg/L as $CaCO_3$) while the values were generally in the range of 14.4-35.2 mg/L as $CaCO_3$ with the MWW_NF as make-up water in summer 2010.

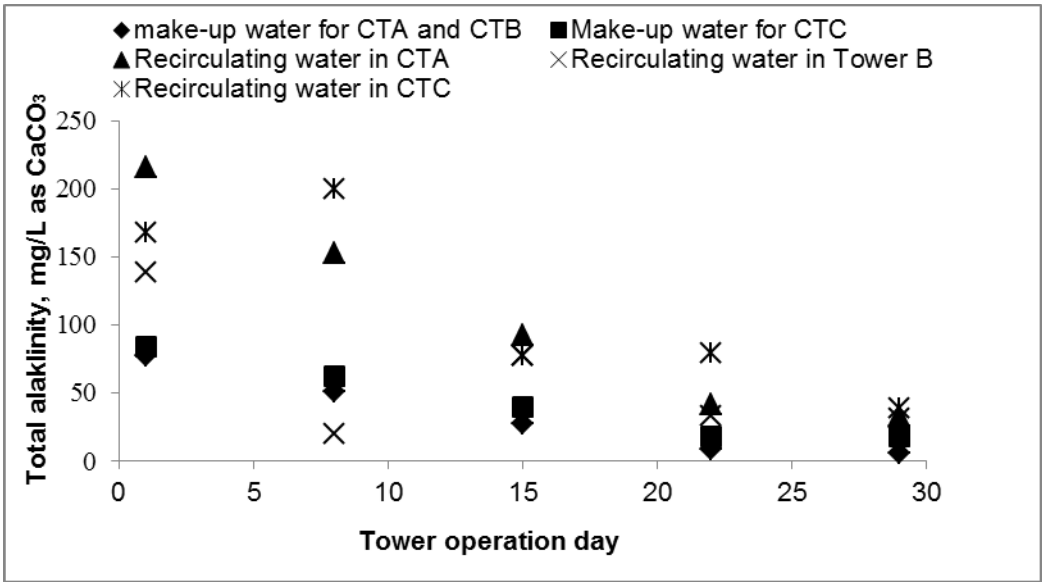


Figure 5.2.7 Total alkalinity profile in pilot-scale cooling tower tests using MWW_NF and MWW_NFG as make-up water, summer 2011

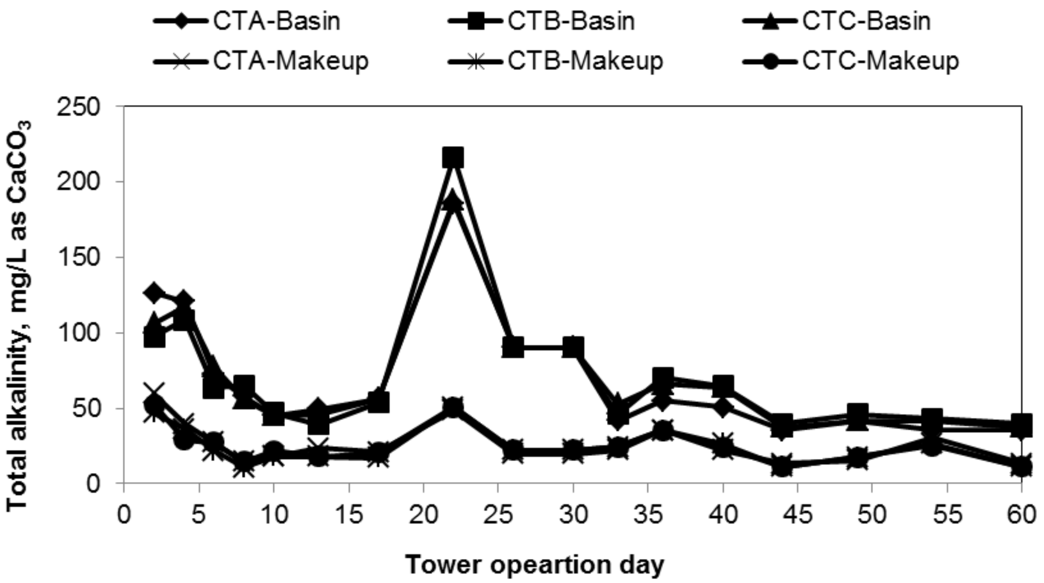


Figure 5.2.8 Total alkalinity profile in pilot-scale cooling tower tests using MWW_NF as make-up water, summer 2010

With regards to the effects of organic removal by activated carbon on the scaling behavior, Tower C with MWW_NFG as make-up water showed the greatest inorganic scale deposition during the first 15 days. Activated carbon adsorption led to an increase in pH of the make-up water (Figure 5.2.9), which enhanced the scaling potential in recirculating cooling tower system. The elevation of pH was understandable since more than half of the organic material in treated municipal wastewater falls in the strong acid or weak acid classification (Bunch et al., 1961). On the other hand, the least inorganic deposition in Tower B was associated with the lowest pH, which could be ascribed to the addition of chlorine dioxide dosed in this tower to control the biofouling.

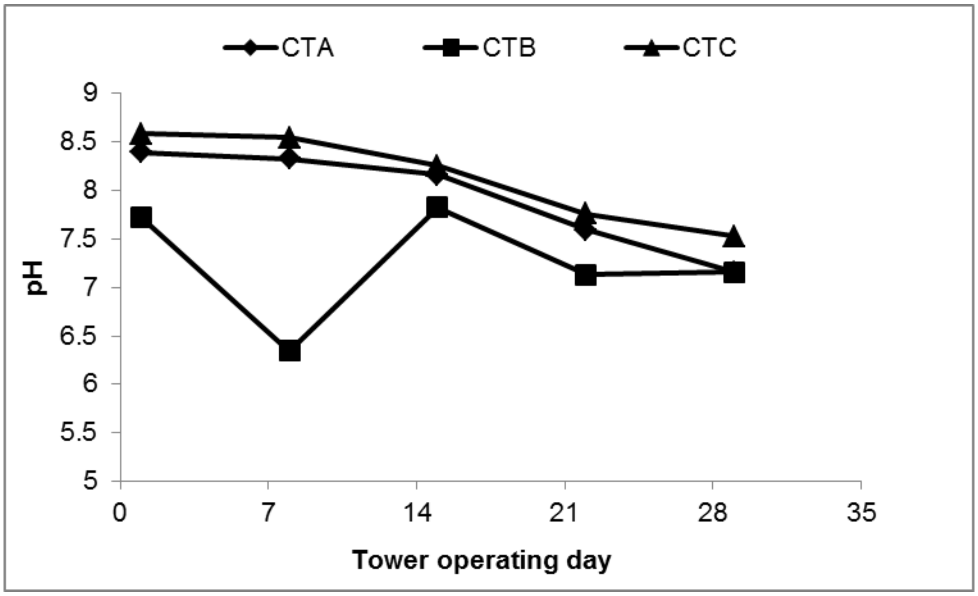


Figure 5.2.9 pH in pilot-scale cooling tower tests using MWW_NF and MWW_NFG as make-up water during the pilot-scale cooling tower test, summer 2011

Another phase of tests was conducted on day 16 when the total alkalinity of make-up water returned to the usual range (5.53-27.63 mg/L as $CaCO_3$). In Phase 2, negligible mass gain was observed in all the three towers as shown in Figure 5.2.1 which was consistent with the pilot-scale cooling tower tests described in Section 4.2. Meanwhile, phosphate concentrations in the recirculating water increased in Phase 2 as shown in Figure 6, indicating lower calcium phosphate formation potential. Besides, much lower LSI meaning less $CaCO_3$

scaling potential was also observed in Phase 2 as shown in Table 1. The above changes were surely related to the reduction in total alkalinity in the make-up water and thus the pH in the recirculating water (Figure 5.2.9).

5.3 Biofouling Control for MWW_NFG Used for Cooling

5.3.1 Testing protocols

Tests were conducted to evaluate the effectiveness of sodium hypochlorite (NaOCl), chlorine dioxide (ClO_2), and monochloramine (MCA) for biological growth control in cooling systems using MWW_NFG as makeup. These disinfectants were studied in batch reactors, a bench-scale recirculating system, and a pilot-scale cooling system according to procedures described in Chapter 4.3.1.

5.3.2 Bench-scale studies with MWW_NFG

5.3.2.1 Free chlorine for biological growth control

MWW_NF samples after filtration through granular activated carbon (GAC) adsorber MWW_NFG) were used for experiments to determine biocide demand, disinfection efficiency and decomposition rate. MWW_NFG contained 2 ppm of non-adsorbable TOC and had a pH of 9.2. Total and free chlorine demand of MWW_NFG at room temperature and 40 °C are shown in Tables 5.3.1 and 5.3.2, respectively, while total chlorine decay and monochloramine formation in MWW_NFG are shown in Figures 5.3.1 and 5.3.2, respectively. Disinfection efficiency of free chlorine and heterotrophic planktonic bacteria counts are shown in Figure 5.3.3.

Table 5.3.1 Biocide demand after 2-hour contact time with MWW_NFG at 23 °C

Dose, ppm	pH	Total chlorine		Free chlorine		Monochloramine[a]	
		Residual (ppm)	Demand, (ppm)	Residual (ppm)	Demand (ppm)	Residual (ppm)	Demand, ppm
0.5	7.5	0.27	0.17	0*	0.46	0.17	N/A
1	7.5	0.28	0.16	0*	0.96	0.06	N/A
2	7.5	0.64	1.30	0.42	1.54	0	N/A
4	7.5	2.30	1.64	2.00	1.96	0	N/A

a: Assumption based on detectable monochloramine residual.

Table 5.3.2 Biocide demand after 2-hour contact time with MWW_NFG at 40 ºC

Dose, ppm	pH	Total chlorine		Free chlorine		Monochloramine[a]	
		Residual, ppm	Demand, ppm	Residual, ppm	Demand, ppm	Residual, ppm	Demand, ppm
0.5	7.5	0.05	0.17	0*	0.26	0.02	N/A
1	7.5	0.07	0.65	0.02	0.76	0	N/A
2	7.5	0.29	1.43	0.13	1.63	0	N/A
4	7.5	1.26	2.46	1.06	2.70	0	N/A

a: Assumption based on detectable monochloramine residual.

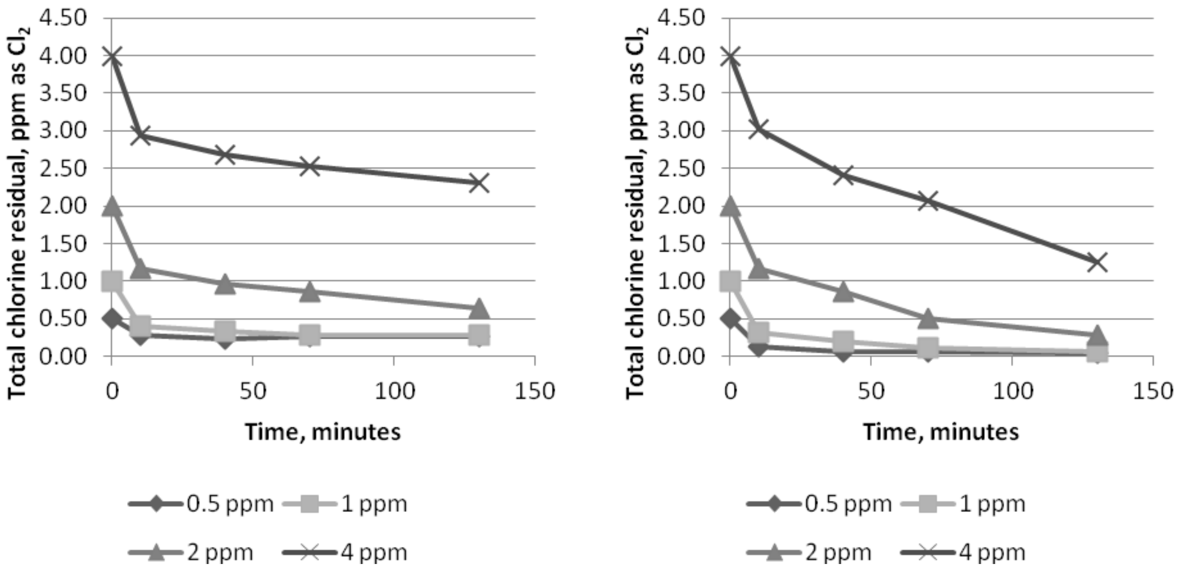


Figure 5.3.1 Total chlorine decay in MWW_NFG with initial free chlorine doses of 0.5, 1, 2, and 4 ppm at 23ºC (left) and 40 ºC (right).

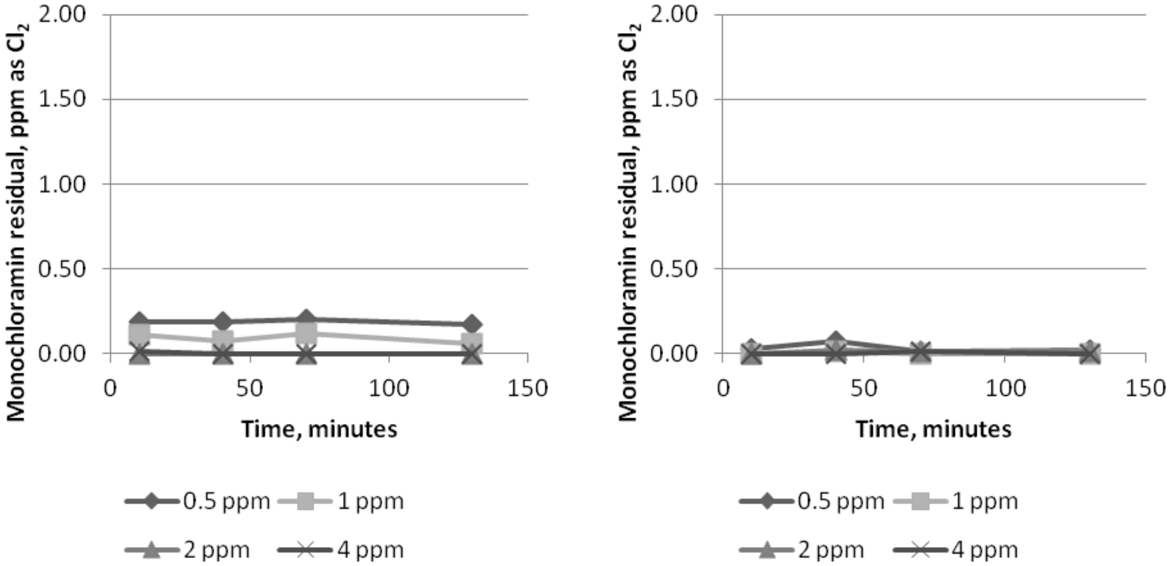


Figure 5.3.2 Monochloramine formation in MWW_NFG with initial free chlorine doses of 0.5, 1, 2, and 4 ppm at 23ºC (left) and 40 ºC (right). The inorganic ammonia concentration in MWW_NFG was 0.5 ppm as NH_3.

Figure 5.3.1 shows that most of the total chlorine added to MWW_NFG at 2 and 4 ppm dose remained as free chlorine because these high free chlorine doses lead to breakpoint chlorination. Increase in temperature to 40 ºC increased the biocide demand. This observation is confirmed by very low monochloramine concentrations shown on Figure 5.3.2. Another interesting finding was that the unidentified portion of total chlorine, suspected as organic chloramines, was consistently observed between 0.05~0.20 ppm as Cl_2. The increase in water temperature to 40 ºC slightly reduced monochloramine formation due to additional ammonia stripping at elevated temperature.

Figure 5.3.3 shows the biocidal efficacy of free chlorine in MWW_NFG. Overall, it appears that breakpoint chlorination is feasible to control the planktonic bacteria growth in MWW_NFG

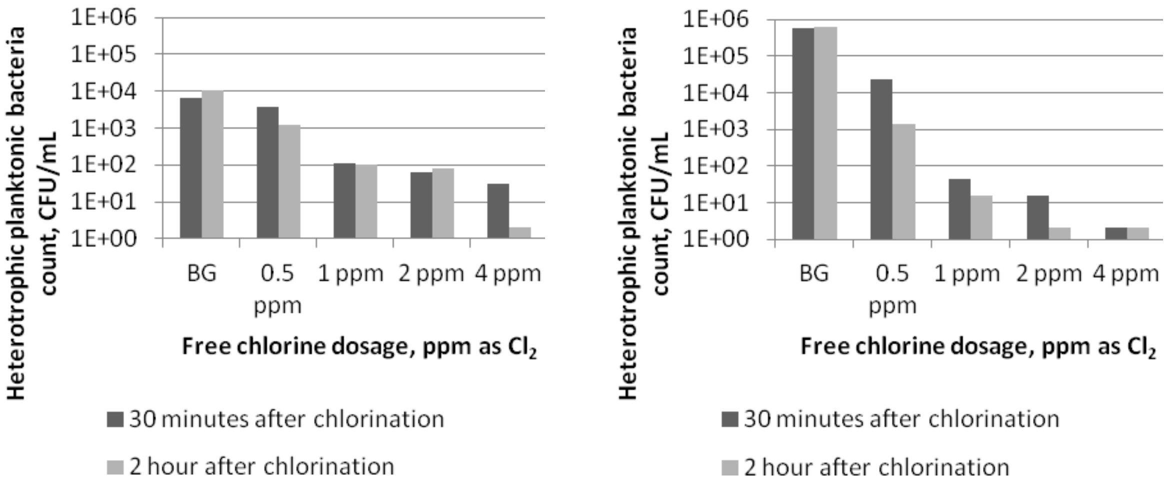


Figure 5.3.3 Heterotrophic planktonic bacteria counts with different free chlorine doses in MWW_NFG at 23 ºC (left) and 40 ºC (right). BG stands for background levels in the absence of any disinfectant.

5.3.2.2 Chloramination for biological growth control

Fate of pre-formed monochloramine and its biocidal efficiency in MWW_NFG was evaluated at 23 and 40 °C using the same procedure as in the tests with free chlorine. The total chlorine and monochloramine demand of MWW_NFG at different temperatures is shown in Tables 5.3.3 and 5.3.4. Monochloramine decay in MWW_NFG is shown in Figure 5.3.4 while its biocidal efficacy is depicted in Figure 5.3.5.

Table 5.3.3 Monochloramine and total chlorine demand after 2 hours of contact with MWW_NFG at 23 ºC

Dose, ppm	**pH**	**Total chlorine**		**Monochloramine**	
		Residual, ppm	Demand, ppm	Residual, ppm	Demand, ppm
0.5	7.5	0.42	0.05	0.33	0.12
1	7.5	0.89	0.08	0.80	0.15
2	7.5	1.79	0.18	1.58	0.37
4	7.5	3.53	0.44	3.39	0.55

Table 5.3.4 Monochloramine and total chlorine demand after 2 hours of contact with MWW_NFG at 40 ºC

Dose, ppm	pH	Total chlorine		Monochloramine	
		Residual, ppm	Demand, ppm	Residual, ppm	Demand, ppm
0.5	7.5	0.20	0.16	0.20	0.08
1	7.5	0.61	0.29	0.58	0.20
2	7.5	1.28	0.58	1.24	0.54
4	7.5	2.92	0.94	2.45	1.33

Monochloramine decay trend in MWW_NFG is shown in Figure 5.3.4. Increase in water temperature from 23 to 40 ºC significantly increased the decay. However, the biocidal efficacy of pre-formed monochloramine in MWW_NFG shown in Figure 5.3.5 shows that pre-formed monochloramine performed better at 40 ºC than at 23 ºC. An initial dose of 2 ppm or higher was required to reduce the planktonic bacteria population below 10^4 CFU/mL in MWW_NFG within a 2-hour contact time at 23 °C. However, the initial dose of 1 ppm was required to accomplish the same criteria when water temperature was adjusted to 40 ºC.

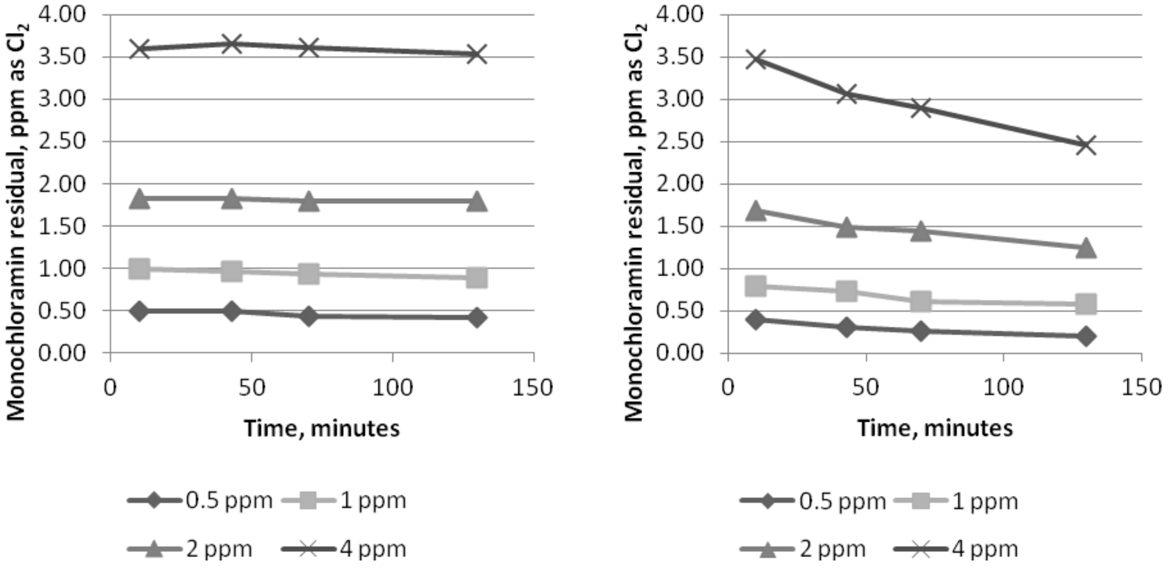


Figure 5.3.4 Monochloramine decay in MWW_NFG as a function of the initial dose at 23ºC (left) and 40 ºC (right)

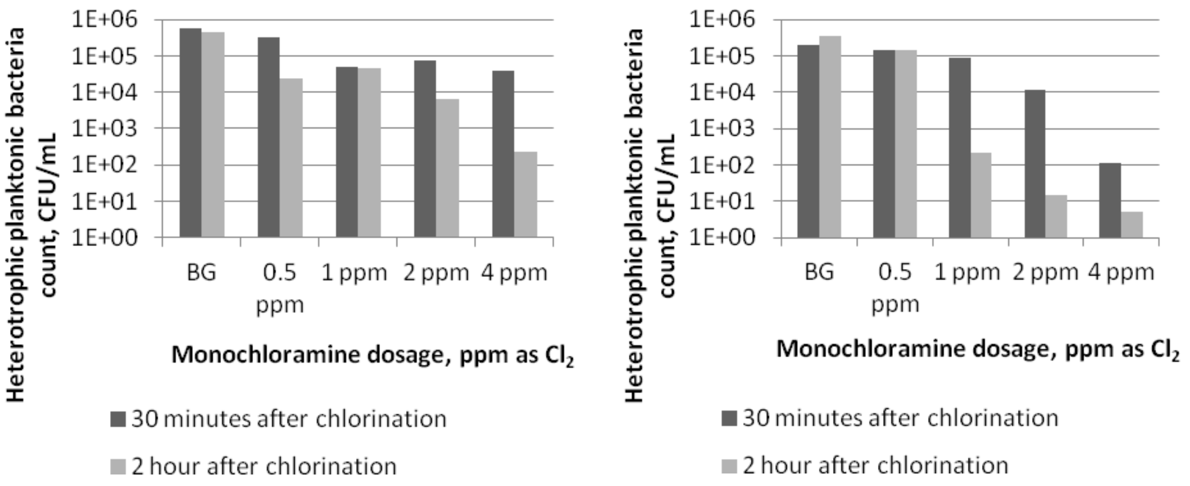


Figure 5.3.5 Heterotrophic planktonic bacteria counts as a function of pre-formed monochloramine dose and contact time in MWW_NFG at 23 ºC (left) and 40 ºC (right)

5.3.2.3 Chlorine dioxide for biological growth control

Biocidal efficiency of chlorine dioxide in MWW_NFG at 23 and 40 ºC was evaluated using the procedure described in previous sections. Chlorine dioxide stock solution was made following Standard Method 4500-ClO_2 (APHA/AWWA/WEF, 2005). The chlorine dioxide demand in MWW_NFG at 23 and 40 ºC is shown in Table 5.3.5, while the chlorine dioxide decay and its disinfection efficiency are shown in Figures 5.3.6 and 5.3.7, respectively.

Table 5.3.5 Chlorine dioxide demands after 2 hours of contact with MWW_NFG at 23 and 40 ºC

Dose, ppm	pH	23 ºC		40 ºC	
		Residual, ppm	Demand, ppm	Residual, ppm	Demand, ppm
0.5	7.5	0.02	0.22	0.00	0.28
1	7.5	0.21	0.53	0.03	0.56
2	7.5	0.91	0.83	0.29	1.14
4	7.5	2.25	1.49	1.21	2.22

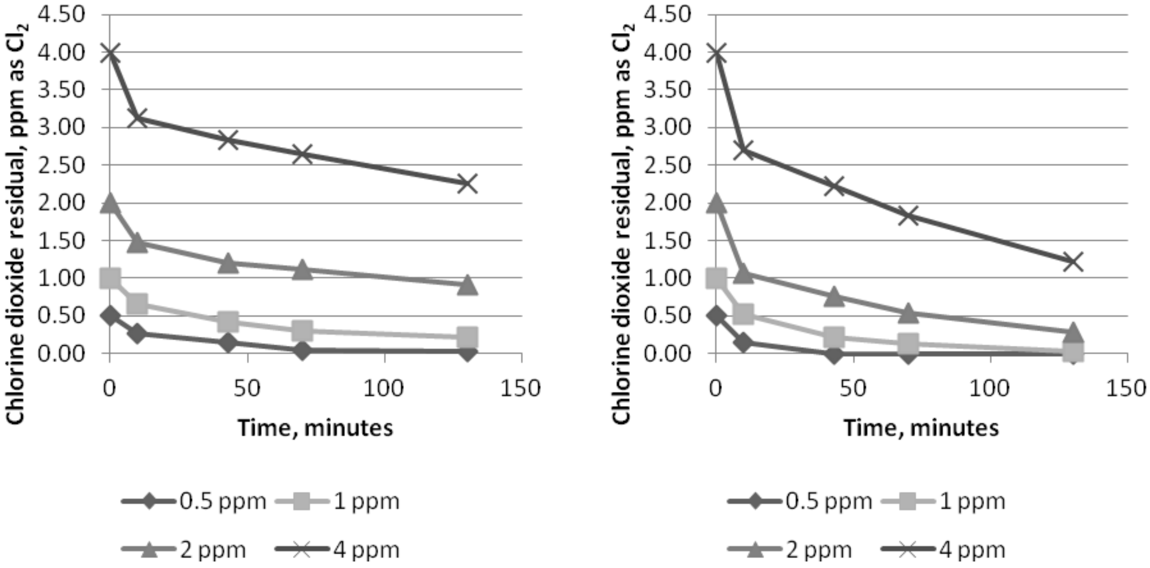


Figure 5.3.6 Chlorine dioxide decay in MWW_NFG as a function of the initial dose at 23ºC (left) and 40 ºC (right)

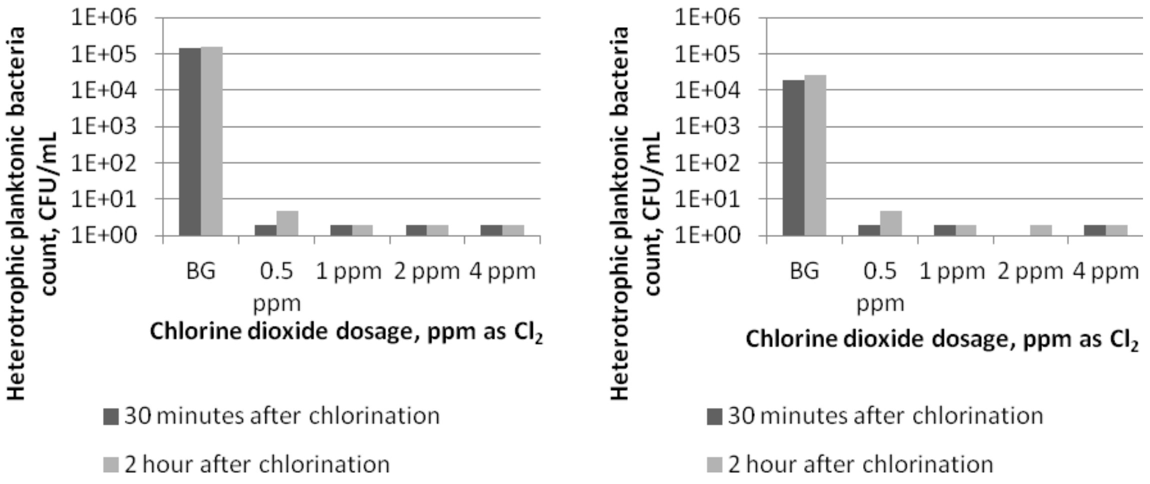


Figure 5.3.7 Heterotrophic planktonic bacteria counts as a function of the initial chlorine dioxide dose and contact time in MWW_NFG at 23 ºC (left) and 40 ºC (right)

Similar to the results with pre-formed monochloramine, water temperature significantly increased the decay rate at 40 ºC. However, experimental results suggest that the disinfection efficiency of chlorine dioxide in MWW_NFG was not impacted by temperature. The bacterial activity can be easily controlled below 10^4 CFU/mL in MWW_NFG with 0.5 ppm of ClO_2 after 2 hours of contact time (Figure 5.3.7).

5.3.3 Bench-scale recirculating studies with MWW_NFG

5.3.3.1 Biofouling potential tests

Control tests of biofouling potential in CoC 4 MWW_NFG were conducted to determine whether the removal of organic matter by activated carbon adsorption would have an impact on bacterial growth rate. Results of the control tests shown in Figure 5.3.8 suggest that the planktonic bacterial population reached exponential growth phase and sessile bacterial population exceeded the biofouling control criterion (10^4 CFU/mL or CFU/cm^2) within 24 hours. Results also suggest that the decrease in TOC did not limit either planktonic or sessile bacterial growth.

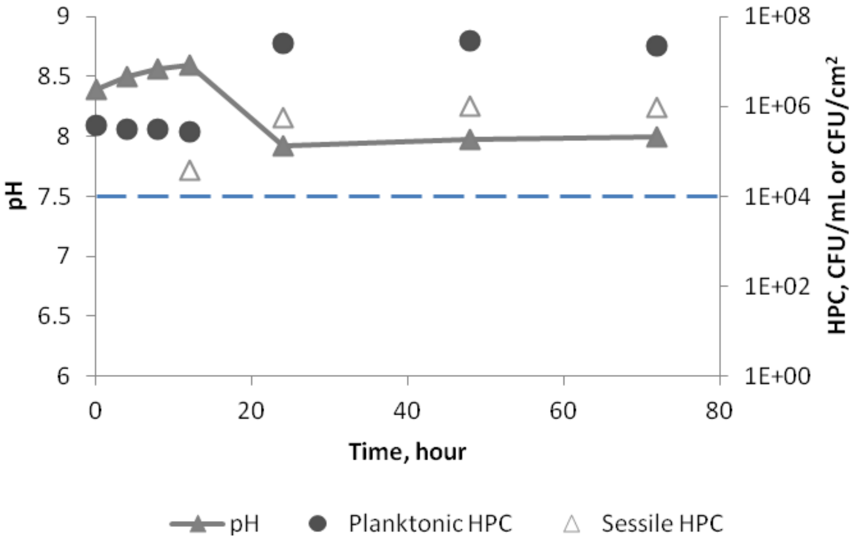


Figure 5.3.8 Planktonic and sessile heterotrophic bacteria growth in CoC 4 MWW_NFG in bench scale recirculating system.

5.3.3.2 Chlorination for biological growth control

Planktonic and sessile bacterial activity in the presence of total free chlorine residual between 3-4 ppm as Cl_2 in the recirculating system is shown in Figure 5.3.9. The pH in the recirculating system was also monitored (Figure 5.3.10) to validate the major chlorine forms in the system. The experiment was repeated due to the large variation of total chlorine residual in the first test. Since the initial inorganic ammonia (0.01 ppm) and organic compounds were low in MWW_NFG (TOC level was 2 ppm), there were no chloramines in the system. The average total chlorine residual was 3.78 ± 1.67 ppm as Cl_2 and free chlorine residual was 3.12 ± 1.63 ppm as Cl_2. In addition, free chlorine residual accounted for 78% of total chlorine residual. This test suggests that maintaining 3-4 ppm of total chlorine residual can control both planktonic and sessile heterotrophic bacteria in the recirculating system. The dosing rate required to maintain 3-4 total chlorine residual in CoC 4 MWW_NFG was 0.35 ppm per hour.

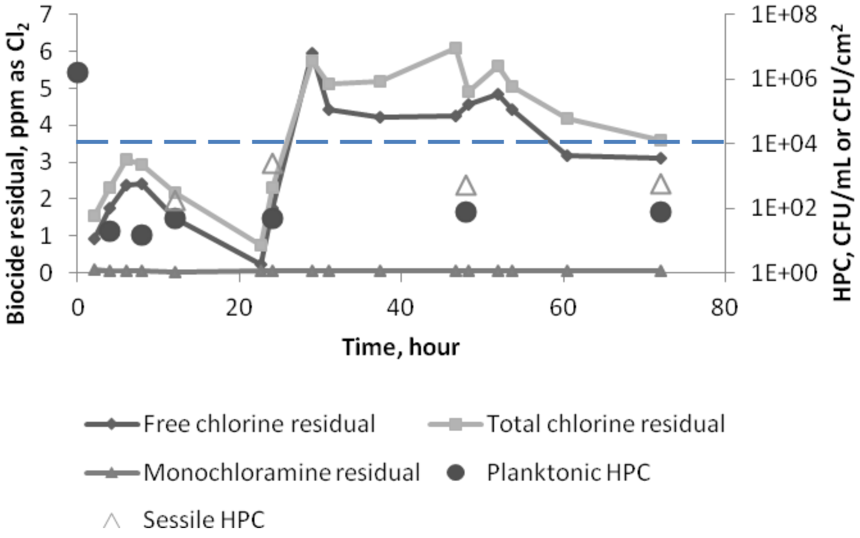


Figure 5.3.9 Total chlorine, free chlorine, and monochloramine residuals and heterotrophic bacteria counts in bench scale recirculating system with CoC 4 MWW_NFG treated with free chlorine

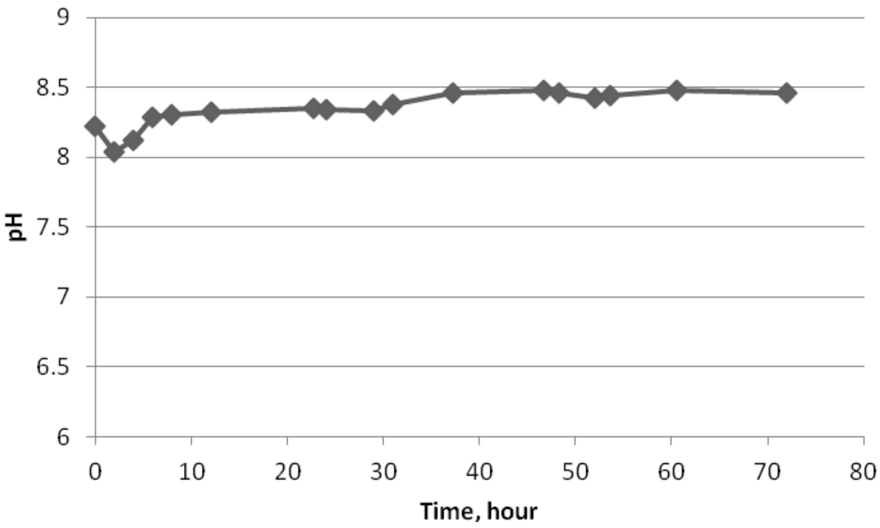


Figure 5.3.10 pH variation in the bench scale recirculating system with CoC 4 MWW_NFG treated with sodium hypochlorite.

5.3.3.3 Chloramination for biological growth control

Planktonic and sessile bacterial activity in the recirculating system in the presence of monochloramine residual ranging between 2-3 ppm as Cl_2 is shown in Figure 5.3.11, while the pH in the recirculating system is shown in Figure 5.3.12. Monochloramine residual averaged 2.90 ± 0.55 ppm of MCA and it was the dominant fraction of the total chlorine residual that averaged 3.30 ± 0.65 ppm as Cl_2 (i.e., approximately 88% of total chlorine). This test showed that maintaining 2-3 ppm of monochloramine residual successfully controlled both planktonic and sessile heterotrophic bacteria in the recirculating system below the industry accepted biofouling control criteria. The dosing rate required to maintain 2-3 monochloramine residual in CoC 4 MWW_NFG was 0.45 ppm per hour.

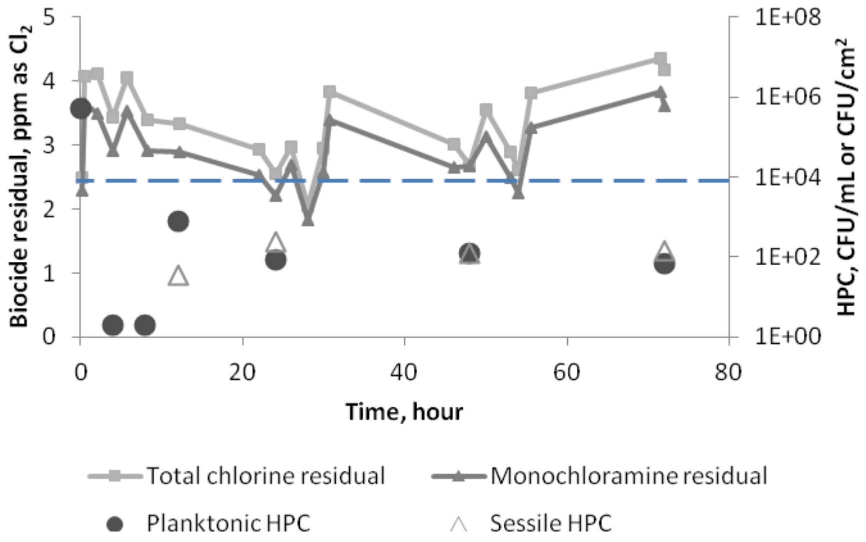


Figure 5.3.11 Total chlorine and monochloramine residual and heterotrophic bacteria counts in bench scale recirculating system with CoC 4 MWW_NFG treated with pre-formed monochloramine

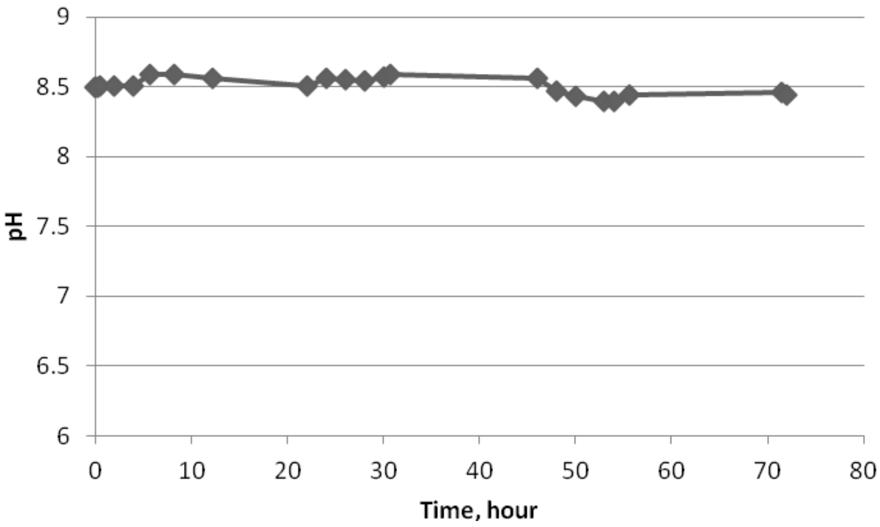


Figure 5.3.12 pH variation in the bench scale recirculating system with CoC 4 MWW_NFG treated with pre-formed monochloramine.

5.3.3.4 Chlorine dioxide for biological growth control

Planktonic and sessile bacterial activity in the recirculating system in the presence of chlorine dioxide residual ranging between 0.25-0.5 ppm as Cl_2 is shown in Figure 5.3.13, while the pH in the recirculating system is shown in Figure 5.3.14. This test demonstrated that maintaining 0.25-0.5 ppm of chlorine dioxide residual successfully controlled both planktonic and sessile heterotrophic bacteria in the recirculating system. The dosing rate required to maintain 0.25-0.5 chlorine dioxide residual in CoC 4 MWW_NFG was 0.70 ppm per hour.

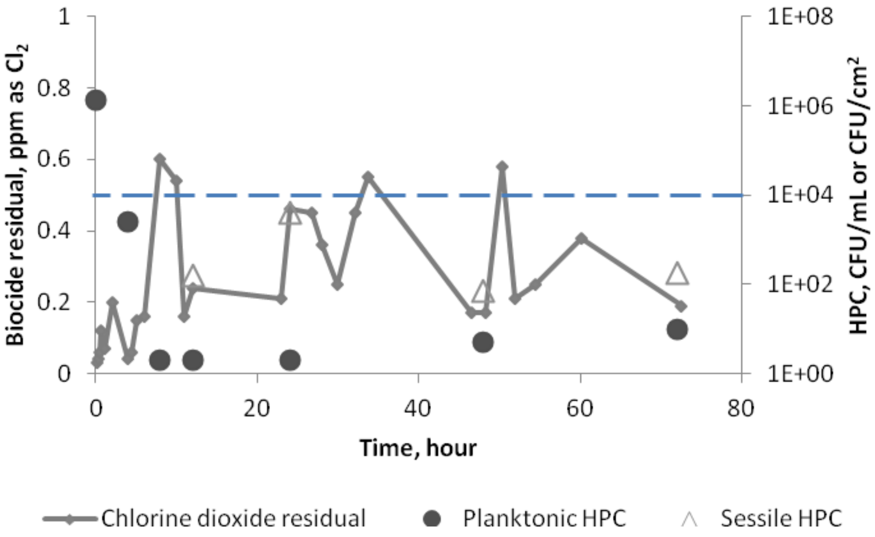


Figure 5.3.13 Chlorine dioxide residual and heterotrophic bacterial counts in the bench scale recirculating system with CoC 4 MWW_NFG treated with chlorine dioxide

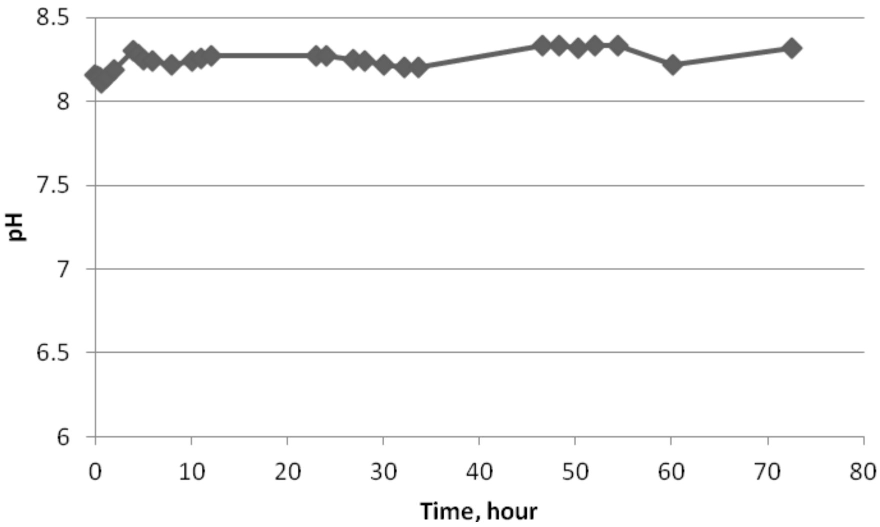


Figure 5.3.14 pH variation in the bench scale recirculating system with CoC 4 MWW_NFG treated with chlorine dioxide.

5.3.4 Pilot-scale studies with MWW_NFG

Biofouling control data for CTC3 during the entire test are shown in Figure 5.3.15. Monochloramine residual in CTC3 was maintained at 2.23 ± 0.64 ppm as Cl_2 throughout the experiment. Monochloramine accounted for 86% of the total chlorine residual and was consistently maintained above 2 ppm throughout the test. Planktonic HPC results were above the biofouling control criterion of 10^4 CFU/ml on Day 13, Day 17, and after Day 25, despite the fact that monochloramine residuals were at reasonable levels on these occasions. The only sessile sample that exceeded the biofouling control criterion was on Day 10. The planktonic HPC appeared to be increasing in the tower but the repeated 10-day sessile sample collected on Day 17 did not reflect the same behavior. In summary, maintaining MCA residual above 2 mg/L was not able to restrain planktonic bacteria below 10^4 CFU/mL while the same MCA residual was able to control sessile bacteria below 10^4 CFU/cm^2 for a period of 30 days. Biocide dosing rate during this test was 2.44 ± 0.72 gallon per day. The monitored MCA concentration in stock solution was 893 ± 66 mg/L throughout the experiment. Similar to results with pre-formed MCA in MWW_pH and MWW_NF, Legionella analysis throughout this test revealed the absence of this microorganism.

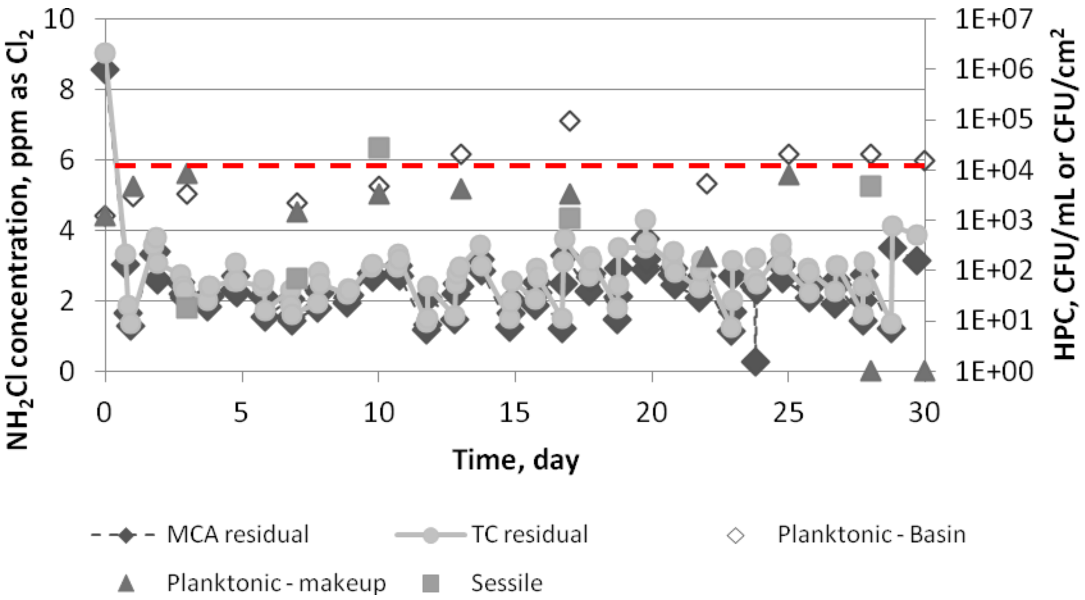


Figure 5.3.15 Disinfectant residual and bacterial counts in CTC3 test (Summer 2011).

5.4 Summary and Conclusions

Corrosion analysis carried out in both bench-scale and pilot-scale studies using secondary treated municipal wastewater with nitrification, filtration, and activated carbon adsorption (MWW_NFG) revealed that MWW_NFG caused pitting corrosion to copper and cupronickel alloys in both bench-scale and pilot-scale studies. Use of 2ppm TTA in the system could not prevent the pitting corrosion of copper and cupronickel alloys in pilot-scale experiments carried out over a period of 28 days. Presence of pitting corrosion in copper and cupronickel alloys (which is unacceptable according to the industrial corrosion control criteria) made the MWW_NFG unsuitable for use in recirculating cooling systems. It was observed that removal of organic matter by granular activated carbon adsorption made the MWW_NFG more corrosive towards copper and cupronickel alloys. Organic matter present in the municipal wastewater provides some corrosion protection through formation of surface complexes with the metal alloys (Choudhury et al., 2012b).

Mild steel corrosion rate was maintained within good levels (according to the industrial corrosion control criteria) after an immersion period of 28 days. The lower corrosion rate of mild steel in the cooling system was due to high scale formation (resulting from higher alkalinity) in MWW at the time period of the experiment.

From the representative information obtained from the second phase, it could be concluded that MWW_NFG showed equally low scaling potential as MWW_NF. The removal of organic matters does not cause significant changes on the scaling characteristics of MWW_NF.

Laboratory scale batch studies revealed that sodium hypochlorite exhibited slightly better results in terms of biocide demand and decay rate in MWW_NFG than in MWW or MWW_NF. A minimum of 0.5 ppm free chlorine residual was required to achieve control of biological growth (10^4 CFU/mL) in MWW_NFG at 40 ºC when the initial heterotrophic planktonic bacteria population was $10^{5.8}$ CFU/mL. Approximately 0.05-0.30 ppm of total chlorine residual that was consistently formed when adding sodium hypochlorite as biocide could not be identified as either monochloramine or free chlorine. Pre-formed monochloramine showed similar results in terms of biocide demand and decay rate in MWW_NFG as that previously observed with MWW_NF. The initial pre-formed monochloramine dose of 1 ppm was required to achieve planktonic bacteria control criteria (10^4 CFU/mL) in MWW_NFG at 40 ºC when the initial heterotrophic planktonic bacteria population was $10^{5.6}$ CFU/mL.

The results of control tests (MWW_NF or MWW_NFG at CoC 4 with no biocide addition) in a recirculating system indicated that the planktonic bacteria population reached exponential growth phase within 24 hours. At the same time, sessile bacterial population exceeded the biofouling control criterion (10^4 CFU/cm^2). The comparison of biological growth between MWW_NF and MWW_NFG indicated that the TOC removal did not limit biological growth.

In bench-scale recirculating tests, removal of TOC was determined to enhance the performance of NaOCl by increasing the active free chlorine residual from 44% to 78% in the concentrated municipal wastewater. For pre-formed monochloramine, the portion of monochloramine did not change significantly as a result of TOC removal. As for chlorine dioxide, it was observed that TOC removal led to lower biocide demand and reduced the dosage required to maintain similar chlorine dioxide residual in a bench scale recirculating system.

Performance of pre-formed monochloramine in pilot-scale cooling systems was more reliable than the other two biocides regardless of TOC level. TOC removal would have greatest benefit for tertiary treated municipal wastewater systems in which NaOCl is used as a biocide if ammonia stripping in the full/pilot scale unit was as effective as in the laboratory unit. However, an undefined portion of total chlorine residual needs to be characterized to ensure that there are no adverse impacts from its release into the environment. Removal of TOC was also determined to be beneficial when chlorine dioxide is used as a disinfectant as it reduces the initial disinfectant demand but not as much as in the case of free chlorine.

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6.0 Life Cycle Costs of Tertiary Treatment Alternatives for Reuse of Secondary Treated Municipal Wastewater in Cooling Systems

Much of the freshwater withdrawal for power production is for power plants employing once-through cooling, which is employed for about 43% of all U. S. power plants (USDOE, 2009). The large water intake for such systems causes negative impacts, including fish impingement and entrainment (USEPA, 2010). Also, blowdown from the once-through cooling returns to the environment at a higher temperature, potentially harming fish and other wild life (Averyt et al. 2010).

To regulate the impacts on aquatic life, the Clean Water Act (CWA) Section 316(b)) has been driving power plants to implement the use of recirculating cooling systems, typically mechanical- or natural-draft wet cooling towers at new and renovated plants (USEPA, 2010). For wet recirculating systems, each kW-hour of electricity generation requires 20-50 gallons of water in once through cooling systems, while only 0.3-0.6 gallons of water is required to generate each kW-hour of electricity in modern re-circulating systems (Veil, 2007).

In areas with limited available freshwater, alternative cooling water sources are of interest. Various alternatives have been explored, including acid mine drainage, ash pond water, oil and natural gas produced water, sea/saline water and industrial and municipal wastewater (Richard, 1964; Paul and Ken, 2003; Veil et al., 2003; Vidic and Dzombak, 2009). Among various alternative sources analyzed for feasibility of use in existing U.S. thermoelectric power plants, secondary treated municipal wastewater (MWW) from POTWs (Publicly Operated Treatment Works) located within 25 miles of power plants could satisfy more than 75% of the cooling water demand for existing power plants assuming that recirculating systems are employed at all plants (Li et al., 2011).

As studied in the previous chapters MWW contains higher concentrations of suspended and dissolved solids, dissolved nutrients and organic matter which increase the challenges of managing cooling water quality in recirculating systems. To avoid scaling, corrosion, and bio-fouling in cooling system piping and heat exchange equipment, the reclaimed water needs to be treated to higher levels. Advanced/tertiary treatment coupled with chemical inhibitor addition to MWW can yield water quality suitable for reuse in cooling systems (Li et al., 2011; Vidic and Dzombak, 2009).

This research focused on conventional tertiary treatment processes including biological ammonia oxidation, chemical precipitation for removal of dissolved hardness, sand filtration for suspended solids removal, granular activated carbon adsorption for organic carbon removal, and disinfection for control of bio-fouling. These treatment methods have significant capital investment and maintenance costs, which is why, except for filtration and disinfection, tertiary treatment processes are not widely adopted by the majority of POTWs in the U.S.

In order to determine the most advantageous tertiary treatment and chemical management alternatives for reusing wastewater in a recirculating cooling system, life cycle cost (LCC) analysis and life cycle inventory (LCI) analysis were employed. Either LCC or LCI can be used for decision making, but there are benefits to using both. A separation of LCI and LCC leaves uncharacterized the important relationships and trade-offs between the economic and life cycle environmental performance of product or process design decision scenarios (Norris, 2001).

Cost estimating procedures for MWW treatment processes were developed by the U.S.EPA in the 1970s (USEPA, 1975), and various proprietary tools building on the U.S.EPA work and other resources have been developed since then. For example, CapdetWorks, a proprietary tool using the U.S.EPA cost estimation algorithms (USEPA, 1975) has been developed to determine budgetary costs for wastewater treatment alternatives (Hydromantis Inc., 2011). Based on the level of estimation defined by Association of Advancement for Cost Engineers (AACE), CapdetWorks and other proprietary tools focus on providing higher-accuracy, construction-level project estimates for precise design take offs. Few methods and tools to estimate conceptual costs at the preliminary stages of a wastewater treatment project are available in the public domain. Conceptual or first-stage estimations are still widely used and needed for economic feasibility decisions. Additionally, the proprietary tools require high level input data that are not easily accessible to academic users and the general public.

In this work a public domain tool was developed for the conceptual, first-stage cost estimation for some conventional tertiary treatment processes for treatment of MWW prior to reuse. In related work, life cycle emissions for the same treatment scenarios have been evaluated (Chapter 7). Life cycle costs (LCC) are those incurred over the life of a project, process or product. Cost estimates at various levels of detail and accuracy are used to support critical decision processes, including the mission need (rough order of magnitude), baseline approval (budgetary), and construction start (definitive or firm). Cost estimation quality should be appropriate for the life-cycle phase and should be documented to provide consistency and standardization considering estimate type, purpose, methodology and contract type (USDOE, 2003).

The primary objective of the research was to develop a life cycle conceptual cost (LC^3) model to estimate LCC for construction and operation of tertiary treatment units and treated water delivery required for MWW reuse as makeup water in power plant cooling systems. For the typical MWW water quality and a particular makeup water flowrate, construction and operational costs calculated using standard databases and first-stage cost estimation methods were applied to compare alternative treatment processes and treatment sequences required to maintain heat exchange efficiency within desired bounds. In addition, sensitivity of the costs to variations in quality and quantity of the influent was studied. Average river water and city potable water supply costs were collected to provide a benchmark for the cost of supplying tertiary treated wastewater as cooling water makeup. For the purpose of validating construction and operational cost estimates from the LC^3 model, results from several test cases were compared with the CapdetWorks software estimates.

6.1 Approach

6.1.1 Goal and scope sefinition

According to the ISO 14040 (2006), first phase of a life cycle study is to define goal and scope of the study, including identifying the intended application, the reasons for carrying out the study, and the intended audience. The scope elucidates the product or process system to be studied, the functions of the system, the functional unit, the system boundary, data requirements, assumptions, and limitations (ISO 14040, 2006)

In this study, life cycle costing for six tertiary treatment alternatives and treated water supply for power plant cooling systems was performed. The methodology adopted to estimate conceptual costs has been represented in the flowchart of Figure 6.1.1. The basic parameters used for design of the treatment units were influent flowrate and water quality characteristics as indicated in Figure 6.1.1. Using defined constants and assumptions from established design manuals for biological nitrification, chemical precipitation, and disinfection and filtration units, design dimensions of the respective units were calculated. The dimensioned concrete structure, and mechanical and electrical equipment required for each treatment unit were then multiplied with the respective unit costs and totaled to obtain the capital costs.

6.1.2 System boundary

The system boundary considered for this study is depicted in Figure 6.1.2. Figure 6.1.2(a) describes the municipal wastewater treatment cycle, and the dashed line outlines the system boundary for this study. Within the boundary, the costs of 1) raw material extraction

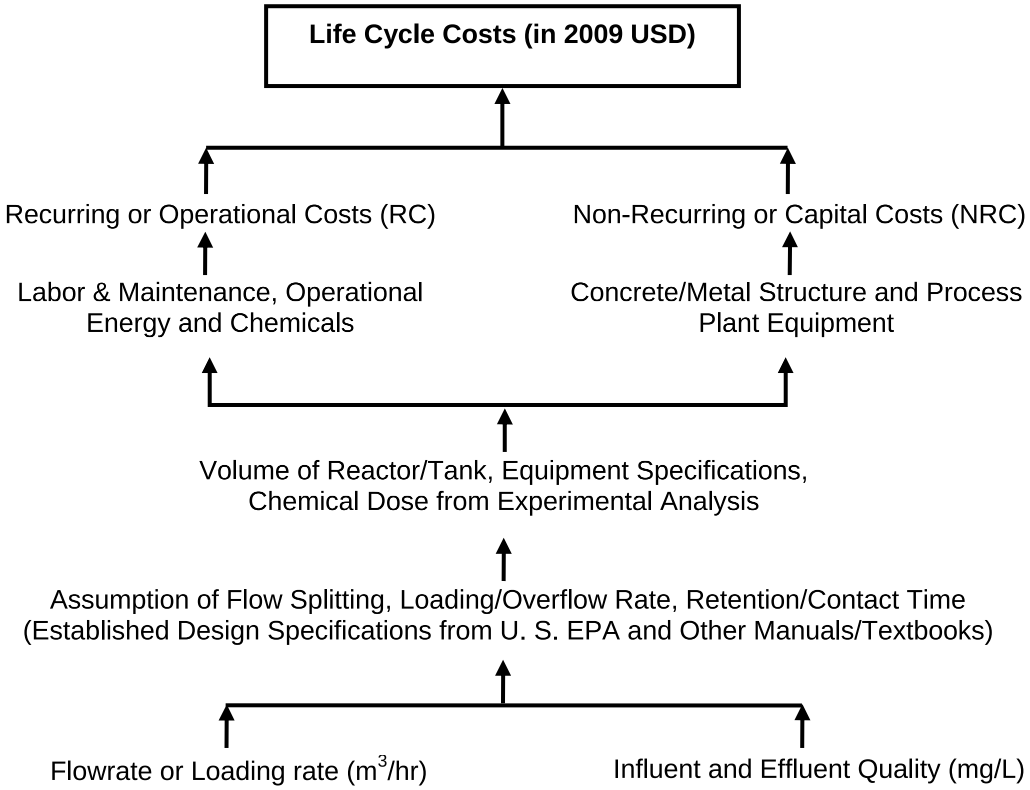

Figure 6.1.1 Flowchart of life-cycle conceptual cost (LC^3) model used for evaluation of municipal wastewater tertiary treatment scenarios

phase for construction and chemical manufacture, 2) fabrication of construction infrastructure and equipment, 3) operation of tertiary treatment processes, 4) production of chemicals for tertiary treatment and conditioning (after tertiary treatment), and 5) treated water delivery up to 10 miles were considered.

Figure 6.1.2(b) expands on the individual tertiary treatment processes considered, including suspended growth nitrification, chemical precipitation for hardness removal, sand filtration for total suspended solids (TSS) reduction, pH adjustment for neutralization, and granular activated carbon adsorption for reduction of TOC. Tertiary treatment alternatives considered were different combinations of the above-mentioned treatment processes; not all treatment processes are required for any chosen tertiary treatment objective.

The individual tertiary treatment processes were selected based on previous findings from bench and pilot scale testing wherein MWW was evaluated for reuse in cooling systems (Vidic and Dzombak, 2009). The above mentioned processes were selectively combined to form various tertiary treatment alternatives/scenarios presented in Table 6.1.1. These alternatives provide varied effluent quality and consumed varied doses of anti-fouling chemicals (to control corrosion, scaling and bio-fouling) as shown in Table 6.1.2. Chemical requirements for conditioning were based on treated water quality as observed during bench and pilot-scale field studies conducted with actual tertiary treated municipal wastewater from Franklin Township Municipal Sanitary Authority (FTMSA) in Murrysville, PA (Chapters 3, 4 and 5). The combined alternative treatment total costs per year (in 2009 USD/kgal) were compared and ranked from the least to most expensive.

6.1.3 Life cycle conceptual cost (LC^3) model development

6.1.3.1 General equations for LC^3 model

Life-cycle costs are those incurred over the life span of a process system, including costs required to construct, equip, and operate the system. The accuracy of the estimate depends on the LCC analysis method used. For each treatment process, a general, annual cost estimation equation was developed consisting of the recurring costs (RC), also known as operation and maintenance costs, and the non-recurring costs (NRC), otherwise known as capital costs, converted to an annual cost basis (Dhillon, 2010). Since the level of estimation was conceptual, the percent contingency applied to the overall annual cost was 35%, which places the LC^3 model estimates within the target accuracy range for conceptual cost estimates (Westney, 1997). The general equation used in the LC^3 model was:

$$\text{Annual LCC} = \text{RC} + \text{NRC} \tag{6.1.1}$$

RC includes annual labor costs, operational energy costs and maintenance (repair) costs plus chemical manufacture and supply costs if the treatment unit requires chemicals for operation. NRC costs include the annualized construction costs and the costs of pumps, motors, auxiliary mechanical and digital equipment; *NRC* can also include initial purchase of activated carbon for the granular activated carbon (G) treatment process. The *NRC* was amortized over the period of the equipment or treatment plant life, i.e., 25 years to estimate the annual life cycle costs.

Amortized costs were calculated by multiplying the net capital investment with the discount rate, which accounts for the time value of money and is also known as the capital recovery factor, *crf*. Amortized capital costs or annual equivalent worth (AEW) represents a stream of *N* constant partial payments *R*, the sum of which equals the capital investment of the project/process. The amortized costs are calculated as follows:

$$\text{NRC (\$)}, R = \text{Initial investment x (crf)} \tag{6.1.2}$$

where, *R* ($) is the amount allocated each year towards capital costs or *NRC* for a period of *N* years representing the service life of the process plant. The service life of the treatment plant and the capital recovery factor used in Eq. (6.1.2) to obtain the results given were considered to be 25 years and 12% (de Neufville, 1990), respectively.

6.1.3.2 Water quality and flowrate considered

The procedure followed for the LC^3 model was initiated with the design of tertiary treatment units shown in Figure 6.1.2(b) using the influent water quality provided in Table 6.1.3 as input. The influent water quality data adopted in this study were average values for actual secondary and tertiary treated effluent quality from FTMSA.

In this study, LCC of the tertiary treatment alternatives were estimated for reuse of MWW in cooling system of a 550 MW baseline pulverized coal power plant cooling system. The functional unit selected for the case study was 7.75 million gallons per day (MGD) of wastewater treated, approximately the daily flow of make-up water required for the 550 MW pulverized coal-fired thermoelectric power plant (USDOE, 2007).

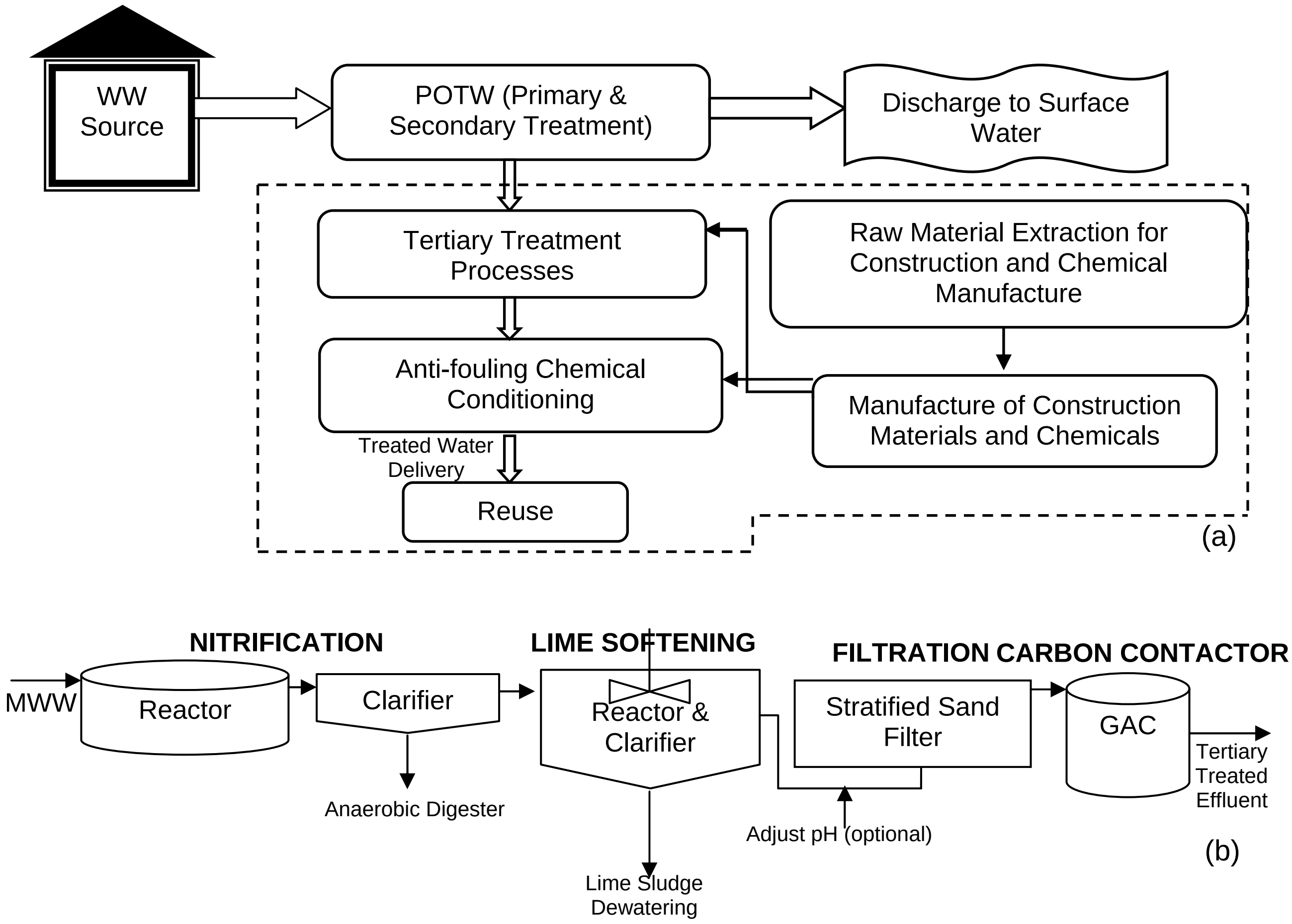

Figure 6.1.2 (a) Treatment of municipal wastewater in POTW prior to reuse. The system boundary for this study is indicated as a dashed line. **(b)** Tertiary treatment processes included in the system boundary

Table 6.1.1 Abbreviation of selectively combined tertiary treatment alternatives/scenarios

Abbreviations	Tertiary Treatment Processes Included
RW	River water pumped from source and filtered
MWW_F	Filtration (F) only with addition of chemical conditioning agents
MWW_NF	Nitrification (N) and filtration (F) with addition of chemical conditioning agents
MWW_SF	Softening (S) and filtration (F) with addition of chemical conditioning agents
MWW_NSF	Nitrification (N), Softening (S) and filtration (F) with addition of chemical conditioning agents
MWW_NFG	Nitrification (N), filtration (F) and granular activated carbon (G) treatment with addition of chemical conditioning agents
MWW_pH	pH adjustment (pH) with addition of chemical conditioning agents
WSI&D	Treated water delivery infrastructure (pipeline and pumps) and energy to pump water to an average 10 mile distance between POTW and TPP
CW	City water which has been treated for commercial and/or industrial use

Table 6.1.2 Doses of anti-fouling chemicals used in bench and pilot-scale testing of tertiary MWW for use in cooling systems (Chapters 3, 4 and 5)

Anti-Fouling Chemicals	Chemical Dose Based on Level of Tertiary Treatment (ppm)						
	RW	MWW _F	MWW _pH	MWW _NF	*MWW _SF	*MWW _NSF	MWW _NFG
Tolytriazole (TTA)	1.0	1.0	1.25	1.0	1.0	0.5	0.5
Poly Maleic Acid (PMA)	1.25	1.25	1.25	1.25	0	0	0
Monochloramine (MCA)	52	64	70	52	52	52	43

* MWW_SF and MWW_NSF bio-fouling is assumed identical to MWW_NF fouling, with the assumption that chemical precipitation of hardness (S) does not decrease total organic carbon (TOC).

6.1.3.3 Unit costs of materials and factors

From design analysis conducted for each treatment unit, a list of major components required to construct the unit was developed, and the design dimensions/specifications were listed. Unit costs for the major equipment list for each treatment unit were collected from construction and process plant cost databases and by contacting vendors. Common cost factors and unit costs for major construction materials and equipment used in the cost calculations are provided in Tables 6.1.4 and 6.1.5.

6.1.3.4 Conceptual cost estimation procedure

All estimated costs were referenced to 2009, the year in which most of the cost data were obtained from vendors and databases, or were adjusted to year 2009 costs using the Engineering News Record (ENR) construction and material cost indices. The cost of each unit was calculated by multiplying the unit cost with the respective unit type and number of the units among which the flow was divided.

Concrete structure costs were calculated by separating the structure into three components: the concrete wall (can be flat or curved) with thickness of 12", the concrete slab/floor with thickness of 18", and the foundation which was assumed to be 25% of the cost of concrete slab with a thickness of 8-12". Foundation costs do not include the site work as the required details of elevation and type of soil, are site specific.

Equipment costs were obtained from vendors selling water/wastewater treatment equipment. If quotes were not available for the required plant capacity, then a process cost estimating rule-of-thumb known as the 6/10th rule (Peters et al., 2003) was used:

$$C_n = C_o\,(S_n/S_o)^{0.6} \qquad (6.1.3)$$

where, C_n = Cost of new equipment,

C_o = Cost of existing equipment with pricing available,

S_n = Size/Capacity of new equipment, and

S_o = Size/Capacity of existing equipment.

The sum of the project capital costs was calculated by totaling the construction or prefabricated metal structure costs with major equipment costs and adding 15% (Peters et al., 2003) of the major equipment cost for auxiliary equipment such as controllers, sensors, and support structures. Additionally, a contingency of 35% of total project capital costs was added to the project capital costs, and then the total including contingency was amortized to obtain annual capital costs.

Operations costs were calculated by including 5 - 10% of total project capital costs (including contingency) for labor and maintenance, plus optional chemical costs and the specific electricity/energy requirements (with no contingency added).

Finally, the amortized capital costs were summed with the annual operational costs to obtain the total annual cost. Detailed cost estimate equations used to design and cost individual treatment units are documented in Appendix E and the LC^3 model can be downloaded from the project website under the ‘Resources’ page/tab (http://cooling.ce.cmu.edu/).

Table 6.1.3 Secondary and tertiary treated water quality from FTMSA used for design of tertiary treatment units (Vidic and Dzombak, 2009)

Parameters (mg/L unless noted)	Secondary	Tertiary
Calcium (Ca)	41.5	39.7
Magnesium (Mg)	10.7	9.8
Manganese (Mn)	0.32	0.34
Ammonia-N (NH_3-N)	21.0	ND*
Nitrate (NO_3-N)	3.6	18.1
Total Phosphorus	4.5	3.8
HCO_3 Alkalinity (mg/L as $CaCO_3$)	177.0	25.4
Total Alkalinity (mg/L as $CaCO_3$)	177.0	25.4
Biochemical Oxygen Demand (BOD)	31.9	5.8
Total Organic Carbon (TOC)	27.0	8.7
Total Dissolved Solids (TDS)	661	473
Total Suspended Solids (TSS)	40.7	20.8
Specific Conductivity (SC) (mS/cm)	1030	739
pH	7.2	6.8
Turbidity (NTU)	16.7	6.2
Total Coliform (No./100 mL)	10^6-10^5	10^4-10^5

*ND – Not Detectable

Table 6.1.4 Factors assumed for conceptual cost estimation

Factors	% Total Costs	Source
Discount factor (%)	12	de Neufville, 1990
Contingency (%)	35	Westney, 1997
Labor and Maintenance (%)	8	Ray and Sneesby, 1998
Foundation Costs (%)	25	Veolia Water North America, 2009*
Auxiliary Equipment Costs (%)	15	Ray and Sneesby, 1998

*Assumed based on personal conversation with cost estimation specialist

6.1.3.5 Potential freshwater alternatives costs – Literature survey data

In order to provide a benchmark for comparison of costs of treated wastewater with costs of other potential sources of cooling water, average costs of potable water supply were collected from a national survey of the 50 largest U.S. cities (Black & Veatch, 2010). As the tertiary treatment cost estimates were referenced to 2009, the survey result was considered as the primary reference for benchmarking or cost comparison. A second benchmark for comparison of treated wastewater costs was pumped river water. River water withdrawal rates or charges were available from three different sources: Delaware River Basin Commission (DRBC, 2011), Minnesota Department of Natural Resources (MDNR, 2011), and Water Management Program advisory committee of the Lower Colorado River Association (WMP-LCRA, 2011).

From Table 6.1.6, the river water withdrawal fee costs for three different sources mentioned above are listed.

6.1.3.6 Validation of LC^3 model estimates

For the purpose of validation, LC^3 model results referenced to 2007 USD using ENR indices were verified against CapdetWorks software (Hydromantis Inc., 2011) estimates. CapdetWorks is a preliminary design and costing program that uses a unit costing approach with an extensive costing database that accounts for the effects of inflation using cost indices. The current version of CapdetWorks (v2.5) references estimates to 1997, 2000 and 2007.

For the validation simulations, design constants and unit cost inputs to the CapdetWorks software were maintained consistent with those used in the LC^3 model, except for assumptions on labor costs, units to split the flow among, type of chemical precipitation unit, and the engineering, overhead/profit fees which are not included in the LC^3 Model. CapdetWorks contains default design equations and accounts for budgetary unit costs to provide higher accuracy estimates. Cost estimates for suspended growth nitrification, lime softening, pH adjustment, sand filtration and granular activated carbon process were simulated using CapdetWorks. Since CapdetWorks doesn't include water supply system and chemical conditioners used in this study, treated water supply system and chemical conditioning process were not validated.

Table 6.1.5 Unit costs of concrete structures, metal equipment and material as obtained from the vendor or cost database

Materials (units)	Unit Costs (in 2009 USD)	Source
Flat concrete wall (/CY)	$387.00	Richardson Cost Data, Ch. 3
Curved concrete wall (/CY)	$793.00	Richardson Cost Data, Ch. 3
Elevated slab (/CY)	$617.00	Richardson Cost Data, Ch. 3
Aerator costs (/SF)	$15.00	Veolia Water North America*
Centrifugal pump for sludge (/50 GPM)	$4,000.00	Richardson Cost Data, Ch. 100
Clarifier scraper (/30 ft diameter)	$50,000.00	Veolia Water North America*
Mixer (/1 HP) & impeller (/ft diameter)	$7,000.00	Richardson Cost Data, Ch. 100
Chemical metering pump(/100 GPH)	$500,000.00	Richardson Cost Data, Ch. 100
Lime milk production and feed system (/unit)	$250,000.00	Con-V-Air Solutions*
Backwash pump/15 gpm(/SF)	$20,000.00	Enviroquip*
Filter sand(/CF)	$4.50	Northern Filter Media, Inc.*
Filter underdrains+air blower(/SF)	$175.00	Enviroquip*
Filtrasorb 300 – activated carbon (/lb)	$1.80	Calgon Carbon*
Filtrasorb 300 – regenerated carbon (/lb)	$0.64	Calgon Carbon*
Cost of natural gas (/1000 CF)	$10.00	U.S. EIA, 2009
20 inch diameter pipe costs (/18 ft)	$127.00	American Cast Iron Pipe Co.*
Excavation and fill (/0.5 CY)	$12.00	2009 R. S. Means
Cost of bedding material (/ft)	$8.00	2009 R. S. Means
Centrifugal pump horizontal split, single stage	$34,600.00	Richardson Cost Data, Ch. 100
Centrifugal pump vertical split, single stage	$21,000.00	Richardson Cost Data, Ch. 100
Sludge dewatering equipment	$150,000.00	Phoenix Process Equipment Co.*
Hauling cost (/hr)	$120.00	Waste Management*
Landfill/disposal cost (/ton)	$35.00	Waste Management*
93% sulfuric acid (H_2SO_4)(/750 lb)	$185.00	Brainerd Chemical Inc.*
Monochloramine (MCA) (/lb)	$0.46	Kroft Chemicals*
Tolytriazole (TTA) (/lb)	$2.75	Kroft Chemicals*
Polymaleic acid (PMA) (/lb)	$3.12	Kroft Chemicals*
Hydrated lime(/short ton)	$136.00	USGS, 2010
Soda ash (/lb)	$0.07	USGS, 2012

Note: CY = cubic yard, SF = square feet, GPM = gallon/minute, HP = horse power, GPH = gallon/hour, CF = cubic feet

*Price quotes obtained during personal conversation

Table 6.1.6 Raw river water rates from various sources

Source of river water rate	Rate ($/kgal)
[a]Delaware River Basin Commission Rates	$0.08
[b]Minnesota Dept. of Natural Resources	$0.42
[c]Lower Colorado River Basin	$0.46
Average River Water Costs	$0.30

[a] DRBC, 2011; [b]MDNR, 2011; [c]WMP-LCRA, 2011

6.1.3.7 Assumptions

To keep the procedure for estimation of costs simplified and to fill in the gaps of missing data, some assumptions were used for the development of the LC^3 model, which is common to the conceptual level of cost estimation in life cycle studies. The assumptions were as follows:

1) A typical and constant MWW quality was assumed for design purposes.

2) Approximate prices of concrete structures as obtained from standardized cost databases such as 2009 Richardson Process Plant Costs and R. S. Means were used to estimate capital costs for each unit.

3) Only major equipment in each treatment unit (e.g., agitators, aerators/blowers, and rotary distributor pump) was considered for calculation of power consumption.

4) The treatment units were assumed to be connected to each other so that the water flow is driven by gravity rather than by pumping.

5) The cost for de-construction of the treatment units at the end of the life cycle/span was not considered.

6) Makeup water demand at the power plant is constant and the accounts for seasonal and daily variations in water availability (EPRI, 2002). Thus, this project assumed that the design peak factor is equal to one.

7) Level of tertiary treatment to be provided for reuse of MWW in cooling system was based on desired feed water quality determined in related pilot scale studies.

8) For transport, a distance of 10 miles was selected as 85% of the existing municipal wastewater treatment plants and thermo-electric power plants are co-located within this distance in the U.S. (Vidic and Dzombak, 2009).

6.2 Results and Discussions

6.2.1 Individual tertiary treatment processes

With the limitation in scope of the study of tertiary treatment for secondary municipal wastewater for reuse in cooling system, cost analyses did not include the primary and secondary treatment process, or the costs incurred post reuse of the treated water. Figure 6.2.1 shows the estimated costs for individual tertiary treatment processes and the treated water delivery costs from the POTW to the point of reuse. The total annualized costs (in 2009 USD/kgal) to treat MWW can be divided into construction infrastructure costs (which include equipment), operational electricity, labor and maintenance, chemical manufacture and supply, and activated carbon regeneration costs for GAC unit.

In processes used for hardness ion precipitation with lime and pH adjustment with sulfuric acid (H_2SO_4), the chemical production and transport costs were determined to be the major component of the total direct costs. For the pH adjustment treatment option, addition of dilute sulfuric acid at regular intervals (0.05 M) was assessed to reduce pH from 8.3 to 7.6, which was shown to prevent precipitation of the dissolved solids, in cooling water recirculation loop (Chapters 3, 4 and 5). For granular activated carbon adsorption, the activation of spent carbon (regeneration costs plus 10% of new carbon to replace loss during regeneration), which includes annualized costs for initial purchase of granular activated carbon, dominates the overall treatment costs. The biological ammonia oxidation was also quite expensive, due to the aeration infrastructure costs and high electricity requirement for maintaining dissolved oxygen level. Filtration was found to be the most economic treatment process due to the low capital investment and low maintenance requirement with automated filter backwash. A treated water supply has approximately equal operational and construction costs due to daily pumping costs.

6.2.2 Cost comparison among tertiary treatment alternatives

Figure 6.2.2 describes the total direct annual costs (in 2009 USD/kgal) for selected tertiary treatment alternatives combined with annual cost for anti-fouling chemicals used for conditioning, and cost to transport water from the treatment plant to the point of reuse. Raw MWW costs were considered for referencing estimated costs with other source water alternatives. The error bars represent the ±35% contingency for conceptual (first-stage) cost estimates to show maximum and minimum range of total costs along with the added lower and higher rates charged for acquisition of MWW.

In a survey shown in Appendix D, the power plant personnel were questioned regarding the amount paid to wastewater treatment plants for reuse of wastewater in the power plant cooling system. Seventy percent of these respondents reported payments below \$0.66 per 1,000 gallons (in 2009 USD) for secondary treated MWW. The price paid for raw secondary treated MWW with no tertiary treatment and minimal supply fee was taken to be 0.18\$/(1000 Gal) in this case study based on one of the relevant responses in the survey.

The error bars for RW and CW represent the lower and higher range of charges paid by consumer as per the literature (DRBC, 2011; MDNR, 2011; and WMP-LCRA, 2011) and national survey data collected (Black & Veatch, 2010), respectively.

As shown in Figure 6.2.2, anti-fouling chemical costs were observed to be relatively high for all tertiary treatment options, and especially for MWW_pH and MWW only (\$0.33/kgal). The MWW_pH scenario costs reflect the acid addition (\$0.25/kgal), rendering it to be the more expensive tertiary treatment alternatives. The anti-fouling chemical management costs decreased with improved water quality as in MWW_NSF ($0.18/kgal), MWW_NFG ($0.21/kgal). However, the higher level of treatment, consumes greater amount of resources and overall treatment costs. Thus, MWW_SF and MWW_NF appear to be the most cost-effective tertiary treatment options, with moderate anti-fouling chemical management costs ($0.22/kgal and $0.26/kgal respectively) and overall treatment costs ($1.10/kgal and $1.11/kgal respectively).

6.2.3 Tertiary treated source cost comparison with other potential freshwater sources

River water withdrawal with minimal treatment is the least expensive ($0.74/kgal) alternative source of feed water to the cooling system, which is why most power plants are located near freshwater sources and once-through cooling systems have traditionally been employed. However, as noted earlier, once-through cooling involves large withdrawals and adverse environmental impacts and is being discouraged by regulations. On the other hand, the CW utility average charge ($2.95/kgal) is quite high and CW does not appear to be a feasible alternative feed water source for use in power plant cooling systems. As shown in Figure 6.2.2, the estimated tertiary treated municipal wastewater cost using the LC^3 model was found to be between the rate charged for river water withdrawal with filtration and chemical conditioning i.e., an average of $0.74/kgal for some areas of the U.S., and the national average rate for potable city water i.e., $2.95/kgal.

Comparing CW costs i.e., $2.95 /kgal, which is the national average for industrial and commercial customers, against the lower cost tertiary treatment alternatives MWW_SF with water supply costs added ($1.10/kgal), the latter source is significantly less expensive and a better alternative for cooling system feed water. Although RW and CW are higher quality alternatives compared to MWW_SF effluent, an important consideration is that the reuse of treated water would lead to preventable surface water withdrawal and cost-efficiency in the long term.

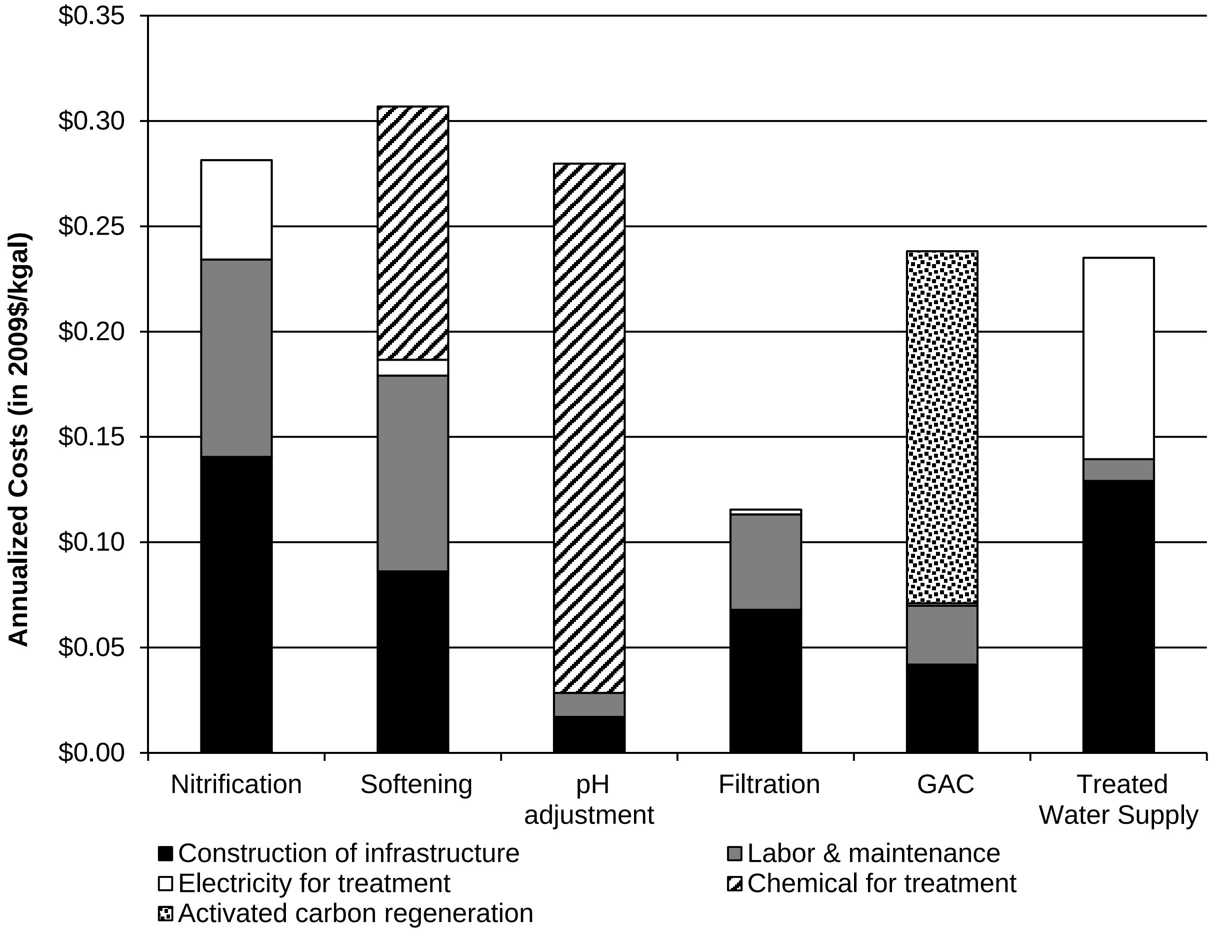

Figure 6.2.1 Total annualized cost (in 2009 USD/kgal) for selected conventional tertiary treatment process used to treat 7.75 MGD secondary treated municipal wastewater
Note: Lime softening includes addition of hydrated lime at 212 mg/L, sludge handling and disposal costs; pH adjustment considers continuous addition of H_2SO_4 at 6.15 mM and; Treated water supply considers delivery of effluent from POTW to the thermo-electric power plant.

6.2.4 Validation of LC^3 model estimates

Results of the validation using CapdetWorks are shown in Figure 6.2.3 where it may be seen that the LC^3 model estimates were in accordance with CapdetWorks estimates for the various tertiary treatment units. The error of approximation was found to lie within the contingency percentage for the respective models. Difference in total costs for units using chemical addition or activated carbon for treatment were observed due to the changes in land costs, type of reactor, regeneration energy used, and the operational labor accounting.

The operation and maintenance (labor, chemicals/activated carbon and electricity) costs in both models contribute most to annualized cost, except in the case of granular activated carbon process estimates, wherein infrastructure costs contributed the most due to type of filter unit (metal or concrete). Labor costs are influenced by the number of personnel, which is in turn dependent on the degree of automation assumed (Tsagarakis et al., 2003). Although automation requires considerable capital outlay as well as specialized personnel to operate the plant, there is a concomitant reduction in number of employees needed (Drake and Page, 1981). Automation and control is an important factor that influences the overall energy demand (Tsagarakis et al., 2003). On a similar note, higher energy requirements in case of activated carbon process are based on the assumption of the regeneration being performed on-site rather than off-site.

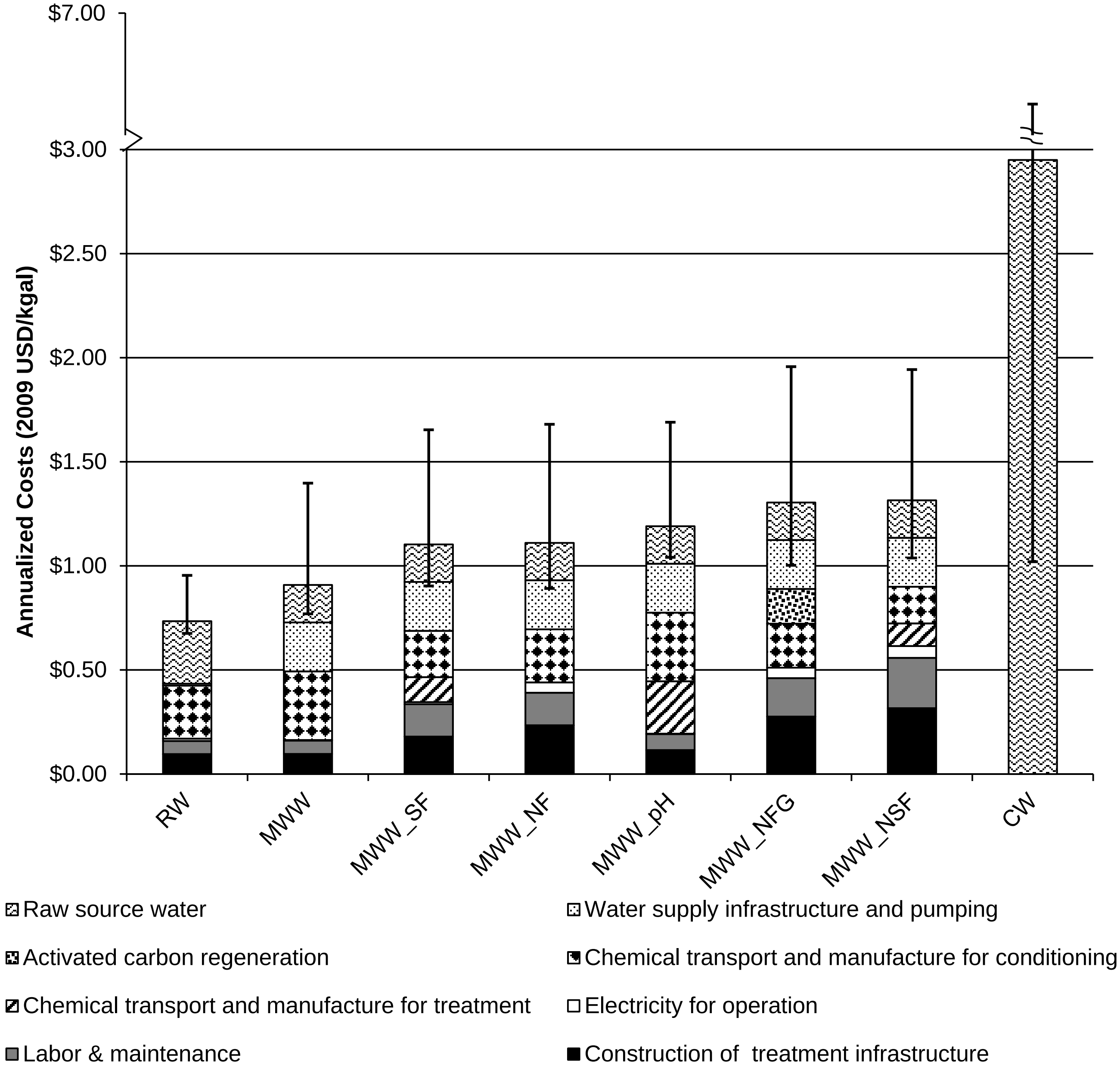

Figure 6.2.2 Comparison of LC^3 model estimates (2009$/kgal) for tertiary treatment alternatives with documented average river water and city water prices referenced to year 2009. Costs developed for treatment capacity of 7.75 MGD

Note: 1)The costs presented in the graph are base treatment costs that include nominal charges for the secondary treated wastewater and supply fees for 10 mile delivery distance and additional chemical management, but do not include additional service or taxes. Secondary treated raw municipal wastewater cost was assumed to have a base value of approximately $0.18/kgal and was added to the tertiary treatment costs when comparing total costs with city water (CW) and river water (RW) costs.
2) kgal = kilo gallon, Nitrification (N), Lime Softening (S), pH adjustment (pH), Filtration (F), Chemical management (C) for anti-corrosive agent –Tolytriazole (TTA), anti-scaling agent – Polymaleic Acid (PMA) and biocide – Monochloramine (MCA)
3) Established RW and CW rates were obtained from state or regulatory agencies (Delaware River Basin Commission, 2011. Minnesota Department of Natural Resources, 2011, WMP Advisory Committee – LCRA Raw Water Rates, 2011 and Black & Veatch, 2010).

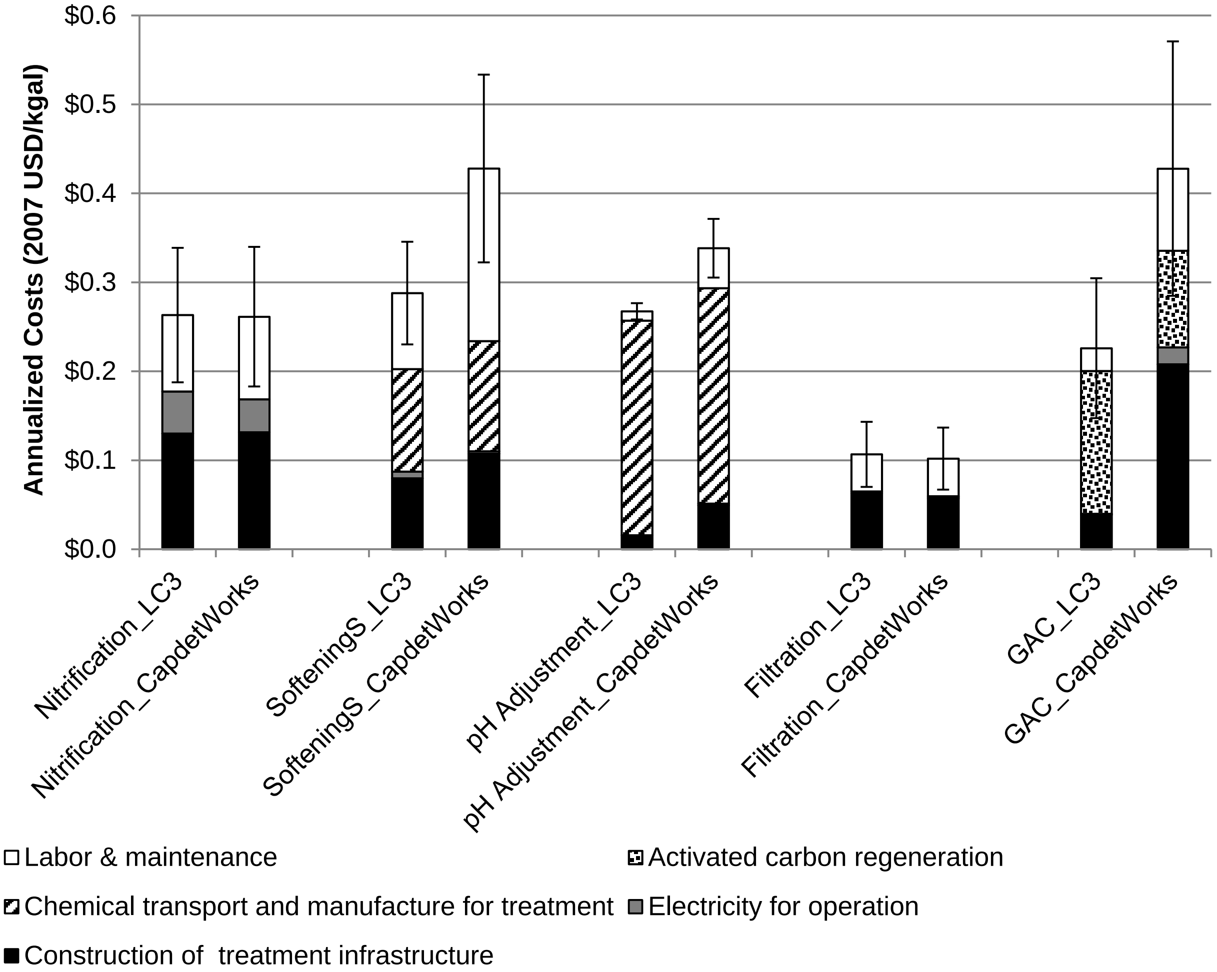

Figure 6.2.3 LC^3 model results (in 2007 USD/kgal) validation using CapdetWorks (CapW) cost estimation software. Note: LC^3 model costs were referenced to 2007 USD using ENR cost indices.

6.3 Conclusions

Life cycle cost (LCC) analysis was performed for selected conventional tertiary treatment processes (Nitrification - N, Lime softening - S, Sand filtration - F, pH adjustment – pH, Granular activated carbon treatment - G, and Chemical conditioning - C) and combined treatment alternatives (MWW_F, MWW_NF, MWW_SF, MWW_pHF, MWW_NSF and MWW_NFG) to treat secondary treated municipal wastewater (MWW) for reuse in thermoelectric power plant cooling. A public domain conceptual costing tool (LC^3 model) was successfully developed and used to determine LCC for selected conventional tertiary treatment processes and combined treatment alternatives to treat MWW for reuse in thermoelectric power plant cooling systems. Estimated costs for the selected tertiary treatment scenarios were compared with established costs of river water and city water supply as alternative sources of cooling water.

MWW with advanced treatment and addition of conditioning chemicals is promising as a power plant cooling system makeup water based on the cost range of $0.91 - $1.32 (in 2009$/kgal), which lies within the river water withdrawal and treatment costs of $0.74 (in 2009$/kgal) and the average city water costs of $2.95 (in 2009$/kgal). By applying the LC^3 model to the case of a MWW effluent with flowrate of 7.75 Mgal/d, the typical make-up water requirement for a 550 MW thermoelectric power plant, it was found that treated water supply and chemical conditioning costs dominated the overall cost. MWW_SF and MWW_NF appear to be the more cost-effective treatment options among the tertiary treatment alternatives considered in this study because of the higher effluent quality with moderate infrastructure costs and the relatively low doses of conditioning chemicals required. It was also found that reuse of MWW would lead to preventable surface water withdrawal and cost-efficiency in the long term when compared to using river water or city water as makeup water source.

Given the uncertainty of source water costs with limited availability of alternative sources for industrial uses and variability of the total costs with tertiary treatment design parameters, conceptual cost estimation needs to be combined with other tools to make better informed decisions. To determine the most cost-effective and environmentally viable tertiary treatment alternatives with addition of conditioning chemicals, the indirect environmental emissions and damages need to be evaluated.

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7.0 Direct and Environmental Costs of Tertiary Treatment of Municipal Wastewater for Reuse in Cooling Systems

Secondary treated municipal wastewater (MWW) reused as makeup water for cooling systems at thermoelectric power plants has been investigated to reduce withdrawal of freshwater for cooling, and to decrease water contamination and withdrawal-related impacts on aquatic life (Li et al., 2011). As shown in the Chapters 3, 4 and 5, though secondary wastewater is a reliable and easily accessible resource, its higher contaminant concentrations increase the scaling, corrosion, and bio-fouling control challenges of managing cooling water quality in recirculating cooling systems (Vidic and Dzombak, 2009). These challenges however can be resolved with advanced treatment of MWW prior to reuse in power plant cooling systems (Li et al., 2011; Vidic and Dzombak, 2009).

Selecting an appropriate level of tertiary treatment of MWW prior to reuse in a recirculating cooling system is an optimization problem. Advanced tertiary treatment processes for industrial water reuse applications is generally chosen based on economic feasibility and effluent quality requirements; rarely is the level of treatment to be imparted decided based on environmental inventory emissions or the economic consequences of those emissions. Construction and operation of advanced or tertiary treatment systems incur not only significant capital and operation and maintenance (O&M) costs, but also lead to consumption of resources such as energy, coal for activated carbon and chemicals, that need to be taken into account when selecting a treatment strategy for MWW reuse (Chapter 6).

Extraction of raw materials and production of equipment, chemicals and energy required for tertiary treatment with conditioning release environmental emissions (such as greenhouse and acidifying gases and particulate matter) that are largely unexamined due to lack of incentives e.g., credits to reduce consumption of treatment chemicals, to study and quantify these environmental emissions. Previous studies have made efforts to evaluate the net impact of treatment technology created in the pursuit of stringent water quality objectives (Antonucci et al., 1975). Tools and/or methods devised to estimate the direct and indirect costs (Hydromantis Inc., 2011), energy and resource consumption (Racoviceanu et al., 2007; Pasqualino et al., 2010; Tillman et al., 1998) and air emissions (Stokes and Horvath, 2009; Meneses, et al., 2010; Pasqualino et al., 2009, 2010; Ortiz et al., 2006; Beavis and Lundie, 2003; Bagley, 2000; Roeleveld et al., 1997) associated with wastewater treatment have been developed. These studies have identified energy consumption, chemical manufacturing, sludge handling and disposal to be the main contributors to costs and environmental emissions (Pasqualino et al., 2009).

Integration of external environmental and human health costs with economic costs is one approach toward a welfare-based measure of pollution output for any industrial process. One study on the environmental accounting for pollution in the U.S. economy (Muller, et al., 2011) showed that the sewage treatment sector had air pollution damages higher than the value added by its service which was because of the lack of means to measure valuable non-market services by prices in the national accounts. Although services for the sewage treatment industry are accounted by the costs of production such as sewage fees landfill, tipping fees for sludge disposal, and land application costs for biosolids disposal, these fees fail to account for some of the non-market services and impacts, e.g., the external costs resulting from air emissions (Muller, et al., 2011). Though this study does not attempt to provide measures for all non-market services and impacts of the tertiary wastewater treatment processes considered, it does make an effort to determine the economic consequence of specific air emissions released during construction and operation of tertiary treatment processes used to treat MWW for reuse in cooling systems.

The objective of this study was to integrate life-cycle cost estimates and life-cycle inventory emissions calculated for tertiary treatment of MWW in terms of monetary cost to aid in selection of a cost-effective and environmentally sustainable approach for tertiary treatment of MWW. The annual external costs due to greenhouse gas emissions (ton CO_2 eq.), acidifying gas emissions (kg SO_2 and kg NO_x) and particulate matter emissions (kg $PM_{2.5}$) from construction (fabrication of treatment infrastructure and equipment) and operation (energy, chemical and other supplies production) of MWW tertiary treatment alternatives were estimated using a hybrid LCA approach and external unit damage cost factors (in 2010$/ton or 2002$/kg). The effects of other emissions to air, water and land, as well as benefits stemming from reduced freshwater withdrawal for cooling systems were not estimated due to lack of established cost factors for those emissions and benefits.

7.1 Materials and Methods

7.1.1 Life cycle inventory assessment

LCA is a technique to assess the environmental aspects and potential impacts associated with a product, process, or service. LCA involves the following components: (a) compiling an inventory of relevant energy and material inputs and environmental releases; (b) evaluating the potential environmental impacts associated with identified inputs and releases;

and (c) interpreting the results to help make a more informed decision (USEPA, 1993). There are four phases in an LCA study (ISO 14040, 2006): 1) the goal and scope definition phase, 2) the inventory analysis phase, 3) the impact assessment phase, and 4) the interpretation phase.

In this study, life cycle inventory and external cost assessment for six tertiary treatment alternatives and treated water supply for power plant cooling systems were evaluated. The methodology adopted to estimate conceptual costs has been represented in the flowchart of Figure 7.1.1. Application of hybrid life-cycle inventory analysis can be advantageous for decision makers to choose among wastewater tertiary treatment alternatives for wastewater reuse in cooling systems.

The combined tertiary treatment processes for which LCI emissions were determined are as listed in Table 6.1.1 and the system boundary for the study is as given in Figure 6.1.2(a) and (b) and include various combinations of units for biological ammonia oxidation, chemical precipitation for removal of dissolved hardness, sand filtration for suspended solids removal, granular activated carbon adsorption for organic carbon removal, and disinfection for control of bio-fouling. Treated water piping from point of treatment to power plant, assuming offsite treatment, was included for analysis.

The LCI assessment for each of the tertiary treatment alternatives was based on the international standards (ISO 14040, 2006) for LCA. The Eco-invent, U.S.LCI and Franklin database in Simapro v7.3 (PRé Consultants, 2009) and economic input-output life-cycle assessment (EIO-LCA) (CMU, 2012) were used to perform a hybrid analysis to estimate the important environmental emissions to air, surface water and land. The system boundary, the functional unit and the models and databases used to determine the emissions during tertiary treatment are discussed in detail below.

7.1.2 Process-based model life cycle assessment tool/database

The Simapro process-based LCA tool was developed by PRé Consultants (2009) and is widely used to collect and analyze complex life cycles of products and services in a systematic way, following ISO 14040 recommendations.

In Simapro, process-based inventories of many common systems/materials are compiled into modules of information to be assembled by the user into a complete inventory. It contains both European and U.S. databases, including Eco-invent, U.S.LCI, BUWAL250, IDEMAT 2001, ETH-ESU 96, and the Franklin Database, and is expandable with new libraries (Frischknecht and Jungbluth 2001; Goedkoop 2003; Norris 2003; Spriensma 2003). It also provides impact assessment tools, including Eco-Indicator 99, to be used to convert inventory

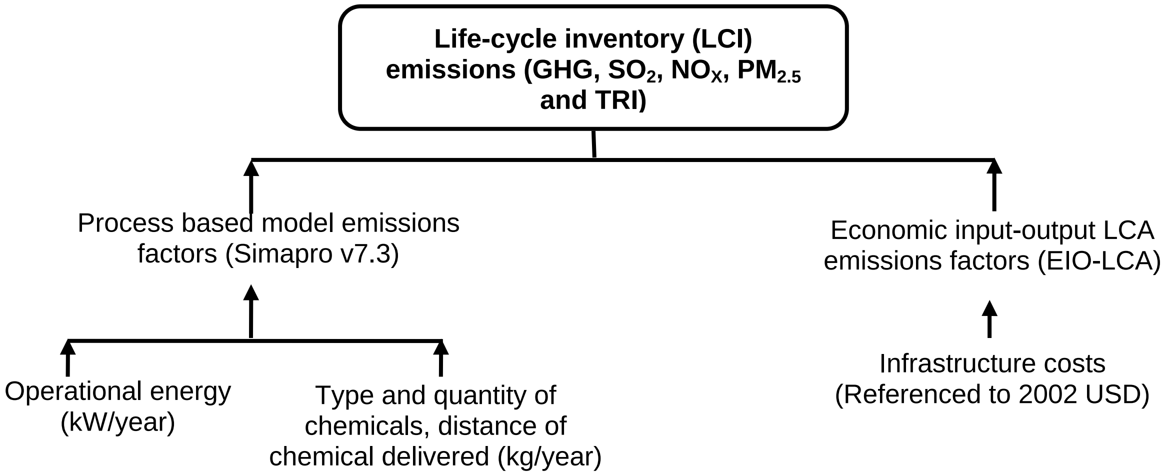

Figure 7.1.1 Flowchart of life-cycle inventory assessment (LCI) model used for evaluation of municipal wastewater tertiary treatment alternatives

data into environmental impacts. As the impact assessment tools were not used in this study; they are not discussed here. The SimaPro software contains data on most basic processes, with significantly more European than U.S. data. Data completeness varies from process to process, because the files are obtained from many data sources, which seldom have the same level of detail (Miller and Thomas, 2006).

The Eco-invent dataset in Simapro covers 2500 industrial processes (Goedkoop and Schryver, 2008). Eco-invent is a Swiss database and contains international industrial life cycle inventory data on energy supply, resource extraction, material supply, chemicals, metals, agriculture, waste management services, and transport services.

Franklin Associates developed a publicly available U.S. database of LCI information that uses a common protocol for all unit processes. The U.S.LCI database project was a cooperative effort of Franklin Associates with the Athena Institute and Sylvatica (Franklin Associates, 2009). Franklin Associates developed many of the key data sets, including data for production and combustion of process and transportation fuels, grid electricity, plastic resins, primary and secondary aluminum, several metal transformation processes, and several commodity chemicals.

7.1.3 Economic input-output life cycle assessment (EIO-LCA) tool

Combining life cycle assessment and economic input-output is based on the work of Wassily Leontief (Hendrickson et al., 2006). Leontief developed input-output models of the U.S. economy and theorized about expanding them with non-economic data. Researchers at

Carnegie Mellon built an EIO-LCA model for the US economy based on these principles (CMU, 2012). This model tracks the interrelations of 428 economic sectors, based on the 2002 U.S. Department of Commerce commodity input-output matrix of the U.S. economy. When economic activity is stimulated within a single sector, the model quantifies the economic impact of all relevant sectors that contribute to the operation of that sector (Miller and Thomas, 2006).

EIO-LCA assigns emissions to each commodity sector and then determines the aggregate emissions of a product by examining the sectors that contribute to that process (Hendrickson et al. 2006). The EIO-LCA model with a detailed description of the process is available on the web at www.eiolca.net (CMU, 2012).

7.1.4 Assumptions and limitations

Assumptions made for the LCI analyses and while converting air emissions released during construction and operation of tertiary treatment units to monetary damages were as follows:

a.) Location for the tertiary treatment plant is not specified and hence U.S. national average electricity grid mix is considered for power generation as shown in Table F.1.1 in Appendix F

b.) The conventional air emission cost factors were taken as average values across the U.S. since specific locations for the treatment plant and the production of materials for construction and operation of the plant were not provided.

c.) Chemical transport is assumed to be mainly domestic via rail, truck and barge; average transport distances are obtained from the EIO-LCA transportation model for $1 million U.S. of inorganic and organic chemicals purchased.

d.) Emissions were assumed to be released mainly during production of materials required for the treatment process, and so the height for release of emissions was assumed to be mid-level (250-500 meter from the ground) and from point sources.

e.) Emissions on-site at the tertiary treatment plant were assumed to be negligible compared to the emissions released during production of material inputs off-site and hence not considered in this study

f.) Only major equipment in each treatment unit e.g., agitators, aerators/blowers, and rotary distributor pump, are considered to calculate energy consumption. Treatment units are assumed to be connected to each other with decrease in elevation so that the flow of effluent (treated water output) is driven by gravity rather than by pumping.

g.) Lime sludge was assumed to be land filled and biological sludge production during nitrification has not been considered.

h.) The de-construction phase of the life cycle for the tertiary treatment units was not considered due to the long (25 years) plant life considered here.

i.) Constructing a constant-price time series would require time series for all cost parameter values and defining price indexes for each of the cost parameters (Muller et al., 2007) which was out of the scope of this study. Hence the external cost estimates were discounted or inflated to the year 2009 based on dollar value inflation (19.3% from 2002 to 2009) or deflation (-1.6% from 2010 to 2009) rates.

Other major assumptions inherent with the LCA tools used in this study include setting boundaries. In the process-based LCA method, two major assumptions are involved with defining the boundary of the analysis: the output inventory emissions are based on the limits of the system boundary and they do not include impacts of second tier of material inputs. Though the EIO-LCA approach eliminates the two major issues of boundary definition and circularity effects of process-based models, it assumes boundaries that are very broad and inclusive i.e., an industry sector represents a collection of several industry types, and this aggregation leads to uncertainty in how well a specific industry is modeled. Also, both process-based LCA and EIO-LCA tools assume a linear relation between the inputs and outputs, which in the real world is not true for all processes or products (CMU, 2012). Table 7.1.1 summarizes the advantages and disadvantages of both models as explained by the documentation for the EIO-LCA model.

In addition to the specific assumptions listed above, the APEEP model incorporates assumptions which might introduce errors to the external cost estimates. Evaluation of these errors was beyond the scope of this study. The assumptions in APEEP include (Muller 2011): 1) Cost estimates are accounting measures and not measures of economic welfare. The economy has many existing distortions other than those from air pollution—such as taxes, distortions from market power, and other externalities—and existing accounts do not attempt to incorporate those. 2) The model includes only the impact of air pollution and excludes other externalities such as those involving water, soil, and radiation. 3) The uncertainties in the external cost factors are particularly large for the value of mortality risks, the relationship of this value to age, the mortality effect of fine particulates, and the social cost of CO_2 emissions.

7.1.6 Inventory inputs

7.1.6.1 LC^3 cost estimates as inputs

Inputs for the two LCA models were based on the available level of detail for input data, and specific process accessibility in the database(s) for each model. EIO-LCA inputs are in terms of costs (2002 USD) for all sectors. Cost estimates were calculated using the LC^3 model

(Chapter 6), which is a public domain conceptual cost model and estimates costs within conceptual estimation accuracy range (15-40%) defined by International Association for the Advancement of Cost Engineering (AACE).

In the LC^3 model, , annual-cost estimates are calculated as the sum of the recurring costs (RC), also known as operation and maintenance costs, and the non-recurring costs (NRC), otherwise known as capital costs, converted to an annual cost basis. For the purpose of this study, NRC estimates were considered as inputs to the EIO-LCA model and include annualized infrastructure construction costs and the costs of pumps, motors, auxiliary mechanical and digital equipment. NRC estimates were dispersed over the period of the equipment or treatment plant life, i.e., 25 years for concrete infrastructure, 10 years for pumps

Table 7.1.1 Advantages and disadvantages of process-based LCA and EIO-LCA models (CMU, 2012)

	Process-Based LCA	EIO-LCA
Advantages	results are detailed, process specific	results are economy-wide, comprehensive assessments
	allows for specific product comparisons	allows for systems-level comparisons
	identifies areas for process improvements, weak point analysis	uses publicly available, reproducible results
	provides for future product development assessments	provides for future product development assessments
		provides information on every commodity in the economy
Disadvantages	setting system boundary is subjective	product assessments contain aggregate data
	tend to be time intensive and costly	process assessments difficult
	difficult to apply to new process design	must link monetary values with physical units
	use proprietary data	imports treated as products created within economic boundaries
	cannot be replicated if confidential data are used	availability of data for complete environmental effects
	uncertainty in data	difficult to apply to an open economy (with substantial non-comparable imports)
		uncertainty in data

and mechanical equipment and 15 years for the digital control equipment. The estimated 2009 costs from the LC^3 model for the current study are provided in Table F.1.1 in Appendix F.

7.1.6.2 Indexing cost inputs to correspond to EIO-LCA model

The EIO-LCA model database is referenced to the year 2002, whereas the LC^3 cost estimates are indexed to 2009. To convert the estimates from 2009 to 2002, indices in Engineering News Record (ENR), U.S. Bureau of Labor Statistics (BLS) and Chemical Engineering Plant Cost Index (CEPCI) were used. Table 7.1.2 lists the various indices and the respective values reported in the above mentioned sources. After the cost conversions, each cost component of the treatment system was provided as an input to the respective sector in EIO-LCA.

7.1.6.3 Inputs to process-based LCA model

Construction costs were used in the EIO-LCA model due to the conceptual level of design and consequent lack of information on the specific quantities of construction material used for each block of infrastructure. Well-defined quantitative details were available for the O&M variables. Chemicals, electricity grid mix, transport of chemicals, and pumping distance and energy for treated water delivery (an average of 10 miles for water delivery was assumed) were inputs to the process-based model. These processes could be assembled together using the specific process of manufacture and/or mix of fuel in the Simapro model. Type and dosage of chemicals for treatment and conditioning, manufacture and regeneration data for granular activated carbon (GAC), U.S. national grid mix, and the distances for transportation obtained from EIO-LCA are provided in Tables F.1.2 to F.1.5 in Appendix F. Table 7.1.3 lists the input parameters and the database and model from which the respective inventory emissions were extracted.

7.1.7 Emissions, emission factors and characterization factors

Data from various sources are used to develop emissions factors in EIO-LCA and in the databases in Simapro. In EIO-LCA, the factors are obtained as follows:

- Conventional pollutant emissions are from the eco-invent, U.S.LCI and Franklin database and from U.S.EPA database as presented in Simapro v. 7.3 and 2002 EIO-LCA model, respectively (USEPA, 1999).

Table 7.1.2 Cost indices used to adjust 2009 cost estimates to the year 2002

Source for Index Values	2002	2009
ENR-Building Construction Cost Index	3580	4780
BLS Metal and Metal Products Consumer Price Index	107.1	191.6
Chemical Engineering Plant Cost Index for Equipment	395.6	509.1
Producer Price Index for Chemicals and allied products	151.9	229.4

Table 7.1.3 Input to the EIO and process-based LCA models to determine the inventory emissions

Input Parameter	Database/Model used for Inventory
Construction of infrastructure (in 2002 million USD)	EIO-LCA
Pump and pumping material manufacturing (in 2002 million USD)	EIO-LCA
Plumbing fixture fitting (in 2002 million USD)	EIO-LCA
Valve and fittings other than plumbing (in 2002 million USD)	EIO-LCA
Fabricated pipe and other pipe fitting manufacturing (in 2002 million USD)	EIO-LCA
Material handling/manufacturing equipment (in 2002 million USD)	EIO-LCA
Metal tank, heavy gauge manufacturing (in 2002 million USD)	EIO-LCA
*Electricity grid mix (in kWh)	US LCI, Franklin, Eco-invent
*Transportation via road (in ton-km)	Franklin
*Transportation via rail (in ton-km)	US LCI, Franklin, Eco-invent
*Transportation via water (in ton-km)	US LCI, Franklin, Eco-invent
Landfill (ton)	Eco-invent
Coal for GAC (kg)	Franklin
Natural gas for GAC preparation (Nm^3)	Eco-invent
Steam for GAC preparation (kg)	Eco-invent
*HCl for GAC preparation (kg)	Eco-invent
*Hydrated lime for softening (kg)	Eco-invent
*Sulfuric acid (100%) (kg)	Eco-invent
*Maleic anhydride for Polymaleic acid preparation (kg)	Eco-invent
*Monochloramine (kg)	Eco-invent
*Soda ash (kg)	Eco-invent
[a]Chemical manufacture for TTA (kg)	EIO-LCA

[a]Assumed as a general chemical (due to lack of specific data on manufacture process)
*Sensitivity/uncertainty analysis conducted for GHG gas emissions, using available inventory data from other databases such as ETH-ESU 96, BUWAL250, etc., to avoid process variability and account for influence of the inventory data on environmental profile

- Greenhouse gas emissions were calculated using the IPCC 2007 100a characterization method (IPCC, 2007) using U.S.EPA AP-42 fuel use emissions factors for CO_2 and methane (USEPA 1995).
- Toxics releases are derived from the US EPA's 1995 toxics release inventory (TRI) and 1995 value of shipments from the 1995 Annual Survey of Manufacturers. (CMU, 2012)

In the process based LCA, the inventory emissions factors for Franklin and U.S.LCI were obtained from the U.S.EPA, and National Renewable Energy Laboratory (NREL) sources.

7.1.8 Estimation of externality costs for emissions

A number of studies have been conducted to try to quantify the environmental impact costs of air emissions. Using data on the social costs of production, attempts have been made to make integrated "green" national accounts by adjusting them for environmental (and other) costs (Matthews, et al., 2000; Nordhaus and Tobin, 1972 and; Daly and Cobb, 1989). A 2010 summary of CO_2 equivalent units costs from approximately 50 studies revealed a median cost of \$10/ton, mean cost of \$30/ton, and 5th and 95th percentile costs of \$1 and \$85/ton (NRC, 2010). The mean \$30/ton cost was implemented in this study for emissions of CO_2 eq. For the other, conventional, air emissions considered in this study, average cost factors of \$5.70/kg SO_2, \$1.60/kg NO_x and \$12/kg $PM_{2.5}$ from the APEEP (Air Pollution Emission Experiments and Policy) model for mid-level point sources (effective height between 250 and 500 meters) was used. The environmental impact cost estimates were discounted or inflated to the year 2009 based on dollar value inflation (19.3% from 2002 to 2009) or deflation (-1.6% from 2010 to 2009) rates (U.S.BLS, 2012).

APEEP (Muller, 2007) is designed to calculate the marginal (incremental) human health and environmental damages corresponding to marginal emissions of $PM_{2.5}$, VOC, NO_x, NH_3, and SO_2 on a dollar-per-ton basis (NRC, 2010; Muller, 2007). APEEP estimates the damages stemming from emissions at nearly 10,000 sources in the contiguous U.S. Damages include adverse effects on human health, reduced yields of agricultural crops and timber, reductions in visibility, enhanced depreciation of man-made materials, and damages due to lost recreation services. The model focuses on damages at the margin in order to weigh marginal damages against the marginal costs of abatement (Muller et al., 2007). APEEP employs the results from numerous, peer-reviewed studies that apply these methods to value the physical effects of air pollution (Muller, et al., 2007). It then evaluates emissions at different release heights.

The mid-level subset of emissions in the APEEP database was used to assess emissions associated with tertiary treatment of MWW. County based weighted average APEEP cost factors determined for the U.S. was used for this study (Mashayekh, 2011). Due to accounting of spatial variability in the cost factors, the uncertainty introduced to the external costs is estimated to be very high. This uncertainty can be reduced if the spatial location of the input inventory production or use is specified by an approximate regional boundary i.e., rural or urban location and by accounting for the 5th and 9th percentile values for the social costs than the entire range of factors (which can include outliers or higher or lower extremes). To study the rural and urban only differences three alternate scenarios were analyzed as given in Appendix F, Figure F.3.1 to Figure F.3.4.

7.1.9 Integrating LCI with LCC

A total cost assessment tool completed by a collaborative effort of ten multinational companies and the American Institute of Chemical Engineers' Center for Waste Reduction Technologies defined five cost types which include direct, indirect, contingent, intangible and external costs (Norris, 2001). The direct costs include the capital investment, labor, raw material and waste disposal; indirect costs include overhead; contingent costs include fines and penalties, property damage and personnel injury liabilities; intangible costs include difficult to measure costs such as customer loyalty, work wellness, corporate image, union and community relations and; external costs are costs borne by the society (CWRT, 1999).

This study integrated the first and last type of costs among the five cost types explained above. After calculating the environmental costs for the air emissions, the costs were inflated to 2009$/kgal and then added to the life-cycle conceptual costs obtained from the LC^3 model estimates. The sum of the above mentioned costs present the total costs of tertiary treatment (in 2009$/kgal) normalized for a given loading rate of the treatment plant.

7.1.10 Uncertainty in inventory emissions

In this study, the main sources of uncertainty were: 1) variability of emission factors, within a database due to process variation and due to averaging between databases, 2) quantifying energy and transport distances based on average grid mixes and aggregated delivery distances, respectively.

Process variability (when one or more processes are available for a single product or service) is seen in the process-based model databases in Simapro, and was represented in this study for chemicals using box plots in Figure 7.2.2. Though the variability due to averaging between processes has been often observed when multiple databases are used, in this study individual inputs were derived from a single database and so the probability of added error due to process variability from combining databases was assumed to be negligible. In Ecoinvent, the probabilistic mean values obtained were seen to differ slightly from the deterministic mean values, but the current database provides deterministic values based on reproducibility and reliability (Frischknecht et al., 2007). For construction material emission factors which were obtained from a single inventory source (EIO-LCA), uncertainty analysis was not considered.

The uncertainty of averaging travel/water delivery distances, energy grid mix and their emission factors have been addressed using Monte Carlo simulation for greenhouse gas emissions in this study and have been reported in the GHG emissions graph.

Among the results of the LCI and external cost analysis study, only greenhouse gases, followed by conventional air pollutants such as SO_2, NO_x and $PM_{2.5}$ are discussed here as external costs are well established methods for these air pollutants. The TRI emissions to air water and land are presented in Appendix F.

7.2 Results and Discussion

Among the results of the LCI and external cost analysis study, only greenhouse gases, followed by conventional air pollutants such as SO_2, NO_x and $PM_{2.5}$ are discussed here as external costs are well established methods for these air pollutants. The TRI emissions to air water and land are presented in Appendix F in Figures F.2.1 to F.2.14.

The emissions presented in the results include a spatial distribution bias due to lack of knowledge regarding the location of raw material extraction, production of inventory (energy, chemicals and equipment), and treatment plant operation. Hence, to fully understand the local and global impacts of adopting tertiary treatment alternatives for reuse, it is necessary to incorporate spatial resolution limitations and address implications of all category emissions.

7.2.1 Greenhouse gas emissions

Figure 7.2.1 represents the annual GHG emissions in terms of t CO_2e/yr. It can be observed from Figure 7.2.1 that chemical manufacture and transport for treatment and conditioning chemicals, followed by electricity consumption contributes most to GHG emissions. The results also show that tertiary treatment with nitrification, softening and filtration (MWW_NSF) is the alternative with the highest annual GHG emissions, due to higher energy consumption by the nitrification unit, hydrated lime and soda ash dosing during the softening treatment process, and added infrastructure for three tiers of treatment. Similarly, higher emissions were estimated for MWW_SF due to dosing of hydrated lime. For MWW_NFG, activated carbon production and regeneration also causes high GHG emissions due consumption of coal and natural gas (consists of burning 2 kg of coal for every 1 kg of new activated carbon produced and 10% loss of carbon per regeneration cycle; 4.9 Nm^3kg and 2.7 Nm^3/kg of new activated and regenerated carbon). RW, MWW and MWW_pH were estimated to have lower overall GHG emissions, but relatively higher emissions for chemical conditioning as they require higher anti-scalant, anti-corrosion, and biocide agent addition due to lower effluent quality. Water supply infrastructure and delivery GHG emissions are mainly due to pumping energy consumption. Overall, the curtailment of GHG emissions can be achieved through the efficient use of chemicals and selecting less energy intensive treatment processes, consistent with findings of others (Racoviceanu et al., 2007).

The error bars shown in Figure 7.2.1 represent the uncertainty range of emissions as evaluated using the statistical analysis method with - Monte Carlo simulation to understand the variability of results with changes in emission factors (excluding the chemicals emission factor range) and input inventory quantity. The uncertainty considered was that due to electricity grid mix emission factors, piping distance for treated effluent delivery, and maximum and minimum distance for transport of chemicals with range for transportation emission factors. Direct cost estimation uncertainty has been evaluated in Chapter 6; in this chapter the indirect variability due to cost variation is incorporated via changes in input inventory data mentioned above.

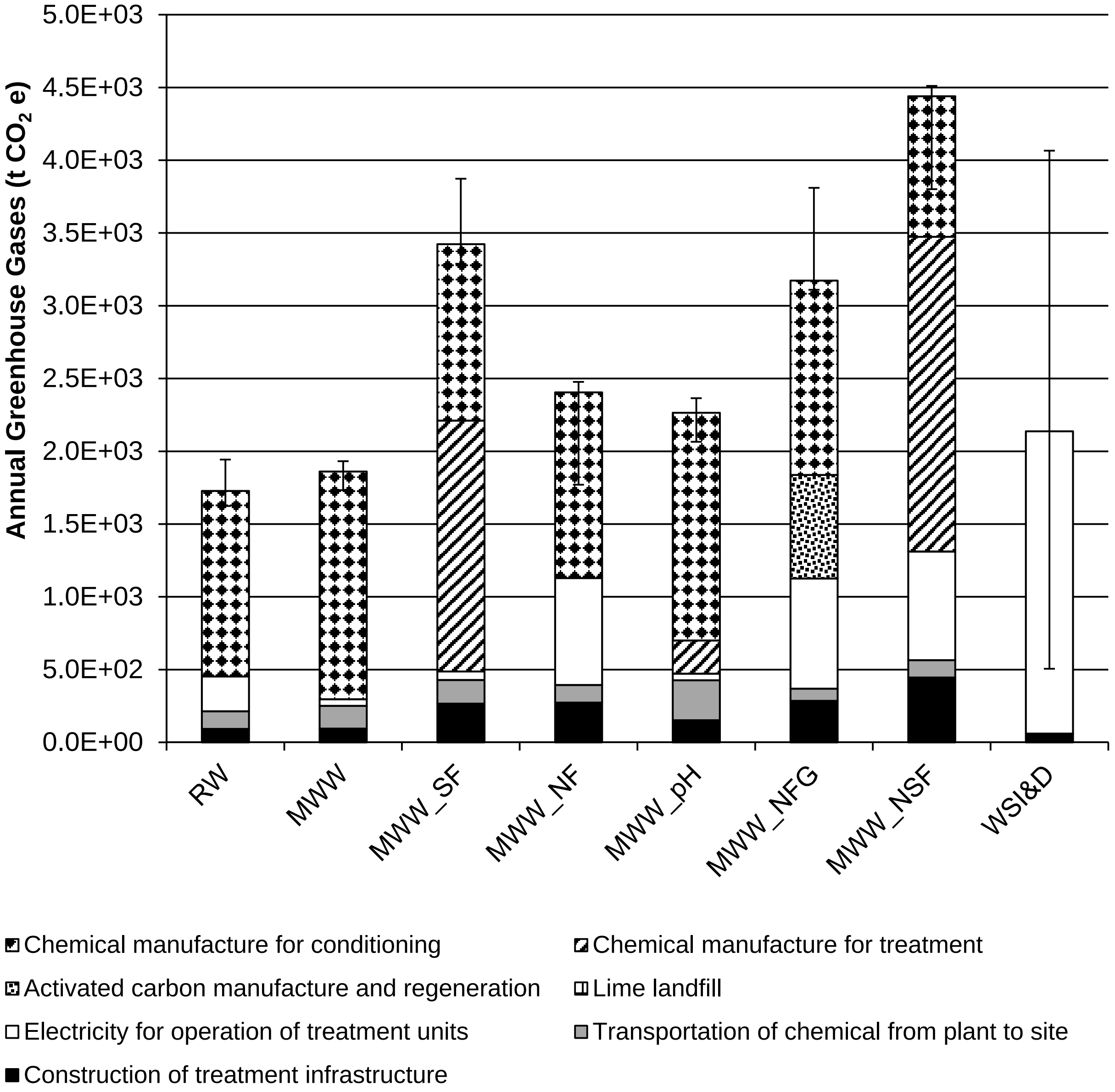

Figure 7.2.1 Greenhouse gas emissions (in t CO_2 e./yr) of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD. Error bars indicate uncertainty due to electricity grid mix emission factors, piping distance for treated effluent delivery, and maximum and minimum distance for transport of chemicals with range for transportation emission factors.

7.2.2 Uncertainty analysis for chemical GHG emissions factor

As seen in Figure 7.2.1, chemicals used for treatment and conditioning contribute significantly to overall GHG emissions. The range in Figure 7.2.1 doesn't account for variability due to chemical manufacturing as accounted in various databases, as chemical emissions factor values in the study were taken from only Ecoinvent database.

The range of values for GHG emissions from manufacturing of chemicals obtained from all available databases is shown in Figure 7.2.2. Hydrochloric acid (HCl) used for cleansing the activated carbon, hydrated lime used for chemical precipitation, and ammonia used to prepare monochloramine are most variable. Chlorine and maleic anhydride (partial input for monochloramine and polymaleic acid, respectively) emissions further vary with database, although this variation was not considered in this study. This plot provides insight into the possible sensitivity of the reported results in this study due to chemical emission factors and indicates the significance of uncertainty analysis if deterministic values are considered in future.

7.2.3 Acidifying (SO_2 and NO_x) and particulate matter ($PM_{2.5}$) emissions

Figures 7.2.3 and 7.2.4 indicate the emissions for acidifying pollutants and particulate matter, respectively. Similar to GHG emissions, emissions of SO_2 in Figure 7.2.3 (bar on the left for each represented category) are mainly from chemical and GAC production and regeneration and electricity generation for operation. Sulfuric acid used in the MWW_pH process is the primary contributor for SO_2 emissions. MWW_NFG and MWW_NSF have higher SO_2 emissions since they require GAC, and regeneration and dosing of soda ash respectively.

Annual NO_x emissions (bar to the right for each category represented on the x-axis) are higher than annual SO_2 emissions for most tertiary treatment alternatives, except for MWW_pH. Sources of the high NO_x emissions are transportation of chemicals, activated carbon manufacture and regeneration, and electricity generation for operation. MWW_NSF, followed by MWW_NFG with soda ash and GAC manufacture emit the most NO_x emissions, respectively. RW and MWW have the lowest SO_2 and NO_x emissions among all the treatment alternatives.

Particulate matter emissions <2.5 micron in size, presented in Figure 7.2.4, are highest from chemical and GAC manufacture and construction processes. MWW_NSF, as a three-tiered tertiary treatment process, requires more infrastructure and two different types of chemicals (hydrated lime and soda ash) to be added, which makes it the highest $PM_{2.5}$ emission contributing alternative. MWW_NFG, MWW_SF and MWW_pH also emit relatively high amounts of particulate matter among all the treatment processes.

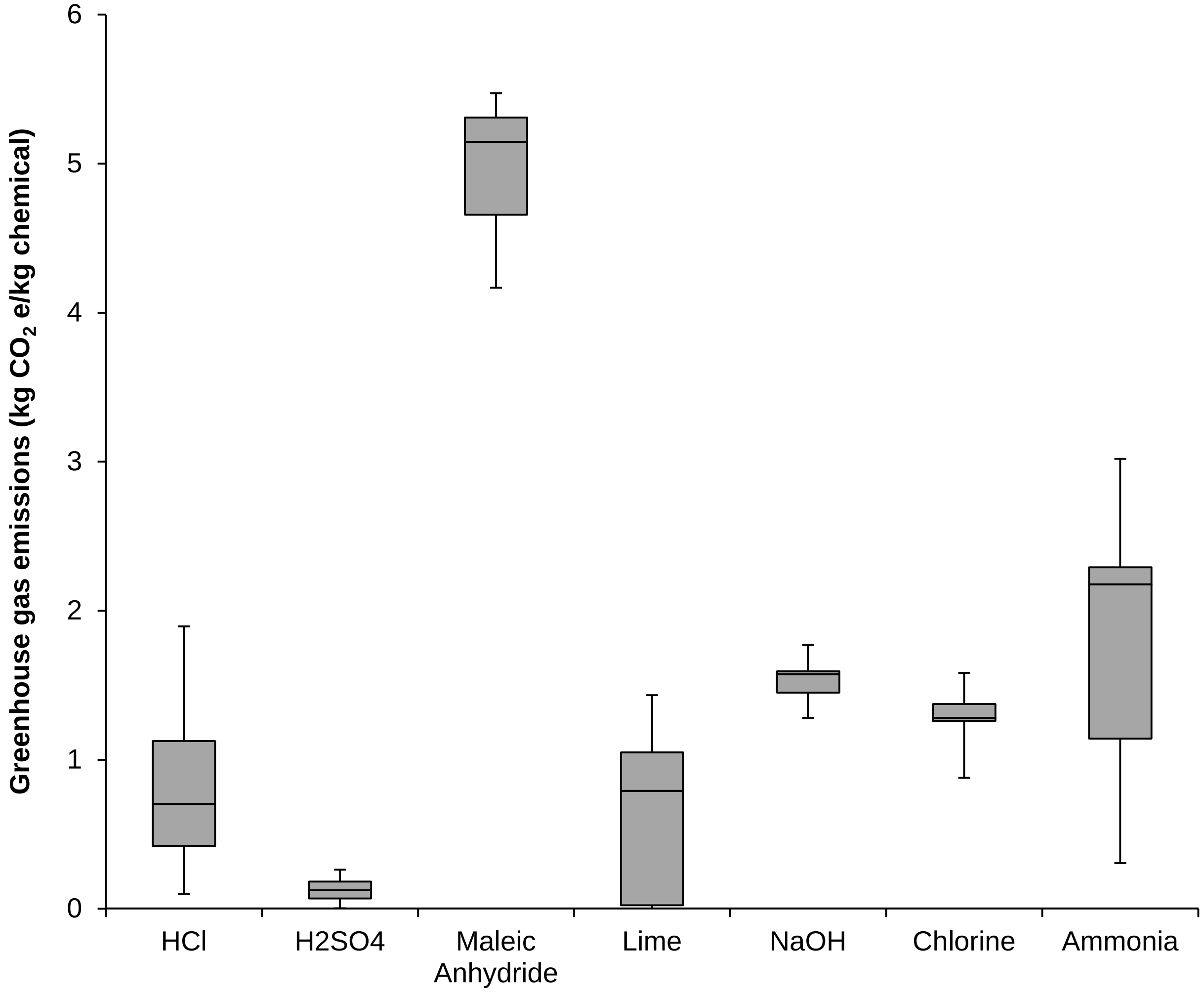

Figure 7.2.2 Range of GHG inventory emissions (kg CO_2 e/kg of chemical) reported from all chemical databases in Simapro. All chemicals used for treatment and conditioning in the various tertiary treatment alternatives are shown in the plot.

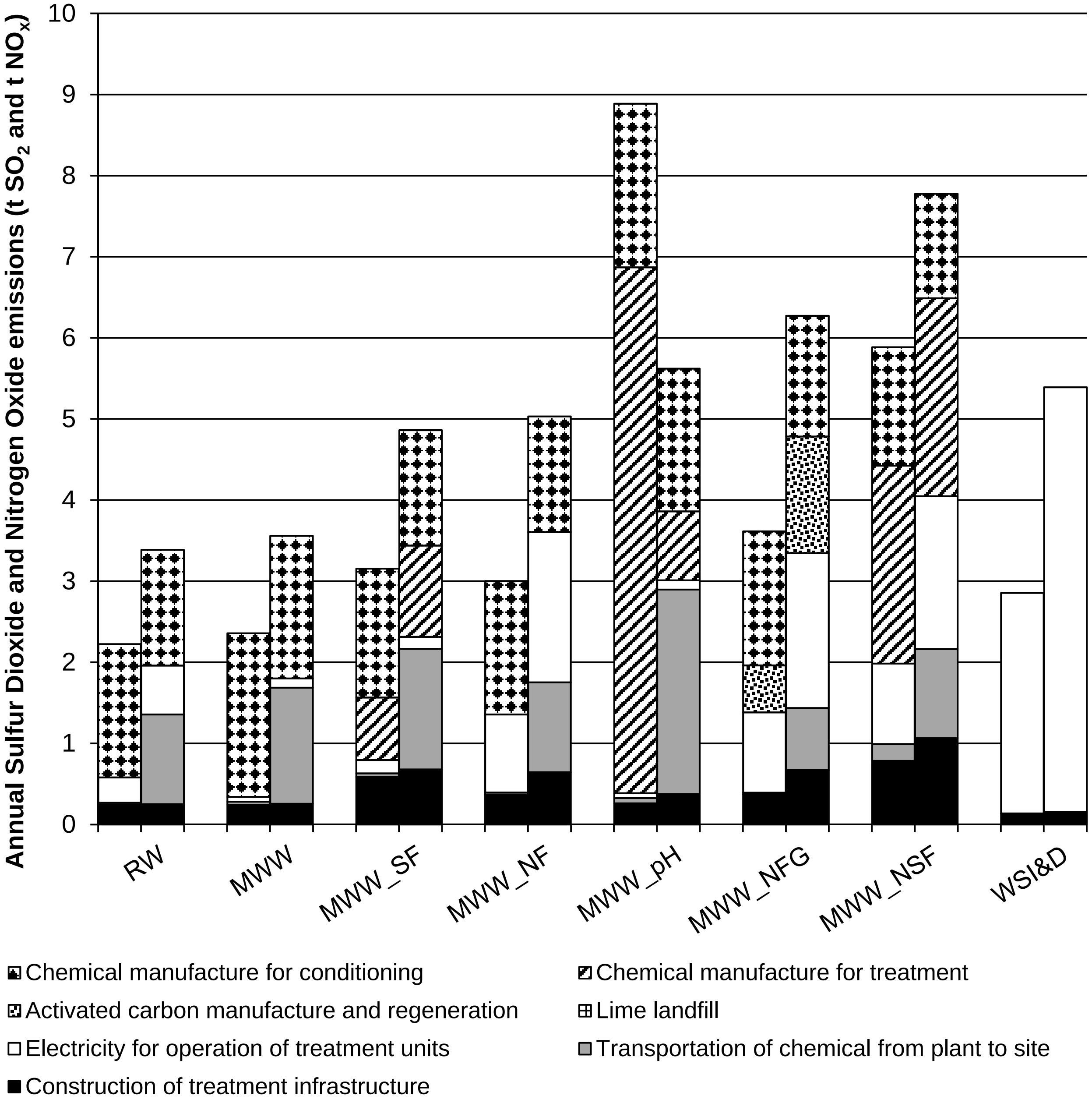

Figure 7.2.3 Sulfur dioxide (SO_2) (bar on the left for each category on the x-axis) and nitrogen oxide (NO_x) (bar on the right for each category on the x-axis) emitted (in t SO_2/yr and t NO_x/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

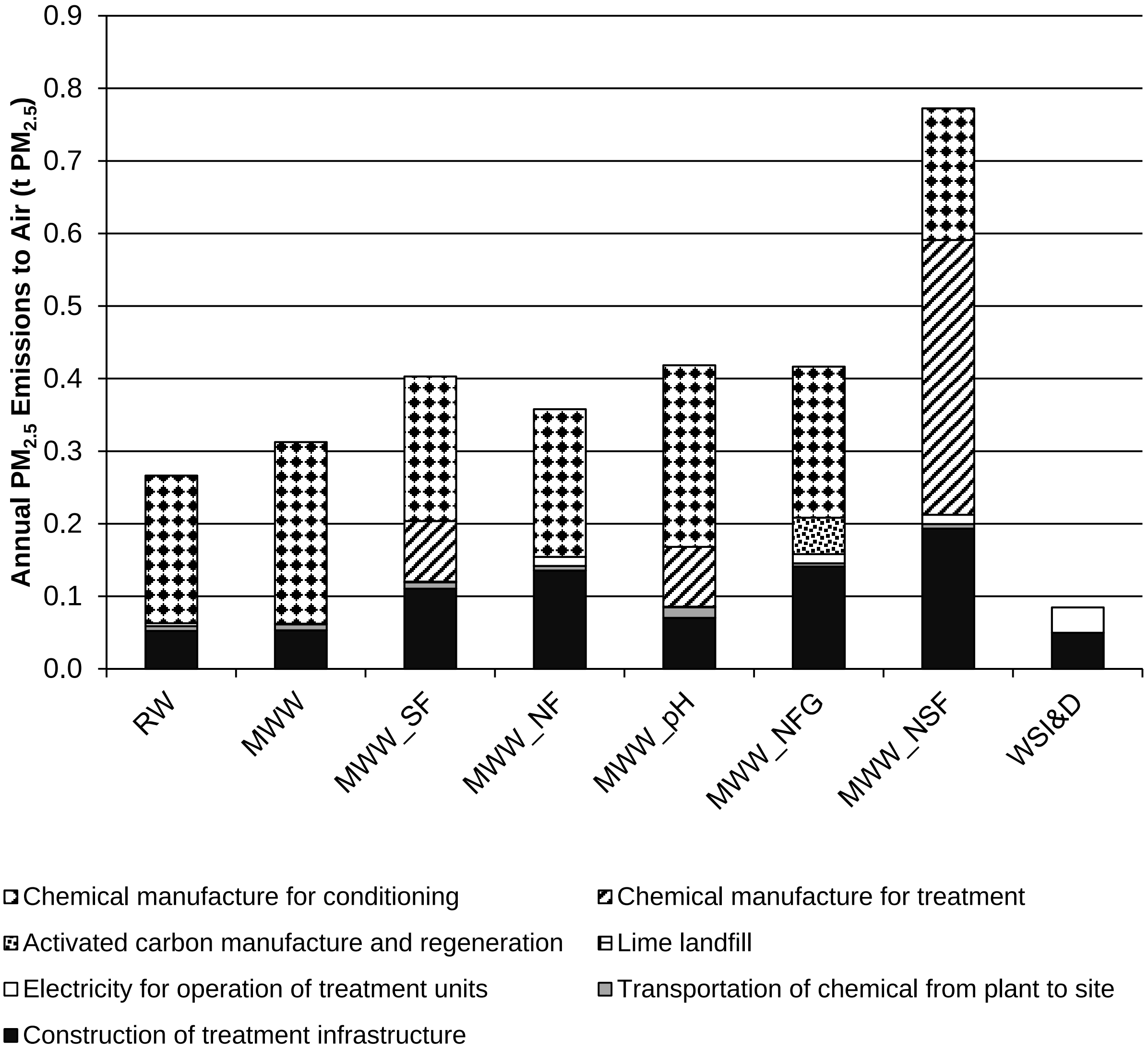

Figure 7.2.4 Particulate matter ($PM_{2.5}$<2.5 microns in size particulates) emissions (in t $PM_{2.5}$/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

7.2.4 Integrated costs of combined tertiary treatment alternatives

Table 7.2.1 shows the collection of unit costs of external (or social) damages obtained from the literature (Matthews et al., 2000; NRC, 2010) and the model (Muller, et al., 2007) for each inventory emission in terms of $1992 and $2002 (except for CO_2 eq. for which $2010 values are shown). Figure 7.2.5 shows the temporal variation (range for U. S. counties) of cost factors. From values presented in the Table 7.2.1, it may be seen that there is a significant increase in the dollar-per-ton values from 1992 to 2002 for all inventory emissions, which may be due to inflation in market goods (the loss of which is used to evaluate non-market services) or due to increased value applied to the damages caused from air emissions released.

$PM_{2.5}$, SO_2 and CO_2 eq. emissions values across the country vary widely for mid-level point sources as seen in Figure 7.2.5. NO_x emissions show the minimum value to be negative, although its range of variation is relatively small. The average dollar-per-ton cost factors used in this study were close to the 5% percentile costs shown in Figure 7.2.5 of the range for all emissions considered here, which is well justified by the literature sources (Mashayekh, 2011).

Results of the marginal damage costs and the integrated costs are shown in Figures 7.2.6 and 7.2.7, respectively. Figure 7.2.6 splits the costs for individual emissions per treatment alternative to examine the individual allocation of emissions of the total external costs calculated. It can be seen that the GHGs are primary contributors of impact costs for all alternatives. For MWW_pH, the SO_2 emissions allocated to the manufacture of sulfuric acid is almost equal to the GHG emissions. MWW_NSF and MWW_NFG have noticeably high marginal external damage costs because of the three-tiered treatment infrastructure, electricity for aeration, addition of moderate conditioning and treatment chemicals, and regeneration of spent carbon. Among the two-tiered treatment alternatives, MWW_NF has lower external costs compared to the MWW_SF, because the latter alternative requires regular dosing of chemicals during tertiary treatment. RW and MWW_F have lower marginal damage costs since they involve only sand filtration (F) as a tertiary treatment step, followed by addition of higher doses of chemical conditioners, to which a major portion of the existent emissions costs can be allocated.

Among emissions, though $PM_{2.5}$ has high average unit costs, the overall costs of those emissions for the treatment units is estimated to be low because the quantity of $PM_{2.5}$ released is much lower compared to other emissions. On the other hand, though GHGs are relatively emitted in much higher amounts, since their cost factors are not very high, their impact costs remain moderate.

Figure 7.2.7 illustrates the total annual costs (in 2009$/kgal) which is the sum of direct and impact related costs of emissions. The average impact costs of GHGs, acidifying gas emissions and particulate matter (<2.5 microns in size) emissions vary between 5.3 – 7.2% of the total annual costs, with higher impact costs for MWW_NSF, followed by MWW_NFG, MWW_SF and then MWW_pH. Considering the maximum unit cost factors that can be used to determine the maximum impact costs ($105/t CO_2e, $139/kg SO_2, $15.65/kg NO_x and $420/kg $PM_{2.5}$) as seen in the upper margin of the cost bar (indicated with error bar) in Figure 7.2.7, the maximum impact costs would be in the range of 45-70% of the total annual costs, which is almost equal to or greater the direct costs incurred from tertiary treatment alternatives.

Considering the rural and urban only case scenarios and replacing the maximum and minimum range cost values with 5th and 9th percentile error margins for these scenarios, the damage-based costs for the urban only scenario range from 6-14%of the total annual costs. Similarly for the rural only scenario, the damage-based costs are 5-11% of the total annual costs. With 5th and 9th percentile error margins, the range for average values across all the counties of the U.S. is 5.3 – 20% of the total annual costs. The external costs values were seen to be slightly lower if the inventory production and use were located in rural areas compared to urban areas.

Table 7.2.1 Unit external (or social) damage estimates ($1992, $2002 and $2010) from air emissions of environmental externalities

Species	**Estimated external costs ($/kg of air emissions)**			
	min	**median**	**mean**	**max**
[a]**CO_2 eq. ($1992/kg)**	0.002	0.014	0.013	0.023
[b]**CO_2 eq. ($2010/kg)**	0.001	0.01	0.03	0.105
[a]**SO_2 ($1992/kg)**	0.77	1.8	2	4.7
*[b]**SO_2 ($2002/kg)**	0.61	5.20	5.71	139.40
[a]**NO_x ($1992/kg)**	0.22	1.06	2.8	9.5
[c,d]**NO_x ($2002/kg)**	-5.71	1.31	1.60	15.65
[a]**PM_{10} ($1992/kg)**	0.95	2.8	4.3	16.2
[c,d]**$PM_{2.5}$ ($2002/kg)**	0.54	7.61	11.43	420.12

[a] Source: Matthews et al., 2000

[b] Source: NRC, 2010

[c] Source: Muller, 2007

[d] Mid-level point source emissions

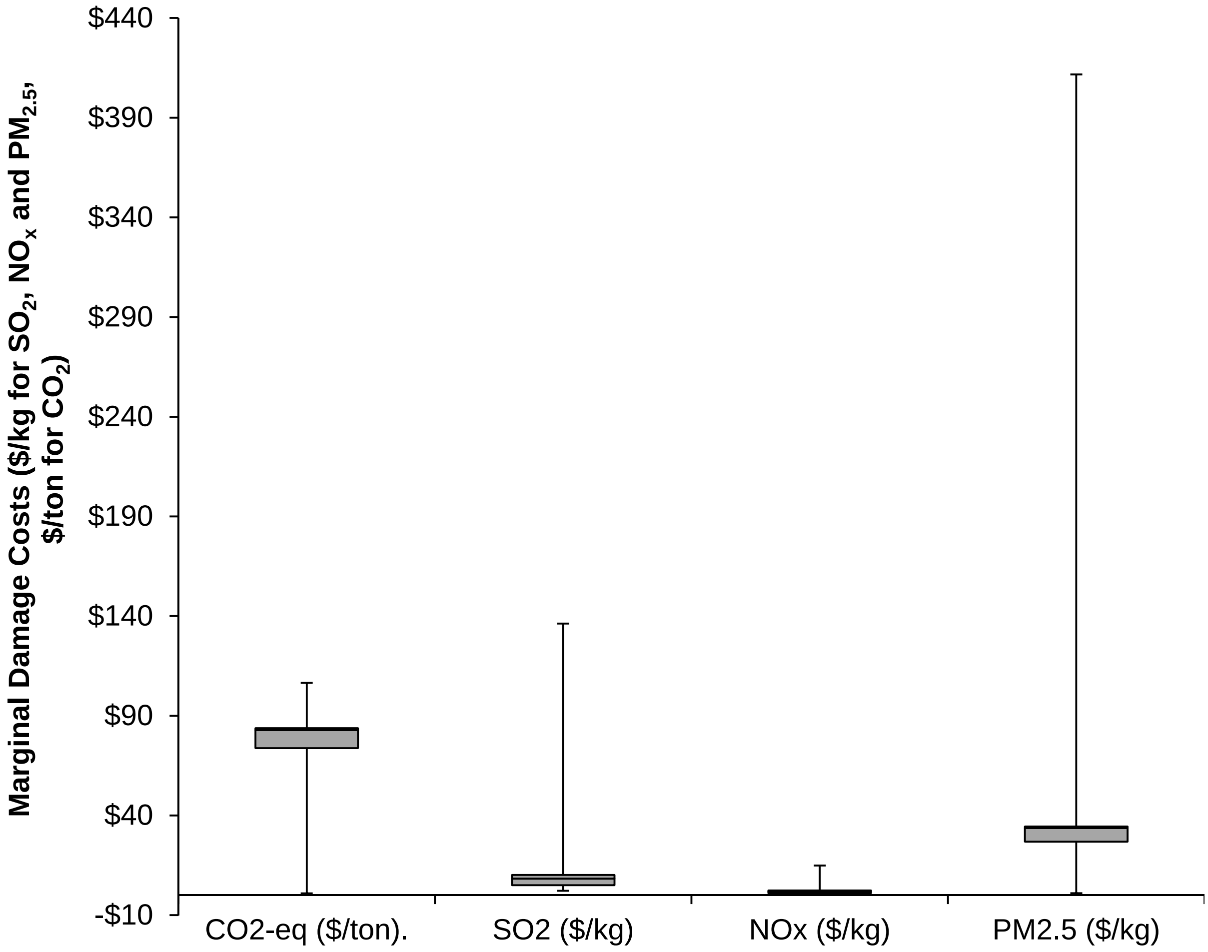

Figure 7.2.5 Range of marginal damage costs (in 2002$/kg for SO_2, NO_x and $PM_{2.5}$, 2010$/ton for CO_2) the U. S. taken from NRC study for greenhouse gases (CO_2 eq.), from APEEP for mid-level pollutants emissions of acidifying gases (SO_2 and NO_x) and particulate matter ($PM_{2.5}$)

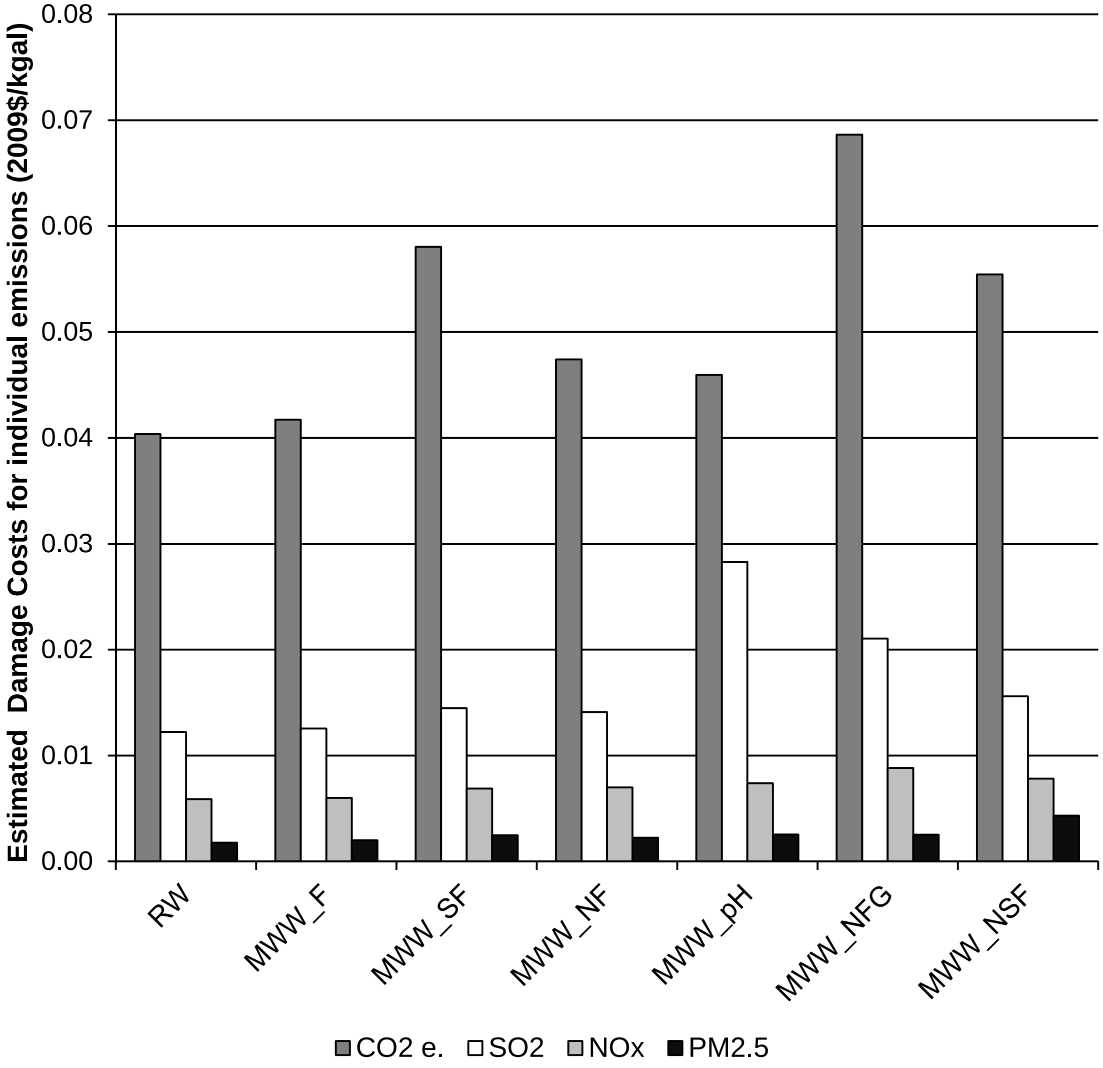

Figure 7.2.6 Estimated environmental costs (in 2009$/kg) of individual air emissions for tertiary treatment alternatives. Average external damage costs per t CO_2 eq. = $30, per kg SO_2 = $5.7, per kg NO_x =$1.6 and per kg $PM_{2.5}$ =$12 were used for estimation of impact costs
Note: kgal = kilo gallon, Nitrification (N), Lime Softening (S), pH adjustment (pH), Filtration (F), Chemical management (C) for anti-corrosive agent –Tolytriazole (TTA), anti-scaling agent – Polymaleic Acid (PMA) and biocide – Monochloramine (MCA)

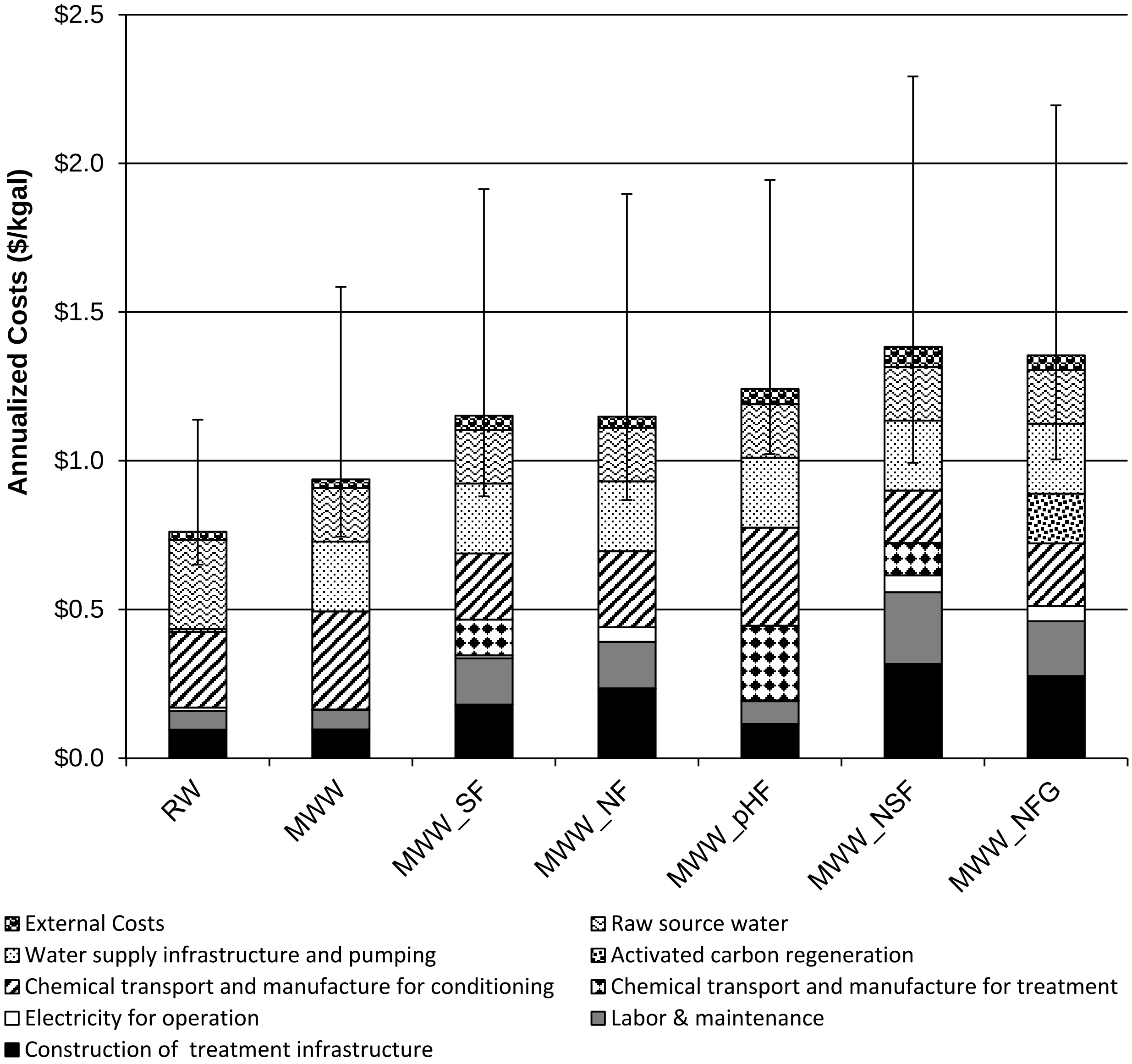

Figure 7.2.7 Total annual costs (in $/kgal) of tertiary treatment alternatives, inclusive of direct and external costs. Average external damage costs per t CO_2 eq. = $30, per kg SO_2 = $5.71, per kg NO_x =$1.51 and per kg $PM_{2.5}$ =$12 were used for estimation of impact costs

Note: 1)The costs presented in the graph are base treatment costs that include nominal charges for the secondary treated wastewater and supply fees for 10 mile delivery distance, additional chemical management and external costs due to impacts of emissions on the environment but do not include additional service or taxes. Secondary treated raw municipal wastewater cost was assumed to have a base value of approximately $0.18/kgal and was added to the tertiary treatment costs when comparing total costs with river water (RW) costs. 2) kgal = kilo gallon, Nitrification (N), Lime Softening (S), pH adjustment (pH), Filtration (F), Chemical management (C) for anti-corrosive agent –Tolytriazole (TTA), anti-scaling agent – Polymaleic Acid (PMA) and biocide – Monochloramine (MCA). 3) Established RW rates were obtained from state or regulatory agencies (Delaware River Basin Commission, 2011; Minnesota Department of Natural Resources, 2011; and WMP Advisory Committee – LCRA Raw Water Rates, 2011). 4) Error bars signify the ±35% contingency range plus the 5th and 95th percentile range of county based external damage costs

7.3 Summary and Conclusions

Life-cycle inventory analysis along with integration of external costs of emissions with direct costs was estimated to evaluate relative emissions to the environment and external costs associated with construction and operation of tertiary treatment alternatives for reusing secondary municipal wastewater (MWW) in thermoelectric power plant cooling systems. Output inventories of greenhouse gas emissions, sulfur dioxide, nitrogen oxides, and particulate matter (<2.5 micron) to air show that manufacture of chemicals for tertiary treatment and conditioning, and electric power generation are the main processes that contribute to these emissions. Transportation of chemicals and construction of added treatment infrastructure to obtain higher quality effluent cause the NO_x and $PM_{2.5}$ emissions to be high, respectively.

Tertiary treatments involving three tiers, i.e., MWW_NSF and MWW_NFG, have additional expensive infrastructure compared to other treatment alternatives and they consume more types of resources, soda ash and activated carbon, respectively. Also, the higher level of treatment doesn't substantially offset dose of conditioning chemicals, which causes high inventory emissions ranking them as the least environmentally viable tertiary treatment alternatives.

MWW_pH has relatively high NO_x, $PM_{2.5}$ and BTEX emissions to air, but remaining emissions are relatively moderate to air, water and land. MWW_SF and MWW_NF have moderate emissions, except for greenhouse gas emissions, wherein the former alternative emits more greenhouse gases compared to the latter alternative, mainly due to the addition of lime in MWW_SF. MWW_NF has somewhat higher potential to cause ecotoxicity compared to MWW_SF. Among the three-tiered treatment alternatives, MWW_NSF and MWW_NFG cause higher emissions of each category, with MWW_NSF having the highest emissions for all categories except for the BTEX emissions to water and soil/land, which are higher for MWW_NFG. The least emissions were estimated for RW and MWW, as there is not much infrastructure constructed and mainly addition of chemical conditioners is required. Water supply infrastructure and delivery showed high conventional pollutant emissions based on the pumping distance. Thus, if the distance between treatment plant and power plant is minimized, so can these emissions.

Results for TRI air emissions shown in Appendix F for the various tertiary treatment processes were in agreement with the GHG, SO_2, NO_x and $PM_{2.5}$ emissions estimated for the processes. One difference was that the emissions of lead, mercury, cadmium and their compounds (heavy metals) were higher to land than to air due to construction. Though MWW_SF and MWW_NF have relatively lower and comparative TRI emissions to all compartments, MWW_SF has some high TRI releases to air and land compared to MWW_NF, due to addition of lime. Therefore, MWW_NF would be environmentally preferred over MWW_SF. As was the case for the analysis of conventional air pollutant emissions, RW and MWW only with addition of chemical conditioners were estimated to have the least amount of TRI releases.

With the given inventory data and emission factors, the results of this study indicate that higher tiers of tertiary treatment cause higher environmental emissions to air, surface water and land, and lead to greater consumption of resources with not much offset in treated effluent quality. Tertiary treatment options with more construction, chemical and energy requirements such as MWW_NSF and MWW_NFG inherently are associated with more environmental impacts such as acidification, respiratory effects and ecotoxicity. Thus, use of MWW with chemical conditioning only or with less intense tertiary treatment is recommended to avoid outweighing the value added from reuse of treated wastewater in cooling systems.

Integration of external costs of emissions with direct costs of construction and operation for various tertiary treatment alternatives for secondary treated municipal wastewater (MWW) was conducted. Accounting for damages caused by the release of air emissions along with the direct economic costs can help balance the external damage and the value added by a particular service, such as tertiary treatment to provide water that can be reused in power plant cooling. The attempt to account for impact costs due to release of greenhouse gases, acidifying gas emissions and particulate matter emissions in this study provides an aid to selecting a treatment alternative. From a better perspective of the total costs incurred from tertiary treatment of MWW for reuse in power plant cooling systems, a treatment alternative can be selected based on cost-effectiveness and overall sustainability.

The results of the LCI and LCC integration indicate that three-tiered treatment alternatives such as MWW_NSF and MWW_NFG, with regular chemical addition for treatment and conditioning and/or regeneration for tertiary treatment, tend to increase the impact costs and in turn the overall costs of tertiary treatment, even though their percent contribution to the total annual costs is moderate to low. RW and MWW_F alternatives with a single step of tertiary treatment have lower impact costs due to single level of tertiary treatment, but the contribution of impact costs to overall annual costs is higher than all other treatment alternatives and this indicates that these alternatives are less cost-effective when the impact costs are integrated. MWW_NF and MWW_SF alternatives with two-tiered tertiary treatment have moderate external impact costs with moderate infrastructure and chemical conditioner dosing, which makes them (especially the former alternative) better treatment alternatives from the environmental sustainability perspective since they contribute to minimal environmental damage from emissions. To further expand the study with a complete environmental sustainability and cost-effective analysis, environmental costs from other emissions to air, water and land, as well as benefits stemming from offsetting freshwater withdrawal by reusing treated effluent in power plant cooling systems need to be examined.

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8.0 Establish Critical Economic, Technical, and Social Factors Associated with Municipal Wastewater Reuse

In order to determine the most advantageous tertiary treatment and chemical management alternatives for reusing wastewater in a recirculating cooling system, life cycle cost (LCC) analysis and life cycle inventory (LCI) analysis have been employed. Either LCC or LCI can be used for decision making, but there are benefits to using both. A separation of LCI and LCC leaves uncharacterized the important relationships and trade-offs between the economic and life cycle environmental performance of product or process design decision scenarios (Norris, 2001).

An important objective of the research was to identify critical economic, technical, and social factors associated with municipal wastewater reuse in power plant cooling systems.

Economic and environmental factors associated with tertiary treatment of secondary municipal wastewater were evaluated by LCC and LCI, as discussed in Chapters 6 and 7. Specific aims of the LCC and LCI analyses included determining the most economical treatment strategy and the amount of greenhouse gases, particulate matter and acidifying gas emissions during construction of treatment units, manufacture and transport of chemicals used in treatment processes, and generation of the electric energy required to operate the treatment units.

Another important objective of this research was to evaluate the social acceptance of municipal wastewater reuse in power plant cooling systems. This chapter focuses on public acceptance of wastewater reuse in power plant cooling systems, and the social and cultural issues involved. Results of surveys and case studies were analyzed to identify and characterize public concerns about wastewater reuse for various purposes.

8.1 Methodology

Public participation is required for successful implementation of many environmentally based projects, but the meaning of “public” varies and needs to be defined for particular applications. For a wastewater facility and reuse planning, public may be identified as general public, potential users, environmental groups, special interest groups, home owners associations, farmers, regulators and/or regulating agencies, educational institutions, political leaders, and business/academic/community leaders. Different groups have different interests with respect to wastewater reuse (USEPA, 2004). Accordingly for specific water reuse programs like use of reclaimed water for industrial cooling and processing, EPA recommends regulatory, technical, and health experts, as well as representatives from the prospective user and its employees as public. To identify potential impacts on the public, it is necessary to first identify the total environmental impacts from the reclamation and reuse of wastewater for power plant cooling.

A potential impact of primary community concern with the use of treated wastewater in cooling systems is health impact due to air contaminants released from cooling towers. Aerosols that may contain human enteric pathogens and emissions of volatile organic compounds and particulate matter are of primary concern. Worker health impacts are also of interest as workers could be exposed to treated wastewater in liquid and aerosolized form in power plants utilizing reclaimed wastewater in cooling systems (EPRI, 2003).

A wastewater reuse project will be acceptable to the public only if the public health and safety concerns are addressed. Wastewater reuse projects have not gone forward in many instances because of incomplete and unsatisfactory examination of health and safety issues. An example of wastewater reuse implementation failure due to these reasons is discussed in the results section.

In this part of the research, social implications of wastewater reuse were investigated through historical case studies and survey of education and outreach strategies needed for wastewater reuse project success. Specific objectives included the determination of critical social factors that need to be considered in the comparative evaluation of secondary and tertiary treatment of make-up water, as well as the acquisition of documents and published data from surveys and literature for previous cases in which wastewater reuse has been successfully implemented. Historical case studies will serve to elucidate potential social implications and their impacts on the use of wastewater for cooling systems. The case studies and surveys were used to identify the social factors to be addressed during reclaimed water use. The results will help improve the strategies used to educate the public about wastewater reuse.

8.2 Results and Discussion

Review of papers in peer-reviewed journals and reports from regulatory and research institutions were performed to identify key issues and challenges involved with reuse of treated wastewater. Reports from a wide range of organizations were examined, including the Australian Commonwealth Scientific and Research Organization (CSIRO) (Po et al., 2004); the Land and Water program of the Electric Power Research Institute (EPRI, 2003); U.S. Environmental Protection Agency (USEPA, 1980, 2004); and Kennedy/Jenks Consultants (DuBose, 2009). This review of the literature yielded valuable information on the importance of public perceptions as well as case studies which have addressed potential problems involved in reusing wastewater in cooling systems.

Various surveys have been conducted to evaluate public views of wastewater reuse, and some of these are reviewed below. In addition, some specific cases in which treated municipal wastewater has been reused for power plant cooling are presented. Factors involved with public acceptance of treated wastewater reuse are examined.

8.2.1 Studies employing survey and modeling methods to examine public concerns to be addressed in relation to reclaimed water use

8.2.1.1 Clark County Sanitation District (Las Vegas, NV)

A survey on wastewater reuse benefits and concerns was conducted in Las Vegas, Nevada, and 838 people participated. The survey results, summarized in Figure 8.2.1, indicated that 78% of respondents believed that reuse of wastewater via industrial cooling is very beneficial to the community (USEPA, 2004).

8.2.1.2 University of Arizona (Tuscon, AZ)

A survey on reuse and recharge of treated water and wastewater conducted at the University of Arizona (Rock et al., 2009) studied public perception towards terminology, support for potential uses of reclaimed water, concerns about using reclaimed water, support of implementation strategies and how to reduce concerns about reclaimed water. It was found that when negative terminology such as “effluent”, tertiary treated wastewater”, “wastewater”, “toilet to tap” were replaced by positive terminology such as “water reuse”, “recycled water”, “re-purified water”, “reclaimed water”, the public accepted the concept of reuse more readily. This result implies that public outreach and communication plays a key role and can address concerns of the public regarding reusing treated wastewater. Public education is the key to overcoming public fears about a reuse system, particularly fears that relate to public health and water quality (McKenzie, 2005). Additionally, the survey confirmed that by providing stronger oversight of treatment plants, better wastewater treatment, and better information about reclaimed water could help reduce concerns about reclaimed water (Rock et al., 2009).

8.2.1.3 Modeling of socio-demographic factors

The Australian Commonwealth Scientific and Research Organization (CSIRO) collaborated with Australian Water Association (AWA) to conduct a literature review of factors

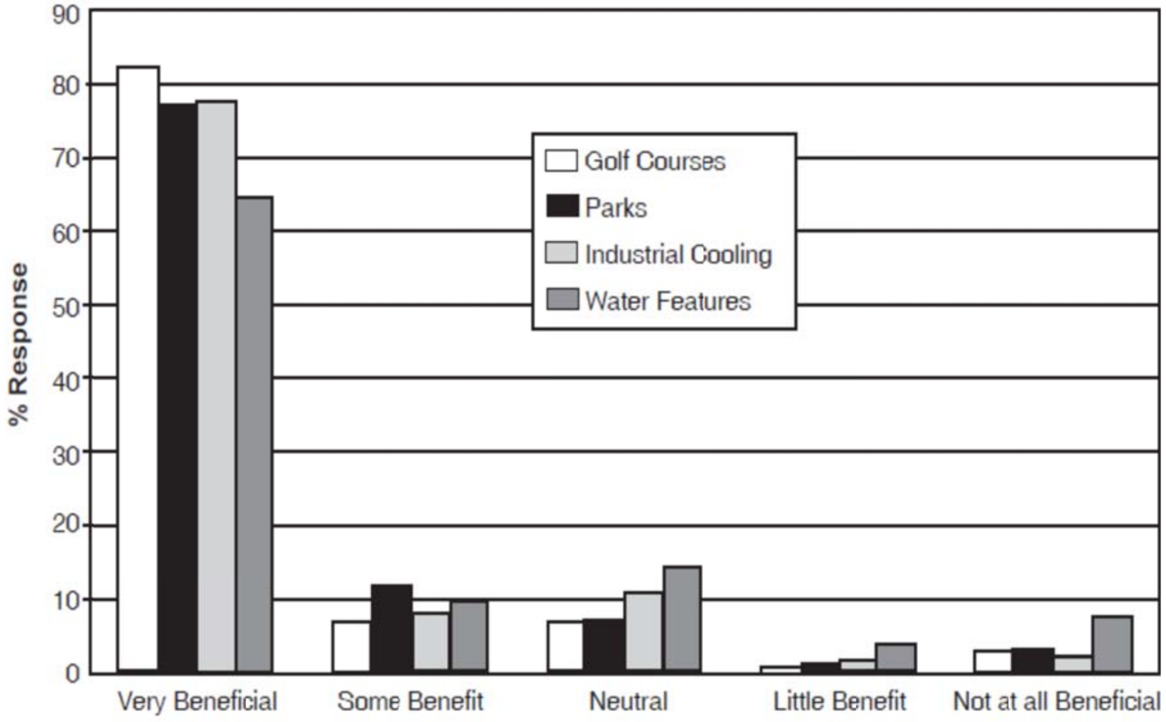

Figure 8.2.1 Survey results for different Reuse of treated Wastewater (USEPA, 2004)

influencing public perceptions of wastewater reuse. An important paper reviewed in this study was that of Po et al. (2004), in which Ajzen's theory of planned behavior (Figure 8.2.1.3) was applied to model the different factors that would influence people's willingness to use recycled water for horticultural purposes.

This model proposes that people's willingness to use recycled water (i.e. behavioral intention) is dependent on: (1) their attitudes towards using the water; (2) their perception of what their significant others think about using recycled water (i.e. subjective norm) and; (3) their perceived ease or difficulty in using recycled water (i.e. perceived control). Attitudes of people towards water reuse are in turn determined by their beliefs about the outcomes of using recycled water and their evaluation of the expected outcomes (i.e., behavioral beliefs and outcome evaluation) (Po et al., 2004).

Overall, the approach of using Ajzen's model in water reuse allows the differing factors to be mapped along with their possible influences on people's willingness to use recycled water. This model has been applied successfully in understanding conservation behaviors (e.g. water conservation) and has also been used to formulate successful conservation programs over the past decades (Po et al., 2004).

While the importance of community acceptance for a successful reuse program is widely acknowledged, there is a lack of social research in understanding the basis of public perceptions of water reuse and the psychological factors governing individual decision making (Po et al., 2004).

8.2.1.4 Survey of public opinion for water reuse (City of Corvallis, OR)

A survey of 1200 randomly selected registered voters in Corvallis was administered by Oregon State University and the Institute for Water and Watersheds during November-December 2008 . A total of 518 responses were received (46% response rate) (DuBose, 2009).

Some of the major questions addressed by the survey results were:

- What is the level of awareness in the community about water reuse?
- What kinds of information does the community need in order to feel comfortable with water reuse?
- What areas of education will the city need to focus on for the community to understand water reuse?
- What deal breakers would make water reuse unacceptable to the public?
- What is the best way to communicate the city's plans to the public and receive feedback? (DuBose, 2009)

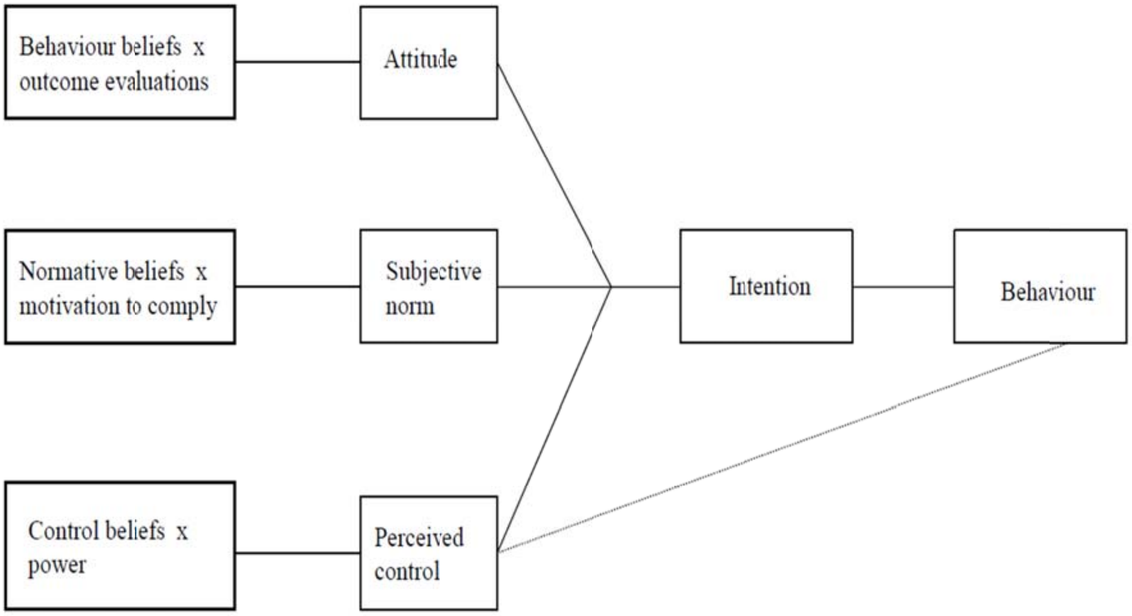

Figure 8.2.1.3 Ajzen (2001) theory of planned behavior (Po et al., 2004).

The results of the survey showed that:

- Significant public support for a water reuse program can be obtained, especially if it adheres to the principles of sustainability. Though the cost of a program would still be a consideration, according to the survey response many respondents appeared willing to pay higher fees for a more sustainable program. Comments on the survey responses indicated that if reuse required building of extensive and expensive infrastructure the support would be reduced.
- Low contact uses are likely to receive the most support from the public. The degree of contact respondents can expect to have with the water is linked to their support for a particular use.
- Public involvement and education efforts should be comprehensive, and include many different communication methods. Respondents clearly favor written communication that they can get in their homes such as mailed pamphlets and articles in local newspapers.
- Education efforts should focus on increasing awareness of the reason the city is proposing water reuse, and increasing knowledge of wastewater treatment. Comments written on the survey indicated a significant level of uncertainty about the safety of water reuse, perhaps for lack of knowledge about treatment technologies and contaminants that may remain in reclaimed water. Language used to discuss wastewater and water reuse must be carefully chosen (DuBose, 2009).

8.2.1.5 Public perception and participation in water reuse

The Water Environment Research Foundation in the United States funded an interdisciplinary and integrative social science study on public perception and participation in wastewater reuse within the US. It employed a three-phased research protocol consisting of: 1) literature review and three comprehensive case studies, including interpretive white papers from five different social science disciplines and public health and environmental engineering scientists; 2) a multi-stakeholder workshop to promote integrative, interdisciplinary analysis of the literature and case study findings; and 3) peer-review among twenty-one social science and water resource management experts (Hartley, 2006).

This study found that technical and scientific challenges that introduce greater uncertainty, and differences of opinions among scientists and engineers, intensify the public debate. Also, opinions about reuse change from non-potable to potable use. A summary of challenges and opportunities faced by water reuse professionals in regard to wastewater reuse is provided in Table 8.2.1.

Table 8.2.1 Challenges and opportunities faced by water reuse professionals in the US(a)

Challenges	Opportunities
• Decline in public trust and confidence in public agencies and officials	• Public interest in being meaningfully involved in water reuse decisions
• Decline in belief that best technologies can remove all impurities and germs from wastewater.	• Public interest in finding ways to ensure independent and secure water supplies for their community
• While the public tends to trust university-based scientists and the medical community on technical and health issues they trust their own impressions of water quality more.	• While the public is not well versed in the water cycle, they are generally aware that there are water supply problems in many parts of the country
• Public impression of water quality can often be based upon the water's turbidity	• Belief that some form of potable reuse in inevitable, given growth and water supply constraints.
• While education and outreach activities can increase support, they can intensify the extremes – those that oppose become more strongly opposed and those supportive are more strongly supportive	• Information sharing, educational activities opportunities for reflection upon the concepts of water reuse can increase support.

(a) Sources: Hartley, 2006; Lawrence, 2000; Jeffrey, 2001; Putnam, 1995; Broad, 1996; and Bruvold, 1981 and 1991

8.2.1.6 Survey of electric power plants using wastewater as cooling system makeup water

As described in Appendix D of this report, a 2009 survey of electric power plants using treated wastewater as cooling system makeup water revealed that there is little public awareness that the respondent plants are using wastewater, and limited to no efforts by the respondent plants to make the public aware. The reasons for both of these findings are unclear and merit further investigation

8.2.2 Case studies of reclaimed water use

8.2.2.1 Municipal electricity generating plant (Burbank, CA)

The Burbank, CA power plant was one of the first to disinfect and used treated wastewater effluent in a power plant cooling system. The plant conducted a study of bacteria in aerosols discharged from the cooling towers to quantify the numbers of fecal coliform and enteric pathogens. Pathogens did not occur in unacceptable quantities after the disinfection chemical was changed from gaseous chlorine to chlorine dioxide (Adams and Lewis, 1978).

8.2.2.2 Redhawk Power Station (Phoenix, AZ)

The 1060 MW Redhawk Power station is located about 60 miles west of Phoenix. The Redhawk plant is a combined cycle, gas-fired facility, and reuses nearly 1 billion gallons of reclaimed wastewater each year for cooling and other plant water needs. Redhawk was among the first power plants to adopt a zero liquid discharge (ZLD) system, which eliminates liquid waste from leaving the plant. Blowdown from the cooling system is evaporated (Yarbrough, 2006). There do not appear to be significant public concerns about reuse of wastewater at Redhawk, probably due in large part to the isolation of the facility.

8.2.2.3 Panda-Brandywine Power Station (Maryland)

A report by Argonne National Laboratory (Veil, 2007) includes a case study of Panda Brandywine plant near Washington. The 230 MW Panda Brandywine combined cycle gas-fired plant has used treated wastewater for cooling system makeup since 1996. The Mattawoman wastewater treatment plant in Charles County provides reclaimed water to Brandywine power plant, which uses about 1.5 million gallons a day of tertiary-treated water. The effluent from Mattawoman is delivered through a 17-mile-long pipeline. The water is chlorinated for biomass growth control, and chemicals are added to the water for corrosion control. No reports of public concern about the use of treated wastewater at Panda Brandywine were found.

8.2.2.4 Natural gas power plant (Brockton, MA)

A 350-megawatt natural gas power plant using treated wastewater was approved to be built by the Commonwealth of Massachusetts in Brockton, but 10 out of 11 city councilors opposed the project (Alspach, 2009). One of the reasons for this disapproval was concern about adding to the air pollution, as the wastewater treatment plant biosolids combustion unit already was a source of particulate matter emissions. Concern about additional emissions from the cooling towers if treated wastewater was used in the cooling system led the council to decline to sell the wastewater to the power plant (Alspach, 2009).

Brockton Clean Energy is now proposing to buy nearly 2 million gallons of drinking water a day to cool the plant's towers, after being refused access to treated waste water. However, city officials and other opponents say the company's latest proposal is an effort to avoid local zoning guidelines for the unpopular proposed project (Bolton, 2010).

8.2.2.5 Eraring Power Station (New South Wales, Australia)

The Eraring Energy Water Reclamation Plant (WRP) completed 14 years of successful operation in March 2005. The plant produces over 9 billion liters of reclaimed water annually. Eraring Energy's Water Reclamation Plant uses microfiltration and reverse osmosis technology to treat secondary effluent produced by the Hunter Water Corporation's Dora Creek Sewage Treatment Plant. The site is able to treat approximately 4.7 million liters of secondary effluent each day and saves up to 3.75 million liters of potable water daily. (Deans and Masson, 1995; Eraring Energy, 2009). The treated wastewater is used for cooling at the Eraring Power Station. There is strong community acceptance of wastewater reuse at the Eraring Power Station because of the reduced demand on the potable water source for the community.

8.2.2.6 Indian Point nuclear power plant (New York, NY)

The New York State Dept. of Environmental Conservation (DEC) on April 2, 2010 denied a water-quality standards certification for Units 2 and 3 of Entergy Corp.'s Westchester County-based Indian Point nuclear power plant. The certification, under Clean Water Act guidelines, is required by the U.S. Nuclear Regulatory Commission to renew the operating licenses for the units. The two units, which generate 1078 MW and 1080 MW, respectively, take in 2.5 billion gallons of water per day from the Hudson River and discharge the untreated, heated water back into the river. Due to concern about impacts of the discharge on the larvae of two fish species, the DEC recommended that the utility install a closed-cycle cooling system. Entergy claims that building such a system is not reasonable and would cost over $1 billion (ENR, 2010). This case study indicates that conversion to a recirculating cooling system is an expensive endeavor, but can be forced due to regulatory pressures. Use of wastewater as makeup water for a recirculating cooling system in such a situation could relieve regulatory pressure regarding discharges but add to the overall cost of converting.

8.3 Summary

Results of studies on behavioral analysis and public perception regarding wastewater reuse indicate the importance of outreach programs and positive communications in changing public perception towards wastewater reuse. The results of the review of community acceptance of wastewater reuse indicate that initial acceptance hinges in large part on the public's awareness of whether or not the reuse applications reflect sustainability principles, improve local water supply, and involve sufficient quality of reclaimed water. Also, the public seeks assurance that the reuse application being considered involves minimal risk of accidental exposure of workers and the public. Opinions about wastewater reuse are different for non-potable and potable uses. Public perception is largely influenced by the potential for human contact; factors such as health and extent of treatment are very important.

Case studies of reuse of wastewater in power plant cooling systems and other applications have revealed that the public is wary of using wastewater in cooling systems. There are several concerns typically voiced, but potential for human exposure is primary. The survey results from Las Vegas, NV indicate that the public views the reclamation of wastewater as both beneficial and as relatively environmental friendly. Studies done at a Burbank, CA power plant on use of treated wastewater for cooling has shown minimal health risks associated with aerosols from cooling towers. However, it was also seen that a Brockton, MA power plant has not been able to implement wastewater reuse due to opposition from local authorities and the public.

Efforts to communicate with the public about wastewater reuse and to build public acceptance should be conducted in the early stages of a project to provide sufficient time to address public concerns and needs. Addressing social challenges involved with wastewater reuse is most effective when information is imparted continuously during the decision making process, and in effective iterations during the multiple stages of the project to develop public confidence and trust.

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9.0 Summary and Conclusions

The overall objective of the study was to evaluate the benefits and life cycle costs of implementing tertiary treatment of secondary treated municipal wastewater prior to use in recirculating cooling systems versus an expanded chemical regimen for managing the quality of the cooling water when secondary treated municipal wastewater is used as makeup.

Specific objectives of the research were as follows:

(a) Determine the benefits and costs of subjecting secondary treated municipal wastewater to additional treatment to remove NH_3 by nitrification, BOD by GAC adsorption, and particles by filtration prior to use as cooling water makeup.

(b) Determine different chemical treatment regimens required to manage cooling water quality for different levels of tertiary treatment ranging from no tertiary treatment to inclusion of nitrification and treatment for the removal of particles and residual organic matter.

(c) Perform comparative life-cycle cost analyses for different levels of tertiary treatment and the corresponding chemical treatment regimens.

(d) Determine critical economic, technical, and social factors that need to be considered in comparative evaluation of tertiary treatment alternatives for secondary treated municipal wastewater when used as cooling system makeup water.

The study comprised bench- and pilot-scale experimental studies with three different tertiary treated municipal wastewaters, and life cycle costing and environmental analyses of various tertiary treatment schemes. Sustainability factors and metrics for reuse of treated wastewater in power plant cooling systems were also evaluated. The three tertiary treated wastewaters studied were: secondary treated municipal wastewater subjected to acid addition for pH control (MWW_pH); secondary treated municipal wastewater subjected to nitrification and sand filtration (MWW_NF); and secondary treated municipal wastewater subjected to nitrification, sand filtration, and GAC adsorption (MWW_NFG). Key findings and conclusions from the experimental studies, life cycle analyses, and sustainability analysis are presented in this section.

9.1 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: pH Adjustment (MWW_pH)

Reuse of MWW_pH water was carried out in both bench-scale and pilot-scale systems with appropriate chemical management programs for corrosion, scaling, and biofouling control. pH control of secondary treated municipal wastewater (MWW) by acidification increased the corrosiveness of the cooling water. Lower pH reduced scaling potential of MWW and resulted in the exposure of metal alloys to the flowing cooling water. Ammonia present in the MWW_pH water was aggressive towards metal alloys. Addition of 5ppm tolyltriazole (TTA) successfully reduced the corrosion rate of copper and cupronickel alloys within excellent levels according to industrial corrosion control criteria. Mild steel corrosion control was also significantly reduced after TTA addition, and it was lowered just within acceptable limits according to industrial corrosion control criteria.

Results from batch tests indicated that lower pH resulted in less precipitation. Scale formed when pH of the synthetic secondary-treated MWW was controlled at 8.4 and 7.8. Bench-scale recirculating tests showed that the addition of 5 ppm PMA delayed the scale deposition process but had minimal impact on the final total mineral deposits at pH 8.4, while the combination of pH control at 7.8 and the addition of 5 ppm PMA not only reduced the scaling rate but also decreased the final mass gain on the sampling specimens. Addition of PMA at pH 7.8 inhibited the formation of magnesium calcite and slowed the kinetics of calcium phosphate precipitation. The pilot-scale cooling tower tests were consistent with conclusion from bench-scale studies, i.e., pH adjustment to 7.8 was needed besides the addition of 5 ppm PMA to reduce the scaling rate significantly. With this scaling mitigation method, the formation of calcium carbonate was inhibited and calcium phosphate was the primary form of mineral scale on both unheated and heated surfaces.

Biofouling control was difficult and unpredictable when using MWW_pH as cooling tower makeup water for an extended testing period (i.e. up to 60 days). Although pH adjustment helped to mitigate the scaling problem, significant input of organic matter into the system still resulted in high variation in biocide residual and low active monochloramine concentration. Pilot-scale tests demonstrated that the biofilm formation can get out of control once monochloramine residual decreases below 3~4 ppm as Cl_2. Application of free chlorine at a dose below that required for breakpoint chlorination at pH 7.7 resulted in total chlorine residual above 4 mg/L but was unable to generate sufficient monochloramine residual to control biological growth. It can be concluded that maintaining 3~4 ppm of total chlorine residual is much less effective for biofouling control than maintaining 3~4 ppm of pre-formed monochloramine residual when MWW_pH is used as makeup water. Test results indicated that pH control at 7.7 yielded a slight increase in active monochloramine residual in cooling water but had limited impact on biological growth control. A single shock dose of 1000ppm glutaraldehyde reduced planktonic bacteria in the recirculating water for about 24 hours. However, the planktonic bacterial population recovered in the absence of adequate primary biocide residual. Sessile biological growth was not affected by a one-time shock dose of glutaraldehyde. Several consecutive 100-ppm doses of glutaraldehyde effectively reduced both planktonic and sessile bacterial growth in the cooling systems.

9.2 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: Nitrification and Filtration (MWW_NF)

Bench-scale studies carried out for corrosion analysis revealed that tertiary treatment with nitrification and sand-filtration increased the corrosivity of secondary treated municipal wastewater. Removal of the corrosion-causing constituent ammonia by nitrification was helpful for corrosion management of the system. Sand filtration removed a significant amount of total solids in the system. As a result there was lower scaling in the system compared to MWW, and the metal alloys exposed to recirculating cooling water experienced higher corrosion rate when immersed in MWW_NF than in MWW.

Tolyltriazole successfully inhibited the corrosion rate of copper and cupronickel alloys in both the bench-scale and pilot-scale studies. TTA also lowered the corrosion rate of mild steel alloys significantly, and was able to control it within acceptable levels according to the industrial corrosion control criteria. The polyphosphate based corrosion inhibitor sodium hexametaphosphate (SHMP) was found to be less effective than TTA in mild steel corrosion control. Most of the SHMP precipitated out of the solution and contributed to scaling. Precipitation of SHMP in the system lowered the corrosion rate of mild steel and cupronickel alloys. However, the reduction in corrosion rate was not significant compared to that using TTA as a corrosion inhibitor.

Bench-scale studies were carried out with 3ppm monochloramine dosing as a biocide in the recirculating cooling water system. It was found that use of 4ppm TTA was able to lower the corrosion rate of mild steel, copper, and cupronickel alloys in the presence of monochloramine.

Pilot-scale experiments were carried out with a control-cooling tower with no addition of corrosion inhibitor TTA, and two other cooling towers with different concentrations (2ppm, and 4 ppm) of TTA dosed for corrosion management. It was observed that even without any corrosion inhibitor, corrosion rates of copper and cupronickel alloys were within acceptable limit (after 28 days, and 56 days) according to the industrial corrosion control criteria. Addition of TTA reduced the corrosion rates of copper and cupronickel alloys to excellent levels. Mild steel corrosion rate was unacceptable in the control-cooling tower according to the industrial corrosion control criteria. However, use of 2ppm, and 4ppm TTA reduced the corrosion rate of mild steel significantly (higher reduction achieved with 4ppm TTA dose) and brought it down just within acceptable limits according to industrial corrosion control criteria.

Based on the results from batch tests, calcium phosphate was the main precipitate formed when MWW_NF was used for the recirculating cooling towers operated at typical operation conditions. Both bench- and pilot-scale studies showed that scaling was not a significant issue on unheated surfaces and no antiscalant was needed for the scaling control at relevant hydrodynamic conditions. However, on heated surface, it was necessary to control the pH in a range around pH 7.8 since comparatively lower pH would retain more soluble calcium and phosphate in the solution and yield crystalline fouling in the high temperature ranges. This study also showed that the flow velocity should be maintained high enough to prevent the sedimentation of bulk precipitates to both the unheated and heated surfaces.

Laboratory scale batch studies revealed that monochloramine (MCA) could control biological growth below 10^4 CFU/mL in MWW_NF at the initial dosage of 2 mg/L and 2 hours of contact time. Biocidal efficacy of MCA increased dramatically when the water temperature increased from 23ºC to 40 ºC. Under these conditions, free chlorine (NaOCl) and chlorine dioxide could achieve the same results at lower dosages of 0.5 mg/L and 1 mg/L, respectively because of a fairly low organic content in MWW_NF. However, an initial chlorine dioxide dose above 4 ppm is required to achieve detectable ClO_2 residual.

Control tests with MWW_NF at CoC 4 with no biocide addition in a recirculating system indicated that the planktonic bacteria population reached exponential growth phase within 24 hours while the sessile bacterial population easily exceeded the biofouling control criterion of 10^4 CFU/cm^2. Bench-scale studies in a recirculating system with continuous biocide addition demonstrated that all three biocides evaluated in this study could achieve biofouling control criteria for 7 days. Tests in the recirculating system revealed that chlorine dioxide residual of 0.5 ppm could easily achieve biofouling planktonic control criteria (10^4 CFU/mL) with minimal impact of water temperature.

Pilot-scale tests with MWW_NF as makeup water (i.e., CTA1, CTB1, and CTC1 tests) indicated that biofouling in the cooling tower could be controlled by maintaining MCA residual above 2 ppm in the recirculating water for 56 days. However, the growth curve of sessile heterotrophic bacteria indicated that biofilm growth exceeded biofouling control criteria after 56 days. Strict control of biocide dosing rate is the major operating requirement to achieve reliable biofouling control in cooling systems using tertiary treated municipal wastewater. Use of pre-formed monochloramine was more reliable in controlling biological growth compared to free chlorine and chlorine dioxide. Maintaining 4-5 mg/L of total chlorine residual by the addition of sodium hypochlorite could achieve target biofouling control criteria. Also, chlorine dioxide residual between 0.5-1 mg/L ClO_2 successfully achieved biofouling control criteria during the extended pilot-scale test.

Pilot-scale tests with MWW_NF as makeup water in the recirculating cooling system with continuous addition of monochloramine, free chlorine or chlorine dioxide revealed complete absence of Legionella species in the system. This result is significant in terms of public health protection for those that work or live around cooling systems using tertiary-treated municipal wastewater.

9.3 Reuse of Tertiary Treated Municipal Wastewater as Alternative Makeup Water for Cooling Systems: Nitrification-Filtration and Carbon Adsorption (MWW_NFG)

Corrosion analysis carried out in both bench-scale and pilot-scale studies using secondary treated municipal wastewater with nitrification, filtration, and activated carbon adsorption (MWW_NFG) revealed that MWW_NFG caused pitting corrosion to copper and cupronickel alloys in both bench-scale and pilot-scale studies. Use of 2ppm TTA in the system could not prevent the pitting corrosion of copper and cupronickel alloys in pilot-scale experiments carried out over a period of 28 days. The presence of pitting corrosion in copper and cupronickel alloys (which is unacceptable according to industrial corrosion control criteria) made the MWW_NFG unsuitable for use in recirculating cooling systems.

Mild steel corrosion rate was maintained within good levels (according to the industrial corrosion control criteria) after an immersion period of 28 days. The lower corrosion rate of mild steel in the cooling system was due to high scale formation (resulting from higher alkalinity) in MWW at the time period of the experiment.

Representative scaling experiments revealed that MWW_NFG showed equally low scaling potential as MWW_NF. The removal of organic matter did not cause significant changes in the scaling characteristics compared to MWW_NF.

Laboratory scale batch studies revealed that sodium hypochlorite exhibited slightly better results in terms of biocide demand and decay rate in MWW_NFG than in MWW or MWW_NF. A minimum of 0.5 ppm free chlorine residual was required to achieve control of biological growth (10^4 CFU/mL) in MWW_NFG at 40 ºC when the initial heterotrophic planktonic bacteria population was $10^{5.8}$ CFU/mL. Approximately 0.05-0.30 ppm of total chlorine residual was consistently formed when adding sodium hypochlorite as biocide but could not be identified as either monochloramine or free chlorine. Pre-formed monochloramine showed similar results in terms of biocide demand and decay rate in MWW_NFG as observed with MWW_NF. An initial pre-formed monochloramine dose of 1 ppm was required to achieve planktonic bacteria control criteria (10^4 CFU/mL) in MWW_NFG at 40 ºC when the initial heterotrophic planktonic bacteria population was $10^{5.6}$ CFU/mL.

The results of control tests (MWW_NF or MWW_NFG at CoC 4 with no biocide addition) in a recirculating system indicated that the planktonic bacteria population reached exponential growth phase within 24 hours. At the same time, sessile bacterial population exceeded the biofouling control criterion (10^4 CFU/cm^2). Comparison of biological growth between MWW_NF and MWW_NFG indicated that the TOC removal did not limit biological growth.

In bench-scale recirculating tests, removal of TOC was determined to enhance the performance of NaOCl by increasing the active free chlorine residual from 44% to 78% for the same dose in the concentrated municipal wastewater. For pre-formed monochloramine, the portion of monochloramine did not change significantly as a result of TOC removal. As for chlorine dioxide, it was observed that TOC removal led to lower biocide demand and reduced the dosage required to maintain similar chlorine dioxide residual in a bench scale recirculating system.

Performance of pre-formed monochloramine in pilot-scale cooling systems was more reliable than the other two biocides regardless of TOC level. TOC removal would have greatest benefit for tertiary treated municipal wastewater systems in which NaOCl is used as a biocide, if ammonia stripping in the full/pilot scale unit was as effective as in the laboratory unit. However, a portion of total chlorine residual needs to be characterized to ensure that there are no adverse impacts from its release into the environment. Removal of TOC was also determined to be beneficial when chlorine dioxide was used as a biocide as it reduces the initial biocide demand but not as much as in the case of free chlorine.

9.4 Life Cycle Conceptual Cost (LC^3) Model Estimates for Tertiary Treatment Alternatives for Secondary Municipal Wastewater Used as Cooling System Makeup Water

Life cycle cost (LCC) analysis was performed for the tertiary treatment processes MWW_pH, MWW_NF, and MWW_NFG, examined in bench- and pilot-scale experiments in this study, and also for MWW with sand filtration (MWW_F), MWW with lime softening and sand filtration (MWW_SF), and MWW with nitrification, lime softening, and sand filitration (MWW_NSF). A public domain conceptual costing tool (LC^3 model) was developed and used to determine LCC for each tertiary treatment process. Estimated costs for the selected tertiary treatment scenarios were compared with established costs of river water and city water supply as alternative sources of cooling water.

MWW with advanced treatment and addition of conditioning chemicals is promising as a power plant cooling system makeup water based on the cost range of \$0.91 - \$1.32 (in 2009\$/kgal), which is between the river water withdrawal and treatment costs of \$0.74 (in 2009\$/kgal) and the average city water costs of \$2.95 (in 2009\$/kgal). By applying the LC^3 model to the case of a MWW effluent with flowrate of 7.75 Mgal/d, the typical make-up water requirement for a 550 MW thermoelectric power plant, it was found that treated water supply and chemical conditioning costs dominated the overall cost.

MWW_SF and MWW_NF were the most cost-effective treatment options among the tertiary treatment alternatives considered in this study because of the higher effluent quality with moderate infrastructure costs and the relatively low doses of conditioning chemicals required. It was also found that reuse of MWW would prevent surface water withdrawal and improve the cost-efficiency in the long term when compared to using river water or city water as makeup water source.

9.5 Life Cycle Inventory Emission Estimates for Tertiary Treatment Alternatives for Secondary Municipal Wastewater Used as Cooling System Makeup Water

Life cycle inventory analysis along with integration of external costs of emissions with direct costs was performed to evaluate relative emissions to the environment and external costs associated with construction and operation of tertiary treatment alternatives for reusing secondary municipal wastewater in power plant cooling systems. Output inventories of greenhouse gas, sulfur dioxide, nitrogen oxides, and particulate matter (<2.5 micron) emissions to air showed that manufacture of chemicals for tertiary treatment and conditioning, and electric power generation associated with process operation are the main processes that contribute to these emissions. Transportation of chemicals and construction of tertiary treatment infrastructure to obtain higher quality effluent cause the increase in NO_x and $PM_{2.5}$ emissions, respectively.

MWW_pH exhibited relatively high NO_x, $PM_{2.5}$ and BTEX emissions to air, but remaining emissions to air, water, and land were relatively moderate. MWW_SF and MWW_NF exhibited moderate emissions, except for greenhouse gas emissions, wherein the former alternative emits more greenhouse gases compared to the latter alternative, mainly due to the addition of lime in MWW_SF. MWW_NF has somewhat higher potential to cause eco-toxicity compared to MWW_SF. The three-tiered treatment alternatives, MWW_NSF and MWW_NFG were associated with higher emissions in each category, with MWW_NSF having the highest emissions for all categories except for the BTEX emissions to water and soil/land, which were higher for MWW_NFG. The least emissions were estimated for RW and the reference case MWW, as there is not much infrastructure constructed and mainly addition of chemical conditioners is required. Water supply infrastructure and delivery showed high conventional pollutant emissions which are correlated with the pumping distance.

Relative toxic release inventory (TRI) air emissions for the various tertiary treatment processes followed the same trend as observed for relative GHG, SO_2, NO_x and $PM_{2.5}$ emissions estimated for the processes. One difference was that the emissions of lead, mercury, cadmium and their compounds (heavy metals) were higher to land than to air due to construction. Though MWW_SF and MWW_NF exhibited relatively lower and comparative TRI emissions to all compartments, MWW_SF had high TRI releases to air and land compared to MWW_NF, due to addition of lime. Therefore, MWW_NF would be environmentally preferred over MWW_SF.

Accounting for damages caused by the release of air emissions along with the direct economic costs can help balance the external damage and the value added by a particular service, such as tertiary treatment to provide water that can be reused in power plant cooling. From the perspective of total costs incurred from tertiary treatment of MWW for reuse in power plant cooling systems, a treatment alternative can be selected based on cost-effectiveness and overall sustainability.

LCI and LCC analysis integration indicated that three-tiered treatment alternatives such as MWW_NSF and MWW_NFG, with regular chemical addition for treatment and conditioning and/or regeneration, tend to increase the impact costs and in turn the overall costs of tertiary treatment, even though their percent contribution to the total annual costs is moderate to low. RW and MWW_F alternatives with a single step of tertiary treatment were associated with lower impact costs due to a single level of tertiary treatment, but the contribution of impact costs to overall annual costs was higher than for all other treatment alternatives. This result indicates that the RW and MWW_F alternatives are less cost-effective when the impact costs are integrated in the overall analysis. MWW_NF and MWW_SF alternatives with two-tiered tertiary treatment exhibited moderate external impact costs with moderate infrastructure and chemical conditioner dosing, which makes them (especially MWW_NF) better treatment alternatives from the environmental sustainability perspective since they exhibited minimal incremental environmental damage from emissions.

9.6 Sustainability Metrics for Reuse of Treated Municipal Wastewater in Cooling Systems

Analysis of the sustainability aspects of wastewater reuse in power plant cooling systems revealed that the sustainability of alternative approaches related to biofouling, scaling, and corrosion control and wastewater delivery can be partially assessed using available sustainability tools and metrics. However, the available tools and metrics are inadequate to address public perception of wastewater reuse, and valuation of freshwater versus wastewater. The primary element that makes treated wastewater reuse in power plant cooling systems sustainable, i.e., reuse of wastewater, is not adequately captured by existing sustainability tools.

Review of literature on behavioral analysis and public perception regarding wastewater reuse indicated the importance of outreach programs and positive communications in changing public perception towards wastewater reuse. The review of community acceptance of wastewater reuse indicate that initial acceptance hinges in large part on the public's awareness of whether or not the reuse applications reflect sustainability principles, improve local water supply, and involve sufficient quality of reclaimed water. Also, the public seeks assurance that the reuse application being considered involves minimal risk of accidental exposure of workers and the public. Opinions about wastewater reuse are different for non-potable and potable uses. Public perception is largely influenced by the potential for human contact; factors such as health and extent of treatment are very important.

Case studies of wastewater reuse in power plant cooling systems and other applications revealed that the public is wary of using wastewater in cooling systems. There are several concerns typically voiced, but potential for human exposure is primary. Survey results from Las Vegas, NV indicate that the public views the reclamation of wastewater as both beneficial and as relatively environmental friendly. Studies done at a Burbank, CA power plant on use of treated wastewater for cooling has shown minimal health risks associated with aerosols from cooling towers. However, it was also learned that a Brockton, MA power plant was not able to implement wastewater reuse due to opposition from local authorities and the public.

Efforts to communicate with the public about wastewater reuse and to build public acceptance should be conducted in the early stages of a project to provide sufficient time to address public concerns and needs. Addressing social challenges involved with wastewater reuse is most effective when information is imparted continuously during the decision making process, and in effective iterations during the multiple stages of the project to develop public confidence and trust.

Use of Treated Municipal Wastewater as Power Plant Cooling System Makeup Water: Tertiary Treatment versus Expanded Chemical Regimen for Recirculating Water Quality Management

September 2012

Final Technical Report - APPENDICES

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Pilot-scale cooling towers installed at FTMSA, PA

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APPENDIX A

Pilot-Scale Cooling Tower Performance Data

Field testing of cooling tower performance with three impaired waters involved the use of three pilot-scale cooling towers. The three towers were operated together at the Franklin Township Municipal Sanitary Authority (FTMSA) up to 2 months with tertiary treated municipal wastewaters. During the testing periods, the corrosion rates of various metal alloys, the scaling potential of the recirculating water, and biological growth were investigated for the treated water types and operational conditions studied. The pilot-scale cooling towers were cleaned and disinfected between test runs. The protocols that have been developed to recover the systems from severe fouling problems are described in a previous report (Vidic et al., 2009). This chapter describes the operational performance data of the pilot-scale cooling systems and ambient weather at FTMSA during the tests performed in this project.

Starting in May 2010, the three pilot-scale cooling towers were operated with tertiary effluents, namely nitrified and sand filtered wastewater and acidified secondary wastewater. The tests were performed at FTMSA for two consecutive 2-month periods. Additional testing was conducted using nitrified, sand filtered, and GAC treated wastewater as cooling tower makeup. During the pilot-scale tests, detailed information on tower operation was recorded, including temperature of water entering/exiting the heating section, temperature of water entering/exiting cooling section, airflow rate inside the cooling column, conductivity of makeup water, recirculating water, and blowdown discharge, makeup water volume, blowdown volume, water flowrate, and ambient conditions (weather, temperature, relative humidity).

Between experiments, the pilot scale cooling towers were cleaned with acid solution and disinfected with sodium hypochlorine. The heating section, especially the copper coils, was treated separately with 10% hydrochloric acid to ensure complete removal of scaling/biofouling material accumulated during the first test.

A.0 Review of Design Criteria for Pilot-scale Cooling Tower

Designated operational parameters are summarized in Table A.1. The fundamental design criteria for the pilot scale cooling towers include 3 GPM water flowrate in the recirculating system and 10 F cooling capacity across the cooling tower. Theoretical airflow rate and temperature at different locations are calculated to achieve the design criteria. Airflow rate of 150 cubic feet per minute inside the column is required to provide 10 F cooling capacity and the temperature of 150 F in the heat exchanger is required to make up the heat loss across the tower and maintain the temperature of water in the basin at 95 F and the temperature of water at the top of the tower at 105 F.

Table A.1. Operational parameters for the pilot-scale cooling tower unit

Parameter	Value	Unit
Water flowrate	3	GPM
Temperature differential	10	ºF
Airflow rate	150	CFM
Temperature of water in the heater	150	ºF
Temperature of water in the basin	95	ºF
Temperature of water at the nozzle	105	ºF
Cycles of concentration	4	-

A.1 Pilot-Scale Cooling Tower Performance When Using Nitrified and Sand Filtered Wastewater as Makeup (MWW_NF RUN)

The first experiment where tertiary effluent (MWW_NF run) was used as makeup water started on May 20th 2010 and finished on July 18th 2010; the second set of experiment where secondary effluent (MWW_pH run) used as makeup water started on August 1st 2010 and finished on October 6th 2010 (described in A.2.). Preliminary tests revealed that tertiary effluent had an average conductivity of 0.84 mS/cm. Therefore, target conductivity for the recirculating water was set between 3.5~4.2 to ensure the system is operated at CoC 4~5. Since the secondary effluent had an average conductivity of 0.93 mS/cm, target conductivity for the recirculating water was set between 3.7~4.6 mS/cm. Besides monitoring the conductivity for CoC control, cycles of concentration were also validated based on blowdown discharge volume and chloride concentration in the recirculating water.

The chemical inhibitor regimen for the pilot-scale tests with the tertiary treated wastewater MWW_NF from May-July 2010 were selected to examine variable doses of the corrosion control agent tolytriazole (TTA). Due to the relatively low pH and ammonia concentration in MWW_NF, corrosion is the major concern when using this water as cooling tower makeup. The chemical inhibitor dosing used in tests with MWW_NF and three pilot-scale cooling towers are shown in Table A.2. Cooling Tower A (CTA1) was a control test, with biofouling control only. In the tests with Cooling Tower B (CTB1) and Cooling Tower C (CTC1), TTA concentrations of 2ppm (CTB1) and 4ppm (CTC1) were studied. Concentrations of antiscaling agent polymaleic acid (PMA) and the biofouling control agent monochloramine (MCA) were held constant in CTB1 and CTC1.

Table A.2. Chemical inhibitor regimen for the pilot-scale cooling tower tests with MWW_NF

Chemical	CTA1	CTB1	CTC1	Function	Dosing location
TTA, ppm as dose	0	2	4	Anti-Corrosion	Makeup water tank
PMA, ppm as dose	0	5	5	Anti-scaling	Makeup water tank
MCA, ppm as residual	2~3	2~3	2~3	Biocontrol	Basin

Note: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine)

A.1.1 Water flowrate in the recirculating system

In the MWW_NF run, the water flowrate in the recirculating system was designated as 3 GPM. An inline flowmeter with scales between 1~5 GPM (Acrylic Flowmeter, 7511212B-08, King Instrument Company, CA) was used to monitor the variance of flowrate. The inline flowmeters were cleaned every 20 days. Flowrates measured in three towers throughout the test are shown on Figure A.1. In the first 7 days, the makeup water was withdrawn from the water channel between the nitrification process and inlet of a sand filtration process. Water withdrawal point was then changed to discharge from the sand filtration and was maintained there for the duration of the test. The average water flowrates in CTA1, CTB1, and CTC1, were 2.81 ± 0.05, 2.85 ± 0.05, and 2.88 ± 0.07 GPM, respectively. The water flowrate barely varied during the 60 day experiment. By the end of the MWW_NF run, the recirculating system was cleaned with acid. It is observed that very little scaling or biofouling formed inside the heat exchanger.

Although minor variance of water flowrates were observed in the MWW_NF run, it is determined that the water mass flow in all three cooling systems were well controlled and thus providing stable water velocity around 6 m/s when using tertiary effluent as cooling tower makeup.

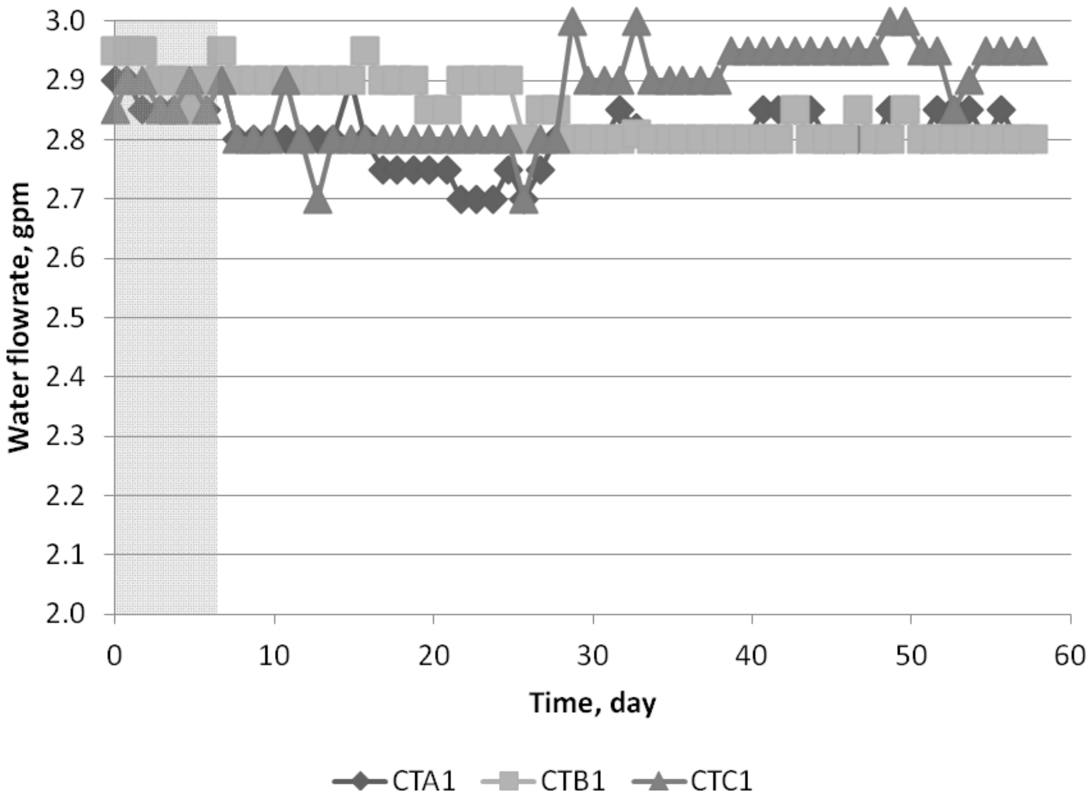


Figure A.1. Water flowrates in the recirculating system recorded in three cooling systems during the MWW_NF run in FTMSA (target flowrate was 3 GPM). Shaded area indicates the period of using unfilterated MWW_NF as cooling tower makeup.

A.1.2 Ambient condition, temperature profile, and air flowrate through the cooling tower

The ambient temperature used in the design of the cooling system was average wet bulb temperature in Pittsburgh, which is 79ºF (26.1ºC) in summer. Ambient temperature and relative humidity may vary daily with the local weather condition. Figures A.2 and A.3 show the daily ambient temperature and relative humidity in FTMSA, Murrysville, PA during the MWW_NF run. Ambient temperatures were acquired from AcuWeather (State College, PA) and are presented as daily highs and lows. Average high and low ambient temperatures through the 60 day experiment were 82 ± 5.8 ºF and 62.2 ± 6.3 ºF, respectively. Due to the lack of reliable humidity database, relative humidity was measured daily at the test site. The average relative humidity through the MWW_NF run was 61.10 ± 11.92 %.

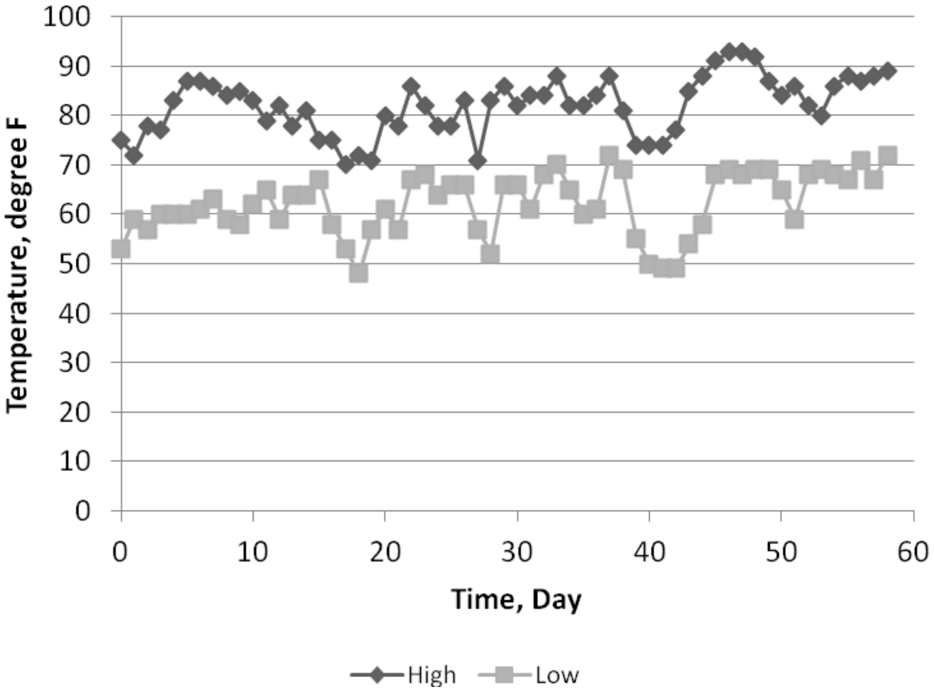


Figure A.2. Daily temperature recorded in FTMSA during the MWW_NF run. The experiment lasted for 60 days from May 21st to July 18th.

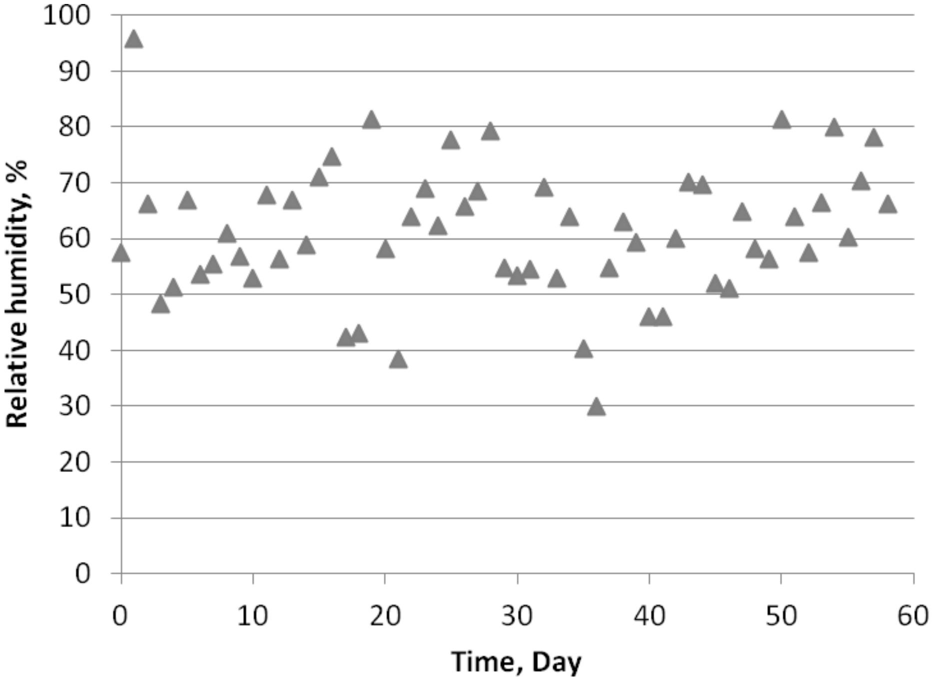


Figure A.3. Daily relative humidity in FTMSA during the MWW_NF run. Average relative humidity during this run was 61.10%.

The temperature differential (cooling capacity) is mainly affected by the air flowrate through the cooling column and temperature in the heat exchanger. Therefore, adjustment of the system is required in order to compensate for the influence of ambient conditions. Fine adjustment of input air flowrate and heating output were performed continuously to ensure desired water temperature values at different points in the system.

The air flowrate is measured by using anemometer at 3 different measuring points. Point X is near the center of column close to the nozzle; point Y is close to the wall; point Z is at the middle between points X and Y. Previous study (Vidic et. al., 2009) revealed that changing the fan speed only shifts the temperature differential instead of increasing the cooling capacity. Therefore, different fan speed settings were applied to all three towers to best meet desired temperature profile. Table A.3 shows the average air flowrate in all towers during MWW_NF run.

Figure A.4 shows the temperature differential (cooling capacity) and air flowrate for all cooling towers in MWW_NF run. Average temperature differentials in CTA1, CTB1, and CTC1 were 10.5 ± 1.3 ºF, 12.1 ± 1.3 ºF, and 10.6 ± 2.4 ºF. The temperature differentials in CTA1 and CTB1 were close to 10 ºF throughout the experiment. However, CTC1 has a larger variation because of the lower air flowrate. Results indicate that the cooling towers required higher airflow rate if the ambient temperature and relative humidity are comparatively higher.

Table A.3. Average air flowrate in pilot-scale cooling systems during the MWW_NF test

MWW_NF run	CTA1	CTB1	CTC1
Air flowrate, CFM	233 ± 34	216 ± 22	152 ± 46

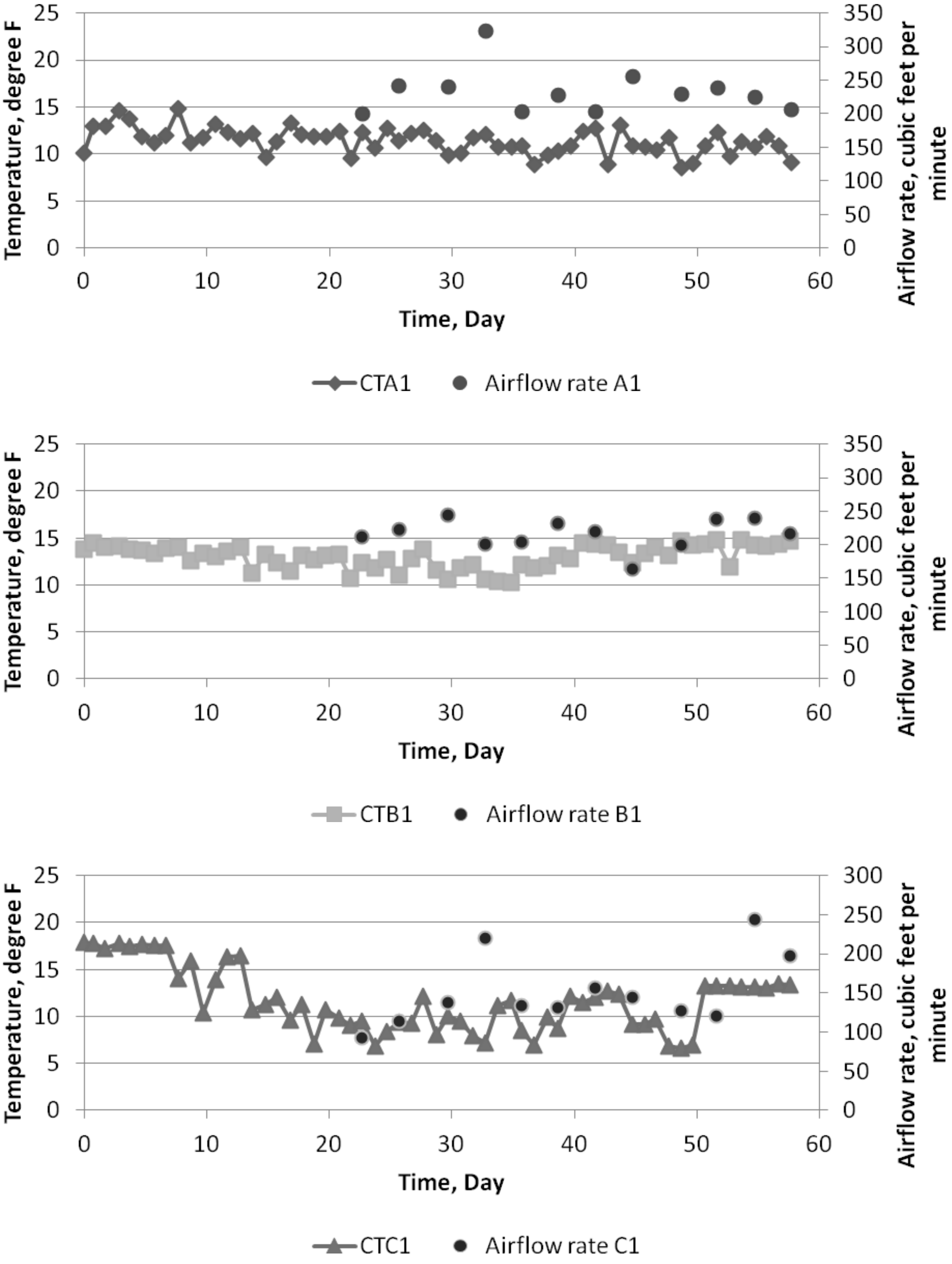


Figure A.4. Temperature differential across cooling towers and air flowrate during the MWW_NF run.

Besides the 10 ºF temperature differential across the tower, maintaining the temperature of water at spray nozzle at 105 ºF was also required to simulate the real situation in the power generating facilities. Table A.4 summarizes the average water temperature at nozzle for all towers during MWW_NF. It is shown that the water temperatures at the nozzle were maintained around 105 ºF in all of the cooling towers throughout the experiment.

Table A.4. Average water temperature at spray nozzle in MWW_NF run.

MWW_NF run	CTA1	CTB1	CTC1
Temperature of water at nozzle (ºF)	103.7 ± 2.4	103.9 ± 2.7	104.0 ± 3.7

The operating procedure used in the field was able to fulfill the design criteria for pilot scale cooling towers listed in Table A.1. Temperature profile can be controlled by adjusting the air flowrate and heat input. In addition, the ambient temperature is equally important for cooling tower operation since it affects the air flowrate required to reach target temperature differential.

A.1.3 Total makeup water volume and blowdown volume

The average daily water consumption rate is influenced by ambient temperature, relative humidity, temperature of water in the recirculating system, and air flowrate through the cooling column. Variations in these parameters will inevitably cause variations in daily makeup water consumption. The blowdown rate is controlled by an inline conductivity meter. When the conductivity of recirculating water exceeds the set point of using 3.8 mS/cm, the conductivity meter will send the signal to open the solenoid valve and initiate blowdown.

The makeup water rate and blowdown rate were recorded by inline totalizers and the evaporation rate can be calculated by subtracting the blowdown rate from the water consumption rate. Data collected during the MWW_NF run are summarized in Table A.5. The average makeup water consumption in the MWW_NF run was 46.4 gallons/day, while average daily blowdown and evaporation were 9.6 gallons and 37.2 gallons, respectively.

Table A.5. Average daily water consumption rate, blowdown rate, and evaporation rate for all three towers during MWW_NF run.

MWW_NF	CTA1	CTB1	CTC1
Daily Water Consuming rate (gallon/day)	43.3 ± 4.1	50.2 ± 5.8	45.7 ± 9.6
Daily Blowdown rate (gallon/day)	10.5 ± 2.8	9.0 ± 3.3	9.4 ± 3.8
Daily Evaporation rate (gallon/day)	32.9 ± 3.5	41.2 ± 5.5	37.4 ± 9.5

Figure A.5 shows the detailed daily water consumption and blowdown of the cooling towers during MWW_NF test. No mechanical failures were observed in CTA1 during these 2 months. In CTB1, it was observed that inline conductivity electrode was fouled and gave improper reading and thus causing much less blowdown on Day 12, 24, and 43. In CTC1, it was observed that the solenoid valve for blowdown control was clogged and allowed very low flow out of the basin between Day 24 and Day 30. The conductivity electrode in CTC1 was also fouled on Day 49, 50, and 51, which caused the increase of CoC and the decrease in water consumption rate.

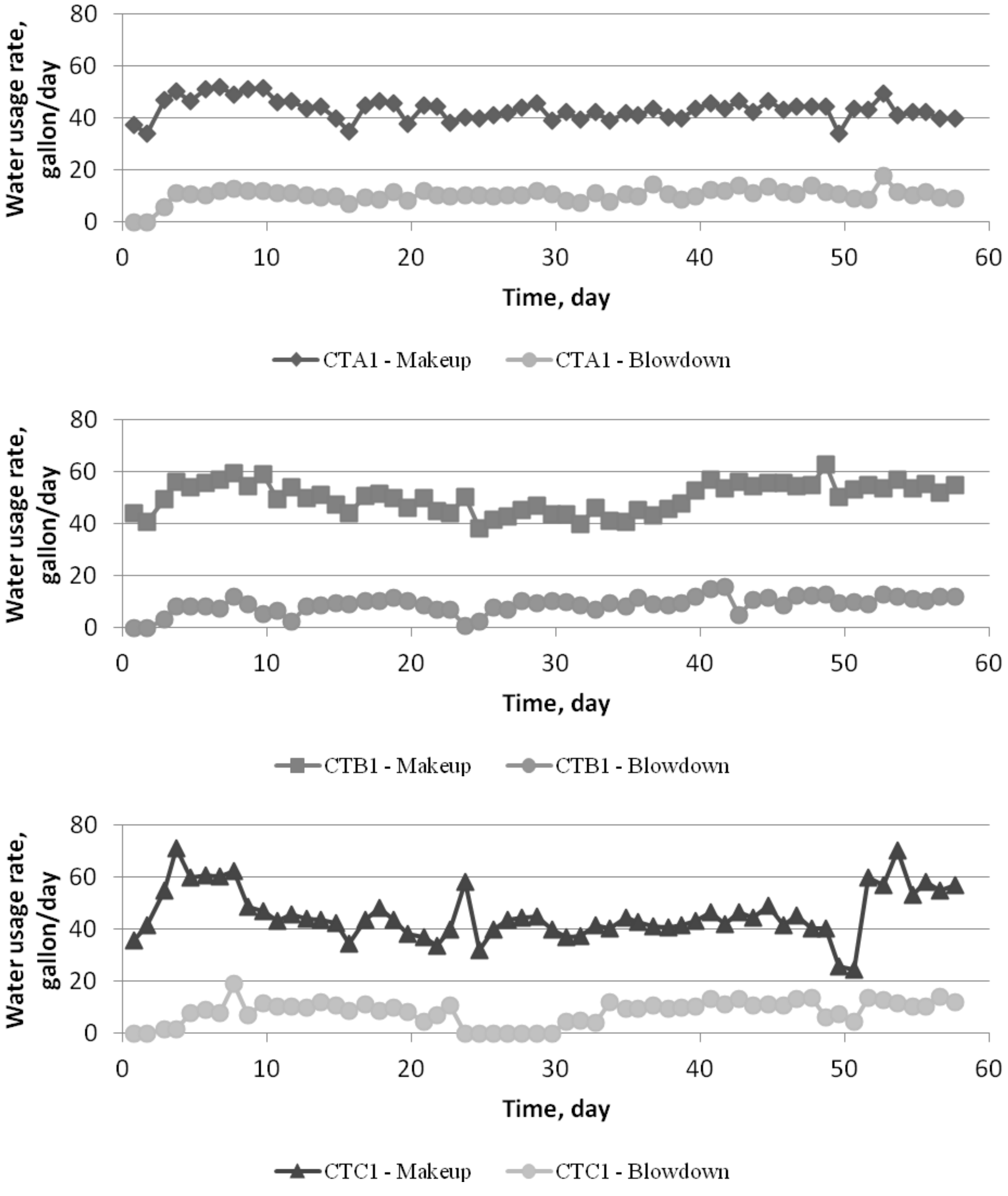


Figure A.5. Daily makeup water and blowdown volume during the MWW_NF run. The amounts of makeup water usage and blowdown discharge in CTA1, CTB1, and CTC1 are shown from top to bottom, respectively.

Generally, the observed cooling tower performance in the field corresponded well to design criteria. The performance of the three cooling towers was fairly stable with few mechanical failures of solenoid valves and fouled conductivity electrodes.

A.1.4 Cycles of concentration in the recirculating system

a) Based on conductivity

Conductivity of the makeup water was recorded daily with portable conductivity meter (Figure A.6). The average conductivity of makeup water (disinfected tertiary effluent) during the MWW_NF run was 0.87 ± 0.1 mS/cm. It was observed that all three towers reached 4 CoC after 48 hours from the beginning of the experiment. The average conductivity of recirculating water in CTA1, CTB1, and CTC1 were 3.56 ± 0.25, 3.83 ± 0.29, and 3.59 ± 0.52 mS/cm, respectively. Conductivity of recirculating water in CTA1 and CTB1 were well maintained in target range throughout the MWW_NF run. It was observed that the conductivity of recirculating water in CTC was below the target range from Day 3 to Day 20. Abnormal makeup water consumption rate also reflected on the conductivity measurement on Day 23, Day 50, and Day 51.

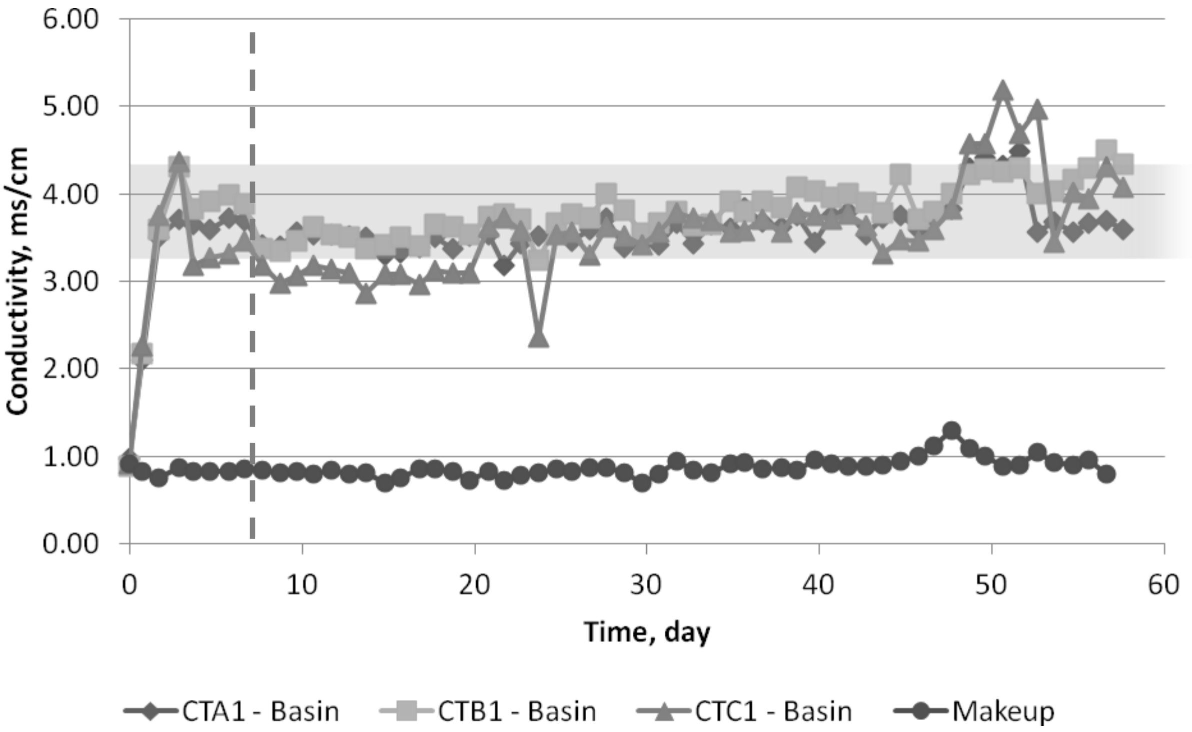


Figure A.6. Conductivities of MWW_NF and recirculating waters in the pilot scale cooling systems. Makeup water source was changed from MWW_N to MWW_NF after 7 days, which is indicated by a vertical dash line. Shaded area indicates the target range for CoC 4~5.

Table A.6 summarizes the conductivity data during the MWW_NF run. Conductivity of makeup water, water in the basin, and blowdown were recorded daily and are shown as 60-day average values. The last two columns show the calculated CoC based on different methods.

CoC(1) is calculated by dividing conductivity of recirculating water measured in the basin by the conductivity of raw tertiary effluent. CoC(2) is calculated by dividing the conductivity of blowdown by the conductivity of raw tertiary effluent. Results show that both the CoC(1) and CoC(2) were controlled in the range of 4-5 in all three towers. The reason that CoC calculations based on blowdown conductivity were higher is because of the sensitivity of online conductivity electrode and the blowdown mechanism. Turbulence in the basin can cause fluctuation of water quality, thus delaying the response time of the blowdown control system. In addition, the blowdown drainage was induced by gravity with a low flow velocity, while the recirculating water was still being evaporated (concentrated). Therefore, the conductivity of blowdown increased as a result of these effects.

Table A.6. Conductivity measurement in CTA1, CTB1, and CTC1 for MWW_NF run and calculated CoC based on two different methods.

Cooling system	Raw **(mS/cm)**	Basin **(mS/cm)**	Blowdown **(mS/cm)**	CoC (1)* **60-day Average**	CoC (2)** **60-day Average**
CTA1	0.87 ± 0.10	3.56 ± 0.25	4.05 ± 0.15	4.1	4.7
CTB1	-	3.83 ± 0.29	4.20 ± 0.29	4.4	4.8
CTC1	-	3.59 ± 0.52	3.78 ± 0.33	4.1	4.3

* The CoC(1) is calculated by dividing conductivity of recirculating water by conductivity of makeup water.
** The CoC(2) is calculated by dividing conductivity of blowdown by conductivity of makeup water.

b) Based on blowdown volume

Besides using conductivity of the recirculating water and of the blowdown to define CoC, the CoC can also be derived by dividing the total water consumption by the total daily blowdown volume (CoC_{BD}). Results are shown in Table A.7 for the MWW_NF run period. Based on this method, it is determined that CTA and CTC were operated inside the target CoC 4~5, while CTB was operated at CoC_{BD} 5.6. Although the conductivity values in all three towers were similar, the higher water consumption rate caused by larger temperature differential in CTB inevitably led to the increase in CoC_{BD}. Figure A.7 shows the daily CoC_{BD}. Results of daily CoC calculation based on blowdown volume indicate that CTA was comparatively better controlled than CTB and CTC. However, an extremely low conductivity of recirculating water can significantly decrease the blowdown volume in the next day, thus leading to an extremely high CoC_{BD} value.

Table A.7. CoC calculated with volumetric based method for the MWW_NF run of the pilot-scale cooling systems.

Tower	Water consumption rate (Gal/Day)	Blowdown rate (Gal/Day)	CoC_{BD} **60-day Average**
CTA1	43.3 ± 4.1	10.5 ± 2.8	4.1
CTB1	50.2 ± 5.8	9.0 ± 3.3	5.6
CTC1	45.7 ± 9.6	9.4 ± 3.8	4.9

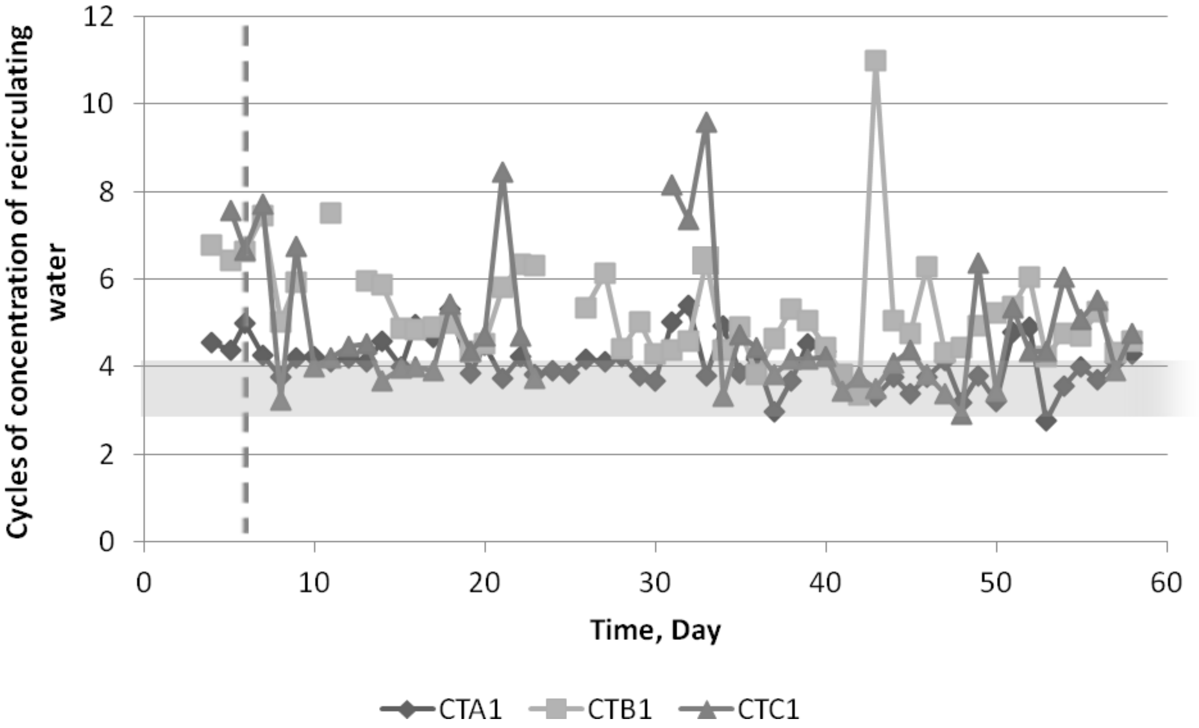


Figure A.7. Daily CoC based on blowdown volume during the MWW_NF run. Makeup water source was changed from MWW_N to MWW_NF after 7 days, which is indicated by a vertical dash line. Shaded area indicates the target range for CoC 4~5.

c) Based on water chemistry

Chloride concentration can also be used to calculate the CoC due to its inert nature (low precipitation tendency). By dividing the chloride concentration in the basin by that in the makeup water, the CoC can be derived on the basis of water chemistry. However, addition of chlorine based biocide affects the mass balance for chloride ions. In order to maintain 2-3 ppm monochloramine in the recirculating system, a biocide dosing rate of at least 50 ppm/day was required. Therefore, calculations of chloride based CoC (CoC_{Cl}) should excluded the amount of chloride contributed by the biocide addition. CoC_{Cl} was then calculated by following equation.

$$\mathrm{CoC_{Cl}} = \frac{C_{RE} - \left(\frac{V_{Biocide} \times C_{Stock}}{V_{MK}} \times \left(1 - \frac{V_{BD}}{V_{MK}} \right) \right)}{C_{MK}}$$

where,
C_{RE} = Chloride concentration in the recirculating system, ppm;
C_{MK} = Chloride concentration in the makeup water, ppm;
C_{Stock} = Chloride concentration in the biocide stock solution, ppm;
$V_{Biocide}$ = Biocide volume, gallon;
V_{MK} = Makeup water volume, gallon;
and V_{BD} = Blowdown volume, gallon.

Table A.8 and Figure A.8 show the 60-day average $\mathrm{CoC_{Cl}}$ and dialy $\mathrm{CoC_{Cl}}$ during the MWW_NF run. All three cooling towers were mostly maintained between 4-5 $\mathrm{CoC_{Cl}}$ throughout the experiment.

Table A.8. $\mathrm{CoC_{Cl}}$ calculated with chloride concentration for the preliminary testing of the pilot-scale cooling systems.

Tower	Chloride concentration in makeup, ppm	Chloride concentration in the system, ppm	Chloride concentration added by biocide, ppm	$\mathrm{CoC_{Cl}}$
CTA1	212 ± 24	920 ± 119	63 ± 22	4.1
CTB1	216 ± 21	1003 ± 114	100 ± 38	4.3
CTC1	222 ± 22	1033 ± 61	75 ± 36	4.7

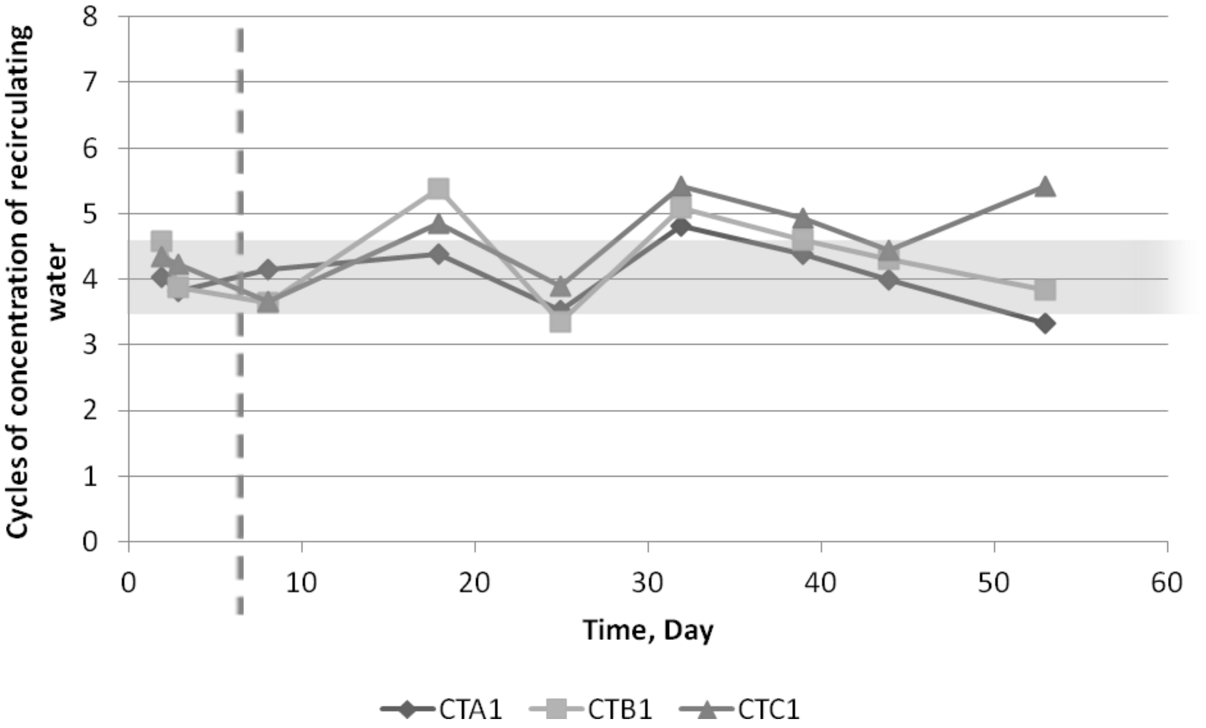


Figure A.8 Daily CoC based on chloride concentration during the MWW_NF run. Makeup water source was changed from MWW_N to MWW_NF after 7 days, which is indicated by a vertical dash line. Shaded area indicates the target range for CoC 4~5.

In conclusion, the cooling towers were appropriately controlled to maintain CoC in a target range based on conductivity measurements, volumetric measurements of makeup and blowdown, or the chloride concentration in the recirculating water.

A.2 Pilot-Scale Cooling Tower Performance When Using Secondary Treated and Acidified Wastewater as Makeup (MWW_pH Run)

The second set of pilot-scale tests conducted in Summer 2010 focused on the effectiveness of pH adjustment for control of scaling when using secondary treated wastewater as cooling system makeup water, and on a direct comparison of monochloramine versus hypochorite for biofouling control. The previous related study (Vidic et al., 2009) showed that scaling and biofouling are major concerns when using secondary treated municipal wastewater as cooling tower makeup. The chemical inhibitor regimens for the MWW_pH tests in Summer 2010 were selected to focus on the pH effect for scaling control and on different biocides for biofouling control. The chemical regimens of the three cooling systems are shown in Table A.9

Table A.9. Chemical inhibitor regimen for the pilot-scale cooling tower tests with MWW_pH

Chemical	CTA2	CTB2	CTC2	Function	Dosing location
TTA, ppm as dose	2	5	5	Anti-Corrosion	Makeup water tank
PMA, ppm as dose	5	5	5	Anti-scaling	Makeup water tank
MCA, ppm as residual	3~4	3~4	-	Biocontrol	Basin
FC, ppm as TC residual	-	-	3~4	Biocontrol	Basin
pH control	NC	7.7	7.7	Anti-scaling	Basin

Notes: TTA (Tolyltriazole); PMA (Polymaleic acid); MCA (Monochloramine); FC (Free chlorine); TC (Total chlorine). NC = no pH adjustment; wastewater used as received.

A.2.1 Water flowrate in the recirculating system

In the MWW_pH run, the water flowrate in the recirculating system was set at 3 GPM. An inline flowmeter with scales between 1~5 GPM (Acrylic Flowmeter, 7511212B-08, King Instrument Company, CA) was used to monitor the variance in flowrate. The inline flowmeters were cleaned every 20. Water flowrates in three towers throughout the testing period of 2 months are shown on Figure A.9. The average water flowrates in CT2A, CTB2, and CTC2, were 2.51 ± 0.19, 2.75 ± 0.05, and 2.71 ± 0.17 GPM, respectively. The water flowrate changes in CTA2 indicated severe scaling problem inside the heating session within 20 days. The copper coil of CTA2 was cleaned with 10% hydrochloric acid for 30 minutes using a side steam to avoid contaminating the recirculating water in the cooling system. First acid wash of CTA2 on Day 22 recovered 100% water flowrate and the second acid wash of CTA2 on Day 41 only recovered 98.2% of water flowrate. CTC2 also had notable decrease to 2.45 GPM in water flowrate after 30 days. Biofouling problem was observed at the same time in CTC2 due to the insufficient amount of monochloramine formation in the MWW. The water flowrate recovered to 2.6 GPM after adding 1,000 ppm glutaraldehyde as shock dose on Day 50. In conclusion, all the results suggest that the water flowrate accurately reflected either scaling or biofouling problem in the recirculating system. With proper treatment, the water flowrate in recirculating system can be recovered to original capacity.

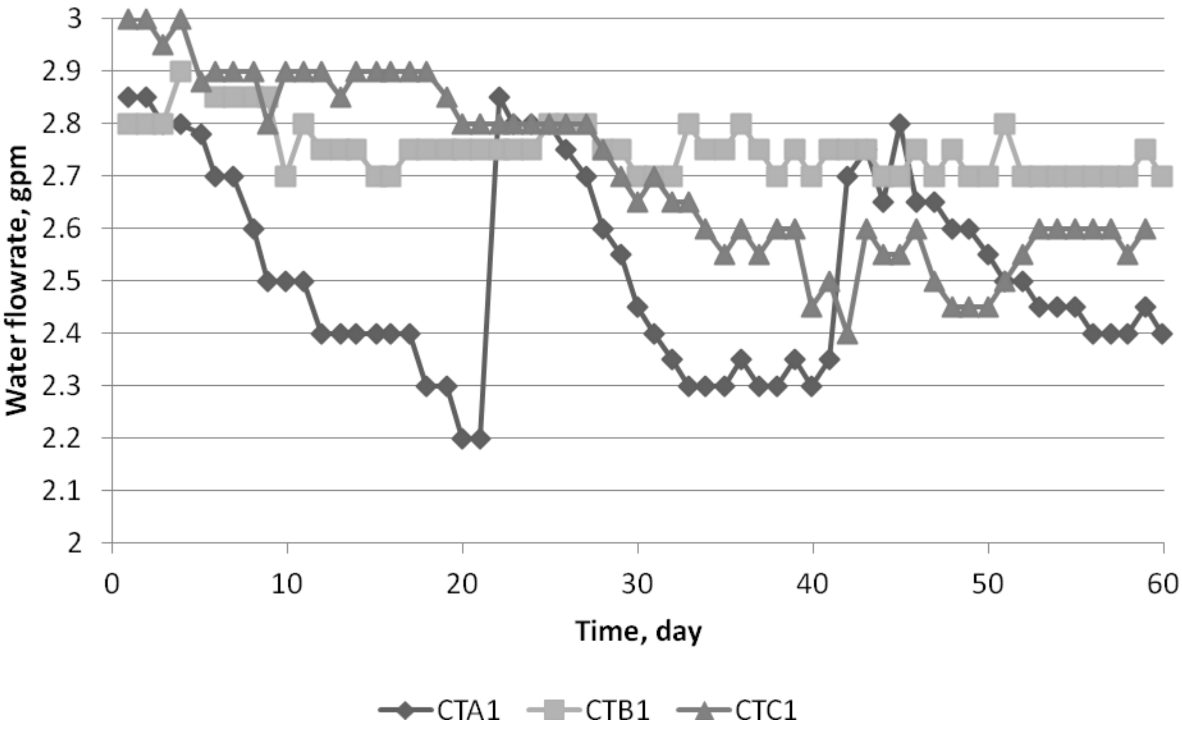


Figure A.9. Water flowrate measured in the system during the MWW_pHrun in FTMSA (target flowrate was 3 GPM). All three cooling towers were using secondary effluent as makeup water source.

A.2.2 Ambient conditions, temperature profile, and air flowrate through the cooling tower

The ambient temperature used in the design of the cooling system was average wet bulb temperature in Pittsburgh, which is 79ºF (26.1ºC) in summer. Figure A.10 and A.11 show the daily ambient temperature and relative humidity in FTMSA, Murrysville, PA during the MWW_pH run. Ambient temperatures were acquired from AcuWeather (State College, PA) and are presented as daily highs and lows. Average high and low ambient temperatures through the 60 day experiment were 80.7 ± 8.5 ºF and 59.6 ± 7.7 ºF, respectively. Due to the lack of reliable humidity database, relative humidity was measured daily on site. The average relative humidity through the MWW_pH run was 65.6 ± 16.0 %. The ambient temperature was lower by the end of the experiment along with frequent precipitations. Due to the low ambient temperature, the water consumption rate was expected to decrease.

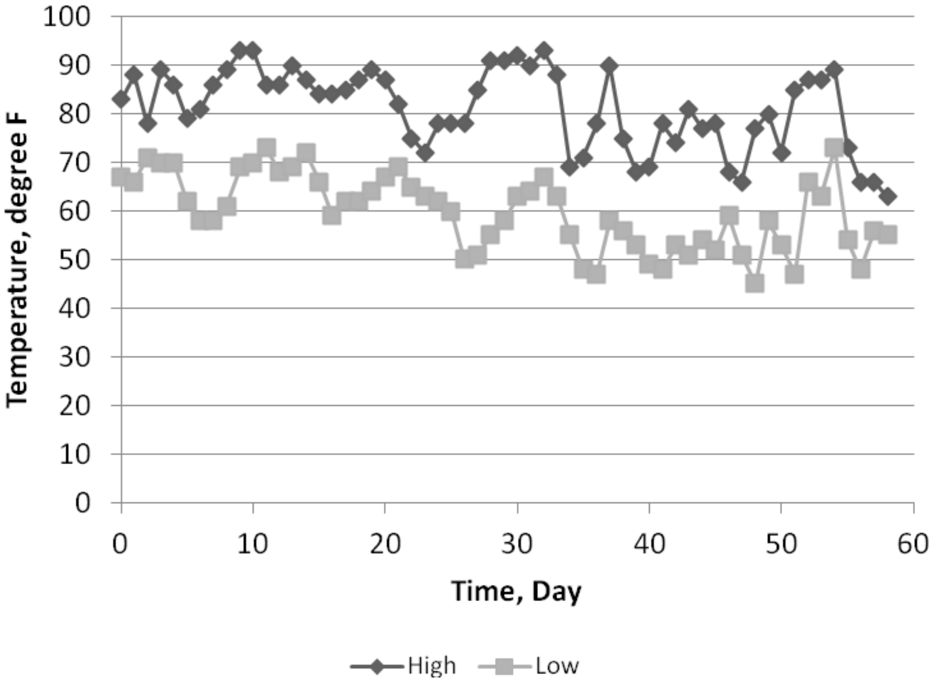


Figure A.10. Daily temperature recorded in FTMSA during the MWW_pH run. The experiment lasted for 60 days from Aug 1st to Oct 6th.

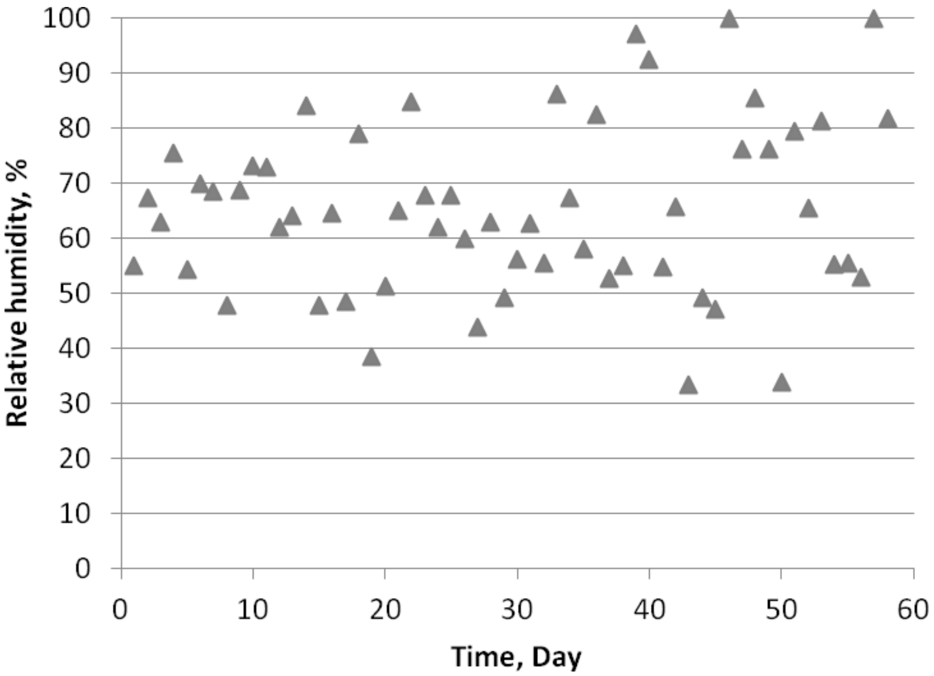


Figure A.11. Daily relative humidity in FTMSA during the MWW_pH run. Average relative humidity during this run was 65.6%.

In order to increase the temperature of water before entering the cooling section to 105ºF, both heater set point and airfan speed were increased. However, this adjustment also directly caused the increase in evaporation rate from the cooling column. Table A.10 shows the average air flowrate in all towers during MWW_pH run. Figures A.12 shows the temperature differential (cooling capacity) and air flowrate for all cooling towers in MWW_pH run. Average temperature differentials in CTA2, CTB2, and CTC2 were 12.4 ± 1.8 ºF, 14.4 ± 1.8 ºF, and 13.5 ± 0.9 ºF. The temperature differentials in all towers were much higher than 10 ºF throughout the experiment due to the high air flowrate and the low ambient temperature.

Table A.10 Average air flowrate in cooling column in all pilot-scale cooling systems during the MWW_pH run

MWW_pH run	CTA2	CTB2	CTC2
Air flowrate, CFM	281 ± 30	274 ± 40	247 ± 45

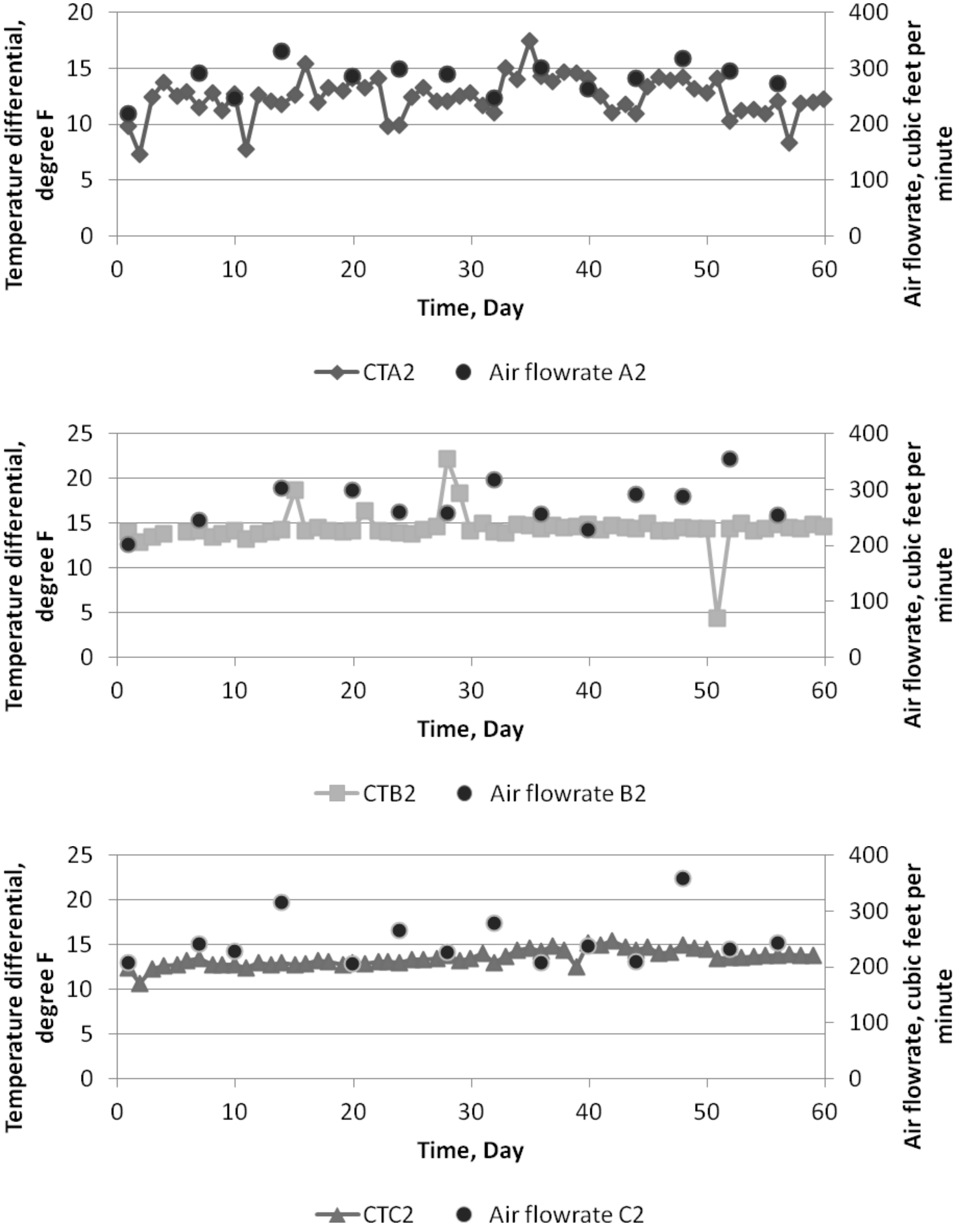


Figure A.12. Temperature differential across cooling towers and air flowrate during the MWW_pH run.

Table A.11 summarizes the average water temperature at nozzle for all towers during MWW. It is shown that the water temperature at the nozzle was maintained around 105 ºF in CTA2, while water temperatures in CTB2 and CTC2 were slightly lower than designated criteria.

Table A.11. Average water temperature at spray nozzle in MWW_pH run

MWW_pH run	CTA2	CTB2	CTC2
Temperature of water at nozzle (ºF)	104.8 ± 2.7	103.8 ± 4.5	103.8 ± 3.3

A.2.3 Total makeup water volume and blowdown volume

The average daily water consumption rate is influenced by ambient temperature, relative humidity, temperature of water in the recirculating system, and air flowrate through the cooling column. The blowdown rate was controlled by an inline conductivity meter, which had conductivity set point at 4 mS/cm for MWW_pH run. The makeup water rate and blowdown rate were recorded by inline totalizers and the evaporation rate can be calculated by subtracting the blowdown rate from the water consumption rate. Data collected during the MWW_pH run are summarized in Table A.12. The average makeup water consumption in the MWW_pH run was 51.3 gallons/day, while average daily blowdown and evaporation were 12.1 gallons and 39.2 gallons, respectively. As mentioned above, water consumption rate did increase along with the low ambient temperature and high air flowrate in the cooling column.

Table A.12. Average daily water consumption rate, blowdown rate, and evaporation rate for all three towers during MWW_pH run.

MWW_pH run	CTA2	CTB2	CTC2
Daily Water Consumption rate (gallon/day)	44.7 ± 5.5	54.8 ± 6.4	55.1 ± 4.8
Daily Blowdown rate (gallon/day)	10.2 ± 2.9	10.9 ± 5.4	11.2 ± 5.3
Daily Evaporation rate (gallon/day)	34.4 ± 4.3	44.5 ± 6.6	43.9 ± 7.0

Figure A.13 shows the detailed daily water consumption and blowdown during the MWW_pH run. Throughout the 2 months, CTA2 had only one mechanical failure on Day 12, when the axial fan was broken and was replaced within one day. However, acid wash was conducted twice for CTA2 on Day 22 and Day 41. In CTB2, conductivity electrode was fouled and was giving improper reading and thus causing much less blowdown on Day 4, and between Day 12~15. The recirculating system in CTB2 also ran out of makeup water on Day 5 and Day 29 due to excess blowdown. In CTC2, the solenoid valve for blowdown control was broken but still let very low amount of water flow out of the basin continuously between Day 3 and Day 8. As a result, conductivity of recirculating water did not increase significantly; however, the blowdown volume was not recorded throughout this period because the blowdown totalizer could not register such a low flowrate. As predicted, the performance of cooling towers was acceptable and in accordance with design criteria. However, the systems still required more frequent examination during the experiment to ensure proper operation.

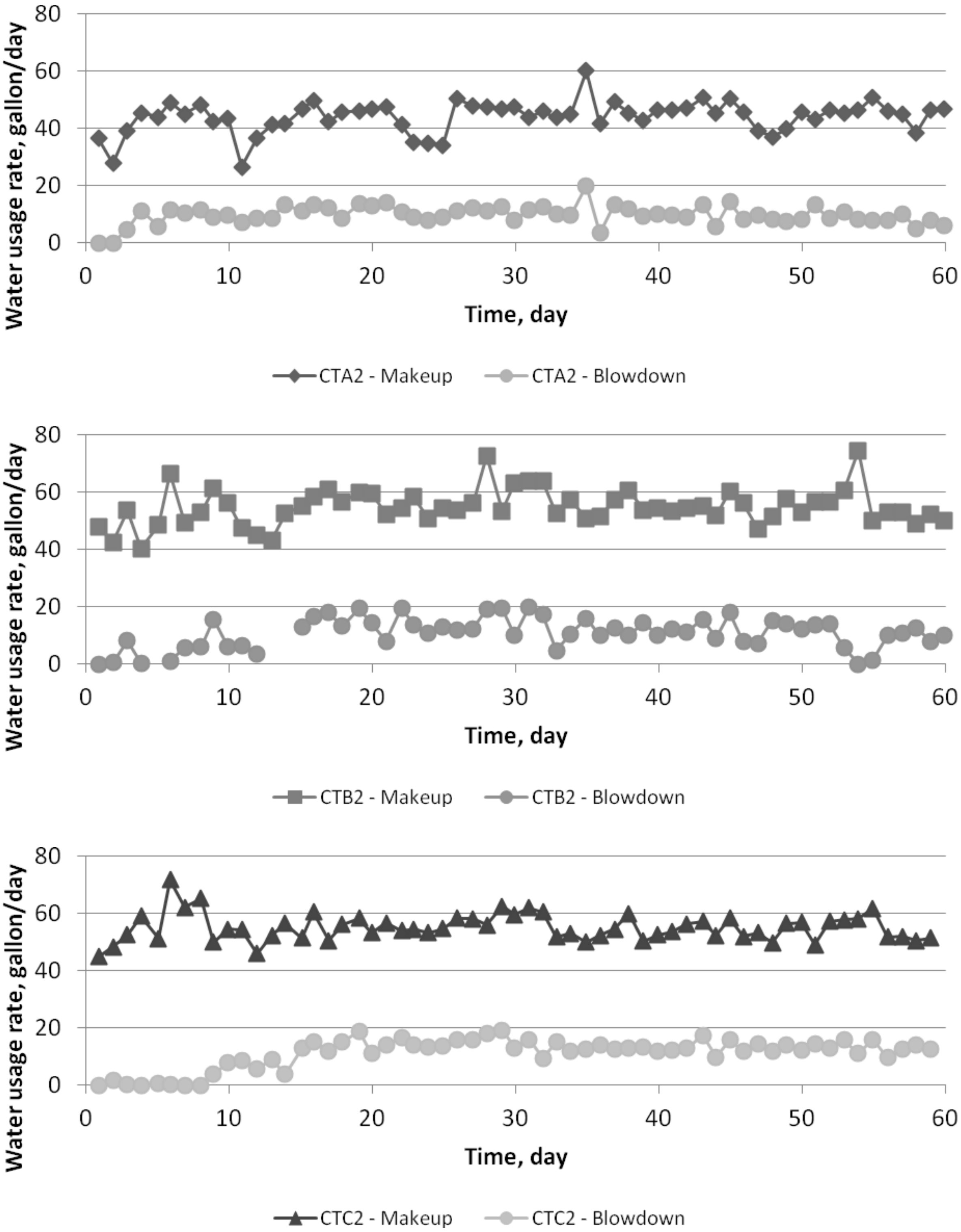


Figure A.13. Daily makeup water and blowdown volume during the MWW_pH run. Makeup water usage and blowdown discharge in CTA, CTB, and CTC is shown from top to bottom, respectively.

A.2.4 Cycles of concentration in the recirculating system

a) Based on conductivity

Conductivity of the makeup water was recorded daily with portable conductivity meter (Figure A.14). The average conductivity of makeup water (disinfected secondary effluent) during the MWW_pH run was 0.93 ± 0.1 mS/cm. It was observed that all three towers reached 4 CoC after 48 hours from the beginning of the experiment. The average conductivity of recirculating water in CTA2, CTB2, and CTC2 were 4.57 ± 0.42, 4.38 ± 1.20, and 3.98 ± 0.39 mS/cm, respectively. Conductivity of recirculating water in CTA2 and CTC2 were fairly well maintained in the target range throughout the MWW_pH run. However, conductivity of recirculating water in CTB2 was beyond the target range from Day 12 to 15 due the malfunctioned conductivity electrode.

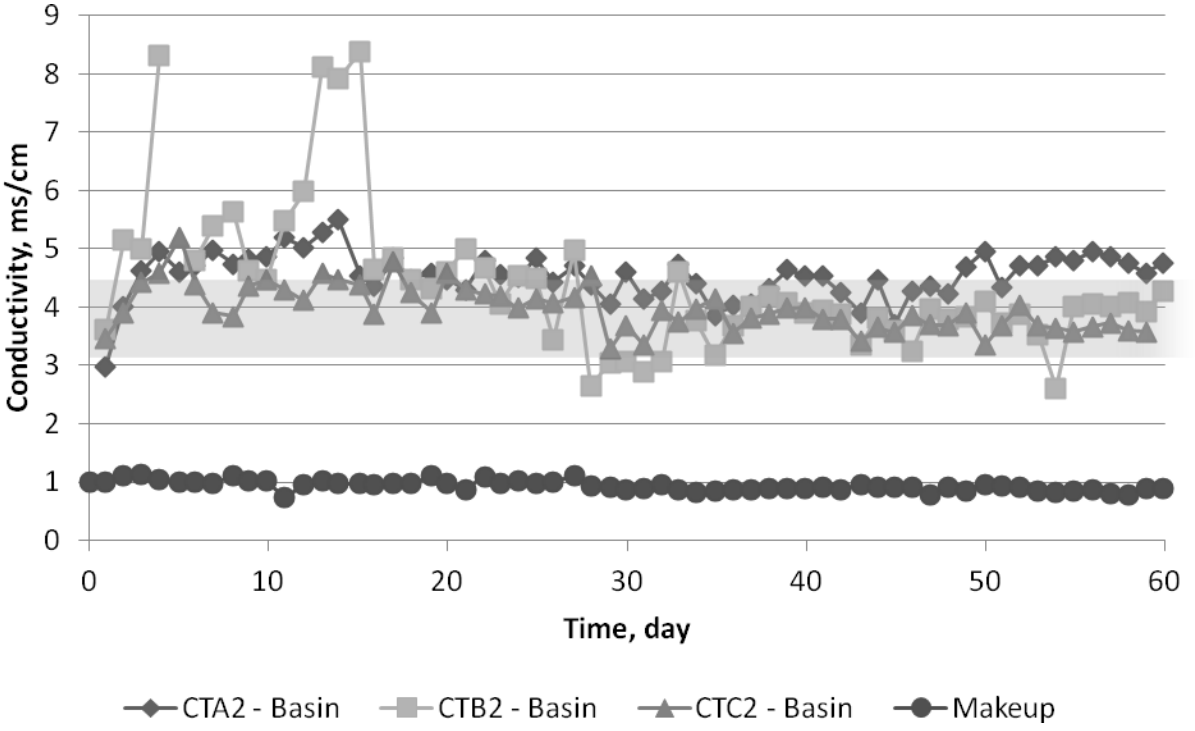


Figure A.14. Conductivities of MWW_pH and recirculating waters in the pilot scale cooling systems. Shaded area indicates the target range for CoC control.

Table A.13 summarizes the conductivity data during the MWW_pH run. Conductivity of makeup water, water in the basin, and blowdown were recorded daily and are shown as 60-day average values. Results indicate that 60-day average conductivity of recirculating water in all towers was inside the design criteria. Because the conductivity in CTA2 and CTB2 were at high end of design criteria, CoCs calculated based on conductivity of blowdown were above 5.

Table A.13 Conductivity measurement in CTA2, CTB2, and CTC2 for MWW_pH run and calculated CoC based on two different methods.

Cooling system	Raw **(mS/cm)**	Basin **(mS/cm)**	Blowdown **(mS/cm)**	CoC (1)* **60-day Average**	CoC (2)** **60-day Average**
CTA2	0.93 ± 0.09	4.57 ± 0.42	5.01 ± 0.66	4.9	5.4
CTB2		4.38 ± 1.20	4.83 ± 0.86	4.7	5.2
CTC2		3.98 ± 0.39	4.26 ± 0.35	4.3	4.6

* The CoC(1) is calculated by dividing conductivity of recirculating water by conductivity of makeup water.

** The CoC(2) is calculated by dividing conductivity of blowdown by conductivity of makeup water.

b) Based on blowdown volume

Results are shown in Table A.14 for the MWW_pH run period. Based on this method, it is determined that all three cooling systems were operated inside the target CoC 4~5. Figure A.15 shows the daily CoC_{BD}. Results of daily CoC calculation based on blowdown volume indicate that CTA2 was comparatively better controlled than CTB2 and CTC2.

Table A.14 CoC calculated with volumetric method for the preliminary testing of the pilot-scale cooling systems.

Tower	Water consumption rate (Gal/Day)	Blowdown rate (Gal/Day)	CoC_{BD} **60-day Average**
CTA2	44.7 ± 5.5	10.2 ± 2.9	4.4
CTB2	54.8 ± 6.4	10.9 ± 5.4	5.0
CTC2	55.1 ± 4.8	11.2 ± 5.3	4.9

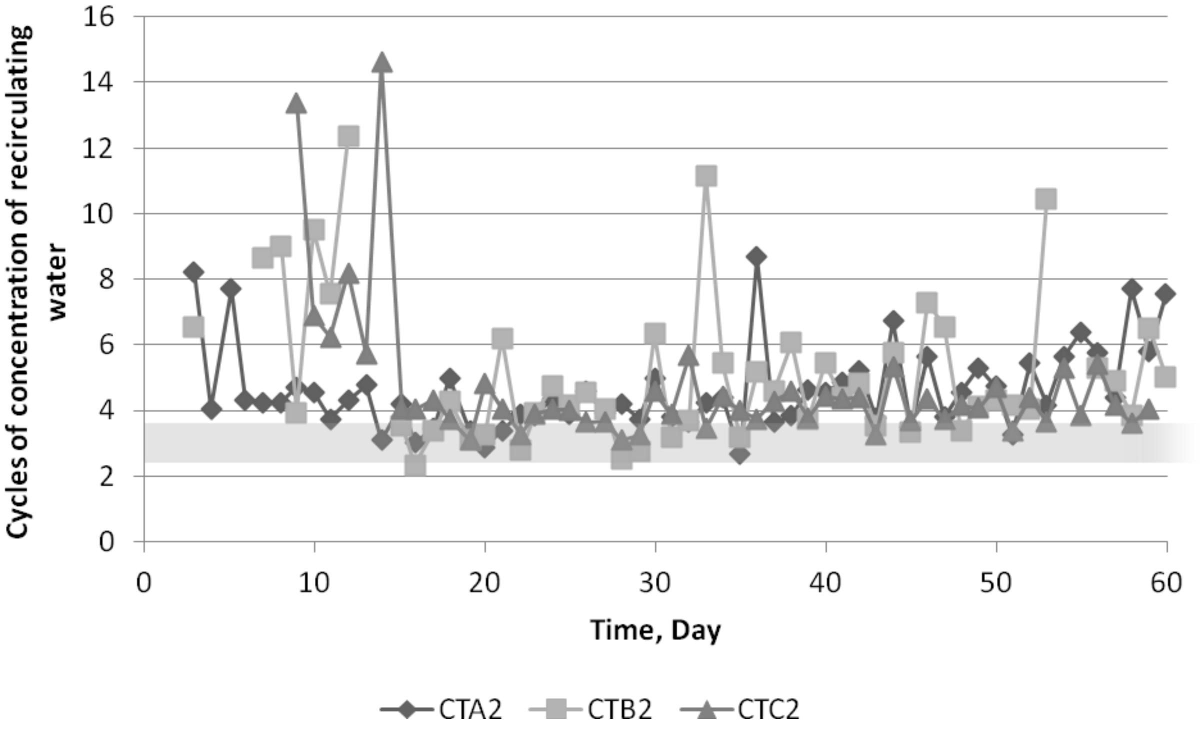


Figure A.15 Daily CoC based on blowdown volume during the MWW_pH run. Shaded area indicates the target range for CoC 4~5.

c) Based on water quality

Chloride concentration can also be used to calculate the CoC due to its inert nature (low precipitation tendency). By dividing the chloride concentration in the basin by that in the makeup water, the CoC can be derived on the basis of water chemistry. However, addition of chlorine based biocide affects the mass balance for chloride ions. In order to maintain 3-4 ppm monochloramine in the recirculating system, a biocide dosing rate of at least 65 ppm/day was required. Therefore, calculations of chloride based CoC (CoC_{Cl}) should excluded the amount of chloride contributed by the biocide addition. CoC_{Cl} was then calculated following the same equation described in section A.1.4 subsection C. Table A.15 and Figure A.16 show the 60-day average CoC_{Cl} and dialy CoC_{Cl} during the MWW_NF run. It was shown that CTA2 and CTB2 were operated at higher CoC than CTC2.

Table A.15. CoC_{Cl} calculated with chloride concentration for the preliminary testing of the pilot-scale cooling systems.

Tower	Chloride concentration in makeup, ppm	Chloride concentration, ppm	Chloride concentration by biocide, ppm	CoC_{Cl}
CTA2	199 ± 25	1091 ± 120	56 ± 29	5.2
CTB2	199 ± 25	1179 ± 326	68 ± 35	5.7
CTC2	199 ± 25	965 ± 109	67 ± 20	4.5

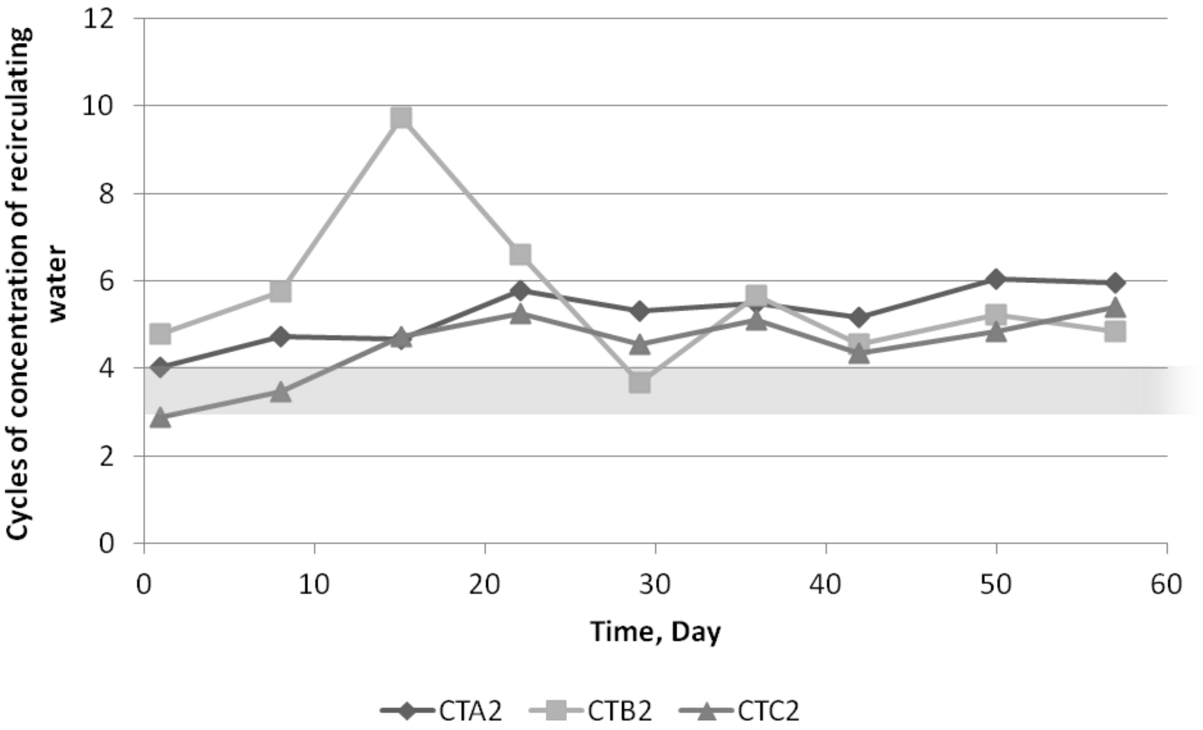


Figure A.16 Daily CoC based on chloride concentration during the MWW_pH run. Shaded area indicates the target range for CoC 4~5.

A.3 Tests with Tertiary Treated Wastewaters as Cooling Tower Makeup (Mix Run)

In May-June 2011, the three pilot-scale cooling towers were operated with tertiary effluents, namely secondary treated plus nitrification and sand filtration (MWW_NF) and the former plus additional GAC adsorption (MWW_NFG), from FTMSA as cooling tower makeup water for one month period. During this pilot-scale test, detailed information on tower operation was recorded, including temperature of water entering/exiting the heating section, temperature of water entering/exiting cooling section, airflow rate inside the cooling column, conductivity of makeup water, recirculating water, and blowdown discharge, makeup water volume, blowdown volume, water flowrate, and ambient conditions (weather, temperature, and relative humidity).

Preliminary tests revealed that both MWW_NF and MWW_NFG had similar average conductivity of 0.84 mS/cm. Therefore, target conductivity for the recirculating water was set between 3.5~4.2 to ensure the system is operated at CoC 4~5. Besides monitoring the conductivity for CoC control, cycles of concentration were also validated based on blowdown discharge volume in the recirculating water.

The three test conditions evaluated in the pilot-scale testing in Summer 2011 were: (1) MWW_NF with addition of sodium hypochlorite as biocide and without corrosion control; (2) MWW_NF with addition of chlorine dioxide as biocide and with tolytriazole (TTA) for corrosion control; and (3) MWW_NFG with addition of pre-formed monochloramine (MCA) as biocide and with TTA for corrosion control. A control test with MWW_NF water containing 2-3 ppm MCA as biocide and TTA for corrosion control was previously conducted in Summer 2010. The pilot-scale test with NaOCl as biocide and without the addition of any corrosion control agent was conducted in order to test the field portable electrochemical corrosion measurement device under aggressive corrosion conditions. The test conditions for the Summer 2011 pilot tests are summarized in Table A.16.

Table A.16. Chemical inhibitor regimen for the pilot-scale cooling tower tests.

Chemical	Pilot-scale Cooling Towers			Function	Dosing location
	CTA	CTB	CTC		
TTA, ppm as dose	0	2	2	Anti-Corrosion	Makeup water tank
NaOCl,ppm as residual	$1\sim2^{(1)}$	0	0	Biocide	Basin
ClO_2, ppm as residual	0	$0.5\sim1^{(2)}$	0	Biocide	Basin
MCA, ppm as residual	0	0	$2\sim3^{(3)}$	Biocide	Basin

(1)Residual as free chlorine (2) Residual as chlorine dioxide (3) Residual as monochloramine

A.3.1 Water flowrate in the recirculating system

The water flowrate in the recirculating system was designated as 3 GPM to provide a turbulent flow inside the recirculating with a Reynolds number of $1.92\text{x}10^4$. An inline flowmeter with a range of 1~5 GPM and a maximal accuracy of 0.2 GPM (Acrylic Flowmeter, 7511212B-08, King Instrument Company, CA) was used to monitor the water flowrate and the measurements are shown on Figure A.17. It was observed that water flowrate decreased sharply in the first two days from 2.9 to 2.8 and then decreased to 2.7 gradually in 10 days.

The average water flowrates in CTA3, CTB3, and CTC3, were 2.79 ± 0.04, 2.76 ± 0.06, and 2.72 ± 0.06 GPM, respectively. Although minor variance of water flowrates were observed in recirculating systems using either MWW_NF or MWW_NFG, it is determined that the water mass flow in all three cooling systems were well controlled and thus providing stable water velocity around 2 ft/s (0.6 m/s), comparing to designated criteria, 2.16 ft/s (0.66 m/s), when using the advanced treated tertiary effluents as cooling tower makeup. This result also suggests that there was no significant scaling/biofouling occurred in the heat exchanger.

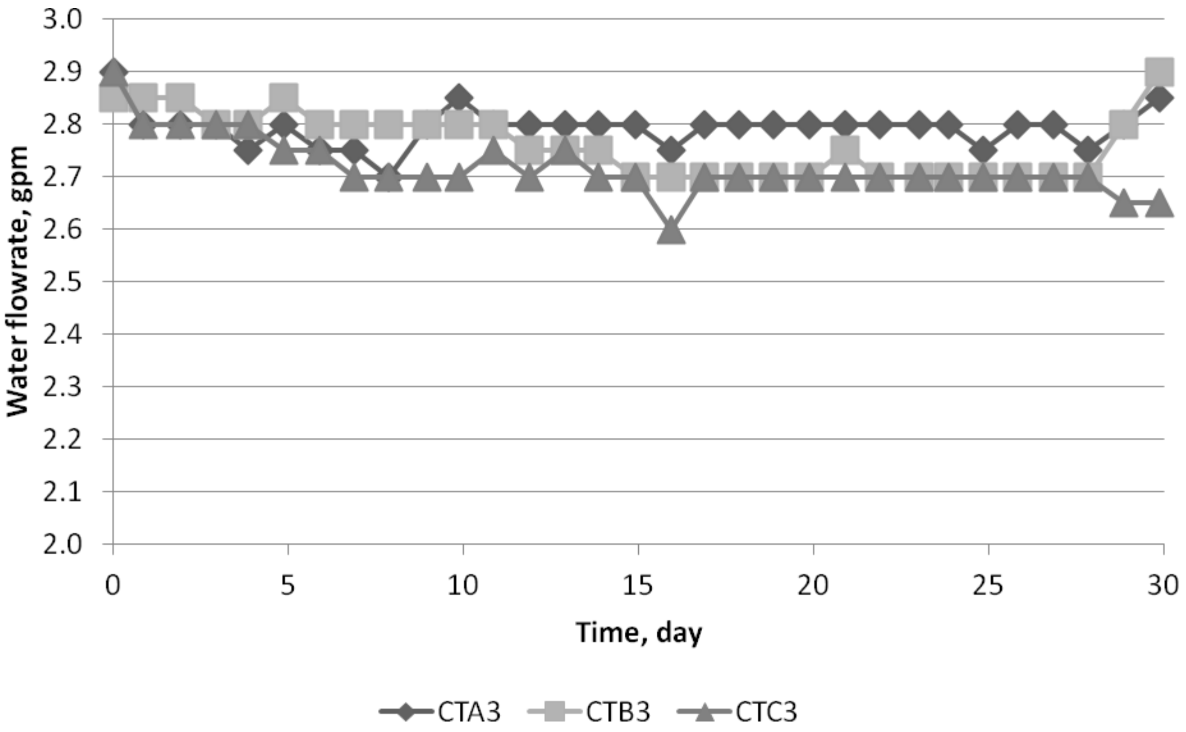


Figure A.17. Water flowrates measured during the pilot-scale testing, Summer 2011. Target flowrate was 3 GPM for the cooling tower systems.

A.3.2 Ambient condition, temperature profile, and air flowrate through the cooling tower

The ambient temperature used to design the cooling system was average wet bulb temperature in Pittsburgh, which is 79ºF (26.1ºC) in summer. Figure A.18 shows the ambient temperature and relative humidity at FTMSA, Murrysville, PA during the Summer 2011 testing. Temperature data were acquired from AccuWeather (State College, PA) and relative humidity was measured manually at FTMSA. Average high and low ambient temperatures through the one month experiment were 81 ± 6.4 ºF and 64.5 ± 4.8 ºF, respectively. The average relative humidity through the MWW_NF run was 64.85 ± 14.51 %. There were three periods of time, Day 10 to 12, Day 18-21, and Day 24, which had daily temperature lower than 80 ºF. Moderate to severe precipitation was observed on Day 1, Day 23, and from Day 25-28.

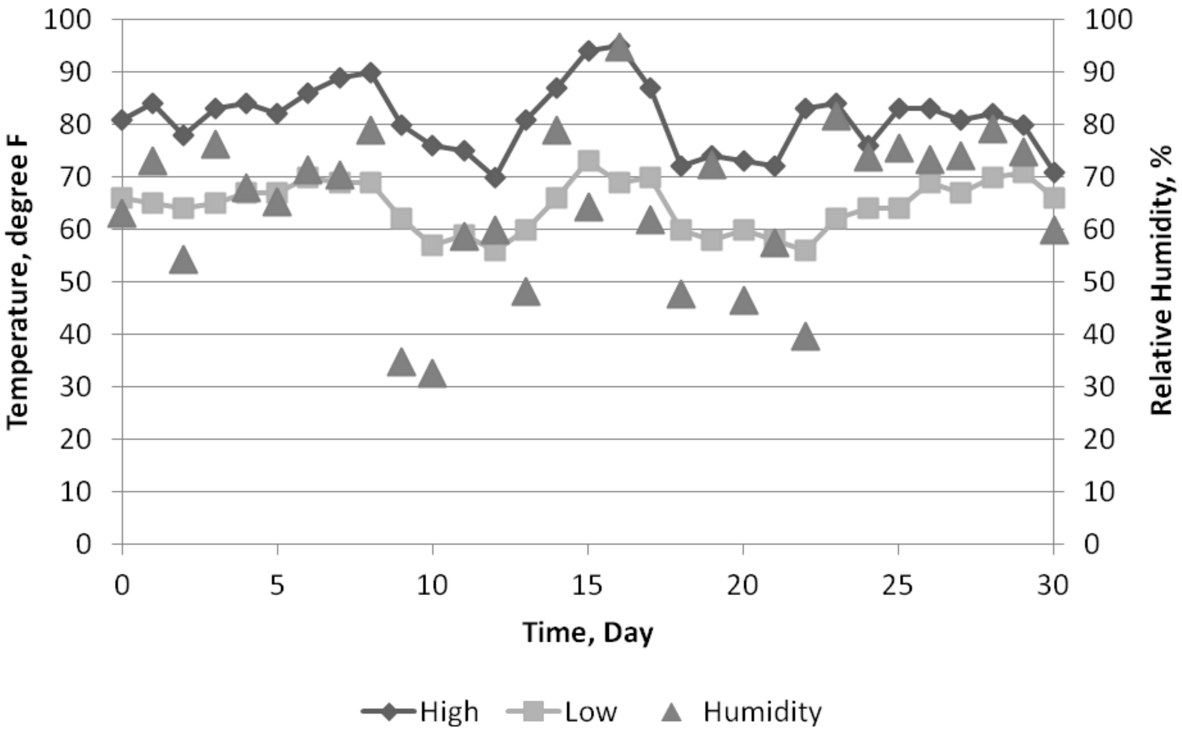


Figure A.18. Daily ambient temperature and relative humidity at FTMSA during the pilot-scale testing, Summer 2011

The temperature differential between water entering the cooling section and the water in the bottom basin (cooling capacity) is mainly affected by the air flowrate through the cooling column and the temperature in heat exchanger. Therefore, adjustment of the system is required in order to compensate the influence caused by the ambient condition. Fine adjustment of input air flowrate and heating output were performed continuously to ensure desired water temperatures at different locations in the system.

The air flowrate within the cooling column is measured by using anemometer at 3 different measuring points. Point X is near the center of column close to the nozzle; point Y is close to the wall; point Z is at the middle between points X and Y. Previous study (Vidic et. al., 2009) revealed that changing the fan speed only shifts the temperature differential instead of increasing the cooling capacity. Therefore, different fan speed settings were applied to all three towers to best meet desired temperature profile.

The temperature differentials and air flowrate in the cooling column in the cooling systems were shown in Figure A.19 and Table A.17. Results depicted in Figure A.19 indicate that the temperature differentials in CTB3 and CTC3 were adjusted and maintained higher than the design criteria in seven days after beginning the experiment. However, severe deposit was observed on the digital thermometer surface, which led to a lower temperature differential average in CTA3. Table A.16 summarizes the average airflow rate and temperature differentials measured in the cooling systems. Fine adjustments of the fan speed and heater setting were performed to achieve the air flowrate above 215 CFM and the temperature difference across all three cooling towers was maintained above 10 ºF.

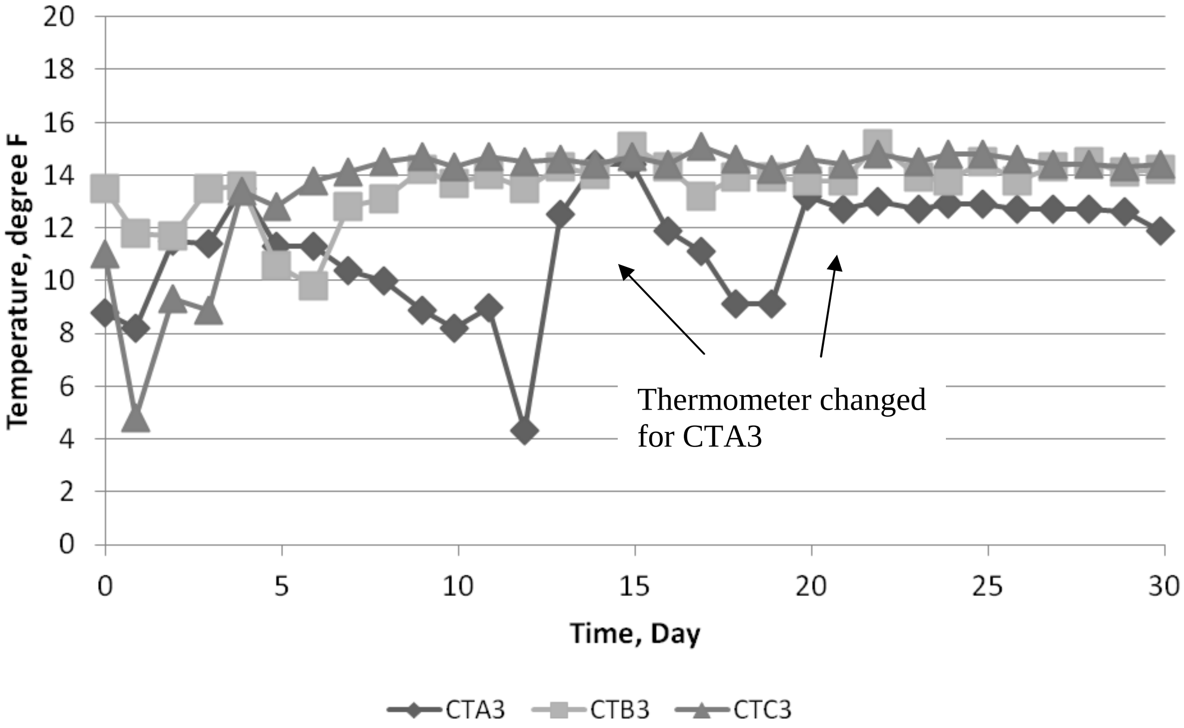


Figure A.19. Temperature difference across the cooling towers CTA3, CTB3, and CTC3 during the pilot-scale testing, Summer 2011. Thermometer in CTA3 was changed on Day 13 and 20 due to severe fouling on the probe surface.

Table A.17. Average air flowrate and average temperature differential in pilot-scale cooling testing, Summer 2011

Cooling Tower	CTA3	CTB3	CTC3
Air flowrate, CFM	260 ± 48	215 ± 55	224 ± 51
Temperature differential, ºF	11.3 ± 2.2	13.6 ± 1.2	13.6 ± 2.2

Besides the 10 ºF temperature differential, maintaining the temperature of water at spray nozzle at 105 F was also requirement to simulate the real situation in the power generating facilities. Table A.18 summarizes the average water temperature at nozzle for all towers. Both CTA3 and CTC3 had water temperature at nozzle lower than 105 ºF, while CTB3 maintained water temperature above 105 ºF throughout the testing period. The higher air flowrate in cooling column may also cause the lower water temperature observed in CTA3. All three cooling towers had similar deviations due to the low ambient temperature between Day 10 to 12 and between Day 18 to 21.

Table A.18. Average temperature of water at nozzle in the pilot-scale cooling tests, Summer 2011

Cooing System	CTA3	CTB3	CTC3
Temperature of water at nozzle (ºF)	101.2 ± 3.1	103.8 ± 3.1	107.4 ± 3.4

A.3.3 Total makeup water volume and blowdown volume

The average daily water consumption rate is influenced by ambient temperature, relative humidity, temperature of water in the recirculating system, and air flowrate through the cooling column. Variations in these parameters will inevitably cause variations in daily makeup water consumption. The blowdown rate is controlled by an inline conductivity meter. When the conductivity of recirculating water exceeds the set point of target criteria, the conductivity meter will send the signal to open the solenoid valve and initiate blowdown. The makeup water volume and blowdown volume were recorded by inline totalizers and the total water consumption volume is the sum of water volume recorded by makeup water totalizer and biocide addition. The evaporation volume can be calculated by subtracting the blowdown volume from the total water consumption volume. Blowdown was also measured manually in case that inline totalizer failed. Both water consumption and blowdown rates were calculated by dividing the volume with the time interval between today and the last arrival time.

Data collected during the first three weeks of operation are summarized in Table A.19. It was observed that the both MWW_NF and MWW_NFG waters had average specific conductivity of 0.85 ± 0.07 mS/cm. Therefore, the target conductivity of the recirculating water at 4~5 CoC was 3.40 ~ 4.25 mS/cm. The amount of biocide added to the system was listed in the table as well. The average makeup water consumption in this run was 53.9 gallons/day, while average daily blowdown and evaporation rate were 8.5 and 45.4 gallons/day, respectively.

Table A.19 Average daily water consumption rate, blowdown rate, and evaporation rate for all three towers during the pilot-scale testing, Summer 2011

Initial phase of the field test	CTA3	CTB3	CTC3
Daily Water Consuming rate (gallon/day)	55.3 ± 3.5	52.2 ± 3.3	54.1 ± 7.3
Daily Blowdown rate (gallon/day)	8.5 ± 1.7	8.0 ± 1.8	9.0 ± 2.5
Daily Evaporation rate (gallon/day)	46.8 ± 3.0	44.4 ± 2.5	45.1 ± 6.1
Daily Biocide Addition rate (gallon/day)	1.5 ± 0.4	7.2 ± 1.0	2.5 ± 0.7

Figure A.20 shows the detailed daily water consumption and blowdown of each cooling tower in this run. No mechanical failures and significant variations of blowdown were observed in CTA3 and CTB3 in this run. As for CTC3, fouled conductivity electrode and water leakage from the cover screen around the cooling column resulted in higher water consumption on Day 12 and 13 and thus causing much less blowdown in the following day.

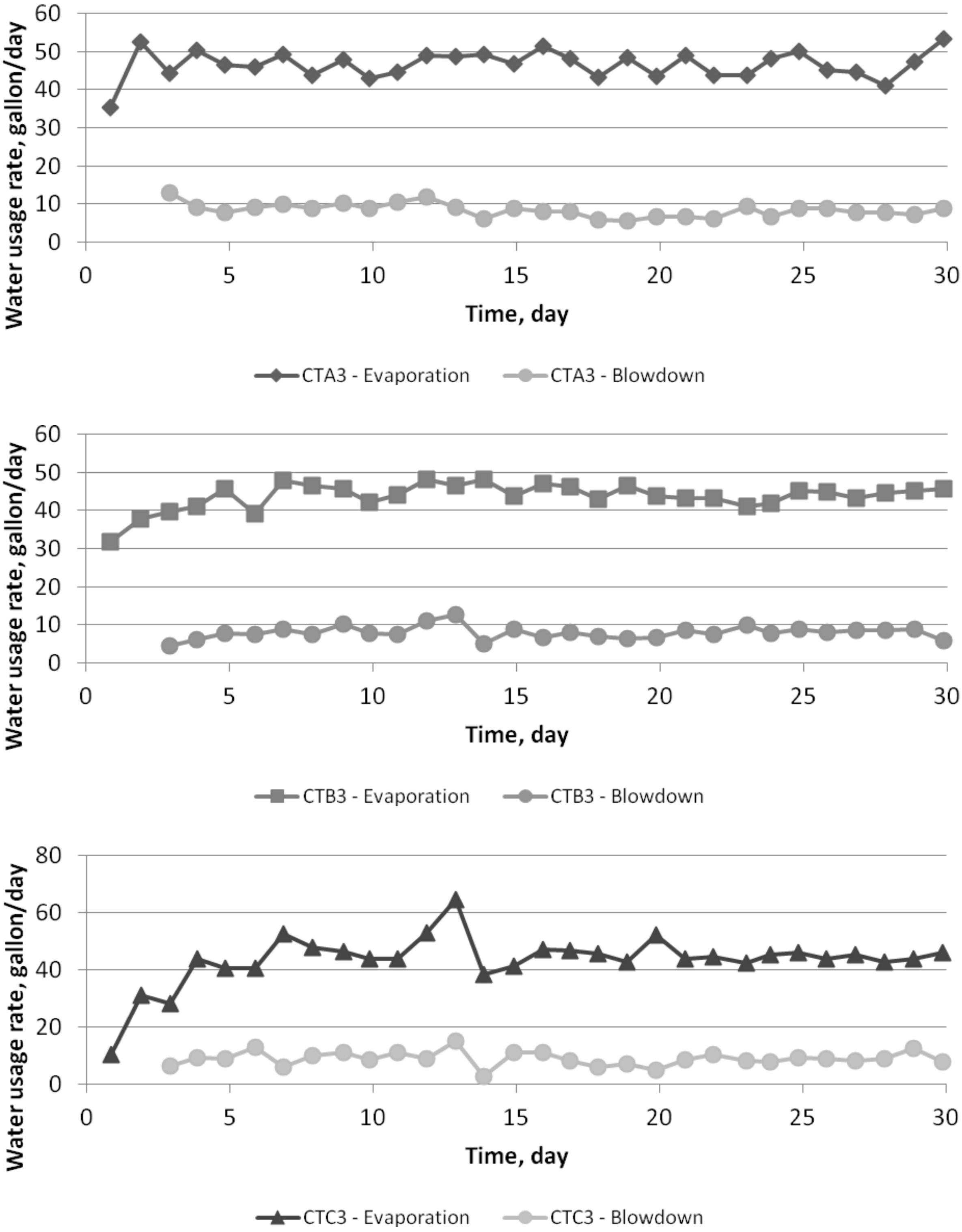


Figure A.20. Daily makeup water and blowdown volume throughout the pilot-scale testing, Summer 2011. The amounts of makeup water usage and blowdown discharge in CTA3, CTB3, and CTC3 are shown from top to bottom, respectively.

A.3.4 Cycles of concentration in the recirculating systems

a) Based on Conductivity

To ensure that the cycles of concentrations in each cooling tower was maintained within a desired range, conductivity of blowdown was measured with a portable conductivity meter (Oakton CON 11 Meter, Fisher Scientific, USA). Conductivity of makeup water and recirculating water are shown in Figure A.21. Both MWW_NF and MWW_NFG had stable conductivities slightly below 0.85 mS/cm throughout the experiment. It was observed that all three cooling towers reached 4 CoC within 72 hours. The average conductivity of recirculating water in CTA3, CTB3, and CTC3 were 4.01, 3.79, and 4.21 mS/cm, respectively. Figure A.22 shows the conductivity of blowdown from cooling towers CTA3, CTB3, and CTC3 in the pilot-scale experiment.

Based on the water evaporation rate shown in Table A.18, the order of conductivities in the recirculating water from high to low should be CTA3, CTB3, and CTC3. However, CTC3 was observed to have the highest conductivity among the three cooling systems due to the larger variation in water consumption rate and comparatively more biocide addition than CTA3. Conductivity of recirculating water in CTB3 was much lower than these in other two systems due to the addition of highly diluted biocide. In general, the conductivities of recirculating were maintained within our designated criteria, 3.5-4.2 mS/cm most time. Minor differences among the three systems were observed and reflected on the cycles of concentration described in next subsection.

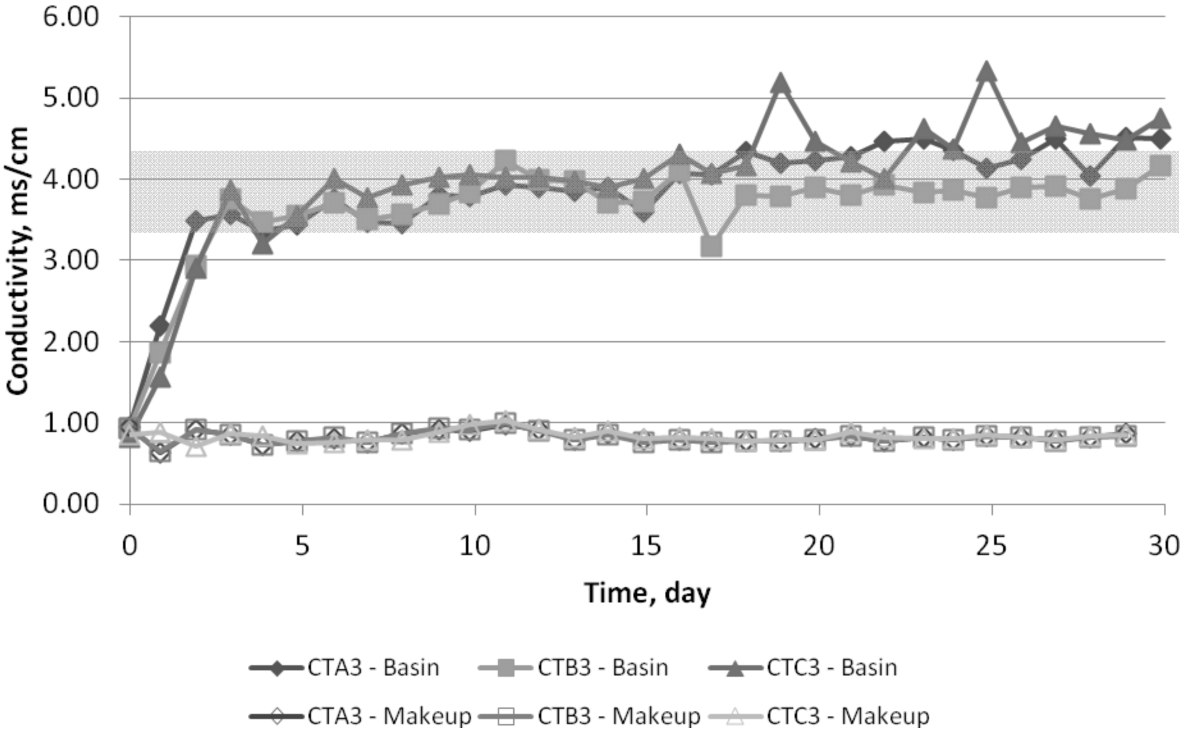


Figure A.21 Conductivity of cooling tower makeup and basin waters in pilot-scale testing, Summer 2011. Shaded area indicates the designate conductivity range for CoC control between 4 and 5.

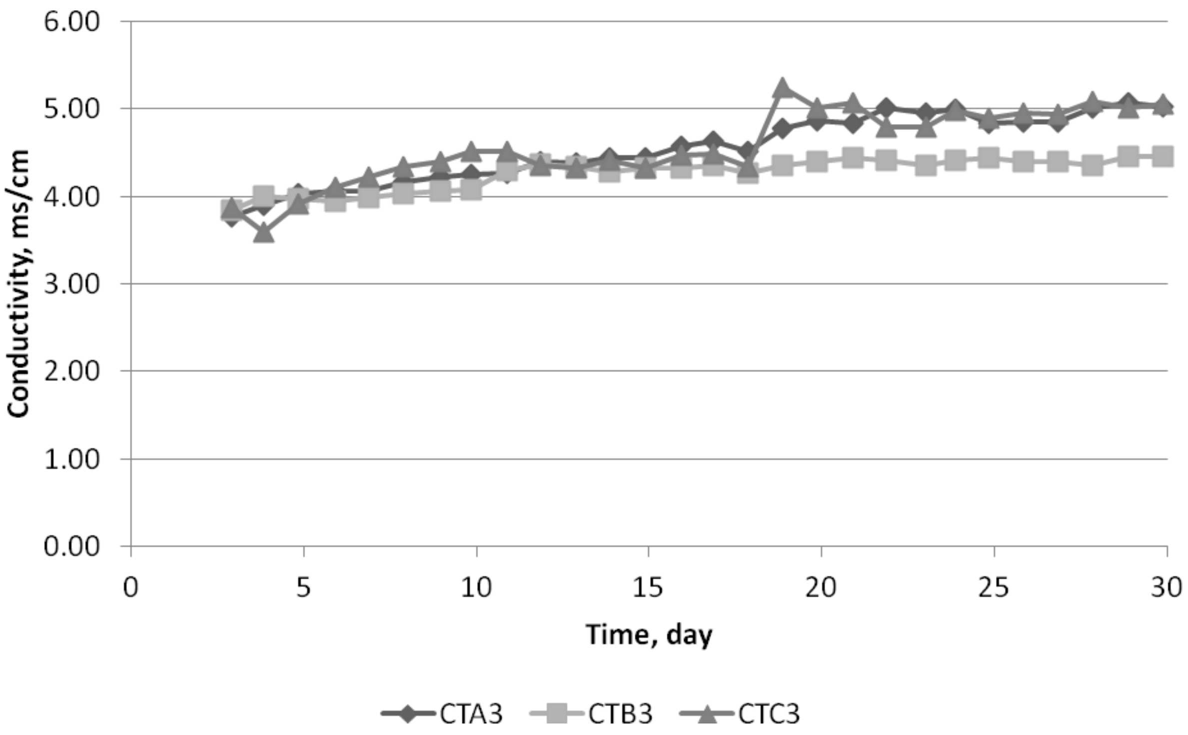


Figure A.22. Specific conductivity of blowdown from cooling towers CTA3, CTB3, and CTC3 during the pilot-scale testing, Summer 2011. Average conductivities were 4.55, 4.27, and 4.57 cm/mS, respectively.

Table A.20 summarizes the conductivity data measured throughout the one month testing and the cycles of concentration of recirculating waters. Conductivity of the makeup water, water in the basin, and blowdown were recorded daily and are shown as 60-day average values. The last two columns show the calculated CoC based on different methods. CoC(1) is calculated by dividing conductivity of the water in the basin by the conductivity of the makeup water. CoC(2) is calculated by dividing conductivity of blowdown by the conductivity of makeup water. Because the conductivity of makeup water was below our designated criteria, 0.85 mS/cm, the CoC(1) in all three cooling towers were pushed to the high end and close to 5. The reason that CoC(2) were higher is the sensitivity of online conductivity electrode. Turbulence in the basin can cause fluctuation of water quality, thus delaying the response time of the blowdown control system. In addition, the blowdown drainage was induced by gravity with a low flow velocity, while the recirculating water was still being evaporated (concentrated). Therefore, the conductivity of blowdown increased as a result of these effects.

Table A.20. Average conductivities and CoC calculations for the cooling towers during the pilot-scale testing, Summer 2011

Cooling System	Raw (mS/cm)	Basin (mS/cm)	Blowdown (mS/cm)	COC (1)* (60-day average)	COC(2)** (60-day average)
CTA3	0.83 ± 0.07	4.01 ± 0.36	4.55 ± 0.35	4.9	5.5
CTB3	0.82 ± 0.07	3.79 ± 0.22	4.27 ± 0.18	4.6	5.2
CTC3	0.83 ± 0.07	4.21 ± 0.45	4.57 ± 0.42	5.0	5.5

* The COC is calculated by dividing conductivity of basin by conductivity of makeup water.
** The COC is calculated by dividing conductivity of blowdown by that of makeup water.

b) Based on blowdown and makeup volume

Besides using conductivity of the recirculating water and of the blowdown to define CoC, the CoC can also be derived by dividing the total water consumption by the total daily blowdown volume (CoCBD). Average daily makeup water and blowdown volume and calculated CoC_{BD} of this run are shown in the last column in Table A.21. Based on this method, it is determined that all three cooling systems were operated above our target CoC 4~5 and reached 6.5 at most. Possible explanation can be an extremely low conductivity of recirculating water, which leads to significantly decrease in blowdown volume for the next day, thus leading to an extremely high CoCBD value. However, this phenomenon only occurred in CTC3 on Day 20 and 25 but did not occur in CTA3 and CTB3. This result may imply that either a great portion of dissolved solids were extracted from the recirculating or there were significant amount of leakage occurred during blowdown process.

Table A.21 CoC in cooling towers based on the volumetric analysis using data from the pilot-scale testing, Summer 2011

Cooling System	Daily Makeup Volume (Gal/day)	Daily Blowdown Volume (Gal/day)	CoC_{BD} (60-day average)
CTA3	55.3 ± 3.5	8.5 ± 1.7	6.7 ± 1.2
CTB3	52.2 ± 3.3	8.0 ± 1.8	6.7 ± 1.3
CTC3	54.1 ± 7.3	9.0 ± 2.6	6.5 ± 2.2

A.4 Summary

Field experiments with pilot-scale cooling systems were conducted at the Franklin Township Municipal Sanitary Authority wastewater treatment facility in 2010 and 2011. The first set of experiments for the 2010 field campaign was conducted with tertiary treated effluent, i.e., secondary effluent subjected to nitrification and filtration as additional treatment. The second set of experiments for the field campaign was initiated with secondary treated effluent in August and was finalized in October. The third set of tests initiated with MWW_NF and MWW_NFG in May 2011. Results of the three sets of experiments are described in this final report. It was demonstrated that performance of the three pilot-scale cooling systems was stable throughout the whole period. Desired operating conditions can be maintained by properly adjusting airflow rate and heating output in the system. The cooling systems were controlled to maintain desired CoC as calculated based on water conductivity measurements, volume of blowdown and makeup water, or chloride concentrations in makeup and system water. However, CoC calculated based on volume of blowdown and makeup water were not in accordance with the one based on conductivity measurements. It is assumed that a loss of dissolved solid in the recirculating system or leakage may occur during the test. In addition, results also suggest that pilot-scale cooling systems designed and built for this study are capable of long term operation without any major deviations from the designed operating parameters.

APPENDIX B

General Water Characteristics of Different Tertiary Treated Municipal Wastewaters Collected From Franklin Township Municipal Sanitary Authority, Murrysville, PA

B.1 Introduction

Three pilot-scale cooling towers were operated for 2 months period using secondary treated (MWW run) and tertiary (nitrified-filtered) treated (MWW_NF run) municipal wastewater as makeup water. Another pilot-scale cooling tower was operated for 1 month using granular activated carbon adsorbed tertiary treated municipal wastewater (MWW_NFG run). During this experiments water samples were collected from the makeup water tank and the cooling tower basins for general water quality analysis. The parameters measured include pH, conductivity, total solids (TS), total suspended solids (TSS), total dissolved solids (TDS), alkalinity, total organic carbon (TOC), chemical oxygen demand (COD), ammonia (NH_3) concentration, anions (chloride (Cl^-), Nitrate (NO_3^-), Sulfate (SO_4^{-2}), Phosphate (PO_4^{-3}) and cations (calcium (Ca), magnesium (Mg), Iron (Fe) and Copper (Cu). These general water characteristics were measured once in every week. Only pH was measured daily.

The first experiment using tertiary (nitrified-filtered) treated wastewater (MWW_NF run) as makeup water started on May 20, 2010 and finished on July 18, 2010; the second set of experiment where secondary treated wastewater was (MWW run) used as makeup water started on August 1, 2010 and finished on October 6, 2010. Pilot-scale cooling tower experiment using granular activated carbon adsorbed tertiary treated municipal wastewater (MWW_NFG run) as makeup water started on May 24, 2011 and finished on June 23, 2011.

Between first and second experiments conducted on 2010, the pilot scale cooling towers were cleaned with acid solution and disinfected with free chlorine. The heating sections, especially the copper coils, were treated separately with 10% hydrochloric acid to ensure complete removal of scaling/biofouling accumulated during the first test. The general water characteristics provided in Appendix-B corresponds to the pilot-scale cooling tower experiments reported in Chapter 3.

B.2 Comparison of Makeup Water Quality with Typical Wastewater Quality Data

The secondary and tertiary (nitrification-filtration) treated municipal wastewater used in the pilot scale experiments were collected from the Franklin Township Municipal and Sanitation

Authority (FTMSA) water treatment plant. Regular monitoring of water quality gave average concentration values for different parameters like TSS, TDS, COD, TOC, Ammonia, Nitrate etc. The average values of these parameters were close to the reference values given in literature. Table B-1 compares some water quality parameters of secondary and tertiary treated municipal wastewater from literature with the average values of the same parameters obtained during the pilot-scale experiments in the FTMSA facility.

B.3 General Water Characteristics Data

B.3.1 pH

Figure B.1 shows the pH profile in the MWW_NF run, MWW run, and MWW_NFG run respectively. The basin water pH values in the three cooling towers CTA, CTB and CTC during the MWW_NF run were 7.79±0.58, 7.75±0.59 and 7.81±0.55 respectively. pH of the CTA, CTB and CTC makeup waters were 6.70±0.63, 6.64±0.60 and 6.63±0.61 respectively. During the MWW run the basin water pH of three cooling towers CTA, CTB and CTC were 8.42±0.28, 7.46±0.49 and 7.71±0.51 respectively. In the MWW run the pH value of CTA makeup water was measured on a daily basis and the pH value of CTA makeup water was 7.16±0.26. From the pH value of CTB and CTC in MWW run it can be said that good pH control (near 7.7) was achieved in the pilot scale experiments. In the MWW_NFG run pH value in cooling tower CTC was 8.09±0.38, and the make up water pH value was 7.94±0.84.

B.3.2 Total solids (TS), total suspended solids (TSS) and total dissolved solids (TDS)

The total solids (TS), total suspended solids (TSS) and total dissolved solids (TDS) were measured on MWW_NF run, MWW run, and MWW_NFG run on a weekly basis. Figure B.2, B.3, and B.4 shows the TS, TSS and TDS profile in the MWW_NF run, MWW run, and MWW_NFG run respectively. Table B.2 summarizes the average TS, TSS and TDS values in the basin waters of the cooling towers and in the makeup waters.

B.3.3 Total organic carbon (TOC)

Figure B.5 shows the Total Organic Carbon (TOC) concentration profile in the basin waters of cooling tower CTA, CTB, CTC and makeup water of CTA during the MWW_NF run and MWW run. Also TOC concentration profile in the basin water of cooling tower CTC and corresponding makeup water is given in Figure B.5. Table B.3 summarizes the average TOC values during these runs.

B.3.4 Alkalinity

The alkalinity of the three cooling tower basin waters were measured at regular intervals during the MWW_NF run, MWW run, and MWW_NFG run. In the MWW_NF run alkalinity of the makeup water from all three towers were measured, where as in the MWW run alkalinity of the CTA makeup water was measured only. Figure B-6 shows the alkalinity profile and Table B.4 summarizes the average value of alkalinity in the system.

B.3.5 Chemical oxygen demand (COD)

The chemical oxygen demand (COD) profile in the cooling water systems during MWW_NF run, MWW run, and MWW_NFG run is shown in Figure B.7. Table B.5 summarizes the average values of COD.

B.3.6 Ammonia concentration

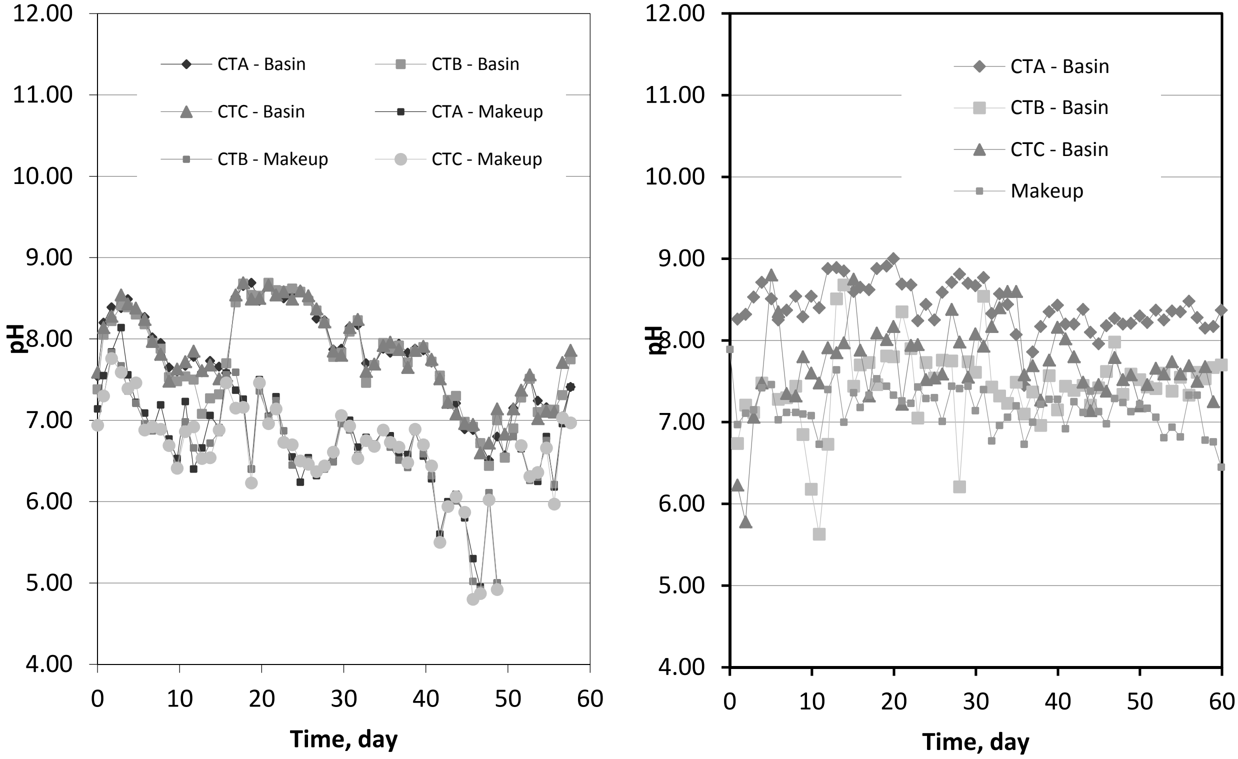
Ammonia concentrations measured in the basin waters of cooling towers and in the makeup waters are shown in Figure B.8. Table B.6 summarizes the average ammonia concentration in the cooling systems.

B.3.7 Anions concentrations

The basin waters from cooling towers and corresponding makeup waters were analyzed for anions like Chloride (Cl^-), Phosphate (PO_4^{3-}), Nitrate (NO_3^-) and Sulfate (SO_4^{2-}) once in every week. Table B.7 summarizes the average concentrations of these anions in the cooling systems for the MWW_NF run, MWW run, and MWW_NFG run. Figure B.9, B.10, and B.11 shows the anions concentration profile during the pilot-scale experiments.

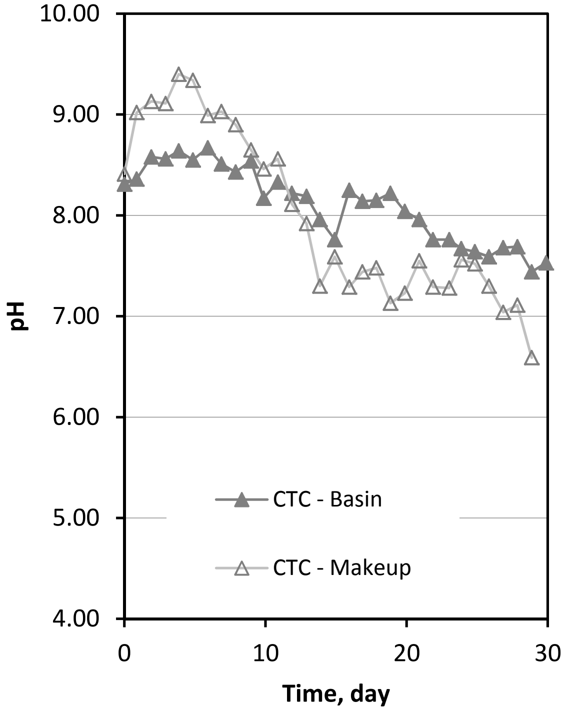
B.3.8 Cations concentrations

The basin waters from cooling towers and corresponding makeup water were analyzed for cations like Calcium (Ca), Magnesium (Mg), Iron (Fe) and Copper (Cu). Total cation and filterable cation concentrations were determined. Table B.8 summarizes the average concentrations of these cations in the cooling systems for MWW_NF run, MWW run, and MWW_NFG run. Figure B.12, B.13, B.14 and B.15 shows the cations concentration profile during the MWW_NF run. Figure B.16, B.17, B.18 and B.19 shows the cations concentration profile during the MWW run. Filterable calcium, and filterable magnesium concentration profile during the MWW_NFG run is given in Figure B.20.



(a)

(b)



(c)

Figure B.1. pH profile in the cooling systems during the (a) MWW_NF run, (b) MWW run, and (c) MWW_NFG run in FTMSA.

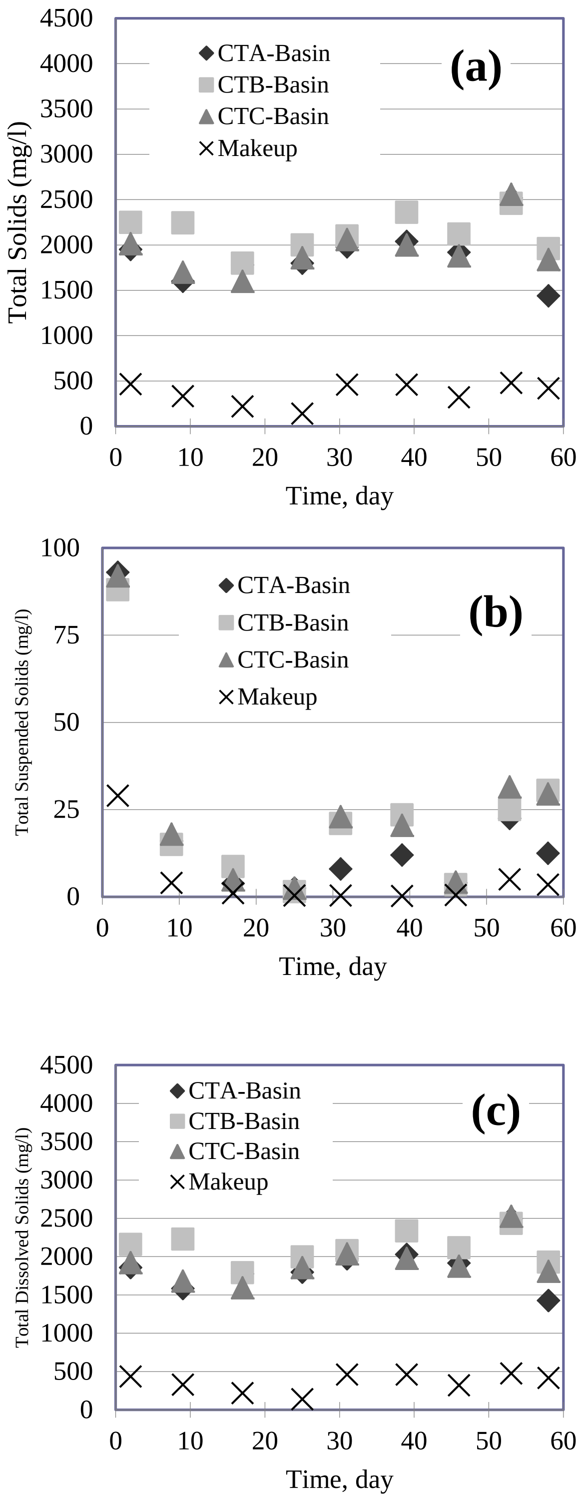


Figure B.2. (a) Total solids (TS), (b) Total suspended solids (TSS), and (c) total dissolved solids (TDS) profile in the three cooling systems during the MWW_NF run in FTMSA.

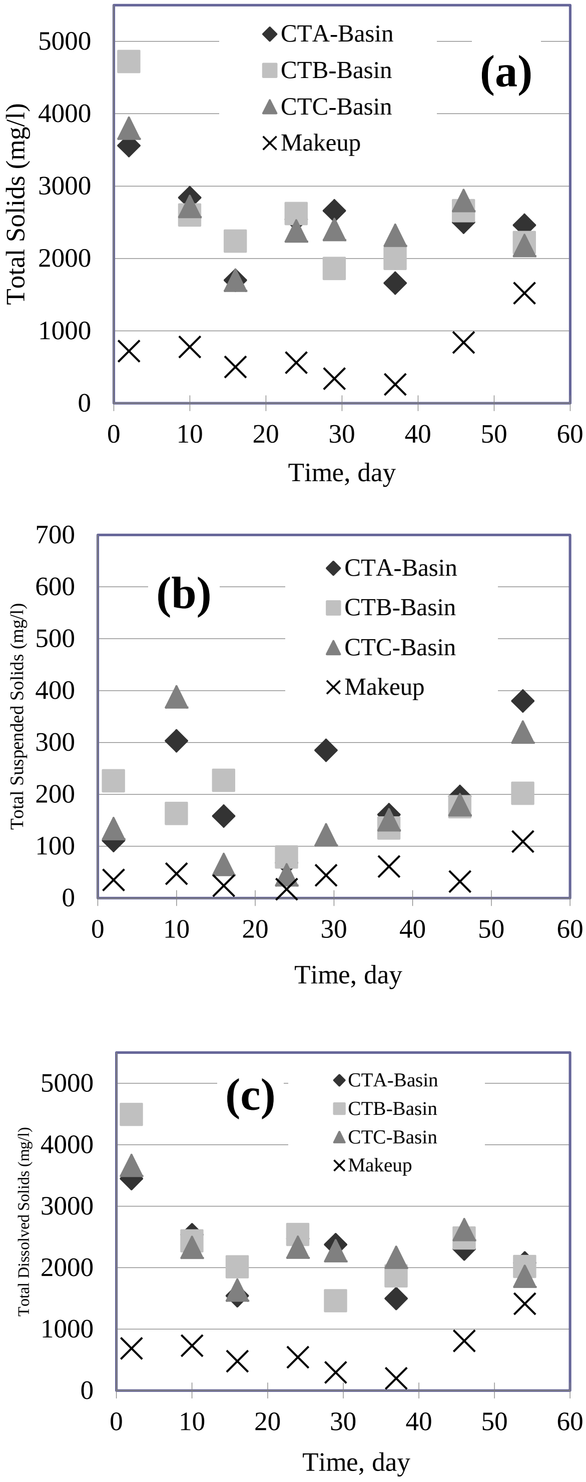


Figure B.3. (a) Total solids (TS), (b) Total suspended solids (TSS), and (c) total dissolved solids (TDS) profile in the three cooling systems during the MWW run in FTMSA.

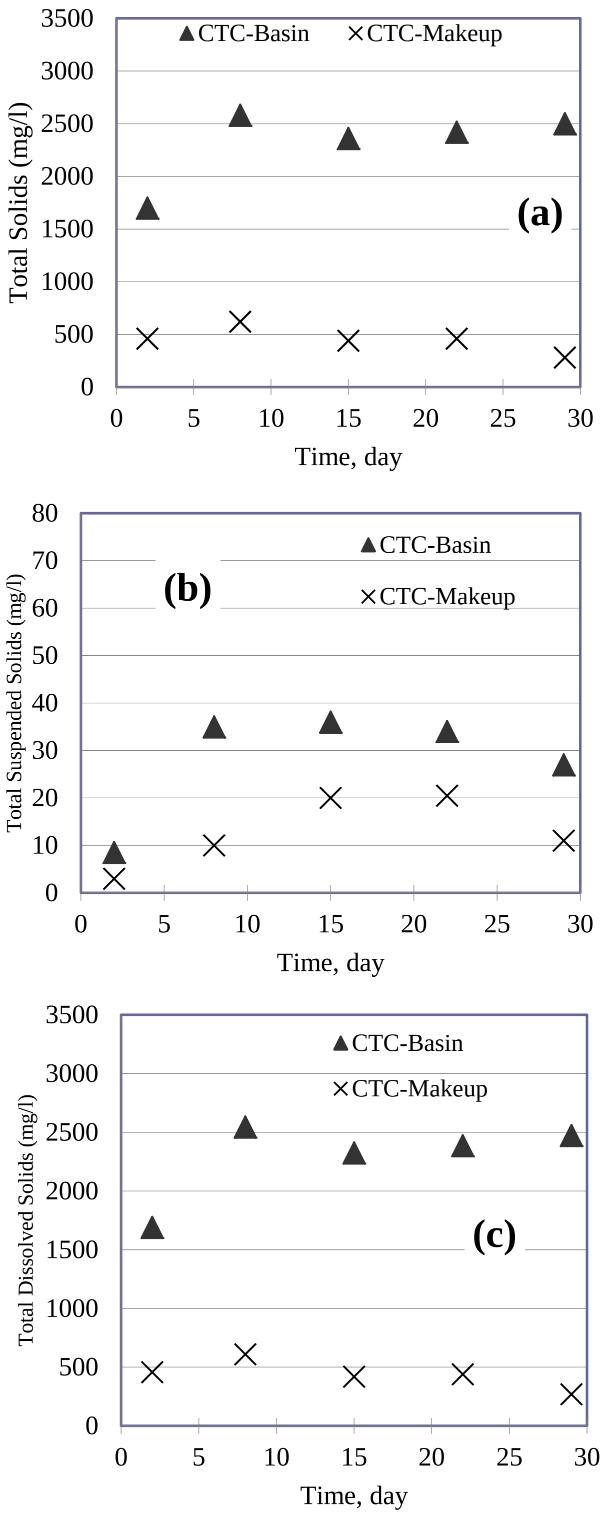
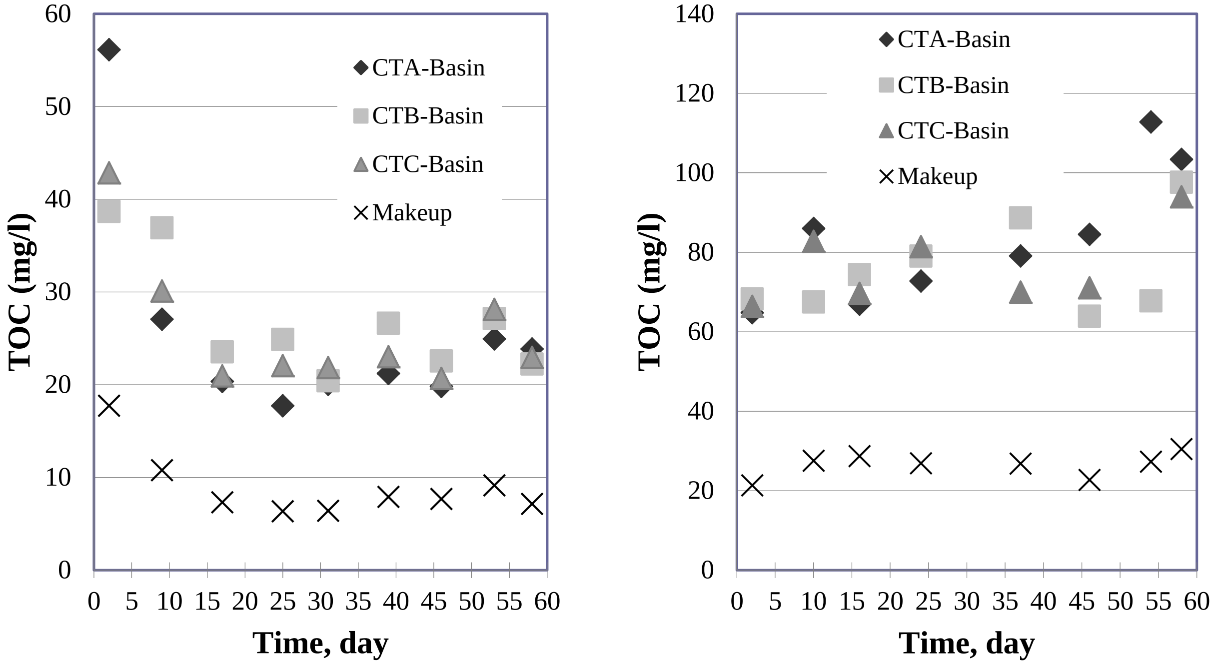
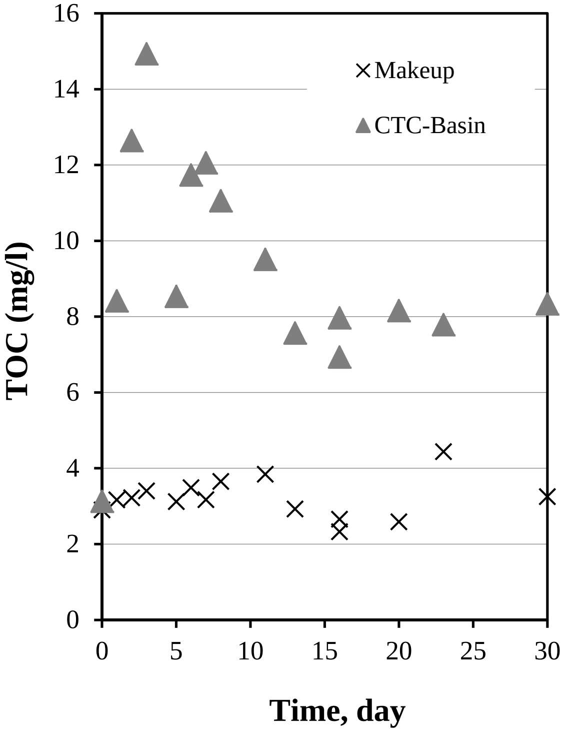


Figure B.4. (a) Total solids (TS), (b) Total suspended solids (TSS), and (c) total dissolved solids (TDS) profile in the cooling system during the MWW_NFG run in FTMSA.



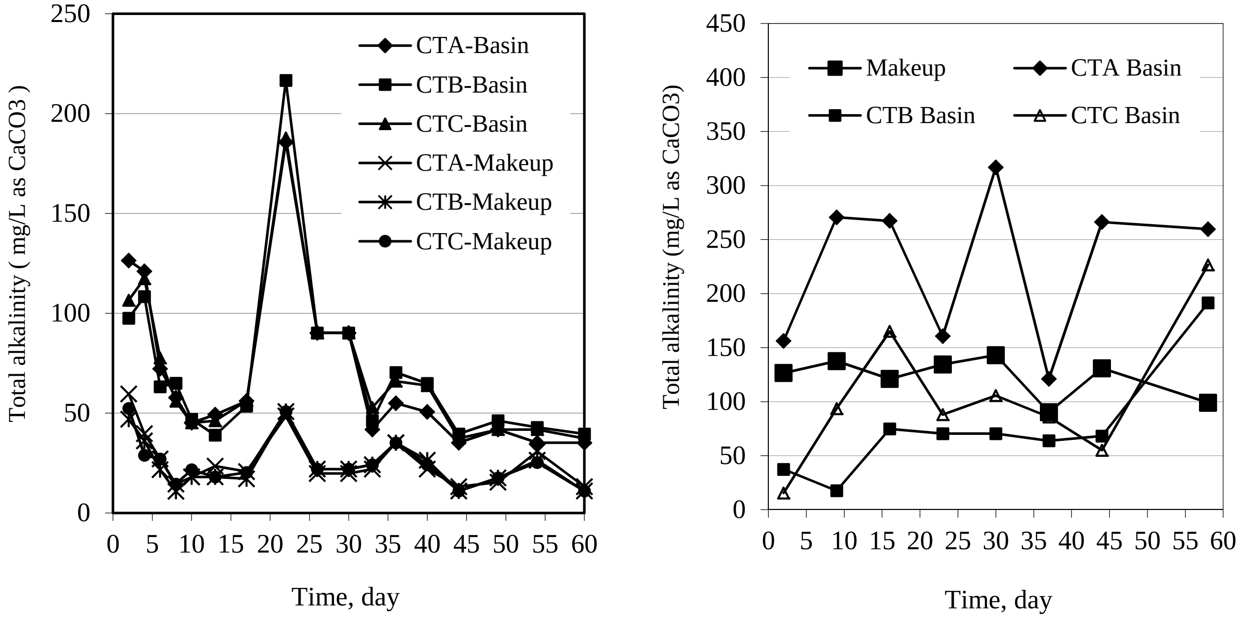
(a)

(b)



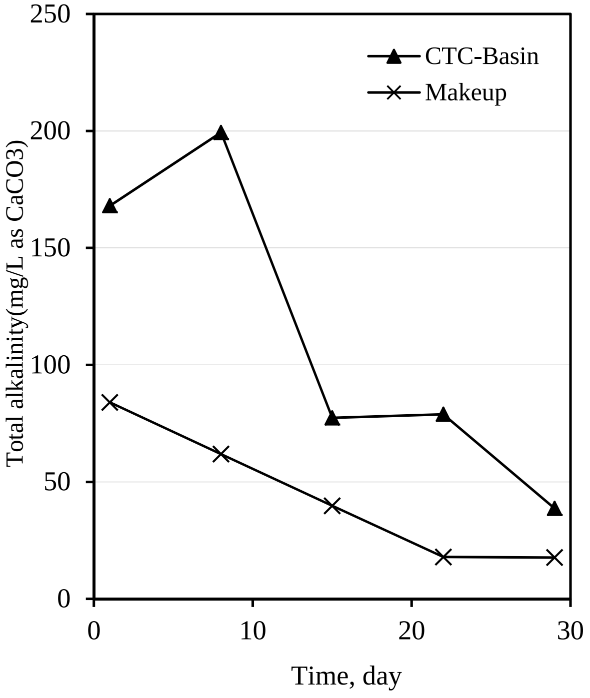
(c)

Figure B.5. Total Organic Carbon (TOC) profile in the cooling systems during the (a) MWW_NF run, (b) MWW run, and (c) MWW_NFG run in FTMSA.



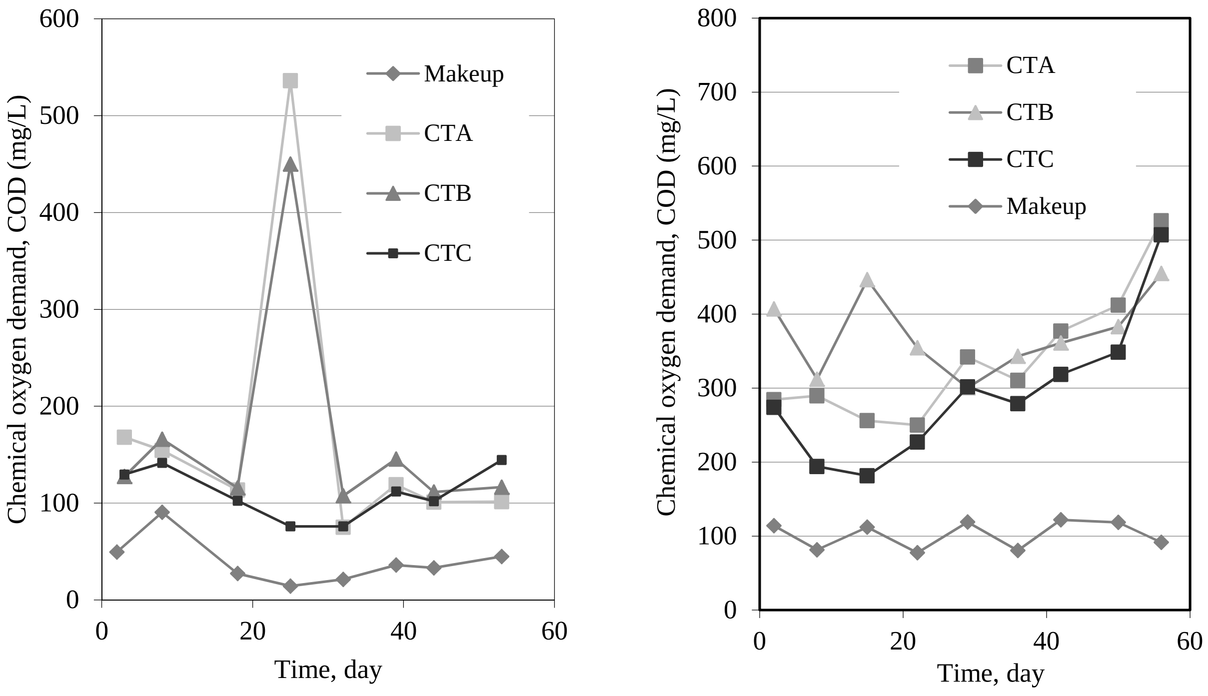
(a)

(b)



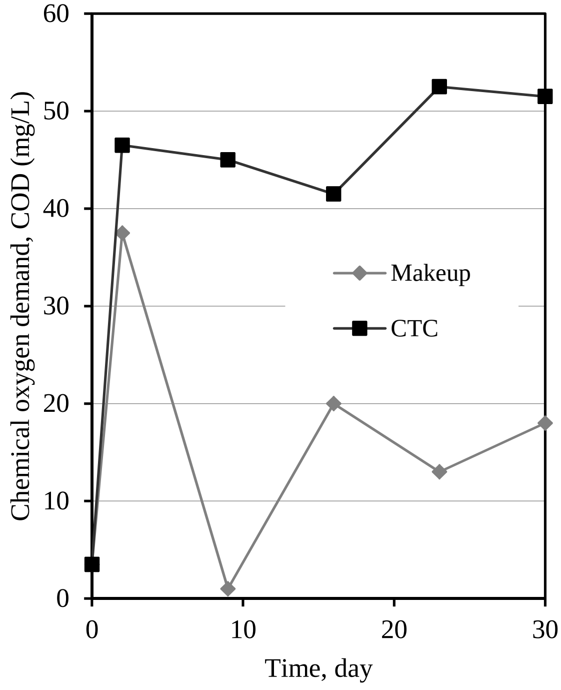
(c)

Figure B.6. Alkalinity profile in the cooling systems during the (a) MWW_NF run, (b) MWW run, and (c) MWW_NFG run in FTMSA.



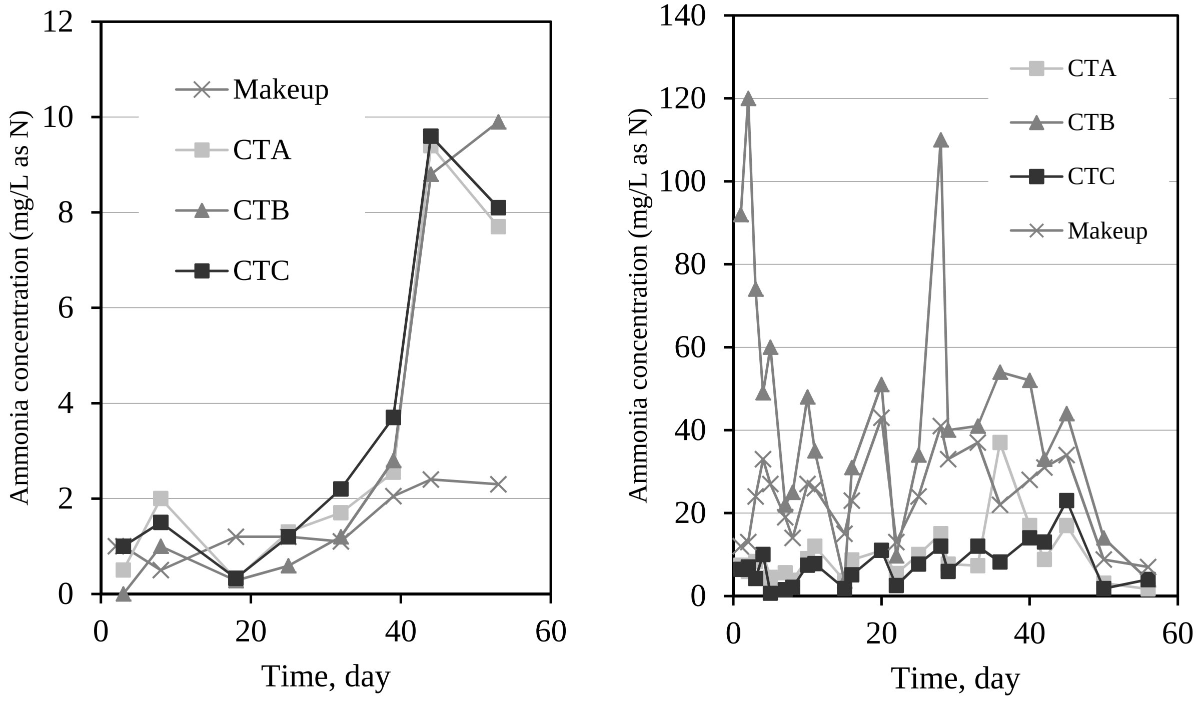
(a)

(b)



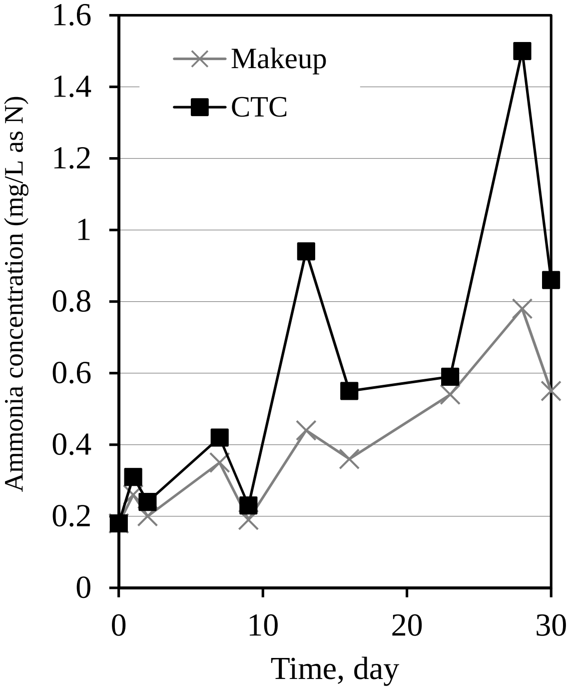
(c)

Figure B.7. Chemical Oxygen Demand (COD) profile in the cooling systems during the (a) MWW_NF run, (b) MWW run, and (c) MWW_NFG run in FTMSA.



(a)

(b)



(c)

Figure B.8. Ammonia concentration (mg/L as N) profile in the cooling systems during the (a) MWW_NF run, (b) MWW run, and (c) MWW_NFG run in FTMSA.

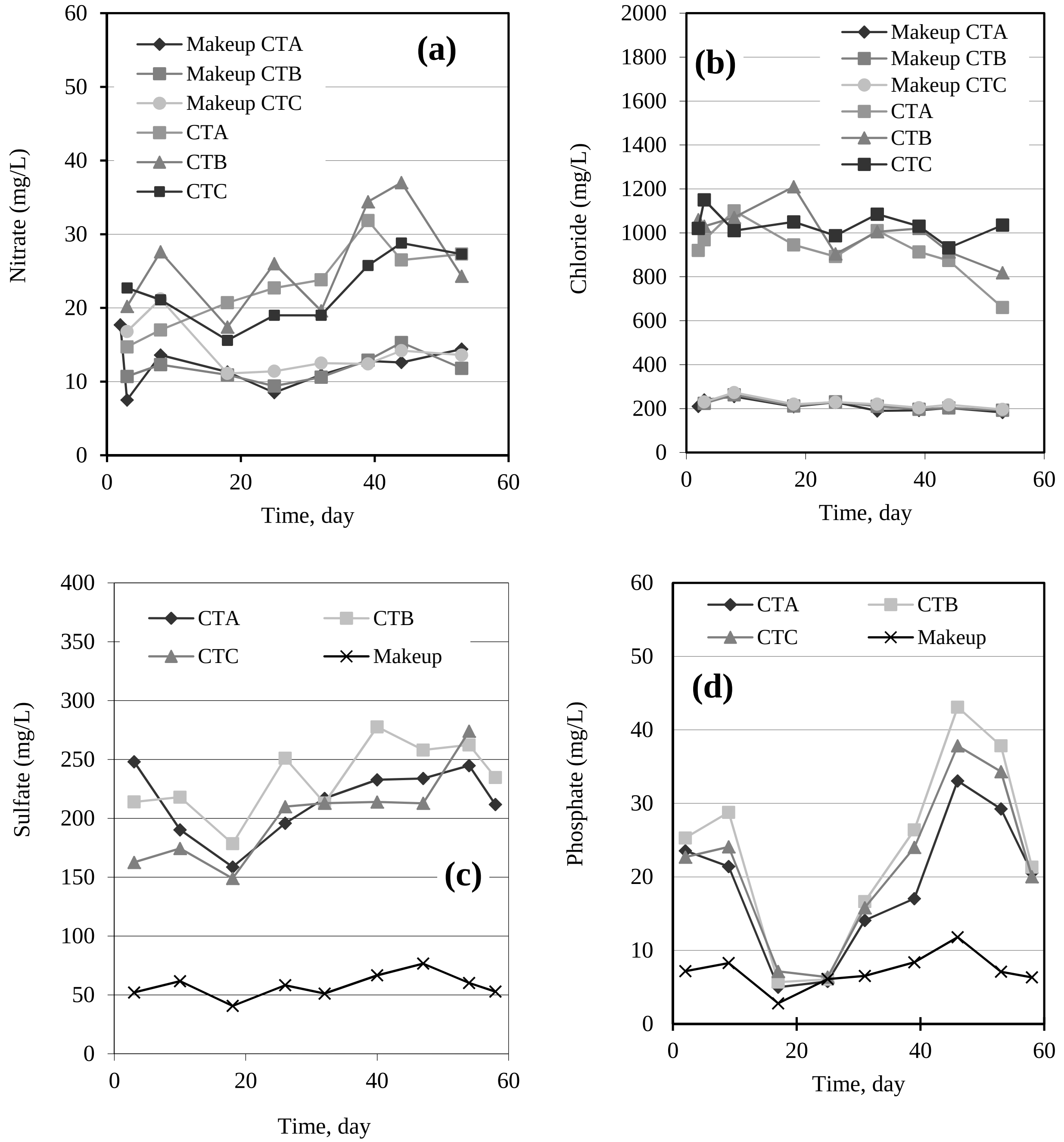


Figure B.9. (a) Nitrate (mg/L), (b) Chloride (mg/L), (c) Sulfate (mg/L) and (d) Phosphate (mg/L) concentration profile in the three cooling systems during the MWW_NF run in FTMSA

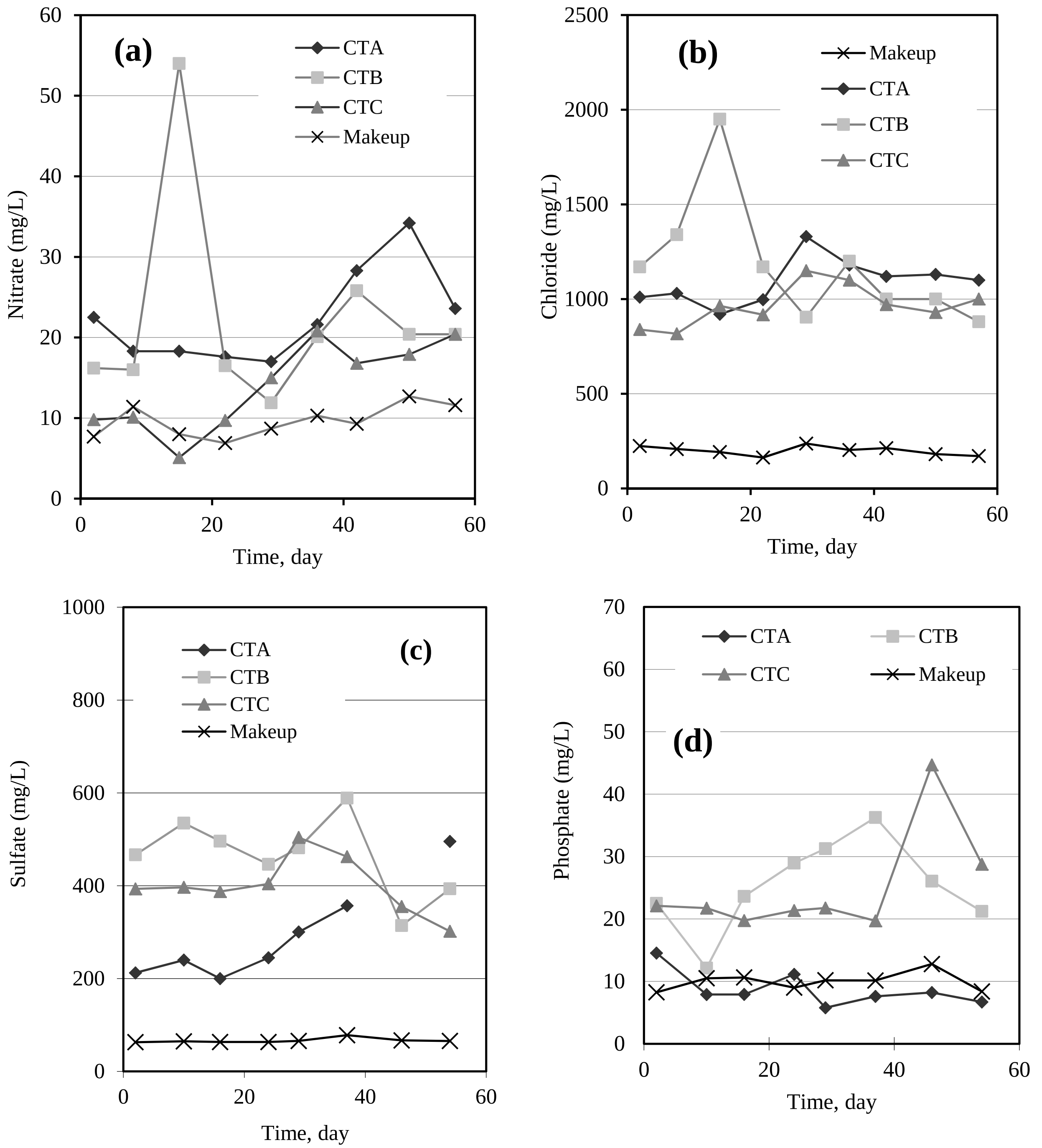


Figure B.10. (a) Nitrate (mg/L), (b) Chloride (mg/L), (c) Sulfate (mg/L) and (d) Phosphate (mg/L) concentration profile in the three cooling systems during the MWW run in FTMSA.

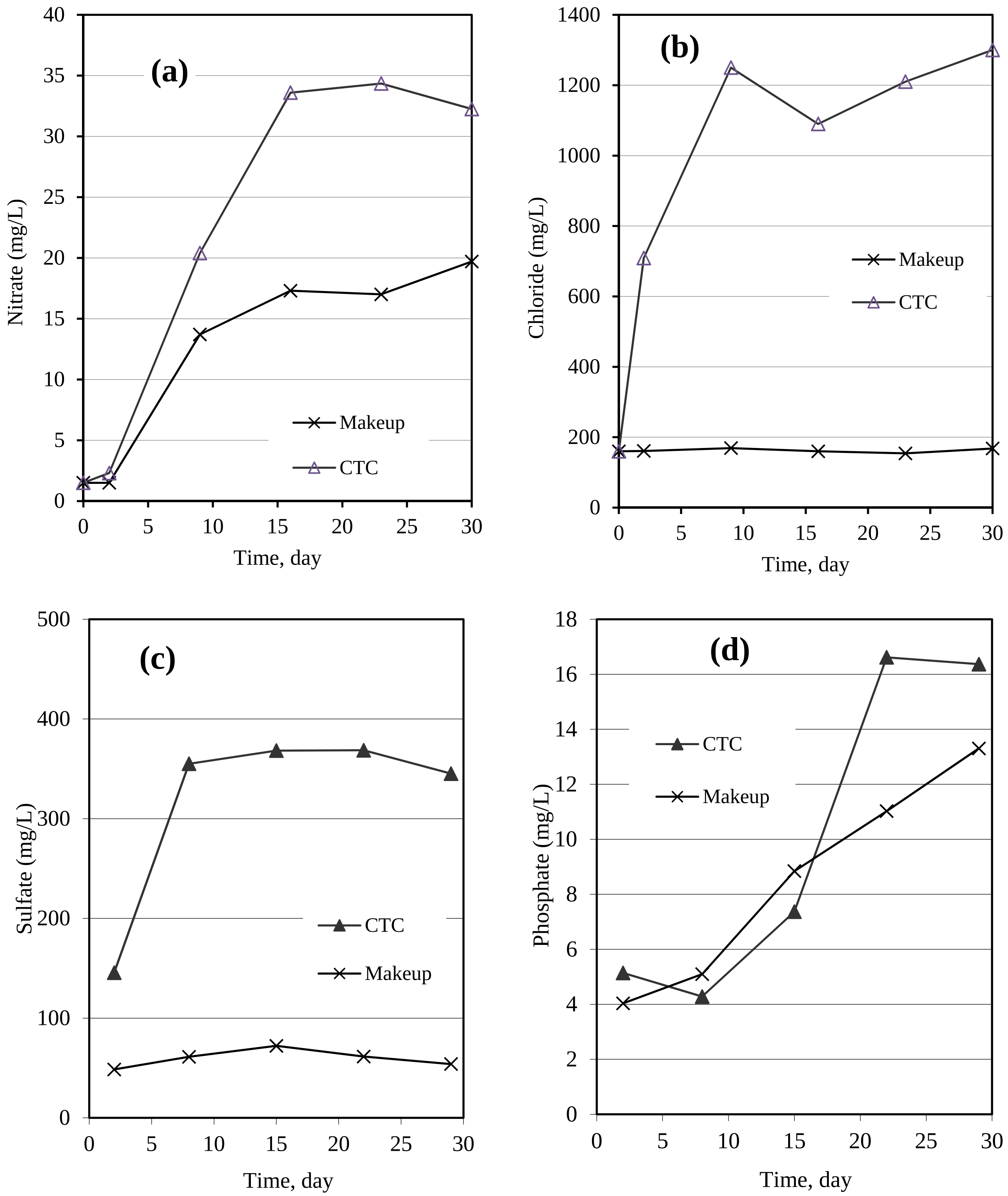
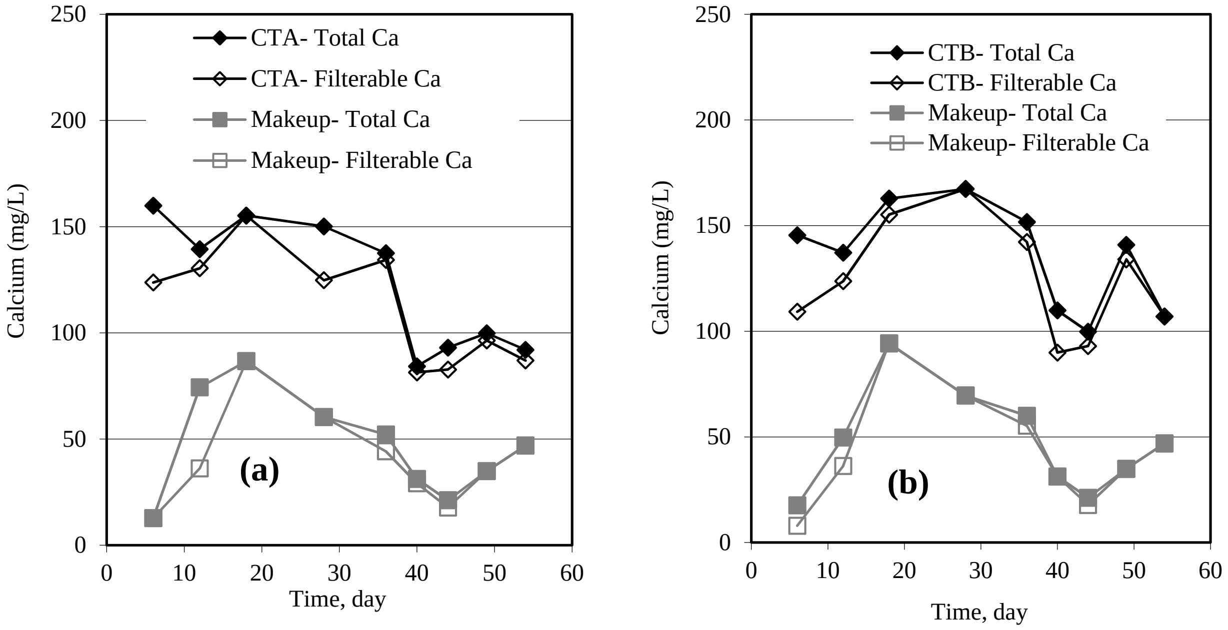


Figure B.11. (a) Nitrate (mg/L), (b) Chloride (mg/L), (c) Sulfate (mg/L) and (d) Phosphate (mg/L) concentration profile in the cooling system during the MWW_NFG run in FTMSA.



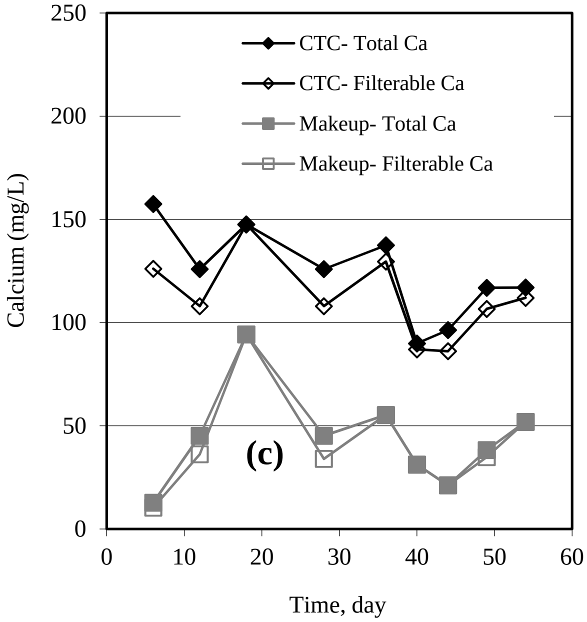


Figure B.12. Calcium (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW_NF run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

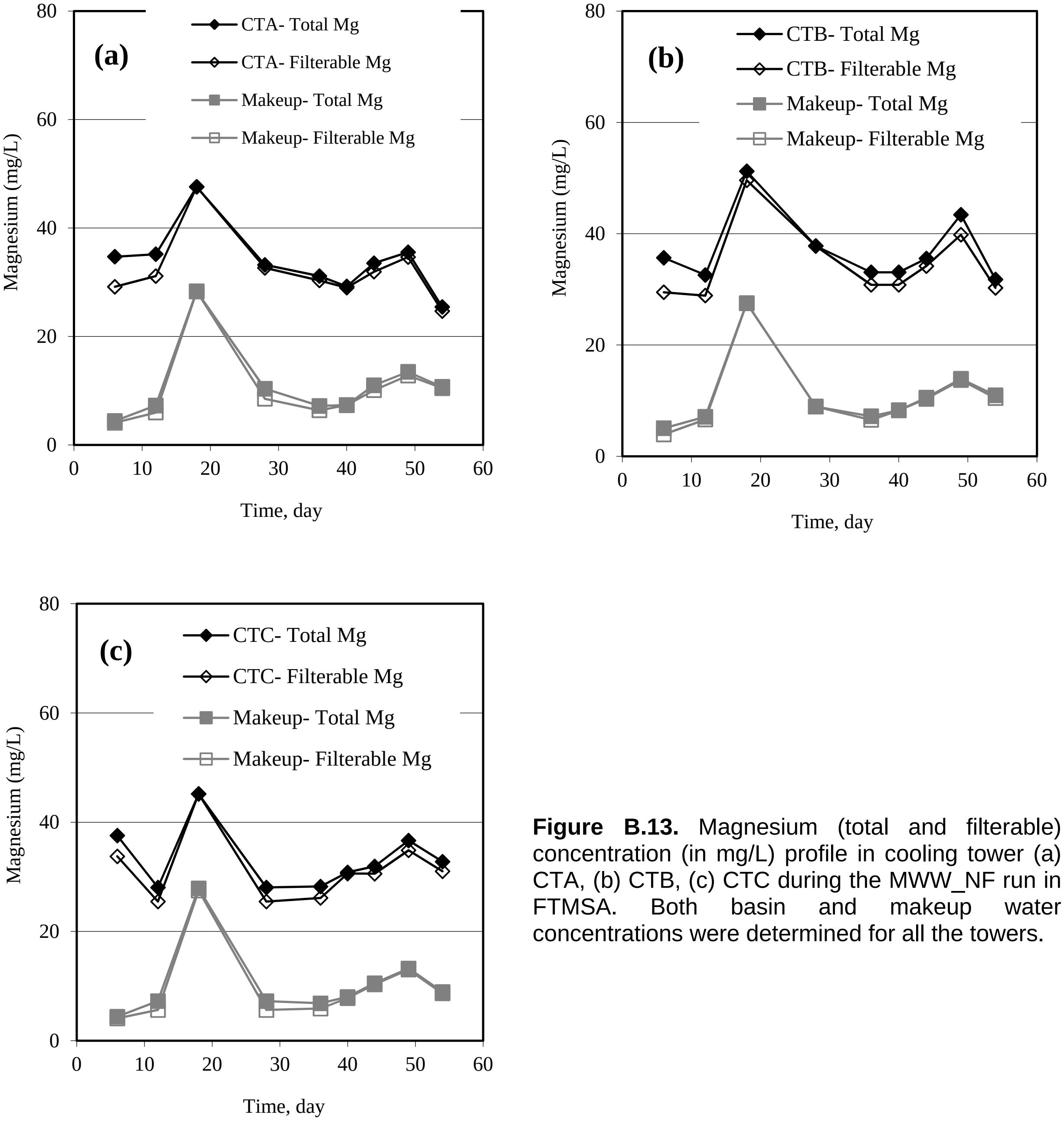


Figure B.13. Magnesium (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW_NF run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

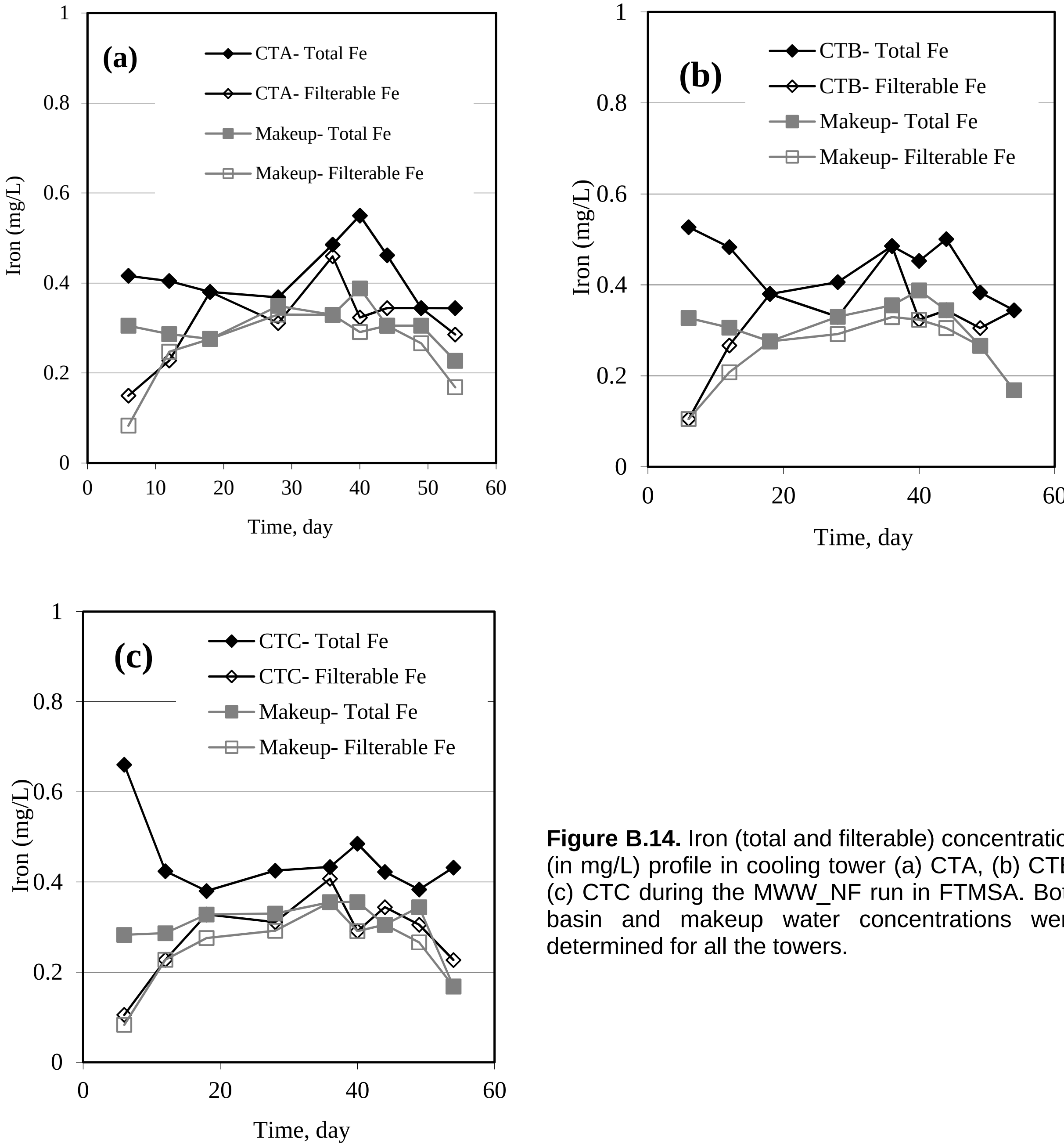


Figure B.14. Iron (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW_NF run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

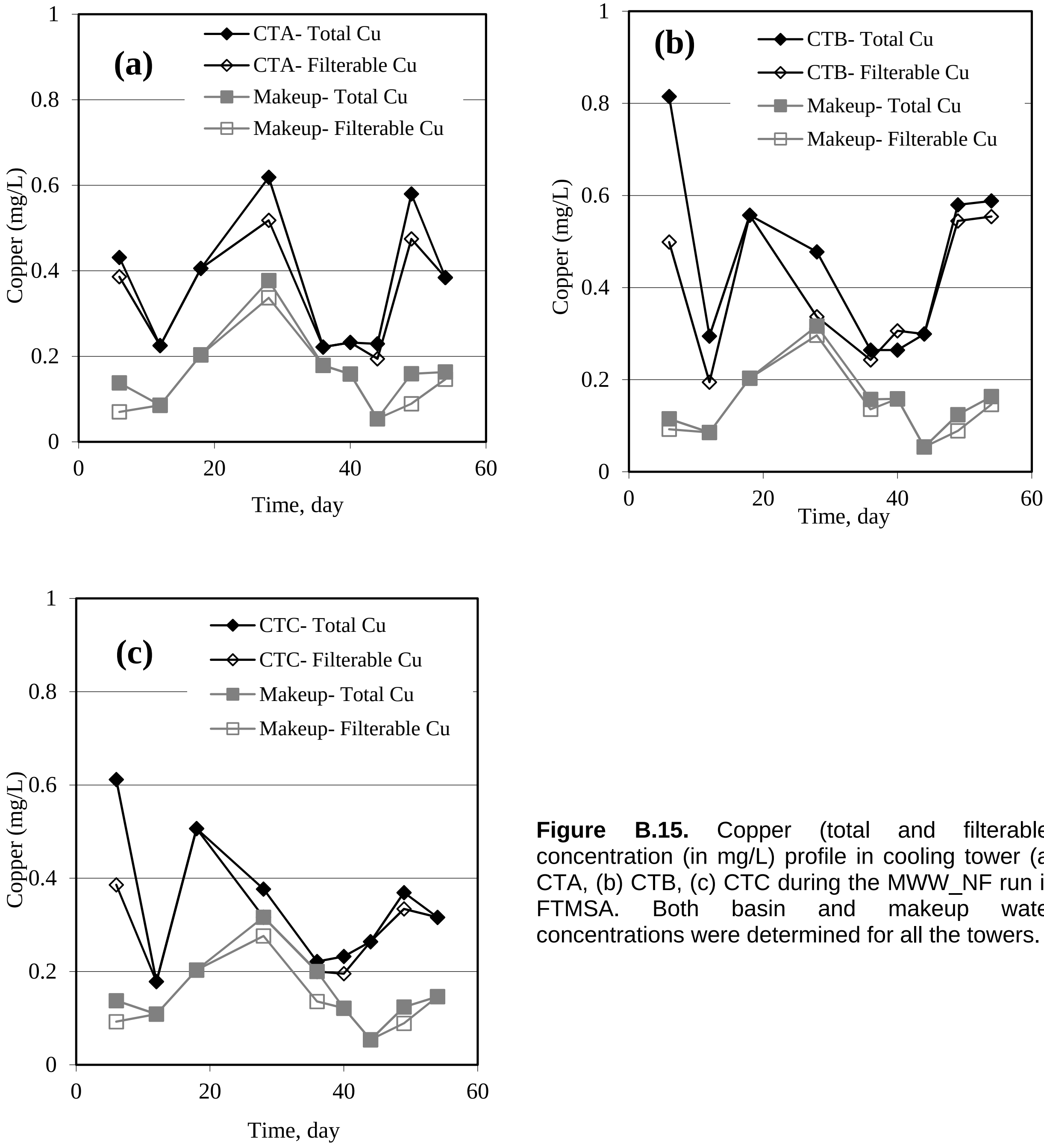
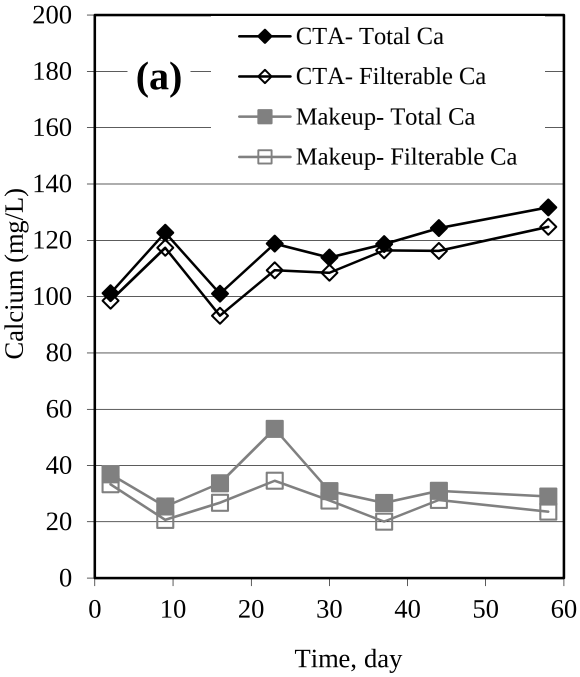
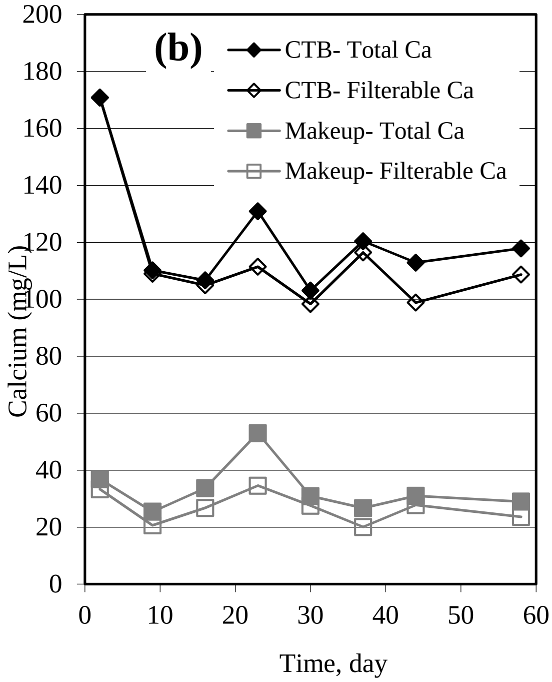


Figure B.15. Copper (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW_NF run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.





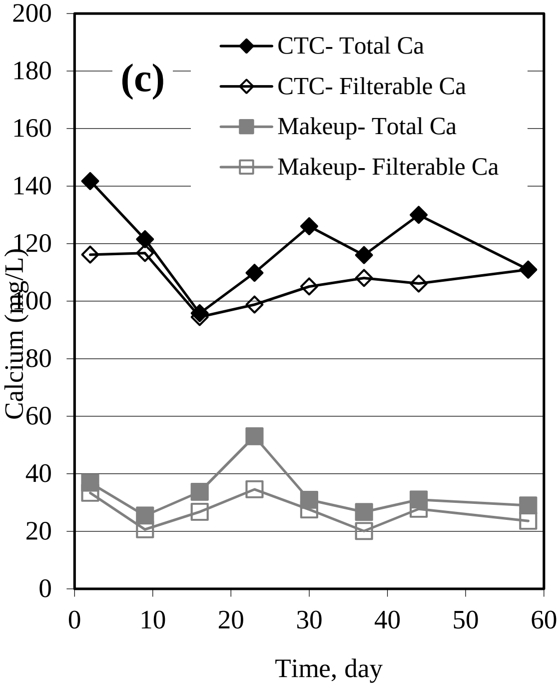


Figure B.16. Calcium (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

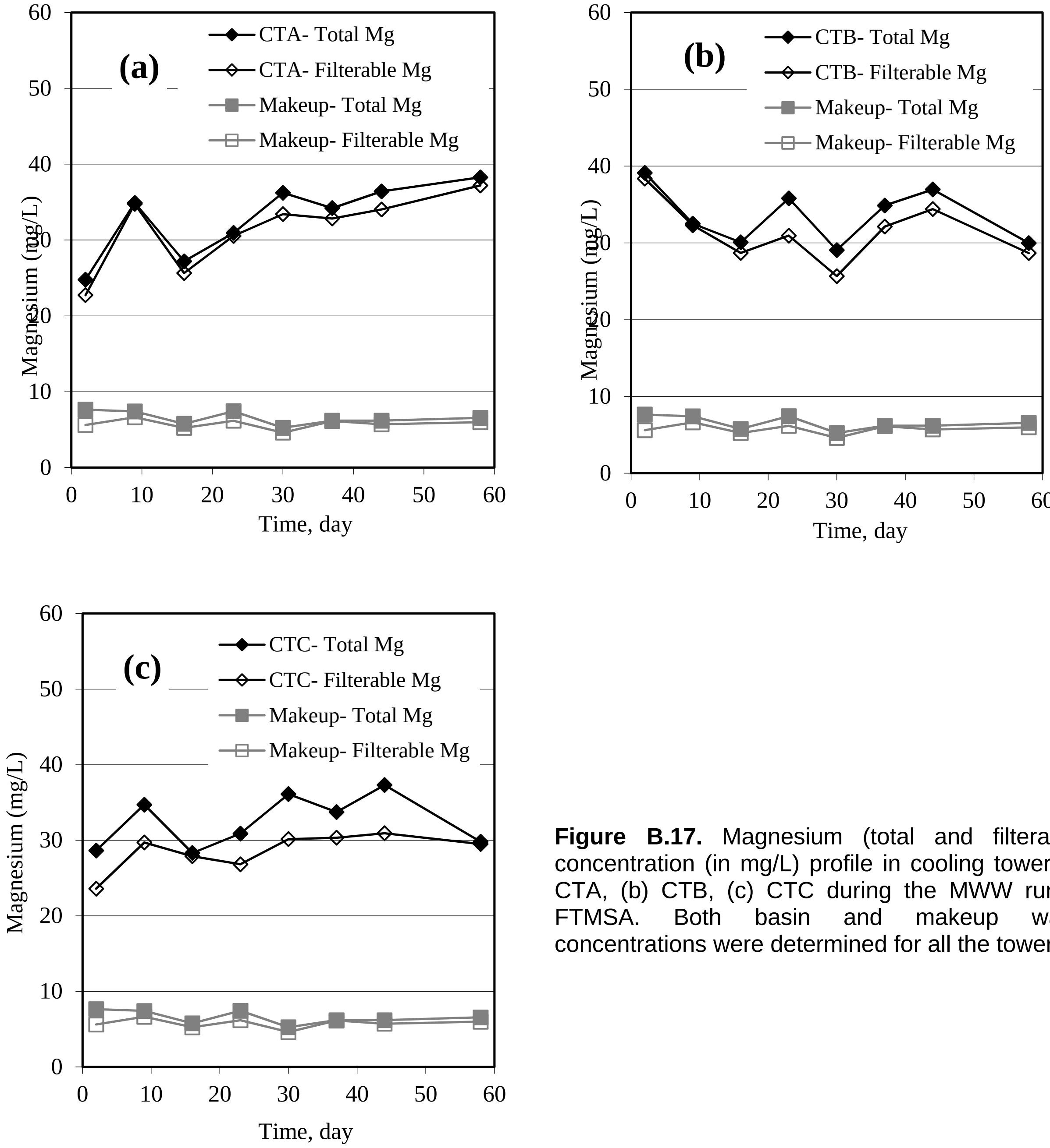


Figure B.17. Magnesium (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

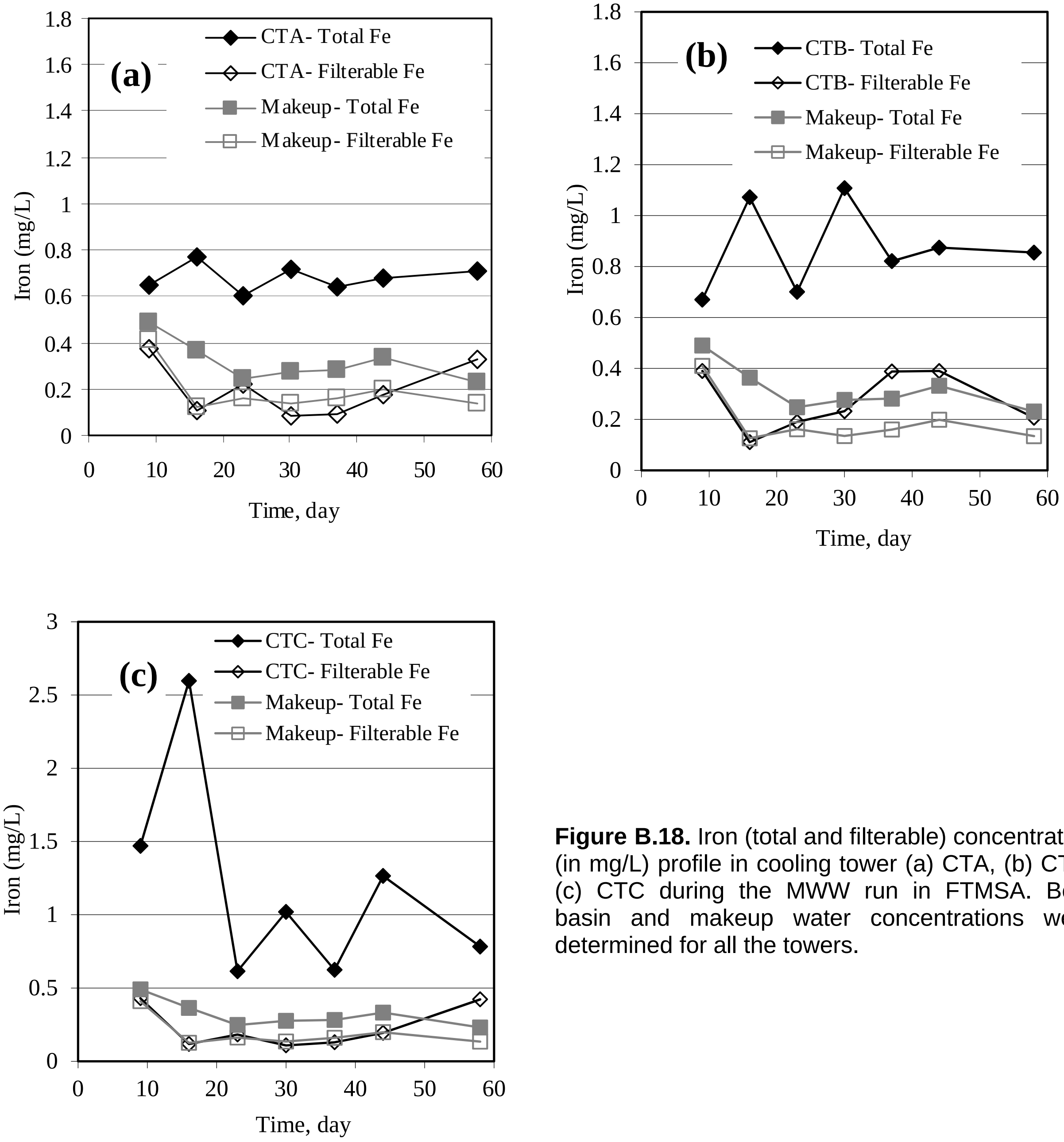


Figure B.18. Iron (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

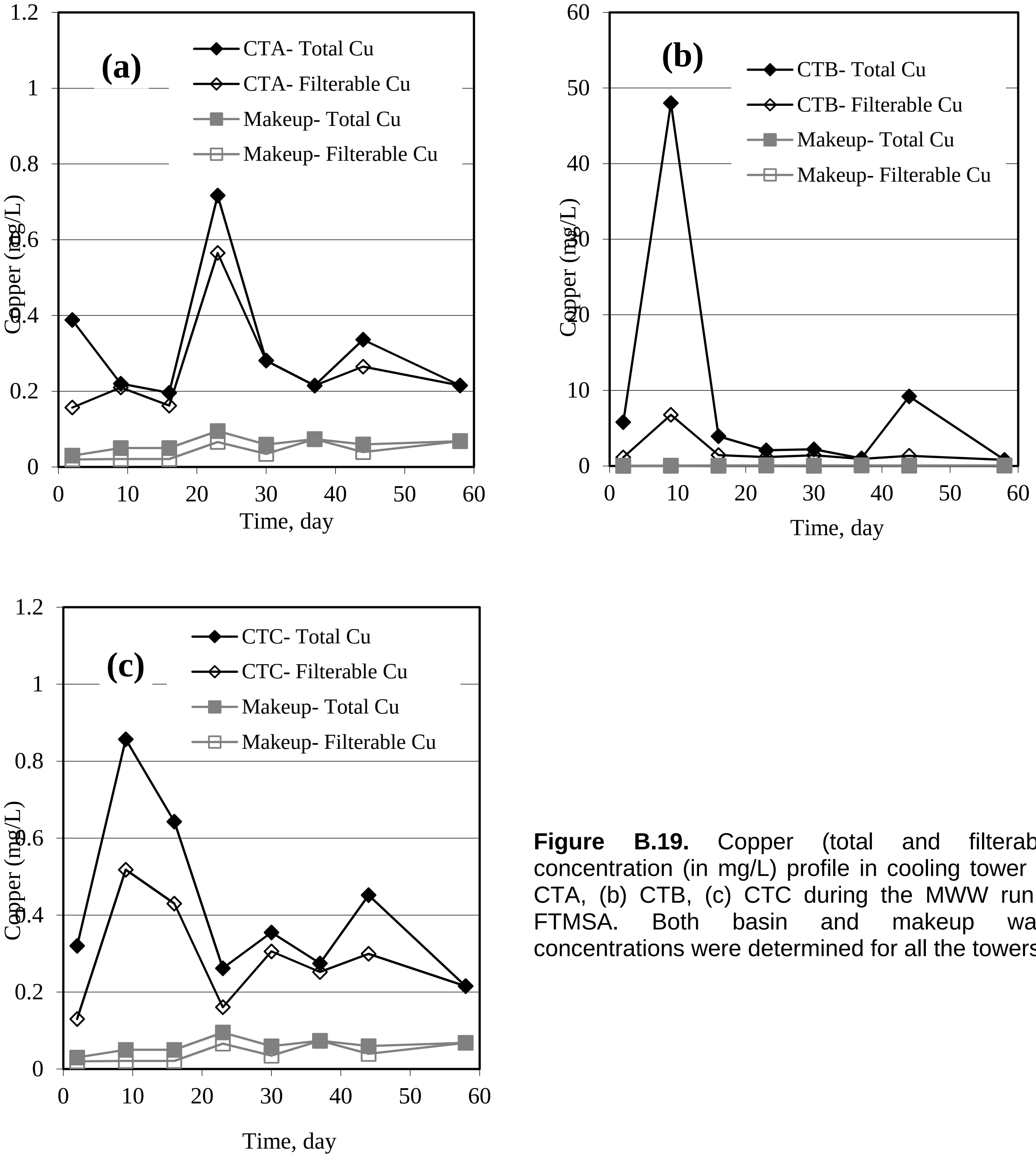


Figure B.19. Copper (total and filterable) concentration (in mg/L) profile in cooling tower (a) CTA, (b) CTB, (c) CTC during the MWW run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

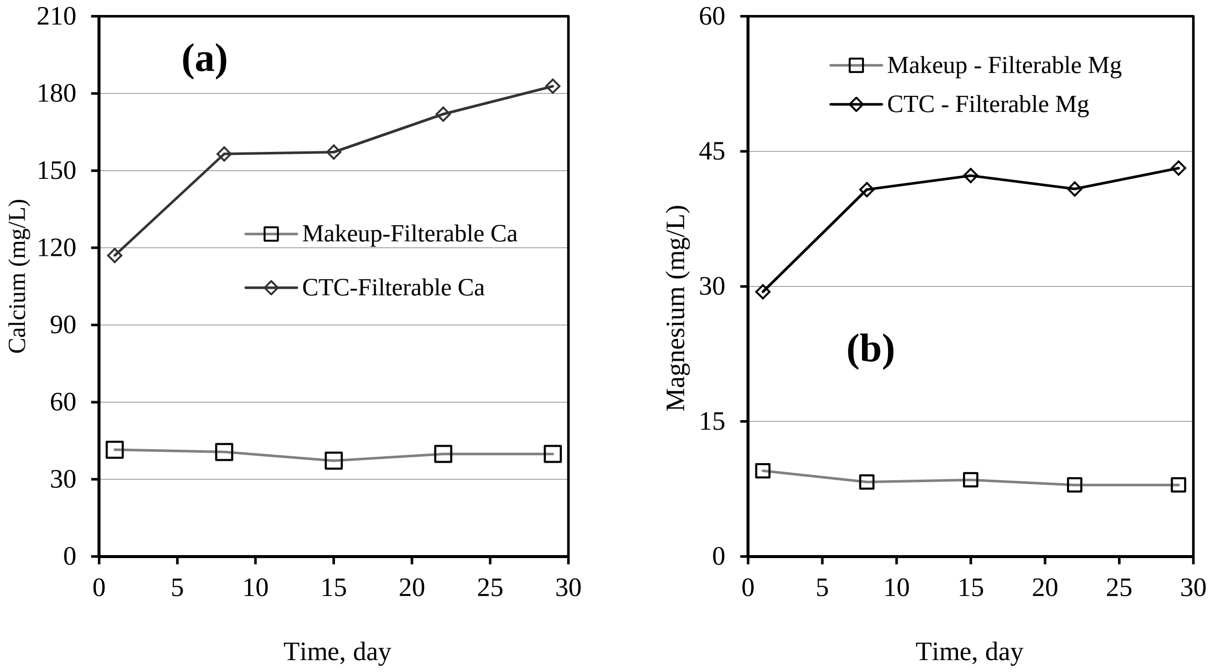


Figure B.20. Calcium (filterable) (a), and Magnesium (filterable) (b) concentration (in mg/L) profiles in cooling tower CTC during the MWW_NFG run in FTMSA. Both basin and makeup water concentrations were determined for all the towers.

Table B.1. Comparison of selected water quality parameters of the secondary and tertiary treated municipal wastewater from FTMSA with reference values found in literature[a].

Parameter	Unit	Secondary Treated Municipal Wastewater from FTMSA	Range of Effluent Quality after conventional secondary treatment [a,b]	Tertiary (nitrified-filtered) Treated Municipal Wastewater from FTMSA	Range of Effluent Quality after conventional activated sludge with nitrification and filtration treatment [a]
Total Suspended Solids (TSS)	mg/L	46 ± 29	5-25	5 ± 9	2-8
Chemical Oxygen Demand (COD)	mg/L	101.83 ± 18.70	40-80	39.61 ± 23.59	30-70
Total Organic Carbon (TOC)	mg/L	26.48 ± 3.02	10-40	8.94 ± 3.58	8-30
Ammonia nitrogen	mg-N/L	24.12 ± 10.19	12-45	3.45 ± 3.50	1-6
Nitrate nitrogen	mg-N/L	2.17 ± 0.45	0 - trace	2.74 ± 0.70	10-30
Total Dissolved Solids (TDS)	mg/L	644 ± 375	500-700	362 ± 119	500-700

[a] Asano, Takashi; Burton, Franklin L.; Leverenz, Harold L.; Tsuchihashi, Ryujiro; Tchobanoglous, George (2007). Water Reuse - Issues, Technologies, and Applications. (pp: 110-111). McGraw-Hill.

[b] Conventional secondary is defined as activated sludge treatment

Table B.2. Total solids (TS), total suspended solids (TSS) and total dissolved solids (TDS) in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Total Solids (TS), mg/L	Total Suspended Solids (TSS) , mg/L	Total Dissolved Solids (TDS) , mg/L
MWW_NF run	CTA – Basin	1892 ± 304	19 ± 28	1873 ± 301
	CTB – Basin	2144 ± 207	24 ± 26	2120 ± 200
	CTC – Basin	1946 ± 274	25 ± 27	1921 ± 266
	Makeup – CTA	366 ± 122	5 ± 9	362 ± 119
MWW run	CTA – Basin	2490 ± 611	208 ± 106	2282 ± 618
	CTB – Basin	2615 ± 901	201 ± 94	2414 ± 916
	CTC – Basin	2537 ± 611	176 ± 120	2362 ± 609
	Makeup – CTA	690 ±3 93	46 ± 29	644 ± 375
MWW_NFG run	CTC – Basin	2312 ± 352	28 ± 12	2284 ± 342
	Makeup – CTC	452 ± 120	13 ± 7	439 ± 121

Table B.3. Total organic carbon (TOC) in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Total organic carbon (TOC), mg/L
MWW_NF run	CTA – Basin	25.68±11.78
	CTB – Basin	27.00±6.49
	CTC – Basin	25.85±7.16
	Makeup – CTA	8.94±3.58
MWW run	CTA – Basin	83.77±16.99
	CTB – Basin	75.89±11.87
	CTC – Basin	89.18±37.10
	Makeup – CTA	26.48±3.02
MWW_NFG run	CTC – Basin	9.25 ± 2.85
	Makeup – CTC	3.21 ± 0.53

Table B.4. Alkalinity in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Alkalinity (mg/L as $CaCO_3$)
MWW_NF run	CTA – Basin	69.93 ± 41.23
	CTB – Basin	71.79 ± 43.34
	CTC – Basin	71.44 ± 38.69
	CTA – makeup	26.07 ± 13.03
	CTB – makeup	24.44 ± 11.75
	CTC – makeup	25.07 ± 11.72
MWW run	CTA – Basin	227.29 ± 70.55
	CTB – Basin	74.25 ± 51.36
	CTC – Basin	104.36 ± 65.14
	CTA – makeup	122.82 ± 18.79
MWW_NFG run	CTC – Basin	112.45 ± 60.72
	Makeup – CTC	44.24 ± 25.73

Table B.5. Chemical Oxygen Demand (COD) in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Chemical Oxygen Demand (COD), mg/L
MWW_NF run	CTA – Basin	171.06 ± 150.46
	CTB – Basin	167.53 ± 115.82
	CTC – Basin	110.50 ± 26.72
	Makeup – CTA	39.61 ± 23.59
MWW run	CTA – Basin	338.60 ± 88.60
	CTB – Basin	373.61 ± 54.48
	CTC – Basin	292.33 ± 97.94
	Makeup – CTA	101.83 ± 18.70
MWW_NFG run	CTC – Basin	40.08 ± 18.39
	Makeup – CTC	15.50 ± 13.20

Table B.6. Ammonia concentration in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Ammonia concentrations, mg/L as N
MWW_NF run	CTA – Basin	3.18 ± 3.42
	CTB – Basin	3.07 ± 3.98
	CTC – Basin	3.45 ± 3.50
	Makeup – CTA	1.42 ± 0.66
MWW run	CTA – Basin	9.56 ± 7.26
	CTB – Basin	45.49 ± 30.49
	CTC – Basin	7.35 ± 5.25
	Makeup – CTA	24.12 ± 10.19
MWW_NFG run	CTC – Basin	0.58 ± 0.42
	Makeup – CTC	0.39 ± 0.19

Table B.7. Anions (Phosphate (PO_4^{3-}), Chloride (Cl^-), Nitrate (NO_3^-) and Sulfate (SO_4^{2-})) concentration in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Phosphate (PO_4^{3-}), mg/L	Chloride (Cl^-), mg/L	Sulfate (SO_4^{2-}), mg/L	Nitrate (NO_3^-), mg/L
MWW_NF run	CTA – Basin	18.86 ± 9.55	920.22 ± 119.36	214.79 ± 29.21	23.07 ± 5.60
	CTB – Basin	23.46 ± 12.76	1003.33 ± 113.87	234.18 ± 31.16	25.81 ± 7.02
	CTC – Basin	21.36 ± 10.70	1033.22 ± 60.94	201.21 ± 39.16	22.40 ± 4.59
	Makeup – CTA	7.16 ± 2.38	212.11 ± 24.33	57.83 ± 10.30	12.14 ± 3.08
MWW run	CTA – Basin	8.72 ± 2.81	1090.78 ± 120.33	292.77 ± 104.46	22.38 ± 5.72
	CTB – Basin	25.25 ± 7.28	1179.33 ± 325.97	465.18 ± 84.27	22.37 ± 12.49
	CTC – Basin	24.98 ± 8.45	965.22 ± 109.025	400.56 ± 61.66	13.96 ± 5.50
	Makeup – CTA	9.98 ± 1.47	199.11 ± 24.53	66.97 ± 5.13	9.62 ± 1.99
MWW_NFG run	CTC – Basin	8.46 ± 3.91	162.0 ± 5.6	59.47 ± 8.94	11.78 ± 8.19
	Makeup – CTC	9.95 ± 6.07	1111.8 ± 238.2	316.55 ± 96.09	20.73 ±15.45

Table B.8. Cations (Calcium, Magnesium, Iron, and Copper) concentration in basin waters, and in makeup waters of cooling towers.

Experiment	Location	Calcium (mg/L)		Magnesium (mg/L)	
		Total	Filtered	Total	Filtered
MWW_NF run	CTA – Basin	123.46 ± 30.64	112.89 ± 26.60	33.94 ± 6.06	32.34 ± 6.36
	CTB – Basin	135.78 ± 24.74	124.63 ± 27.18	37.12 ± 6.38	34.62 ± 6.79
	CTC – Basin	123.86 ± 22.05	112.36 ± 19.76	33.26 ± 5.69	31.46 ± 6.19
	Makeup – CTA	46.73 ± 24.39	41.01 ± 22.48	11.11 ± 7.00	10.43 ± 7.21
MWW run	CTA – Basin	116.53 ± 10.83	110.55 ± 10.46	32.86 ± 4.79	31.39 ± 4.88
	CTB – Basin	121.61 ± 21.70	114.82 ± 23.44	33.54 ± 3.69	31.39 ± 3.90
	CTC – Basin	118.99 ± 14.08	107.06 ± 7.78	32.43 ± 3.48	28.61 ± 2.44
	Makeup – CTA	33.34 ± 8.75	26.80 ± 5.32	6.55 ± 0.87	5.76 ± 0.63
MWW_NFG run	CTC – Basin	Not analyzed	157.10 ± 22.31	Not analyzed	39.28 ± 5.02
	Makeup – CTC	Not analyzed	39.82 ± 1.42	Not analyzed	8.44 ± 0.58

Experiment	Location	Iron (mg/L)		Copper (mg/L)	
		Total	Filtered	Total	Filtered
MWW_NF run	CTA – Basin	0.42 ± 0.07	0.31 ± 0.09	0.37 ± 0.16	0.34 ± 0.12
	CTB – Basin	0.44 ± 0.06	0.32 ± 0.10	0.46 ± 0.19	0.39 ± 0.15
	CTC – Basin	0.45 ± 0.08	0.28 ± 0.09	0.34 ± 0.14	0.30 ± 0.11
	Makeup – CTA	0.31 ± 0.05	0.26 ± 0.08	0.17 ± 0.09	0.15 ± 0.09
MWW run	CTA – Basin	0.68 ± 0.06	0.20 ± 0.12	0.32 ± 0.17	0.26 ± 0.13
	CTB – Basin	0.87 ± 0.17	0.27 ± 0.12	9.13 ± 15.96	1.88 ± 1.99
	CTC – Basin	1.20 ± 0.70	0.23 ± 0.14	0.42 ± 0.22	0.29 ± 0.13
	Makeup – CTA	0.32 ± 0.09	0.19 ± 0.10	0.06 ± 0.02	0.04 ± 0.02

* Iron and copper concentrations were not analyzed in the MWW_NFG run

APPENDIX C

Granular Activated Carbon Treatment Design and Performance Data

C.1 Bench-scale Testing with Granular Activated Carbon for Organics Removal

Another potential tertiary treatment for secondary treated municipal wastewater is with granular activated carbon (GAC) for removal of residual dissolved organic matter. Organic matter in cooling system makeup water promotes biofouling, and increases doses of biocides needed for control of biofouling. In addition, the organic matter can adsorb on metal surfaces and contribute directly to fouling in this manner.

In this quarter, an experimental GAC column apparatus was set up and used for tests with secondary treated municipal wastewater (MWW_F) and tertiary treated municipal wastewater (MWW_NF) to reduce their organic content. Measurement of the total organic carbon (TOC) in raw and filtered water was performed to assess the efficiency of carbon adsorption of dissolved organic matter. Since the secondary treated municipal wastewater (MWW) from FTMSA was collected before the granular media filtration step, filtration through a 0.45 µm membrane filter was performed in the laboratory to generate the MWW_F samples.

C.1.1 Approach

Standard batch adsorption isotherm tests with Filtrasorb 300 and 400 activated carbons (Calgon Carbon Corporation, Pittsburgh, PA) with both types of wastewaters (MWW_F and MWW_NF) were performed to assess the adsorptive capacity of these GACs and determine non-adsorbable fraction of TOC in these water samples. The particular GAC materials were produced from selected grades of bituminous coal and are typically used to remove organic pollutants form potable water and wastewater. After the batch tests, standard breakthrough experiments with a fixed-bed adsorption column were performed with these water samples to determine dynamic adsorptive capacity that may be expected in the field. The bench-scale GAC column was designed using a typical empty bed contact time (EBCT) of 10 minutes.

a) *GAC adsorption isotherm tests:* F300 and F400 granular activated carbons were pulverized to have 95% particles passing through a 325 U.S. mesh screen. A 3-day equilibration period was selected for the batch adsorption isotherm tests. The isotherm test were performed according to the following procedure:

- Secondary and tertiary treated municipal wastewater samples were collected on two separate days at FTMSA. Water samples were kept at room temperature and were used within 6 hours after collection.
- In the first isotherm test, selected weights (0.05, 0.25, 0.5, 2, 4, and 8 grams) of dried, pulverized activated carbon were added to 100 mL of the test solutions (MWW_F and MWW_NF). In the second isotherm test, additional weights (0.05, 0.1, 0.25, 0.5, 1, 2, 4, and 8 grams) were selected to obtain better characterization of the adsorption isotherm.
- The mixtures of GAC and water samples were then continuously agitated for 3 days at room temperature. The samples were considered to be at equilibrium when no further change in solution phase adsorbate concentration was observed.
- After reaching equilibrium, the carbon adsorbent was allowed to settle and then samples of supernatant from each reactor were filtered through a 0.45 µm membrane filter to remove the carbon fines. The aqueous samples were analyzed for residual TOC concentration (TOC analyzer, Ionics, MA).

b) *GAC column adsorption tests:* A bench-scale, glass GAC column (4.0 in. wide and 9.5 in. long) was charged with F300 GAC and used in column testing for development of an adsorption breakthrough curve. The column had an empty bed contact time (EBCT) of 10 minutes and was used to treat about 4 gallons of water. Solution flowrate through the column was set at 120 mL/min based on the calculations shown in Appendix A. Tertiary treated waste water (MWW_NF) was used in the experiments.

C.1.2 Results of bench-scale tests

C.1.2.1 GAC batch isotherm tests

The first test was conducted to validate previous findings. MWW_F and MWW_NF used in this test had the initial TOC of 13.9 and 9.1 mg/L, respectively. Figure C.1 shows the adsorption isotherm of GAC F400 for the two wastewater samples collected on the first sampling in December 2010. In case of MWW_F, 2.45 mg/L of TOC was determined to be non-adsorbable. For MWW_NF, 2.0 mg/L of TOC was found to be non-adsorbable.

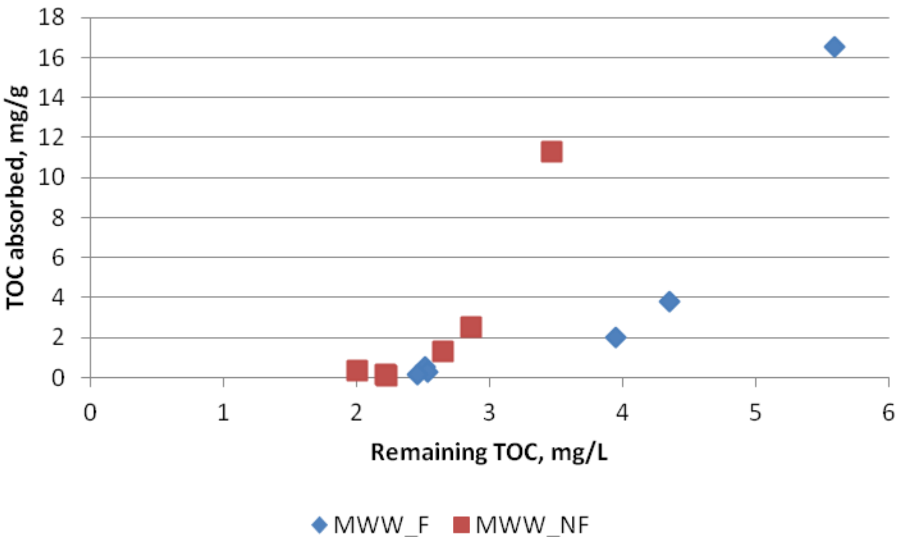


Figure C.1. Adsorption isotherm test results for Filtrasorb 400 with MWW_F and MWW_NF from FTMSA (first test)

The second set of isotherm tests simply repeated the first tests with additional measurement points and water samples from the second sampling event at FTMSA in December 2010. MWW_F had the initial TOC of 16.4 mg/L and MWW_NF had the initial TOC of 13.4 mg/L. Figure C.2 shows the adsorption isotherm for GAC F400 with the two wastewater samples collected on the second sampling day in December 2010. The non-adsorbable TOC fractions in MWW_F and MWW_NF determined in the second set of tests were similar to those obtained from the first tests.

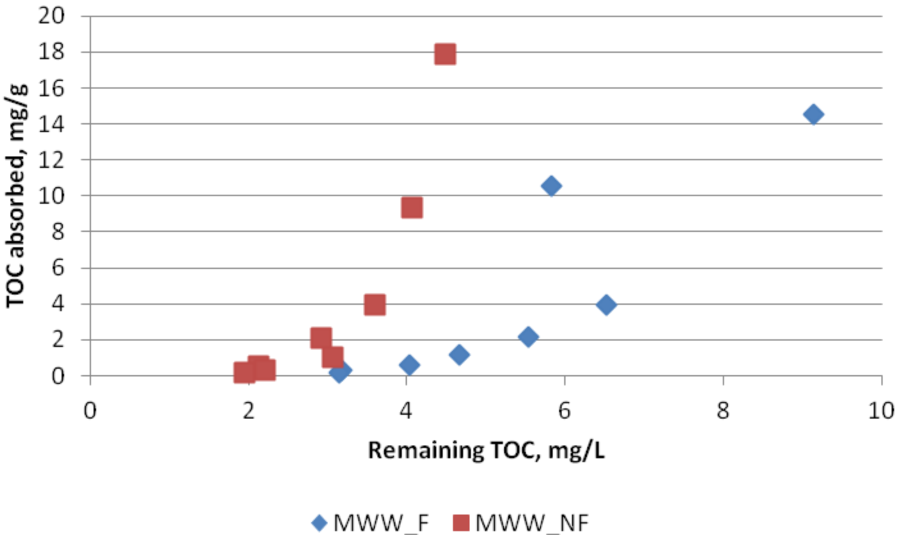


Figure C.2 Adsorption isotherm test results for Filtrasorb 400 with MWW_F and MWW_NF from FTMSA (second test)

The third set of GAC isotherm testing was conducted with F300 and F400 activated carbons and secondary treated waste water without filtration (MWW). The results presented in Figure C.3 indicated that the TOC in MWW could be reduced from 22.01 ± 0.06 mg/L to 4.0 mg/L. It was also determined that the adsorption effectiveness of F300 and F400 are similar. As a result, F300 was chosen for bench-scale GAC column tests.

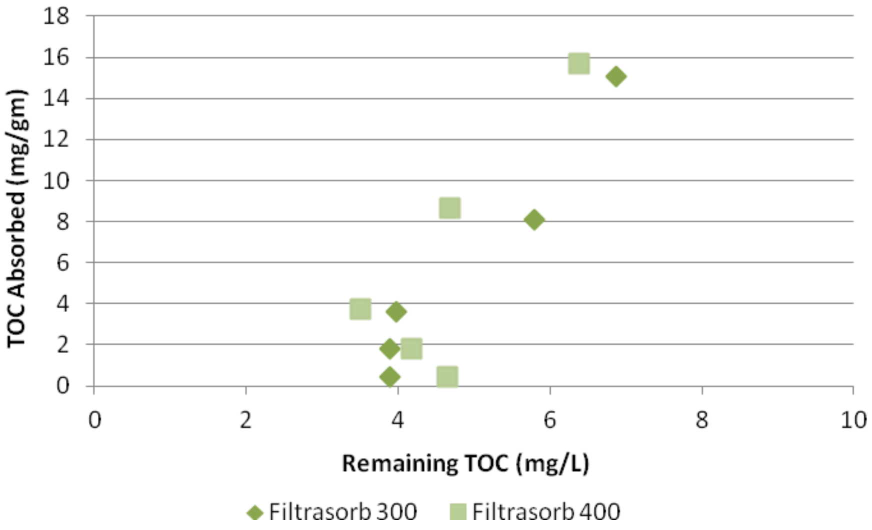


Figure C.3 Adsorption isotherm test results for Filtrasorb 300 and Filtrasorb 400 with MWW from FTMSA

C.1.2.2 GAC dynamic column tests

Dynamic column testing with the F300 GAC was conducted with tertiary treated water (MWW_NF) having an initial TOC of 10.54 mg/L. As can be seen in Figure C.4, initial TOC was reduced to less than 4 mg/ and it remained at that level for the entire duration of the column test. A total of 4 gallons of MWW_NF was filtered through the column in the testing.

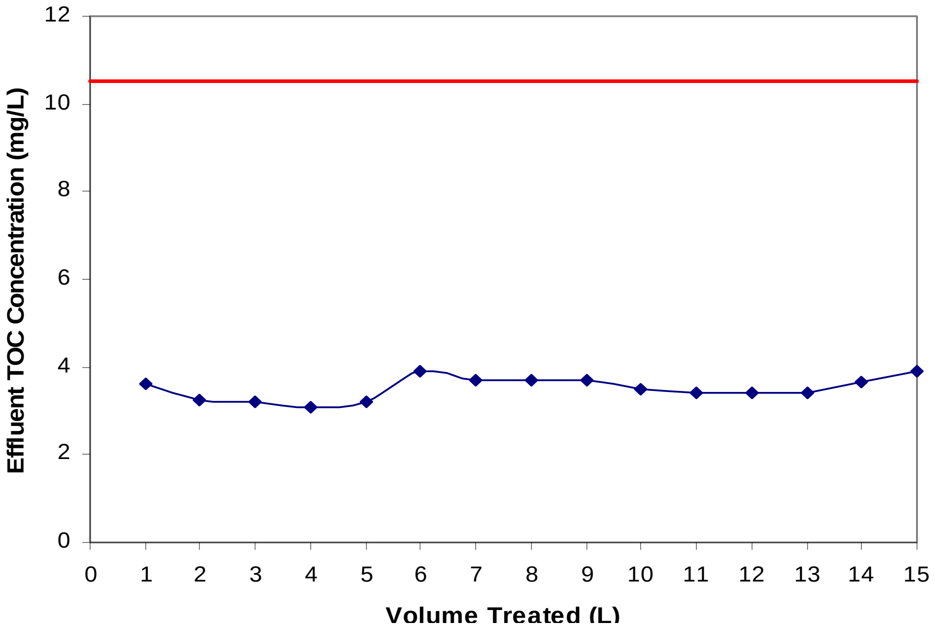


Figure C.4 Breakthrough curve for MWW_NF in column testing with Filtrasorb 300 GAC. EBCT = 10 min

The column effluent TOC level of 4 mg/L represents the non-adsorbable fraction of TOC in the MWW-NF water. Based on these results, it can be estimated that the adsorption column needed in the field to produce 40 gal/day of MWW_NFG (daily make up water for one pilot-scale cooling tower) needs to contain at least 5 gallons of activated carbon. However, the initial TOC level in the tertiary treated water may vary from 5~17 mg/L and this variation will inevitably influence the non-adsorbable TOC in the water.

C.1.3 Summary

- Adsorption isotherm tests showed that Filtrasorb 300 GAC can remove about 80% of TOC from MWW_F and MWW_NF. Non-adsorbable TOC in MWW_F and MWW_NF was determined to be around 2 ppm.
- Dynamic column testing with 10 min EBCT and Filtrasorb 300 GAC reduced the initial TOC in MWW_NF of 10.5 mg/L to about 4 mg/L.
- Variation of the TOC level in the MWW_NF water was observed to be significant from day to day and the efficiency of TOC removal of the GAC column depends on the initial TOC concentration.
- The 10 min EBCT can be utilized to treat the MWW_NF water in a pilot scale column tests.

C.2 Pilot-scale GAC Treatment Setup

C.2.1 Pilot-scale granular activated carbon adsorption column design

Based on the results of bench-scale tests, A column capable of treating 75 gallons per day of MWW_NF water was reassembled in FTMSA. The GAC adsorption column was designed with capacity to remove all of the GAC-adsorbable portion of the TOC in the MWW_NF water (Figure C.5). Like the other waters to be used in the pilot-scale tests, the makeup water tank will be treated with pre-formed monochloramine to achieve a residual of 0.5-1.0 ppm, and with TTA to achieve a concentration of 0.5 ppm.

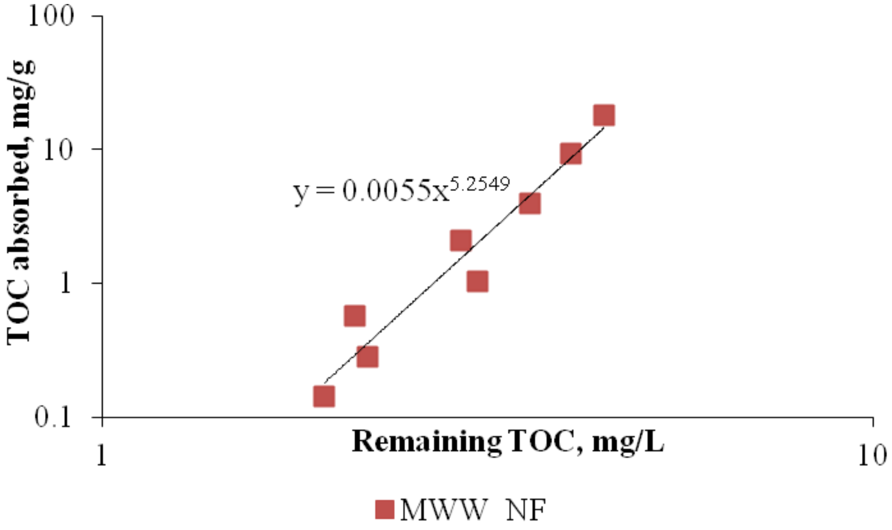


Figure C.5 Freundlich isotherm from equilibrium adsorption experiments with tertiary treated municipal wastewater from Franklin Township Municipal Sanitary Authority, Murrysville, PA

The GAC-treated water was prepared by treating 75 gallons per day of MWW_NF water by passing it through a GAC column (Figure B.6.). The GAC adsorption column was constructed to remove all of the adsorbable portion of the TOC in the MWW_NF water. A PVC column 8-inches in diameter with a gravel and sand support and 33 inches of Calgon Filtrasorb 300 GAC was used in the field. The column was operated with an empty bed contact time of 30 minutes.

Figure C.6. Granular activated carbon adsorption column setup at FTMSA

In order to reduce algal growth in the tank, the pump used to feed the adsorption column was equipped with a timer-based controller to discharge 80-90 gallons of MWW_NFG effluent an hour prior to research group arrival. The water tank was also cleaned routinely with tap water to prevent biological growth on the walls.

C.2.2 Granular activated carbon adsorption column performance data

Total organic carbon concentration was monitored in both influent and effluent from the GAC adsorption column and the data for the duration of the entire experiment are shown in Figure C.7. It was observed that the GAC-adsorption process could consistently reduce the organic matter concentration to about 3 mg/L for one month-long testing without any observable breakthrough. Tertiary water discharged from sand filtration contained 8.73 ± 1.84 mg/L of total organic carbon (TOC), and the same water after GAC treatment contained 2.91 ± 0.51 mg/L TOC.

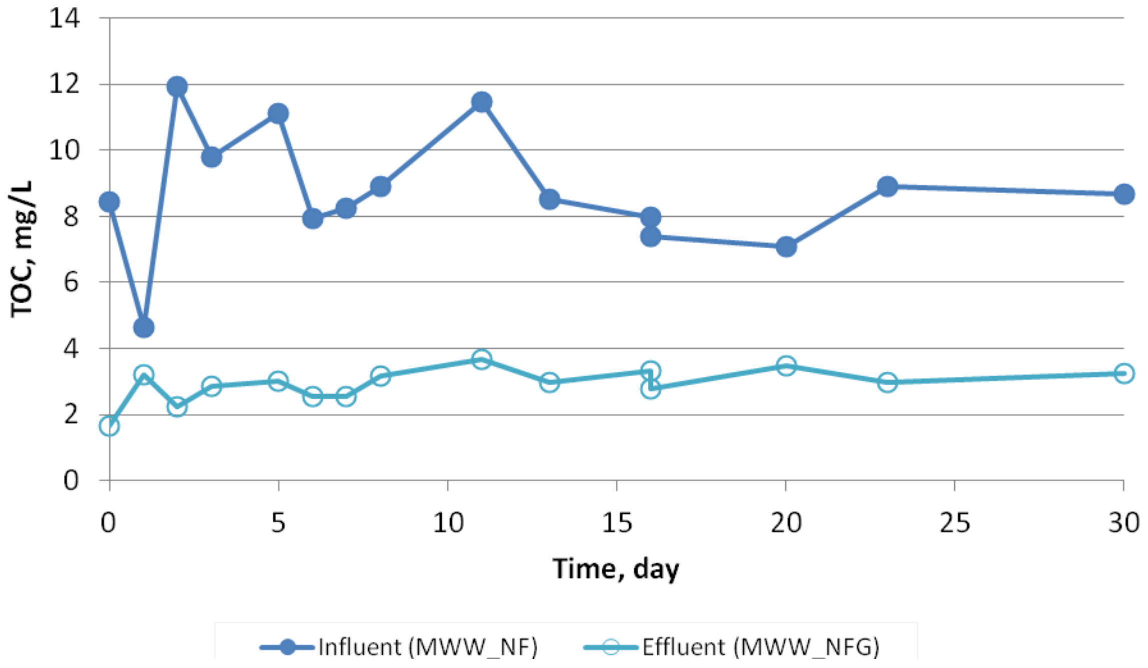


Figure C.7 Total organic carbon concentrations in MWW_NF (before GAC treatment) and MWW_NFG (after GAC treatment) for entire 30 days of pilot-scale testing, Summer 2011.

APPENDIX D

Survey of Existing Practices

There are four main challenges (A through D) associated with the sustainability of reusing wastewater for power plant cooling:

Challenge A – biofouling, scaling, and corrosion,
Challenge B – wastewater delivery,
Challenge C – public perception of wastewater reuse, and
Challenge D – valuation of freshwater versus wastewater.

Sustainability metrics including environmental impacts, water cost and value, and public perception were evaluated through life cycle assessment and survey administration.

Much research has been conducted in the area of wastewater reclamation and reuse, but little work has included sustainability analyses using life cycle assessment. Some authors, such as Feeley at al. (2007), have investigated the rate of freshwater withdrawal in the thermoelectric industry (Feeley, 2007). Others have examined wastewater reclamation from a life cycle perspective (Hospido, Moreira et al., 2004; Chen, Yeh et al., 2005; Ammary, 2006; Hospido, Moreira et al., 2008) and have considered cooling water reuse within power plants (Jin-Kuk Kim, 2004). Few researchers, however, have addressed the broader scenario of freshwater depletion and wastewater recycling from a life cycle assessment perspective (Bayart, Bulle et al., 2010). The sustainability challenges of reusing treated municipal wastewater for power plant cooling have not been examined, nor have collaborations between power plants and wastewater treatment plants on treated wastewater reuse.

D.1 System Description

In 2005, thermoelectric power plants were responsible for 49 percent of total freshwater withdrawals in the United States (Barber, 2009). This water is used primarily for cooling in the steam cycle, via both once-through and recirculating cooling systems. Due to regulations governing freshwater withdrawal, such as Section 316(b) of the U.S. Clean Water Act, more power plants are using recirculating cooling systems.

One alternative source of cooling water is secondary treated municipal wastewater. This reduces freshwater withdrawals, but the levels of biodegradable organic matter, nitrogen, and phosphorus contained in the wastewater, even after treatment, pose significant challenges for power plants with respect to biofouling, corrosion, and scaling in the cooling systems. In most cases, secondary treated municipal wastewater requires additional chemical treatment to control these operational problems for use in power plant cooling (Vidic and Dzombak, 2009). However, chemical treatment can be costly and contribute to increased levels of environmental impacts; both concerns need to be addressed.

A process flow diagram for the coupling of wastewater treatment facilities with power plants for cooling is shown in Figure D.1.1. The wastewater treatment plant includes primary and secondary treatment and sometimes includes tertiary treatment. Upon receipt of the wastewater, the power plant usually further treats the water by adding of chemicals to control biofouling, corrosion, and scaling in the cooling processes.

D.1.1 Challenges to sustainable reuse of wastewater for power plant cooling

Challenge A: Biofouling, Scaling, and Corrosion. Secondary treated wastewater typically exhibits high concentrations of total dissolved solids (TDS), total suspended solids (TSS), organic matter, bacteria, ammonia (NH_3), and phosphate (PO_4) relative to freshwater. These characteristics present certain challenges for reuse and must be addressed through chemical, biological, and physical treatment. As TDS levels increase, rates of corrosion and scaling can also increase. Corrosion often occurs on the surface of the power plant's metal heat exchangers and transport pipes (Li, Chien et al., 2011). Corrosion can ultimately result in equipment failure. Scaling refers to mineral deposition on pipes and heat exchange surfaces and occurs when the recirculating water evaporates, leaving supersaturated salts to precipitate and form scale (Li, Chien et al., 2011). Biofouling occurs when residual amounts of phosphorous, nitrogen, and organic matter still in the treated wastewater foster biological growth on the surfaces of equipment at the power plant (Li, Chien et al., 2011). This research considers a life-cycle sustainability analysis of the chemicals used to treat biofouling, scaling, and corrosion.

Challenge B: Wastewater Delivery. In addition to treatment, the delivery of the wastewater from the wastewater treatment facility to the power plant can be a challenge. In most cases, piping needs to be installed between the plants for conveyance of the water. Manufacture of pipes for this purpose can lead to impacts on the environment.

Challenge C: Public Perception. The public has traditionally been opposed to wastewater reuse, largely due to a lack of information (Friedler and Lahav, 2006; Jimenez and Asano, 2008). Compounding the problem, the information that is made available is often difficult to understand without some technical background knowledge. While the public tends to look favorably upon natural conservation projects, lack of public understanding can often lead to difficulties in obtaining permits and other legal issues (Hartley, 2006).

Challenge D: Valuation of Freshwater vs. Wastewater. It can be difficult to quantify the value of freshwater versus wastewater. In all but arid areas, freshwater may appear to be more accessible as a good of the commons, but given factors related to water rights and environmental regulations, treated wastewater may be less expensive in the long term. In addition, the technology required to treat wastewater can appear to be more expensive in the short term than the cost of leaving wastewater untreated (Romero-Hernandez, 2004). By factoring in the benefits of treating and reusing wastewater, the disparities between water pricing and water value can be better understood and addressed.

D.1.2 Sustainability metrics and tools

Due to the complex nature of the coupled wastewater treatment facilities and power plant system, merely using one sustainability tool or metric will not provide an adequate assessment of the sustainability of the entire system and its challenges. Therefore, this study assesses several tools and metrics for quantifying and evaluating the sustainability associated with the aforementioned challenges. The metrics and tools that can be used to measure sustainability of reusing reclaimed water in power plant cooling system are summarized in Table D.1.1

The sustainability of potential additional treatment processes used to mitigate biofouling, scaling, and corrosion can be evaluated through an LCA of the chemicals and methods used to treat the water (Challenge A). The sustainability metrics available to evaluate Challenge A include feasibility and performance, cost, and environmental impacts. The sustainability of wastewater delivery (Challenge B) can be addressed through and LCA of the conveyance, i.e. piping, of the water. The sustainability metrics relevant for Challenge B include environmental impacts and cost.

Public perception (Challenge C) can be assessed by administering a survey to wastewater treatment and power plant personnel. The survey questions can be designed to elicit responses regarding relationships with customers and other stakeholders by using metrics such as number of complaints and lawsuits filed, willingness of people to live near the plants, the number of public events hosted by the plants, and the number of communications issued by the plants.

Water valuation (Challenge D) considers cost, water volume, water quality, and regional ecosystem quality and can be assessed through metrics such as surveys and literature reviews**.** In this study, environmental impacts, cost, value, and public relations were evaluated as sustainability metrics, and LCA and survey administration were used as the corresponding sustainability assessment tools.

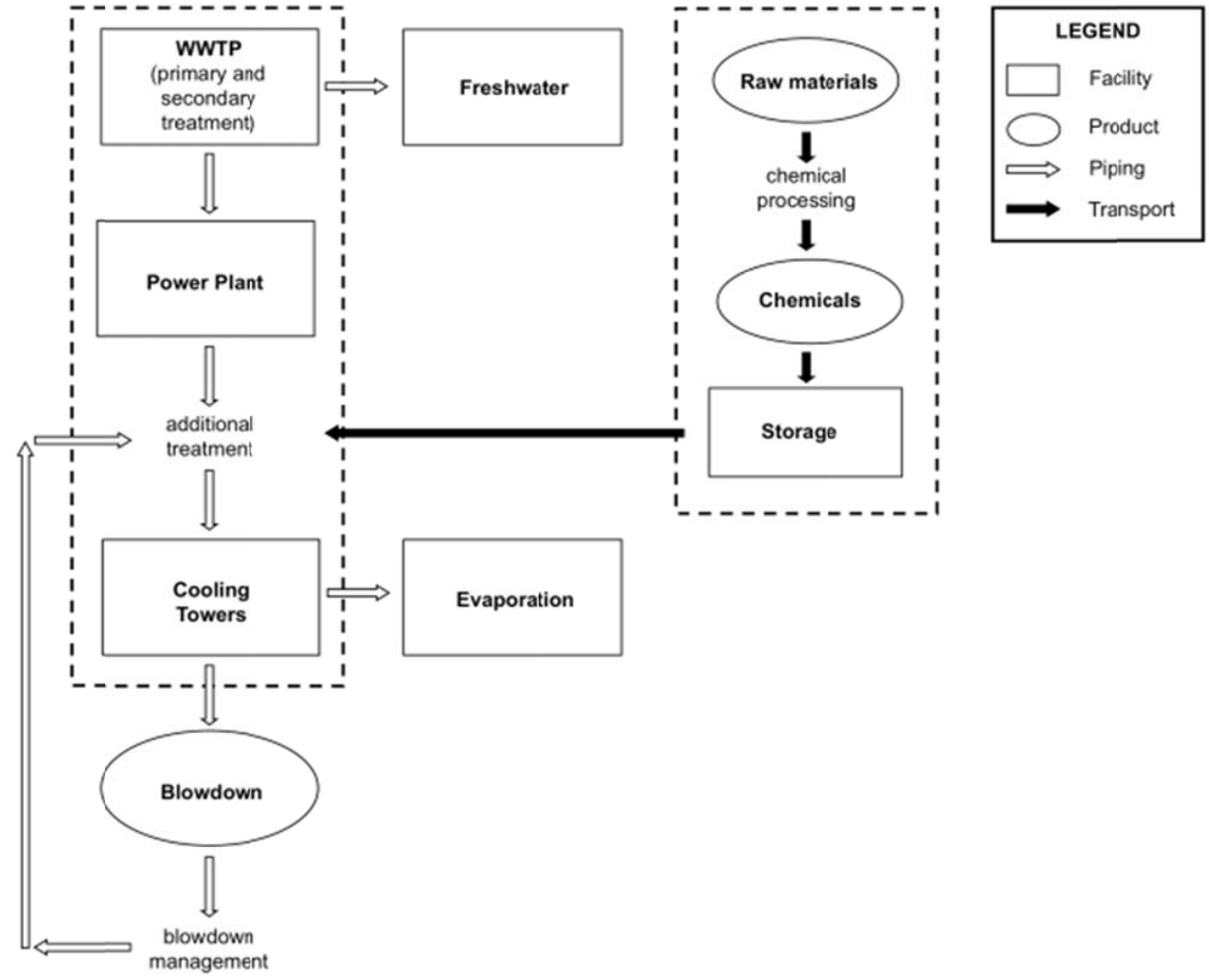


Figure D.1.1 Process flow diagram for coupling wastewater treatment plants with power plants to provide cooling water.
Note:System boundaries for the life cycle assessment are identified with dashed lines (- - -) and include the wastewater effluent site to the cooling water effluent site. WWTP = wastewater treatment plant.

Table D.1.1 Metrics, tools, and data sources to evaluate sustainability challenges of reusing wastewater in power plants.

	Challenges	Potential Sustainability Metrics	Potential Tools
A	Biofouling, corrosion, scaling	Effectiveness of chemical treatment	Treatment records, lab tests, survey
		Environmental impacts [a]	LCA
		Cost	LCC, survey
B	Wastewater delivery	Environmental impacts [a]	LCA
		Cost	LCC, survey
C	Public perception	Complaints, lawsuits, etc.	Survey, public court records
		Public events, communications	Survey
		Willingness to live nearby	Survey
D	Freshwater vs. wastewater valuation	Value	Survey
		Cost	LCC, survey
		Water volume	Water balance, MFA, LCA
		Water quality	LCA, risk assessment
		Regional ecosystem quality	LCA, risk assessment

[a] Denotes metrics evaluated in this research.

D.2 Methods

This study used two main tools to evaluate different metrics for sustainability: LCA and a survey instrument.

LCA. A comparative process-based LCA was conducted in relation to Challenges A and B on a snapshot of the system, the boundaries of which are depicted in Figure D.1.1. The LCA framework of the ISO 14040 was utilized when compiling the LCI data to address the aforementioned challenges to utilizing wastewater for power plant cooling. The system boundaries were drawn from the exit of the wastewater effluent from the wastewater treatment facility to the exit of fluid outputs from the power generation facility. Challenge A (biofouling, corrosion, and scaling) occurs at the power plant, while Challenge B (wastewater delivery) occurs between the wastewater treatment plant and the power plant.

The LCIA tool used in this study is Eco-indicator 99 and its results are reported at the midpoint (Goedkoop and Spriensma, 1999). The categories used in this study include ecotoxicity, respiratory organics, acidification and eutrophication, radiation, land use, carcinogens, fossil fuels, and climate change.

For Challenge A—biofouling, corrosion, and scaling—the system boundaries encompass the chemical production process, extending from the raw material extraction to the storage of the finished product. Transportation to the power plant was not included, nor were use-phase operations at the power plant. Emphasis, instead, was placed on the process of integrating LCA with other sustainability assessment tools. Typical chemical proportions used in industry were assumed for each treatment cycle, as described in Supplementary Table D.2.1. Most of the basic chemical data were collected from the Ecoinvent version 2.0 database (Frischknecht, Jungbluth et al., 2007), except for chlorine (Cl), which was taken from BUWAL 250 data (Spriensma, 2004). Monochloramine (NH_2Cl) was evaluated from a combination of compounds including hypochlorite, ammonia, and sodium hydroxide. These data came from the Ecoinvent version 2.0, Industry data 2.0 (Plastics Europe), and US LCI databases (Pré Consultants, 2004), respectively.

Table D.2.1. Chemical treatment concentrations used in LCA of recycling treated municipal wastewater as cooling makeup water in power plants.

Scenario	Compound	LCI Database	Dose (mg per 1,000 gallons)	References
1ppm Chlorine (Cl_2)	Chorine (Cl_2), pure	BUWAL 250	3,785	[1, 2]
	Sulfuric acid (H_2SO_4), liquid	Ecoinvent 2.0	473,125	125 average ppm assumption [3]
	Sodium tripolyphosphate ($Na_5P_3O_{10}$ or STPP)	Ecoinvent 2.0	18,925	5 ppm assumption [4]
1 ppm Chlorine dioxide (ClO_2)	Chlorine dioxide (ClO_2)	Ecoinvent 2.0	3,785	[1]
	Sulfuric acid (H_2SO_4), liquid	Ecoinvent 2.0	473,125	125 average ppm assumption [3]
	Sodium tripolyphosphate ($Na_5P_3O_{10}$ or STPP)	Ecoinvent 2.0	18,925	5 ppm assumption [4]
3 ppm Monochloramine (NH_2Cl)	Sodium hypochlorite (NaClO), 15% in H_2O	Ecoinvent 2.0	9,063	[5] supplemented with experimental data
	Ammonia (NH_3)	Industry data 2.0	2,266	[5] supplemented with experimental data
	Sodium hydroxide (NaOH)	USLCI	26.8	[5] supplemented with experimental data
	Sulfuric acid (H_2SO_4), liquid	Ecoinvent 2.0	473,125	125 average ppm assumption [3]
	Sodium tripolyphosphate ($Na_5P_3O_{10}$ or STPP)	Ecoinvent 2.0	18,925	5 ppm assumption [4]

[1] Frayne, C. (1999). *Cooling water treatment: Principles and practice*. Chemical Publishing: New York.
[2] Harston, J.D. and Ropital, F. (2004). *A working party report on control of corrosion in cooling waters.* EFC 40. Maney Publishing.
[3] Li, H. (Personal communication). 17 March 2011.
[4] Li, H. (Personal communication). 16 March 2010.
[5] Vidic, R.D., Dzombak, D.A., Hsieh, M.-K., Li, H., Chien, S.-H., Feng, Y., Chowdhury, I. Monnell, J.D. (2009). Reuse of treated internal or external wastewaters in the cooling systems of coal-based thermoelectric power plants: Final technical report.

The LCA for Challenge B—wastewater delivery—included the manufacture of pipes for conveyance between the wastewater treatment plant and the power plant. Figure D.2.1 depicts the flow and system boundaries for the pipes used in this study. These boundaries include the extraction of raw materials, the production of PVC, the production of concrete, and pipe manufacture. Installation, operation, and maintenance of the pipes are not included in this LCA. Material calculations are based on an assumed 12-inch nominal size pipe and a 10-mile average distance between the wastewater treatment plant and power plant. LCI data came from the Industry data 2.0 database for PVC and from the Ecoinvent version 2.0 database for concrete.

Survey. Two surveys were developed to obtain data related to Challenges A, B, and C. One survey was tailored to U.S. power plants and the other survey was tailored to U.S. municipal wastewater treatment plants. Both surveys were administered in 2009 via telephone and e-mail, depending on responder preference, to targeted plant personnel with chemical treatment experience and other experiences pertinent to the survey. The survey questions were designed to match the type of plant (power or wastewater), but the questions were basically equivalent. All of the surveyed plants had been identified as those currently producing or using treated municipal wastewater for cooling system makeup water. Their responses helped to identify cooling water management practices and contributed to the data used in the life cycle inventory. Challenge A questions pertained to wastewater treatment conducted at both plants; Challenge B questions pertained to wastewater delivery distances and methods; Challenge C questions pertained to public awareness, public concerns voiced, and community outreach; and Challenge D questions pertained to the price paid for treated wastewater. Out of the 18 wastewater treatment plants and 32 power plants surveyed, staff at 4 wastewater treatment plants and 6 power plants completed the survey, representing a 22% and 19% response rate, respectively.

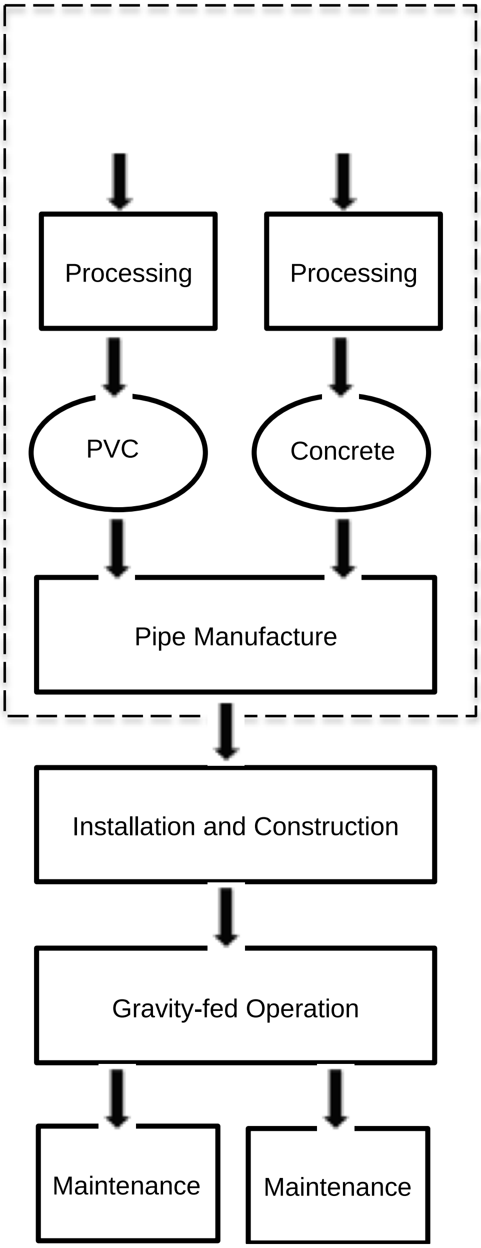


Figure D.2.1 Flow chart of LCA of pipes used for transport of treated municipal wastewater to power plant cooling towers. System boundaries marked with dashed line.

D.3 Results and Discussion

Data from the survey and LCA were analyzed to assess their potential application to sustainability tools for evaluating wastewater reuse practices in power plant cooling systems. Results from the surveys were used to define specific parameters in the LCA. The results are presented based on the four challenges.

Challenge A considers the chemical treatment required to control biofouling, corrosion, and scaling. All three treatment issues inevitably arise when working with reclaimed wastewater and can be effectively addressed through physical, biological, and chemical means.

Both the wastewater treatment plant and the power plant have distinct roles to play in terms of system quality. The wastewater treatment plant conducts secondary and sometimes tertiary treatment, while each power plant conducts additional treatment to address issues more specific to its unique operational equipment and cooling water chemistry. For the wastewater treatment plants, secondary treatment could include the removal of organic matter, nutrients, suspended solids, and pathogenic and biofouling-initiating microbes (Metcalf and Eddy, 2003). For the power plants, additional treatment varies, but usually involves some amount of chlorine added for the control of pathogenic and biofouling microorganisms in the incoming water.

For most of the wastewater treatment plants surveyed, the only tertiary treatment provided was biocide (chlorine) addition as shown in Figure D.3.1. In most cases, the surveyed power plants received the same secondary treated effluent that would otherwise be discharged; thus the burden for additional treatment, in practice, falls to the power plant. The survey responses indicated that reverse osmosis was the most common method of tertiary treatment for the power plants, though other methods were used. Almost all of the power plants surveyed conducted additional treatment of the water with chlorine prior to use in the cooling towers.

The data received for anti-scalants and anti-corrosives were less conclusive than the biocide data, as many plants did not supply information for the former two categories. Out of the responses received, two plants reported using organic polymer for anti-scalant purposes and one plant reported using a phosphorous-based compound for anti-corrosion purposes. Lack of positive responses regarding anti-corrosives and anti-scalants could be due to confidential operating practices or could indicate that these plants are not currently using anti-scalants and anti-corrosives in their systems. Respondents were given the option of entering “None” on the survey to indicate no use of anti-scalants or anti-corrosives. Given the prevalence of scaling and corrosion in power plant systems, it is likely that the lack of positive responses is a result of proprietary operations.

LCA was used to quantify the environmental impacts of the chemicals used to treat biofouling, scaling, and corrosion. The system boundaries focus on the production of the chemicals and extend from the raw material extraction phase to the storage of the finished product. Figure D.3.2 shows impact assessment results for three chemical treatment scenarios, where the chlorine compound treatment is varied within each scenario. The three scenarios are based on three compounds: sulfuric acid (H_2SO_4), sodium tripolyphosphate (STPP), and a chlorine-based compound (Cl_2, ClO_2, or NH_2Cl). H_2SO4 is commonly used to address scaling issues, STPP for corrosion, and chlorine-based compounds for biofouling and general disinfection. The concentrations of H_2SO_4 and STPP were held constant for the comparison while the concentrations of the chlorine-based compounds were varied according to previous research (Chapters 3, 4 and 5). The dosages for each chemical scenario are detailed in Supplementary Table D.2.1.

According to the LCA results presented in Figure D.3.2, no single treatment scenario consistently exhibits the lowest or highest environmental impact in any environmental impact category. There are significant trade-offs, however. For example, while ClO_2 may exhibit, on average, half the impact of Cl_2 with regard to fossil fuel usage and climate change, Cl_2 exhibits a 25 percent greater relative impact than ClO_2 in land use and a 28 percent greater relative impact in carcinogenic potential.

Different impact assessment methods have different emphases and define their categories slightly differently. Eco-indicator 99 defines fossil fuel as oil, gas, or coal. Conventional and unconventional production methods are considered for oil and gas, whereas coal (conventional) for Eco-indicator includes open-pit mining for hard coal or lignite, as well as underground mining. The fossil fuel impact in this study is largely represented by coal and natural gas use at the typical chemical production plant and contributes to climate change via carbon dioxide emissions at the plants. Regarding land use, transformation of fallow land to serve mineral extraction purposes is the primary source of the land use impacts. Monochloramine, when used in place of chlorine dioxide or pure chlorine, contributes an average of 10% additional impact to the categories of land use, carcinogenic potential, and acidification and eutrophication. All three chlorine-based scenarios contribute over 50% to the respiratory organics impact category, largely due to non-methane VOCs and hydrocarbons.

STPP contributes over 60% to the impact categories of total land use and carcinogenic potential. The greatest sulfuric acid impact occurs with respect to the categories of ecotoxicity and acidification and eutrophication. Many different metals used in the production equipment contribute to ecotoxicity, especially chromium and nickel (Wallinder et al., 2006). The acidification and eutrophication impacts can be traced back to emission of sulfur dioxide and nitrogen oxides released in the production phase of STPP.

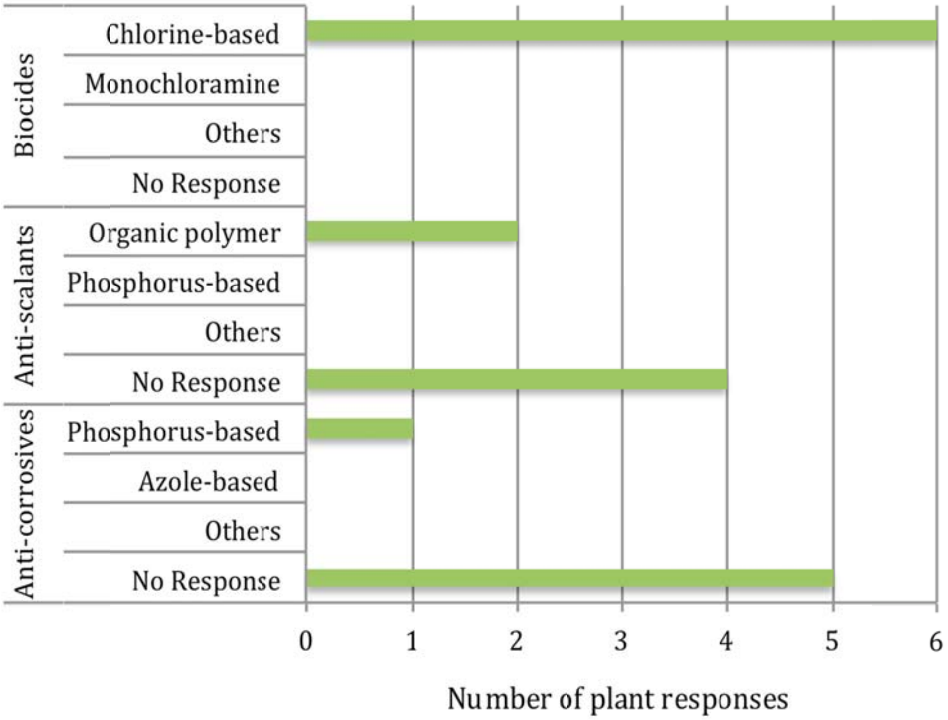


Figure D.3.1 Chemicals used by power plant survey respondents to control biofouling, scaling, and corrosion when reusing treated municipal wastewater for cooling makeup water.

Since this study only considered the raw material extraction and production phases of the chemicals in an effort to show how LCA could be integrated with other sustainability tools, a comprehensive LCA with additional chemical scenarios and risk of leakage would need to be completed to evaluate more comprehensively the system's impacts. Based on the comparative LCA results shown in Figure D.3.2, however, some alternatives for trade-offs can be identified to inform future decisions, especially regarding the use of STPP. To minimize land use or carcinogenic potential, one might look for an alternative to STPP since it contributes the most to these categories. Replacing STPP, however, would involve a trade-off for the category of acidification and eutrophication potential, where impacts from sulfuric acid are nearly as great as those from STPP. Therefore, for the exclusive case of acidification and eutrophication, finding an alternative to sulfuric acid might be more beneficial.

It is important to note that the results in Figure D.3.2 are based on percentage compositions for three sample treatment scenarios; there are potentially many other chemicals and compositions used in cooling water treatment. A plant-specific LCA can help select which chemicals are most appropriate for a particular system. However, LCA should not be mistaken for a full sustainability analysis. Complementary sustainability assessment tools, such as laboratory testing, life-cycle costing, and treatment-effectiveness surveys should be employed in addition to LCA for a more complete analysis.

Challenge B, wastewater delivery, can be assessed using survey results to inform the LCA. Eighty percent of respondents reported plant-to-plant distances of five or fewer miles. Only two plants reported distances greater than five miles: 7 and 18 miles, respectively. Chien et al. (Chien, 2008) calculated an average maximum plant-to-plant distance of 10 miles for 81 percent of proposed U.S. power plants and of 25 miles for 97 percent of proposed U.S. power plants (Chien, 2008). Of existing power plants, approximately 50 percent could use cooling water from wastewater treatment plants within 10 miles and approximately 76 percent could use cooling water within 25 miles (Li et al., 2011). As the distance between plants increases, factors such as piping costs and environmental impacts become more important to consider.

LCA can be used to determine the environmental impacts of materials, construction and maintenance of the piping system. Figure D.3.3 shows a sample comparison of environmental impacts resulting from the production of two types of typical pipe material—PVC (an example of a pressure piping material) and concrete (for conveyance by gravity flow, not under pressure) was conducted —utilizing the average distance of ten miles between plants, which takes both the literature and survey results into consideration. As evident from Figure D.3.3, PVC results in more environmental impacts than concrete in each of the categories shown, particularly with

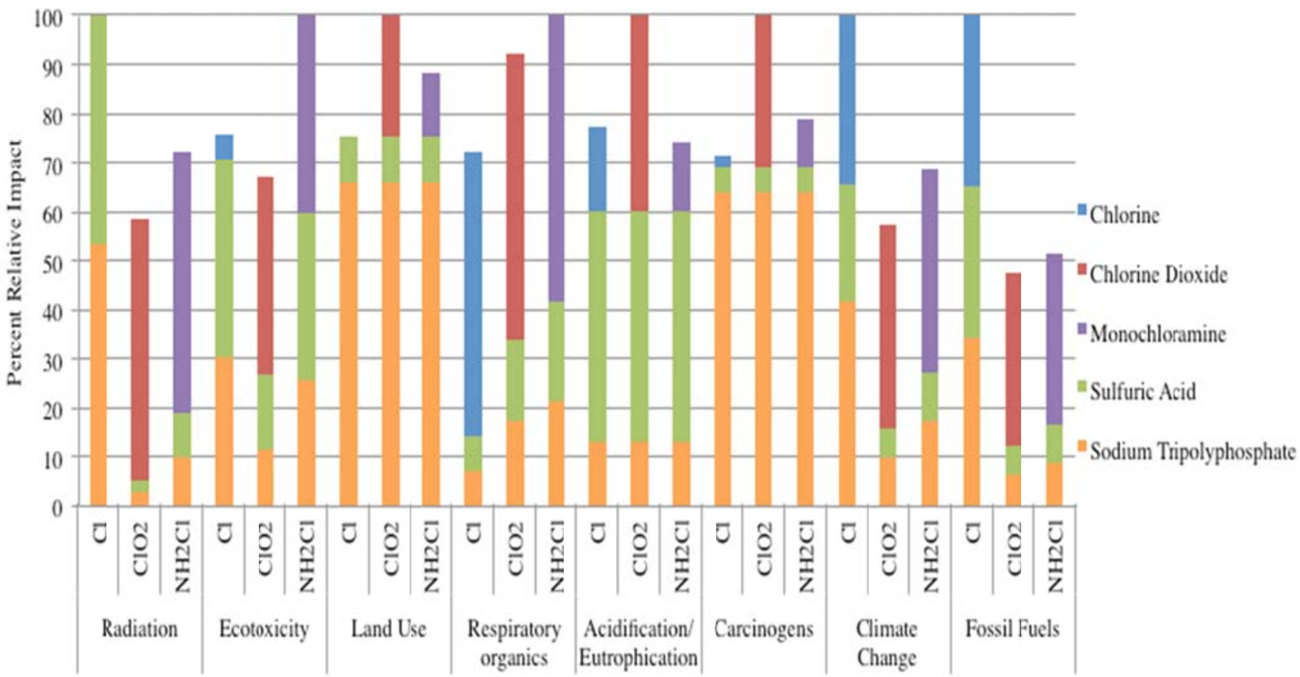


Figure D.3.2 Environmental impacts resulting from raw material extraction and chemical production for three treatment scenarios that address biofouling, corrosion, and scaling.
Note:Results normalized to the highest impact in each category. Cl_2 = chlorine scenario (Cl_2, H_2SO_4, $Na_5P_3O_{10}$), ClO_2 = chlorine dioxide scenario (ClO_2, H_2SO_4, $Na_5P_3O_{10}$), NH_2Cl = monochloramine scenario (NH_2Cl, H_2SO_4, $Na_5P_3O_{10}$).

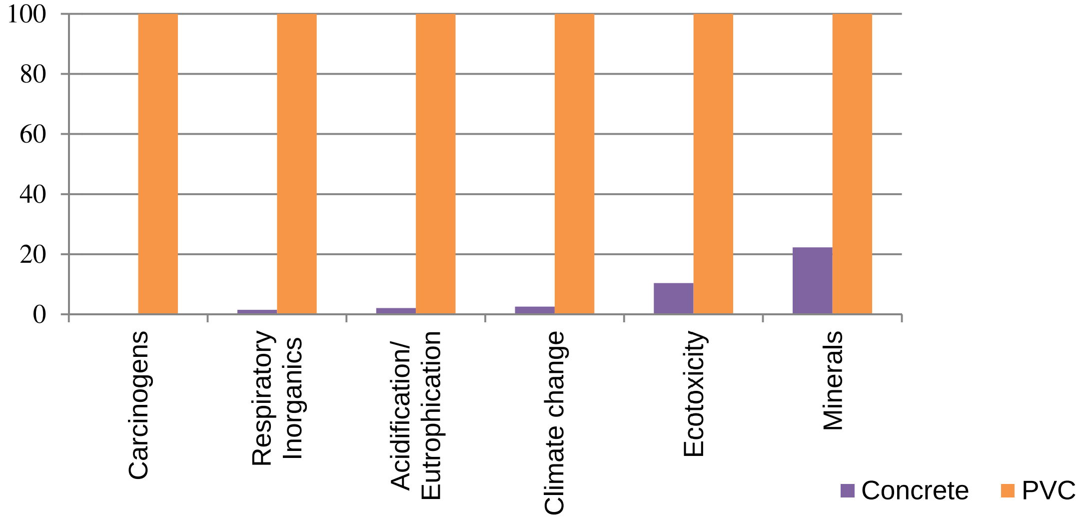


Figure D.3.3 Environmental impacts from production of PVC and concrete piping materials.
Note: Results normalized to PVC.

respect to carcinogens. The carcinogenic human health impacts of emissions during PVC production are widely recognized and apparent in the LCA results. The environmental impacts of PVC in this scenario range three times to 52 times (for respiratory organics) higher compared to the environmental impacts of concrete.

The evaluation of Challenge B does not merely consist of materials selection and distances of piping; factors related to the cost of the materials, the lifetime of the materials, the installation of the system, the maintenance of the piping system, and any necessary energy required for moving the water all contribute to the environmental impacts of wastewater delivery, as well as to the feasibility of wastewater delivery. LCA can be used to analyze the environmental impacts of materials selection, as shown in Figure D.3.3 as well as installation, maintenance, and energy requirements. Maintenance could vary depending on pipe materials, intensity of use, climatic conditions, and many other factors. Energy and emissions from any pumping required could also vary based on pipe diameters, flow rates, and differences in site elevations. Thus, data related to the expected lifetime of the system, the anticipated maintenance of such a system, and the need for pumping of water must be obtained for the specific systems and incorporated into the LCA. Obtaining the land use rights to install piping in a particular location can also be challenging and costly. If permitting becomes an issue, greater distances for trucking or shipping water may be considered. Life cycle costing (LCC) can be used to assess the costs of the various piping and transport scenarios.

Challenge C—public perception—is one challenge that LCA does not adequately address. Therefore, a survey was used to gather information on experiences of power plants and wastewater treatment plants related to public perception and outreach activities with respect to utilizing wastewater for power plant cooling.

Personnel at both the wastewater treatment plants and power plants were asked if the local public was aware that treated wastewater was being sent to or used in the power plant. Some plants allow the public to tour their facilities and other plants inform the public that they are using wastewater to avoid the use of groundwater. Seventy percent of survey respondents, however, reported they were not aware of any outreach efforts; fifty percent reported that the public was not aware of the wastewater reclamation process. As described earlier, low public awareness of the processes and technologies used in wastewater reclamation may lead to negative public perception of both the wastewater treatment plants and the power plants engaged in reclamation activities. Personnel at both plants were also asked if the plants hosted any educational outreach events for the community.

Further research is needed to determine why community engagement is so low and what can be done to improve public awareness and acceptance. Public perception can also be addressed via other tools and metrics as described in Table D.1.1, such as surveys directed at the public.

Challenge D—valuation of reclaimed water—is also inadequately addressed by traditional LCA tools, which have yet to even fully address freshwater consumption (Pfister al., 2009). While current LCIA methods include water quality categories such as eutrophication and ecotoxicity, water withdrawal and consumption categories are largely missing (Goedkoop et al., 2009). Although an LCC code of practice is being developed, the incorporation of water use into LCC still remains to be fully examined as well (Swarr et al., 2011). The major issue to be examined is that the price paid for water does not reflect the actual costs incurred in reusing treated municipal wastewater, nor does it reflect the economic and environmental savings gained from reusing water that would have otherwise been immediately released to the natural waterways.

Power plant personnel were surveyed regarding the amount paid to wastewater treatment plants for reuse of wastewater in the power plant cooling system. Seventy percent of these respondents reported payments below $0.66 per 1,000 gallons (in 2009 USD). For comparison, drinking water in the U.S. typically costs slightly over $2 per 1,000 gallons with treatment accounting for approximately 15 percent of the cost (USEPA, 2004). It is interesting to note that one of the wastewater treatment plants did not charge the power plant anything for its wastewater. Additionally, one respondent reported a dynamic costing scenario. In this case, the power plant paid the wastewater treatment plant a percentage of the cost for potable water, which can vary with broader economic fluctuations, maintenance costs, and any natural events, such as severe storms that require additional treatment of the water.

The price paid for the wastewater may include the services provided to treat it to the secondary or tertiary level, cost of piping, environmental regulation permits, and any internal administrative costs; it does not include the cost of additional chemicals because those are usually handled by the power plants, as reported in Figure D.2.1. When using traditional costing methods such as LCC, wastewater reuse may seem to be more expensive than freshwater withdrawal, since the cost to the environment is not included in LCCs.

Interestingly, the primary element that makes treated wastewater reuse in power plant cooling systems seem sustainable, which is the reuse of wastewater, is not adequately captured by existing sustainability tools. LCA does not distinguish between withdrawal of freshwater versus other sources, LCC does not place any particular monetary value on freshwater versus wastewater, and the prices paid for freshwater versus wastewater would lead the user to choose low-cost freshwater over wastewater.

Sustainability tools and metrics used in addition to LCA and environmental impacts can be used as complementary approaches. As shown in Table D.1.1, additional sustainability metrics include water volume, cost, and value. Water volume here does not refer to a static quantity, but rather the metric used for an input-output water balance conducted over time to identify changes in freshwater levels and potential process improvements. Other possible sustainability metrics for capturing the value of water include water quality and regional ecosystem quality, both of which can be measured using traditional laboratory methods and environmental risk assessment, in addition to LCA.

D.4 Conclusions

Previous research and industry experience has shown that treated municipal wastewater can be used as makeup water for power plant recirculating cooling systems. With the feasibility established, it is useful to evaluate and optimize the overall sustainability of the system. However, one of the main tools used to evaluate sustainability, life cycle assessment, does not currently address the full complexity of wastewater reuse. Therefore, in addition to traditional LCA methods, other tools and metrics must be considered to analyze the sustainability of such a system.

The study employed a survey of personnel at power plants and wastewater treatment plants engaged in cooperative relationships of using the treated wastewater effluent for power plant cooling to inform the LCA and to investigate the value of avoiding freshwater withdrawal for power plant cooling. While the survey intentionally addressed issues such as chemical treatment and public perceptions, which are not always easily captured through traditional research, the responses also highlighted important issues from the perspectives of the plant personnel.

The survey of power plants and wastewater treatment plants served as a source of inventory data used within the LCA. Survey results regarding biofouling, corrosion, and scaling showed that wastewater treatment plants generally do not include any special treatment for effluent transported to the power plants, with the exception of additional chlorination. The survey also revealed that power plants are paying relatively low prices for treated wastewater. Other results showed that most plants surveyed were located within five miles of one another. Finally, few plants have conducted community outreach activities; more research should be conducted on community understanding and concerns.

Analysis of the sustainability aspects of the four challenges to wastewater reuse in power plant cooling systems revealed that the sustainability of alternative approaches related to Challenges A and B can be partially assessed using available sustainability tools and metrics, but the available tools and metrics are inadequate to address Challenges C and D. The primary element that makes treated wastewater reuse in power plant cooling systems seem sustainable, which is the reuse of wastewater, is not adequately captured by existing sustainability tools. New tools and metrics are needed to assess the sustainability of complex water-energy systems like that investigated here.

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APPENDIX E

LC^3 Model User Manual and LCI Emissions

E.1 Model Description

Life-cycle conceptual costing (LC^3) model is a public domain excel-based life cycle conceptual cost estimation model designed for a few conventional tertiary treatment units. The tertiary treatment units were selected based on the contaminants to be removed for reuse of treated water in power plant recirculating cooling systems. The individual treatment units can be combined to form tertiary treatment alternatives based on the target effluent quality desired. The purpose of the tool is to aid decision makers to optimize the level of tertiary treatment based on first-stage cost estimate comparison.

Appendix E provides the procedure manual for the LC^3 model used to estimate costs for constructing and operating an advanced treatment process used to treat the secondary treated municipal wastewater, which will be used as cooling water make-up. The design procedures were put together by following the principles and examples illustrated in textbooks (Metcalf & Eddy, et al., 2003; Asano, et al., 2007; Rittmann, et al., 2001), USEPA (U.S.EPA, 1993) and WEF (WEF, 2010) design manuals. The cost estimation procedure was developed using the principles of conceptual (first-stage) estimates as defined by Association of Advancement for Cost Engineers (AACE) (Westney, 1997) and using standard cost databases referenced to 2009. Estimates calculated using standard databases and conceptual cost estimation methods were applied to compare alternative treatment processes and treatment alternatives required to maintain heat exchange efficiency in the cooling system within desired bounds.

This manual explains the organization of the model and the description of each sheet in the Excel worksheet used to implement the model. The tertiary treatment processes included in this model are suspended growth nitrification in a well mixed reactor (N), lime softening using solids contact clarifier (S), pH adjustment using an acidification unit (A), mono-media sand filtration (F), granular activated carbon adsorption (G) and chemical addition unit (C). Piping or water supply system (WS) designed to deliver treated water from a public owned treatment works (POTW) to point of reuse, and river water withdrawal (RW) cost is also considered. Figure E.1.1 shows the algorithm adopted in this model.

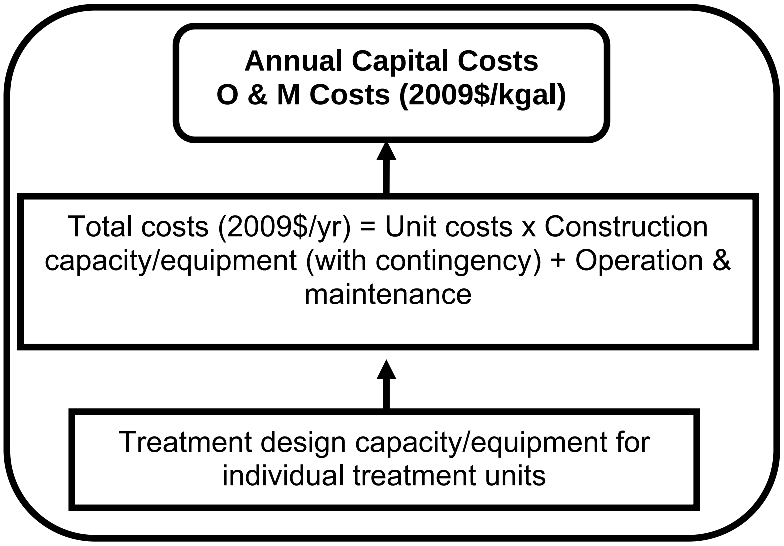


Figure E.1.1 Algorithm of developed life cycle conceptual costing (LC^3) model

E.2 Model Worksheets

The excel worksheet has each tab named with reference to the content of the sheet:

Input Sheet – All treatment process input variables for design and cost estimation are placed in this worksheet and it is in this sheet that the user defines the default values for design parameters, water quality and costs. See Figure E.2.1 and Table E.2.1 which provide descriptions of the various sections of the input sheet.

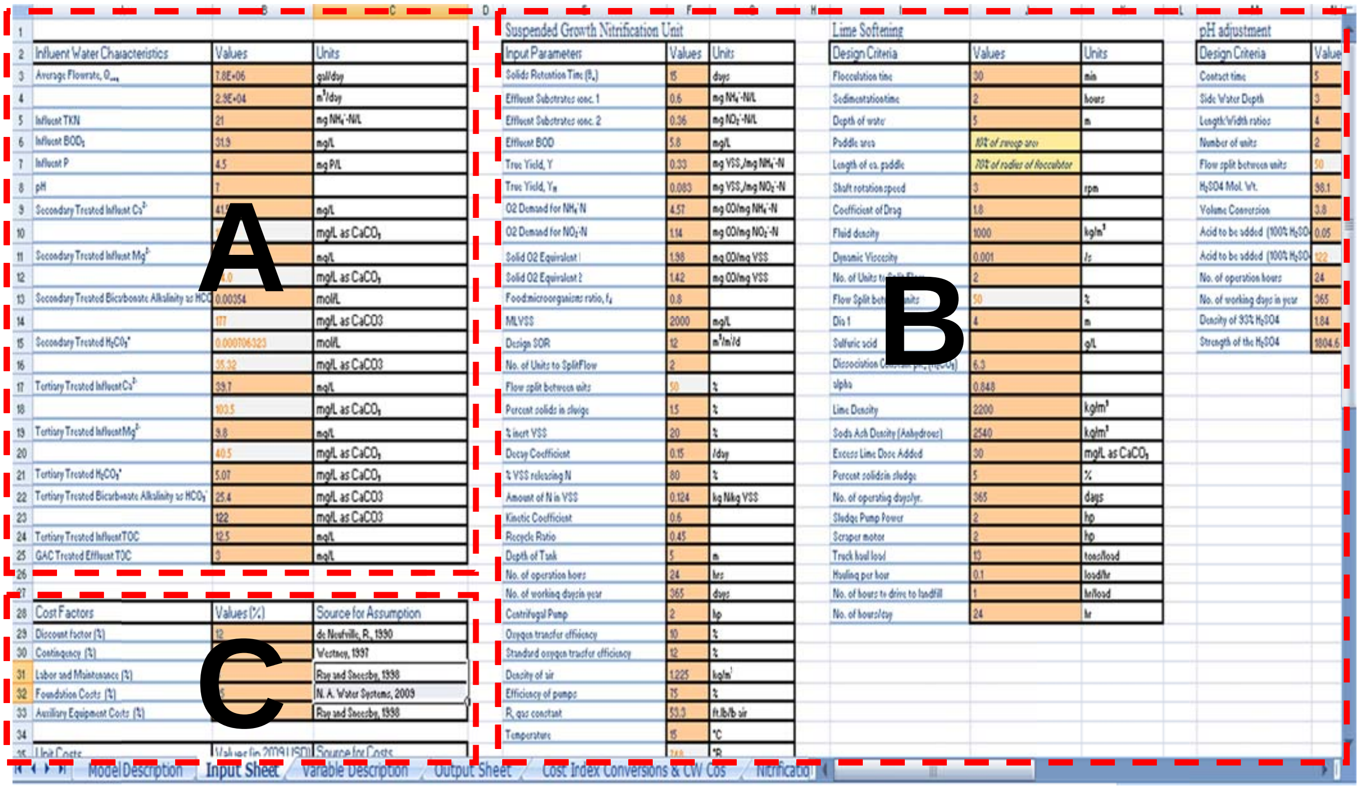
Variable Description - Variables listed or used in all the worksheets given in the model are described with calculation notes; their default values, the range in which they can be varied in their respective worksheets named, and references for the values set are given. Figure E.2.2 is a snapshot of the variable description sheet.

Output Sheet – The end results of cost estimation for individual and combined treatment alternatives in terms of 2009 USD, MM 2009 USD/yr and 2009 USD/kgal, along with the graphical representation of the results can be obtained from this sheet. All outputs are connected to auxiliary input tertiary treatment cells and hence the outputs change automatically when the input is varied. Table E.2.2 describes the different blocks in the output spreadsheet shown in Figure E.2.3.

Cost Index Conversions & CW Cos - Contains cost index values for 2002, 2007 and 2009 and formula cells to help convert annual costs to the respective indexed year. This section also contains CapdetWorks (Hydromantis Inc., 2011) cost estimates for year 2007 for comparison. If the year of reference needs to be varied, then the numbers can be changed to observe variation in output with respect to year of reference.

Nitrification (N) – Consists of design and costing calculations for the suspended growth nitrification unit. The variations were taken from the USEPA (1993) and WEF manuals (2010). For the various sections of the auxiliary worksheet, refer to Figure E.2.4 and Table E.2.3 that describes the different sections on Figure E.2.4.

Softening (S) - Consists of design and costing calculations for the solids contact clarifier. The description of sections of the 'Softening' worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.



Influent Water Characteristics	Values	Units
Average Flowrate, Q_{avg}	7.8E+06	gal/day
	2.9E+04	m^3/day
Influent TKN	21	mg NH_4^+-N/L
Influent BOD_5	31.9	mg/L
Influent P	4.5	mg P/L
pH	7	
Secondary Treated Influent Ca^{2+}	41.[illegible]	mg/L
	1[illegible]	mg/L as $CaCO_3$
Secondary Treated Influent Mg^{2+}	[illegible]	mg/L
	[illegible]4.0	mg/L as $CaCO_3$
Secondary Treated Bicarbonate Alkalinity as HCO	0.00354	mol/L
	177	mg/L as CaCO3
Secondary Treated $H_2CO_3^*$	0.000706323	mol/L
	35.32	mg/L as CaCO3
Tertiary Treated Influent Ca^{2+}	39.7	mg/L
	103.5	mg/L as $CaCO_3$
Tertiary Treated Influent Mg^{2+}	9.8	mg/L
	40.5	mg/L as $CaCO_3$
Tertiary Treated $H_2CO_3^*$	5.07	mg/L as $CaCO_3$
Tertiary Treated Bicarbonate Alkalinity as HCO_3^-	25.4	mg/L as CaCO3
	122	mg/L as CaCO3
Tertiary Treated Influent TOC	12.5	mg/L
GAC Treated Effluent TOC	3	mg/L

Cost Factors	Values (%)	Source for Assumption
Discount factor (%)	12	de Neufville, R., 1990
Contingency (%)	[illegible]	Westney, 1997
Labor and Maintenance (%)	[illegible]	Ray and Sneesby, 1998
Foundation Costs (%)	5	N. A. Water Systems, 2009
Auxiliary Equipment Costs (%)	[illegible]	Ray and Sneesby, 1998

Unit Costs	Values (in 2009 USD)	Source for Costs

Suspended Growth Nitrification Unit

Input Parameters	Values	Units
Solids Retention Time ($θ_c$)	15	days
Effluent Substrates conc. 1	0.6	mg NH_4^+-N/L
Effluent Substrates conc. 2	0.36	mg NO_2^--N/L
Effluent BOD	5.8	mg/L
True Yield, Y	0.33	mg VSS,/mg NH_4^+-N
True Yield, Y_N	0.083	mg VSS,/mg NO_2^--N
O2 Demand for NH_4^+N	4.57	mg O2/mg NH_4^+-N
O2 Demand for NO_2^-N	1.14	mg O2/mg NO_2^--N
Solid O2 Equivalent 1	1.98	mg O2/mg VSS
Solid O2 Equivalent 2	1.42	mg O2/mg VSS
Food:microorganisms ratio, f_d	0.8	
MLVSS	2000	mg/L
Design SOR	12	$m^3/m^2/d$
No. of Units to SplitFlow	2	
Flow split between units	50	%
Percent solids in sludge	1.5	%
% inert VSS	20	%
Decay Coefficient	0.15	/day
% VSS releasing N	80	%
Amount of N in VSS	0.124	kg N/kg VSS
Kinetic Coefficient	0.6	
Recycle Ratio	0.45	
Depth of Tank	5	m
No. of operation hours	24	hrs
No. of working days in year	365	days
Centrifugal Pump	2	hp
Oxygen transfer efficiency	10	%
Standard oxygen transfer efficiency	12	%
Density of air	1.225	kg/m^3
Efficiency of pumps	75	%
R, gas constant	53.3	ft.lb/lb air
Temperature	15	°C
		°R

Lime Softening

Design Criteria	Values	Units
Flocculation time	30	min
Sedimentationtime	2	hours
Depth of water	5	m
Paddle area	*10% of swoop area*	
Length of ea. paddle	*70% of radius of flocculator*	
Shaft rotation speed	3	rpm
Coefficient of Drag	1.8	
Fluid density	1000	kg/m^3
Dynamic Viscosity	0.001	/s
No. of Units to Split Flow	2	
Flow Split between units	50	%
Dia 1	4	m
Sulfuric acid		g/L
Dissociation Constant pK_a (H_2CO_3)	6.3	
alpha	0.848	
Lime Density	2200	kg/m^3
Soda Ash Density (Anhydrous)	2540	kg/m^3
Excess Lime Dose Added	30	mg/L as $CaCO_3$
Percent solidsin sludge	5	%
No. of operating days/yr.	365	days
Sludge Pump Power	2	hp
Scraper motor	2	hp
Truck haul load	13	tons/load
Hauling per hour	0.1	load/hr
No. of hours to drive to landfill	1	hr/load
No. of hours/day	24	hr

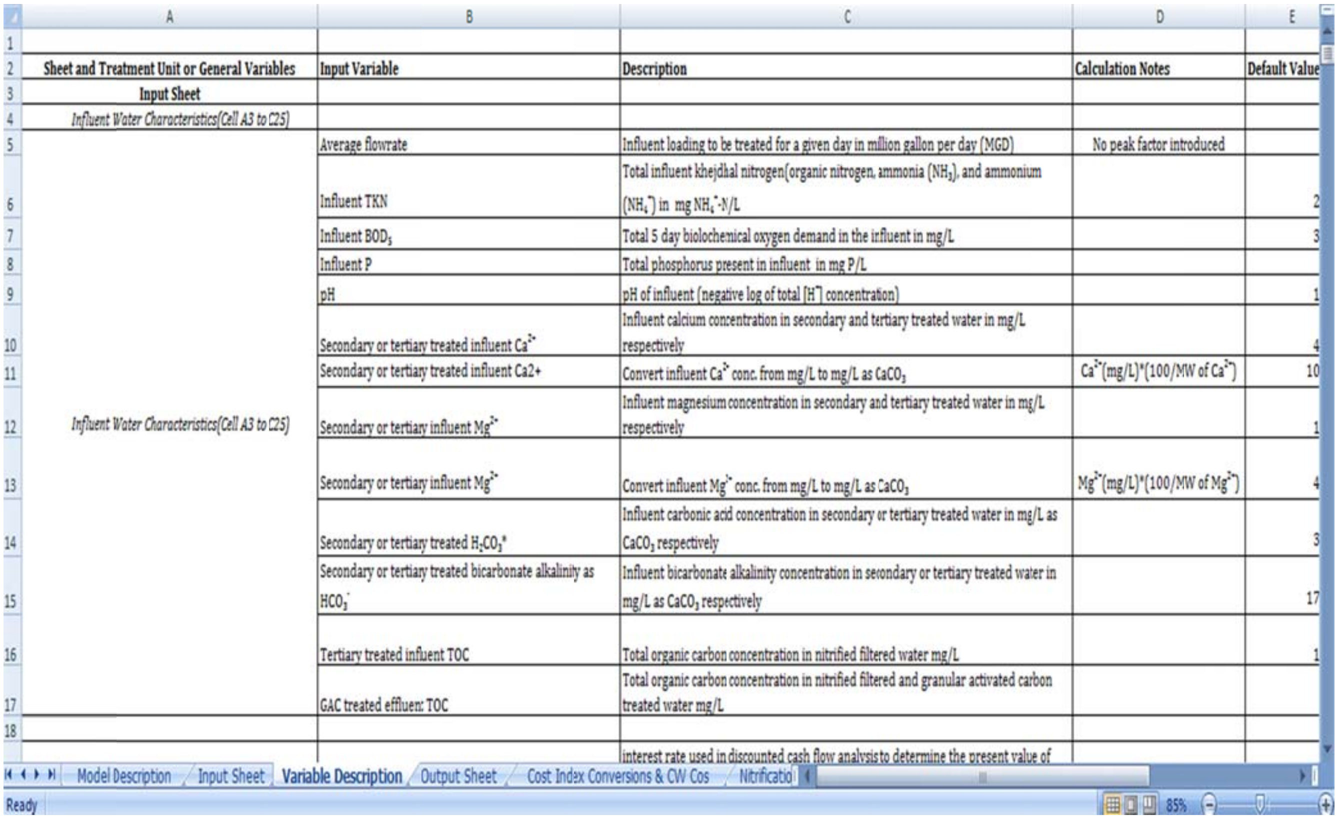
pH adjustment

Design Criteria	Value
Contact time	5
Side Water Depth	3
Length/Width ratios	4
Number of units	2
Flow split between units	50
H_2SO4 Mol. Wt.	98.1
Volume Conversion	3.8
Acid to be added (100% H_2SO4	0.05
Acid to be added (100% H_2SO4	122
No. of operation hours	24
No. of working days in year	365
Density of 93% H_2SO4	1.84
Strength of the H_2SO4	1804.6

Figure E.2.1 Input sheet of the LC[3] model as seen in the excel file

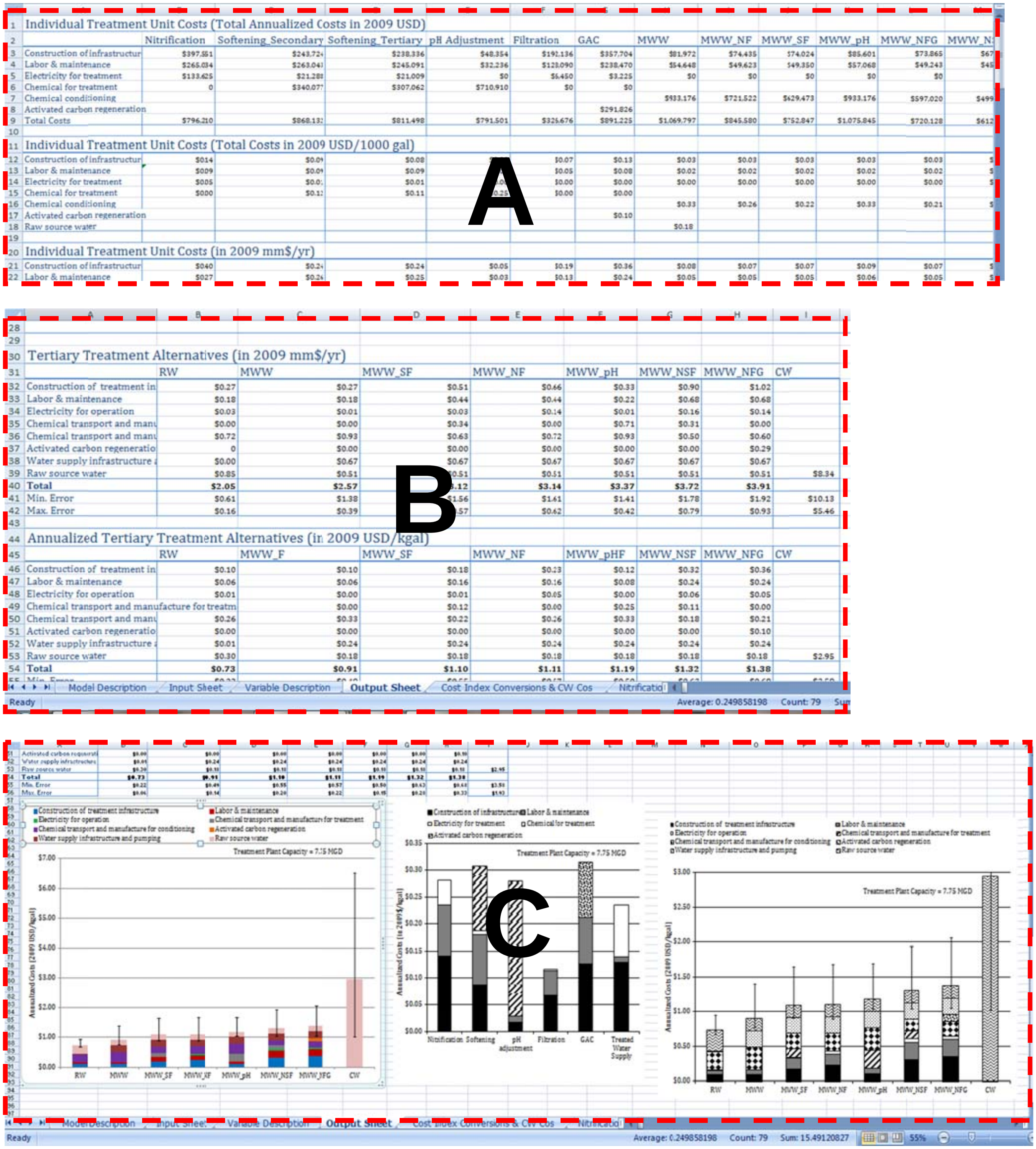
Table E.2.1 Description of the different blocks in the input sheet as shown in Figure E.2.1

Ref #	Section	Description
A	Water Quality Specification	This block includes water quality inputs such as constituent species concentration, pH and BOD. Water quality information is used to calculated and size tertiary treatment units.
B	Tertiary Treatment Inputs	This collection of input blocks provides a number of detailed design parameters for each tertiary treatment unit. Treatment sub-modules and costing will update dynamically as these fields are changed.
C	Cost Factors	A number of cost factors and pricing parameters are collected in this input block. This includes general items such as discount factor and contingency factor, in addition to a number of more specific pricing parameters such as chemical and equipment prices. Note that the cost factors currently built into the model relate to 2009 USD.



	A	B	C	D	E
1					
2	**Sheet and Treatment Unit or General Variables**	**Input Variable**	**Description**	**Calculation Notes**	**Default Value**
3	**Input Sheet**				
4	*Influent Water Characteristics(Cell A3 to C25)*				
5	*Influent Water Characteristics(Cell A3 to C25)*	Average flowrate	Influent loading to be treated for a given day in million gallon per day (MGD)	No peak factor introduced	
6		Influent TKN	Total influent khejdhal nitrogen(organic nitrogen, ammonia (NH_3), and ammonium (NH_4^+) in mg NH_4^+-N/L		2
7		Influent BOD_5	Total 5 day biolochemical oxygen demand in the irfluent in mg/L		3
8		Influent P	Total phosphorus present in influent in mg P/L		
9		pH	pH of influent (negative log of total [H^+] concentration)		1
10		Secondary or tertiary treated influent Ca^{2+}	Influent calcium concentration in secondary and tertiary treated water in mg/L respectively		4
11		Secondary or tertiary treated influent Ca2+	Convert influent Ca^{2+} conc. from mg/L to mg/L as $CaCO_3$	Ca^{2+}(mg/L)*(100/MW of Ca^{2+})	10
12		Secondary or tertiary influent Mg^{2+}	Influent magnesium concentration in secondary and tertiary treated water in mg/L respectively		1
13		Secondary or tertiary influent Mg^{2+}	Convert influent Mg^{2+} conc. from mg/L to mg/L as $CaCO_3$	Mg^{2+}(mg/L)*(100/MW of Mg^{2+})	4
14		Secondary or tertiary treated H_2CO_3*	Influent carbonic acid concentration in secondary or tertiary treated water in mg/L as $CaCO_3$ respectively		3
15		Secondary or tertiary treated bicarbonate alkalinity as HCO_3^-	Influent bicarbonate alkalinity concentration in secondary or tertiary treated water in mg/L as $CaCO_3$ respectively		17
16		Tertiary treated influent TOC	Total organic carbon concentration in nitrified filtered water mg/L		1
17		GAC treated effluen: TOC	Total organic carbon concentration in nitrified filtered and granular activated carbon treated water mg/L		
18					
			interest rate used in discounted cash flow analysis to determine the present value of		

Figure E.2.2 Variable description sheet of the LC^3 model. On the left hand side, the sheet and cells in which the variable is placed and the variable name is given, in the middle the description for the respective variable is provided and on the right, default values, calculation notes, sources for the values are provided



Individual Treatment Unit Costs (Total Annualized Costs in 2009 USD)

	Nitrification	Softening_Secondary	Softening_Tertiary	pH Adjustment	Filtration	GAC	MWW	MWW_NF	MWW_SF	MWW_pH	MWW_NFG	MWW_N
Construction of infrastructur	$397.551	$243.72	$238.336	$48.354	$192.136	$357.704	$81.972	$74.435	$74.024	$85.601	$73.865	$67
Labor & maintenance	$265.034	$263.04	$245.091	$32.236	$123.090	$238.470	$54.648	$49.623	$49.350	$57.068	$49.243	$45
Electricity for treatment	$133.625	$21.28	$21.009	$0	$5.450	$3.225	$0	$0	$0	$0	$0	
Chemical for treatment	0	$340.07	$307.062	$710.910	$0	$0						
Chemical conditioning							$933.176	$721.522	$629.473	$933.176	$597.020	$499
Activated carbon regeneration						$291.826						
Total Costs	$796.210	$868.13	$811.498	$791.501	$326.676	$891.225	$1,069.797	$845.580	$752.847	$1,075.845	$720.128	$612

Individual Treatment Unit Costs (Total Costs in 2009 USD/1000 gal)

	Nitrification	Softening_Secondary	Softening_Tertiary	pH Adjustment	Filtration	GAC	MWW	MWW_NF	MWW_SF	MWW_pH	MWW_NFG	MWW_N
Construction of infrastructur	$0.14	$0.09	$0.08	[illegible]	$0.07	$0.13	$0.03	$0.03	$0.03	$0.03	$0.03	$
Labor & maintenance	$0.09	$0.09	$0.09	[illegible]	$0.05	$0.08	$0.02	$0.02	$0.02	$0.02	$0.02	$
Electricity for treatment	$0.05	$0.0	$0.01	[illegible]	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$
Chemical for treatment	$0.00	$0.1	$0.11	0.25	$0.00	$0.00						
Chemical conditioning							$0.33	$0.26	$0.22	$0.33	$0.21	$
Activated carbon regeneration						$0.10						
Raw source water							$0.18					

Individual Treatment Unit Costs (in 2009 mm$/yr)

	Nitrification	Softening_Secondary	Softening_Tertiary	pH Adjustment	Filtration	GAC	MWW	MWW_NF	MWW_SF	MWW_pH	MWW_NFG	MWW_N
Construction of infrastructur	$0.40	$0.2	$0.24	$0.05	$0.19	$0.36	$0.08	$0.07	$0.07	$0.09	$0.07	$
Labor & maintenance	$0.27	$0.2	$0.25	$0.03	$0.13	$0.24	$0.05	$0.05	$0.05	$0.06	$0.05	$

Tertiary Treatment Alternatives (in 2009 mm$/yr)

	RW	MWW	MWW_SF	MWW_NF	MWW_pH	MWW_NSF	MWW_NFG	CW
Construction of treatment in	$0.27	$0.27	$0.51	$0.66	$0.33	$0.90	$1.02	
Labor & maintenance	$0.18	$0.18	$0.44	$0.44	$0.22	$0.68	$0.68	
Electricity for operation	$0.03	$0.01	$0.03	$0.14	$0.01	$0.16	$0.14	
Chemical transport and manu	$0.00	$0.00	$0.34	$0.00	$0.71	$0.31	$0.00	
Chemical transport and manu	$0.72	$0.93	$0.63	$0.72	$0.93	$0.50	$0.60	
Activated carbon regeneratio	0	$0.00	$0.00	$0.00	$0.00	$0.00	$0.29	
Water supply infrastructure a	$0.00	$0.67	$0.67	$0.67	$0.67	$0.67	$0.67	
Raw source water	$0.85	$0.51	$0.51	$0.51	$0.51	$0.51	$0.51	$8.34
Total	**$2.05**	**$2.57**	**.12**	**$3.14**	**$3.37**	**$3.72**	**$3.91**	
Min. Error	$0.61	$1.38	$1.56	$1.61	$1.41	$1.78	$1.92	$10.13
Max. Error	$0.16	$0.39	.57	$0.62	$0.42	$0.79	$0.93	$5.46

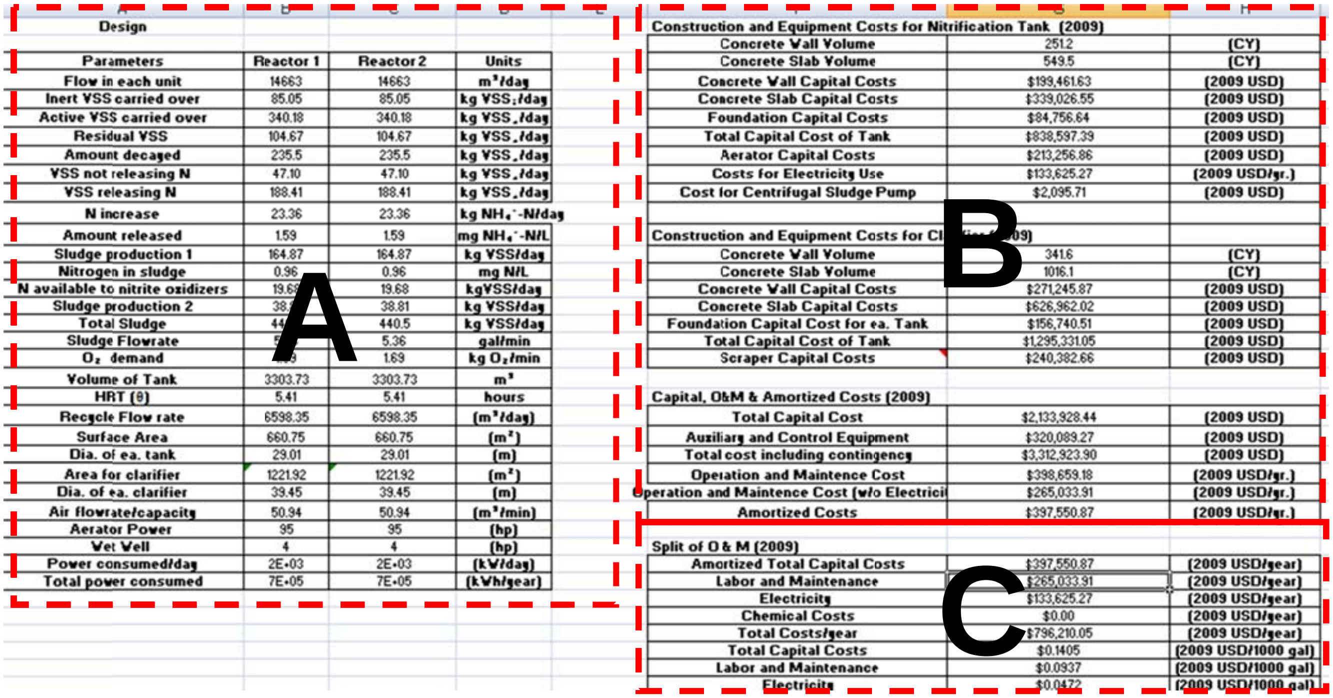
Annualized Tertiary Treatment Alternatives (in 2009 USD/kgal)

	RW	MWW_F	MWW_SF	MWW_NF	MWW_pHF	MWW_NSF	MWW_NFG	CW
Construction of treatment in	$0.10	$0.10	$0.18	$0.23	$0.12	$0.32	$0.36	
Labor & maintenance	$0.06	$0.06	$0.16	$0.16	$0.08	$0.24	$0.24	
Electricity for operation	$0.01	$0.00	$0.01	$0.05	$0.00	$0.06	$0.05	
Chemical transport and manufacture for treatm		$0.00	$0.12	$0.00	$0.25	$0.11	$0.00	
Chemical transport and manu	$0.26	$0.33	$0.22	$0.26	$0.33	$0.18	$0.21	
Activated carbon regeneratio	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.10	
Water supply infrastructure a	$0.01	$0.24	$0.24	$0.24	$0.24	$0.24	$0.24	
Raw source water	$0.30	$0.18	$0.18	$0.18	$0.18	$0.18	$0.18	$2.95
Total	**$0.73**	**$0.91**	**$1.10**	**$1.11**	**$1.19**	**$1.32**	**$1.38**	

Figure E.2.3 Snapshots of the output sheet in the LC^3 model, where, A – individual tertiary treatment costs, B – combined tertiary treatment alternative costs and C – plots of individual and combined costs

Table E.2.2 Description of the different blocks in the output sheet as shown in Figure E.2.3

Ref #	Section	Description
A	Individual tertiary treatment Costs	This block includes total amortized costs for construction and annual O&M of individual tertiary treatment units in 2009$/yr, 2009$/kgal and 2009MM$/yr.
B	Tertiary treatment alternatives costs	This block includes total amortized costs for construction and annual O&M of combined tertiary treatment alternatives in 2009$/kgal and 2009MM$/yr.
C	Output graphs of total direct costs	Plots of total annual direct costs for individual and combined tertiary treatment alternatives in 2009$/kgal are provided.



Design

Parameters	Reactor 1	Reactor 2	Units
Flow in each unit	14663	14663	m^3/day
Inert VSS carried over	85.05	85.05	kg VSS_i/day
Active VSS carried over	340.18	340.18	kg VSS_a/day
Residual VSS	104.67	104.67	kg VSS_a/day
Amount decayed	235.5	235.5	kg VSS_a/day
VSS not releasing N	47.10	47.10	kg VSS_a/day
VSS releasing N	188.41	188.41	kg VSS_a/day
N increase	23.36	23.36	kg NH_4^+-N/day
Amount released	1.59	1.59	mg NH_4^+-N/L
Sludge production 1	164.87	164.87	kg VSS/day
Nitrogen in sludge	0.96	0.96	mg N/L
N available to nitrite oxidizers	[illegible]	19.68	kgVSS/day
Sludge production 2	[illegible]	38.81	kg VSS/day
Total Sludge	[illegible]	440.5	kg VSS/day
Sludge Flowrate	[illegible]	5.36	gal/min
O_2 demand	[illegible]	1.69	kg O_2/min
Volume of Tank	3303.73	3303.73	m^3
HRT (θ)	5.41	5.41	hours
Recycle Flow rate	6598.35	6598.35	(m^3/day)
Surface Area	660.75	660.75	(m^2)
Dia. of ea. tank	29.01	29.01	(m)
Area for clarifier	1221.92	1221.92	(m^2)
Dia. of ea. clarifier	39.45	39.45	(m)
Air flowrate/capacity	50.94	50.94	(m^3/min)
Aerator Power	95	95	(hp)
Wet Well	4	4	(hp)
Power consumed/day	2E+03	2E+03	(kW/day)
Total power consumed	7E+05	7E+05	(kWh/year)

Construction and Equipment Costs for Nitrification Tank (2009)

Concrete Wall Volume	251.2	(CY)
Concrete Slab Volume	549.5	(CY)
Concrete Wall Capital Costs	$199,461.63	(2009 USD)
Concrete Slab Capital Costs	$339,026.55	(2009 USD)
Foundation Capital Costs	$84,756.64	(2009 USD)
Total Capital Cost of Tank	$838,597.39	(2009 USD)
Aerator Capital Costs	$213,256.86	(2009 USD)
Costs for Electricity Use	$133,625.27	(2009 USD/yr.)
Cost for Centrifugal Sludge Pump	$2,095.71	(2009 USD)

Construction and Equipment Costs for Cl[illegible]09)

Concrete Wall Volume	341.6	(CY)
Concrete Slab Volume	1016.1	(CY)
Concrete Wall Capital Costs	$271,245.87	(2009 USD)
Concrete Slab Capital Costs	$626,962.02	(2009 USD)
Foundation Capital Cost for ea. Tank	$156,740.51	(2009 USD)
Total Capital Cost of Tank	$1,295,331.05	(2009 USD)
Scraper Capital Costs	$240,382.66	(2009 USD)

Capital, O&M & Amortized Costs (2009)

Total Capital Cost	$2,133,928.44	(2009 USD)
Auxiliary and Control Equipment	$320,089.27	(2009 USD)
Total cost including contingency	$3,312,923.90	(2009 USD)
Operation and Maintence Cost	$398,659.18	(2009 USD/yr.)
Operation and Maintence Cost (w/o Electrici	$265,033.91	(2009 USD/yr.)
Amortized Costs	$397,550.87	(2009 USD/yr.)

Split of O & M (2009)

Amortized Total Capital Costs	$397,550.87	(2009 USD/year)
Labor and Maintenance	$265,033.91	(2009 USD/year)
Electricity	$133,625.27	(2009 USD/year)
Chemical Costs	$0.00	(2009 USD/year)
Total Costs/year	$796,210.05	(2009 USD/year)
Total Capital Costs	$0.1405	(2009 USD/1000 gal)
Labor and Maintenance	$0.0937	(2009 USD/1000 gal)
Electricity	$0.0472	(2009 USD/1000 gal)

Figure E.2.4 Sample snapshot of the calculation worksheet for individual treatment processes design and costing in the LC[3] model, where, A – Design calculations, B – Costing calculations, C- Conversion of costs to annual or volume of treated basis

Table E.2.3 Description of the different blocks in the calculation worksheet for individual treatment processes design and costing as shown in Figure E.2.4

Ref #	Section	Description
A	Calculation/ Design Block	This block houses design calculations utilized for sizing and operation of the tertiary unit.
B	Costing Summary Block	This block contains calculations for costing of tertiary treatment units. Costing includes construction, pumps, chemicals, and electricity.
C	Output Block	The output block provides costing information in the proper format to interface with the remainder of the combined costing model. Amortized capital costs and fixed O&M are separated from variable O&M costs that vary as a function of makeup water flow.

Acidification (pH) - Consists of design and costing calculations for a tapered horizontal baffled hydraulic tank (for uniform mixing and stabilization), a static mixer is assumed for mixing the acid with the influent, the acidification unit is built to provide sufficient contact time and equalization of the flow. The description of sections of the ‘Acidification’ worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

Filtration (F) – Consists of design and costing calculations for the mono-media sand filtration unit. The description of sections of the ‘Filtration’ worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

Granular Activated Carbon (G) - Consists of design and costing calculations for the activated carbon filtration unit. The description of sections of the ‘Granualar Activated Carbon’ worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

Conditioning (C) - Consists of design and costing calculations for the tapered horizontal baffled hydraulic tank (for uniform mixing and stabilization), a static mixer is assumed for mixing the chemicals with the influent, the conditioning unit is built to provide sufficient contact time and equalization of the flow. The description of sections of the ‘Conditioning’ worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

Treated Water Supply (WS) – Piping of treated water from the wastewater treatment plant to the thermoelectric power plant. Design and costing includes laying of pipeline and pumping of water under gravity flow conditions. The description of sections of the 'Treated Water Supply' worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

River Water Withdrawal (RW) – Costs for withdrawal of river water from source to power plant with an assumption that the power plant is at zero distance from the source. The description of sections of the 'River Water Withdrawal' worksheet are the same as the ones listed in Figure E.2.4 and Table E.2.3 since all worksheets contain the same sections with materials/structural costs varied for the respective treatment processes.

All worksheets, except for the 'Input' and 'Output' sheets are hidden and can be viewed by right clicking on the tab below and choosing the 'unhide' option.

E.3 Equations Used for Design and Costing of Tertiary Treatment Processes in the LC^3 Excel Model

E.3.1 Design and costing of tertiary treatment units

The design and costing equations are provided for systems depicted and listed in Figure 6.2.2(a) and 6.2.2(b) in Chapter 6 of this report.

Design of treatment units is based on the influent wastewater characteristics and the flow rate to be treated. Table 6.1.3 in Chapter 6 of this report provides typical concentration of water quality parameters in secondary and tertiary treated water from water quality analysis carried out for the Franklin Township Municipal Sanitation Authority (FTMSA) treatment plant and typical wastewater characteristics from the literature source (Asano, et al., 2007).

Flowrate to be treated and used to size the treatment units is Q = 7.75 MGD, design of treatment units are not subject to peak hourly flow requirements, since the cooling system make-up water demand includes variations in the temporal attributes (EPRI, 2002), this project assumes that design peak factor is equal to 1.

E.3.1.1 Suspended growth nitrification (N)

Design of Nitrifying Tank and Clarifier

Table E.3.1 provides the inputs and their default values considered for suspended growth nitrification (N) unit design. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual. Equations used to design and cost the 'N' unit are worked out following Table E.3.1.

Given the inputs above, assume no. of reactors 'n' based on flow rate, operational requirements (if standby) and available land area.

Flow split between units = 100%/n (E.3.1)

Flow in each unit = Q_e (m^3/day) = [Q (m^3/day) x (Flow split between units/100)] E.3.2)

To calculate the nitrogen released in the reactor due to decay of "carry-over" active heterotrophs (20% inert, 80% active), we have, inert and active VSS carried over

$(\Delta X^o_i/\Delta t)_{carry\text{-}over}$ (kg VSS_i/day) = (% inert VSS/100) x VSS_{inf} x (Q_e+Q_r)/ (1000mg/kg) (E.3.3)

$(\Delta X^o_a/\Delta t)_{carry\text{-}over}$ (kg VSS_i/day) = 0.8 x VSS_{inf} x (Q_e+Q_r) / (1000mg/kg) (E.3.4)

where,

$\Delta X^o/\Delta t$ = Increase in influent volatile suspended solids during the oxidation process;

i=inert, a=active

VSS_{inf} = Influent volatile suspended solids, and

Recycle flowrate (Q_r, m^3/day) = r x Q_e (E.3.5)

Table E.3.1 Input parameters for design of nitrification unit

Parameters	**Values @ 15^{o}C**	**Units**
Solids retention time (θ_x)	15	days
Influent TKN	21	mg NH_4^+-N/L
Effluent substrates conc.	0.6	mg NH_4^+-N/L
Effluent substrates conc.	0.36	mg NO_2^--N/L
Influent BOD	31.9	mg/L
Effluent BOD	5.8	mg/L
True yield, Y	0.33	mg VSS_a/mg NH_4^+-N
True yield, Y_N	0.083	mg VSS_a/mg NO_2^--N
O_2 demand for NH_4^+-N	4.57	mg OD/mg NH_4^+-N
O_2 demand for NO_2^--N	1.14	mg OD/mg NO_2^--N
Solid O_2 equivalent 1	1.98	mg OD/mg VSS
Solid O_2 equivalent 2	1.42	mg OD/mg VSS
Food:microorganisms ratio, f_d	0.8	
MLVSS	2000	mg/L
Design SOR	12	$m^3/m^2/d$
No. of units to split flow, n	2	
Percent flow split between units	50	%
Percent solids in sludge	1.5	%
% inert VSS	20	%
Decay coefficient	0.15	/day
% VSS releasing N	80	%
Amount of N in VSS	0.124	kg N/kg VSS
Kinetic coefficient	0.6	
Recycle ratio	0.45	
Depth of tank	5	m
No. of operation hours	24	hrs
No. of working days in year	365	days
Centrifugal pump	2	hp
Oxygen transfer efficiency	10	%
Overall oxygen transfer efficiency	12	%
Density of air	1.225	kg/m^3
Efficiency of pumps	75	%
R, gas constant	53.3	ft.lb/lb air
Temperature	15	^{o}C
α	0.283	

The active cells in the carry-over decay over time in the reactor, to find nitrogen released during decay,

$(\Delta X_a/\Delta t)_{residual}$ (kg VSS_a/day) = $(\Delta X^o_a/\Delta t)_{carry\text{-}over}$ x $\{1/[1+b\theta_x]\}$ (E.3.6)

where,

b = decay coefficient for micro-organisms (/day)

θ_x = solids/sludge retention time (SRT, days), and

Amount decayed (kg VSS_a/day) = $(\Delta X^o_a/\Delta t)_{carry\text{-}over}$ - $(\Delta X_a/\Delta t)_{residual}$ (E.3.7)

Now assume that out of the amount decayed, 20% organisms' donot release nitrogen and 80% release nitrogen.

VSS not releasing nitrogen = Amount decayed (kg VSS_a/day) x (1-(% VSS releasing N/100)) (E.3.8)

VSS releasing nitrogen = Amount decayed (kg VSS_a/day) x (% VSS releasing N/100) (E.3.9)

Which implies, nitrogen increase,

$(\Delta N/\Delta t)_{increase}$ (kg NH_4^+-N/day) = VSS releasing N x Amount of N in VSS (E.3.10)

Therefore,

Total N released (mg NH_4^+-N/L) = $[Q_e/(\Delta N/\Delta t)_{increase}]$ x (1 m^3/1000L) x (10^6mg/kg) (E.3.11)

Sludge production during conversion of ammonia to nitrite or sludge production 1 is given by,

$(\Delta X_v/\Delta t)_{sludge1}$ (kg VSS/day) =(Q_e+Q_r) x (Influent TKN–$S_{eff\text{-}NH4+}$) x $Y_1\{[1+(1-f_d)x\ b\theta_x]/[1 + b\theta_x]\}$ (E.3.12)

Nitrogen in sludge (mg N /L) = $(\Delta X_v/\Delta t)_{sludge}$ x Amount of N in VSS x (1/(Q_e+Q_r)x(1 m^3/1000L) x (10^6mg/kg))) (E.3.13)

N available for nitrite oxidizers = Influent TKN - $S_{eff\text{-}NH4+}$ - Nitrogen in sludge (E.3.14)

Sludge production during conversion of nitrite to nitrate or sludge production 2 is given by,

$(\Delta X_v/\Delta t)_{sludge2}$ (kg VSS/day) =(Q_e+Q_r) x (Influent TKN–$S_{eff\text{-}NO2^-}$) x $Y_2\{[1+(1-f_d)x\ b\theta_x]/[1 + b\theta_x]\}$ (E.3.15)

$(\Delta X_v/\Delta t)_{total\ sludge}$ (kg VSS/day)=$(\Delta X_v/\Delta t)_{sludge1}$+$(\Delta X_v/\Delta t)_{sludge2}$ + $(\Delta X^o_i/\Delta t)_{carry\text{-}over}$+$(\Delta X_a/\Delta t)_{residual}$ + VSS not releasing N (E.3.16)

where,

$S_{eff\text{-}NH4+}$ = Target effluent NH_4^+-N concentration (mg/L as N)

$S_{eff\text{-}\ NO2\text{-}}$ = Target effluent NO_2^--N concentration (mg/L as N)

Y_1 = True yield of microbial growth during the oxidation process (mg VSS_a/mg NH_4^+-N)

Y_2 = True yield of microbial growth during the oxidation process (mg VSS_a/mg NH_4^+-N)

f_d = food : micro-organisms ratio in the reactor

To calculate the power required to pump the waste sludge is given as,

Q_{sludge} (gal/min) = $(\Delta X_v/\Delta t)_{total\ sludge}$ x [(1/ρ_{sludge}) x (1000L/1m^3) x (1day/24hrs) x (1hr/60mins) x (1gal/3.784L] (E.3.17)

where,

ρ_{sludge} = density of the sludge solution to be pumped out (kg/m^3)

Total oxygen required, O_2 (kg OD/day) = [(Q_e+Q_r) x (($S_{inBOD\,+}$- S_{effBOD})/1000)]/f-(1.42 x P_x) + (4.57 x (Q_e+Q_r) x (Influent TKN – ($S_{eff\text{-}NH4+}$+ $S_{eff\text{-}NO2-}$))/1000) (E.3.18)

where,

P_x = net mass or of volatile solids (cells) or total sludge produced

f = conversion factor

S_{inBOD} = Influent BOD concentration (mg/L)

S_{effBOD} = Effluent BOD concentration (mg/L)

Volume of the reactor is therefore given as,

V_e(m^3) =[θ_x /MLVSS] x $(\Delta X_v/\Delta t)_{total\ sludge}$ x(10^{6}mg/kg) x (1m^3/1000L) (E.3.19)

where,

MLVSS = mixed liquor volatile suspended solids, (mg/L)

Hydraulic retention time (θ, day) = [V_e /Q_e] x (24hr/day) (E.3.20)

Assuming depth of reactor to be'h' (m),

Surface area, A (m^2) = V_e /h (E.3.21)

d (m) = √ A x (4/π)] (E.3.22)

Assuming surface overflow rate (SOR) for reactor (between 8 – 16 m/day),

Surface area of clarifier, A_c (m^2) = Q_e (m^3/day)/SOR (m/day) (E.3.23)

Diameter of clarifier, d_c (m) = √[A_c x (4/π)] (E.3.24)

Air flowrate/capacity is,

Q_o (kg/min) = O_2 (kg O_2/day) x 36.2 (m^3 of air/kg O_2) x (O_2 eff./Overall O_2 eff.) (E.3.25)

Aerator power = {[(Q_o xρ_{air} x 2.205 (lb/kg)/60) x (53.3 (ft.lb/(lb air) x T (^{o}R))]/[(550 (ft.lb/s.hp) x n x η/100) x (($p_1/p_2)^{0.283}$-1)] } (E.3.26)

where,

ρ_{air} = density of air kg/m^3

T = temperature of influent (^{o}R = ^{o}K + 460)

η = efficiency of pump

p_1 = absolute inlet pressure (lb_f/in^2)

p_2 = absolute outlet pressure (lb_f/in^2)

Wet well hp = 2 x {[Q_e/(24hr/day)/(3960 x 0.5)/ η] x 5.3 x α) (E.3.27)

Power consumed/day, p (kW/day) = [(Aerator hp + Wet well hp) x 0.746kW/hp x No. of hr/day] + [pump hp x 0.746 kW/hp x No. of hr/day x α] (E.3.28)

Total power consumed, P (kW/yr.) = p x No. of days/yr. (E.3.29)

Cost Estimation for Suspended Growth Nitrification

Nitrifying Tank

Structure material quantity is determined prior multiplying with unit costs to determine the total costs.

Concrete wall volume (CY) = [π x d_1 x (0.91 yd/m) x h_1 x (0.91 yd/m) x T_{w1} x 0.33 yd/ft)] + [π x d_2 x (0.91 yd/m) x h_2 x (0.91 yd/m) x T_{w2} x 0.33 yd/ft)] + ….[π x d_n x (0.91 yd/m) x h_n x (0.91 yd/m) x T_{wn} x 0.33 yd/ft)] (E.3.30)

where,

d_1, d_2, d_3….d_n = diameter of reactors (m)

h_1, h_2, h_3….h_n = depth of wall for reactor (m)

T_{w1}, T_{w2}, T_{w3}….. T_{wn}= thickness of concrete wall (ft)

n = no. of reactors or in other words, reactor no. ‘n’

Concrete slab volume (CY) = [A_1 x (0.84 yd^2/m^2) x T_{s1} x 0.33 yd/ft)] + [A_2 x (0.84 yd^2/m^2) x T_{s2} x 0.33 yd/ft)] + ….[A_n x (0.84 yd^2/m^2) x T_{sn} x 0.33 yd/ft)] (E.3.31)

where,

A_1, A_2 ,A_3 ….. A_n = surface area of reactor (m)

T_{s1}, T_{s2}, T_{s3}…. T_{sn} = thickness of concrete slab for reactor (ft)

Concrete wall costs (2009\$) = Concrete wall volume (CY) x Curved Concrete Wall Cost (\$/CY) (E.3.32)

Concrete slab costs (2009\$) = Concrete slab volume (CY) x Elevated slab cost (\$/CY) (E.3.33)

Foundation costs (2009\$) = Foundation costs (%) x Concrete slab costs (2009\$) (E.3.34)

Aerator capital cost (2009\$) = ΣA_n (ft^2) x Aerator costs (\$/ft^2) (E.3.35)

Cost of centrifugal sludge pump (2009\$) = (hp of pump$_1$/50 hp)$^{0.6}$ x Cost of 50 hp pump (2009\$) + ….(hp of pump$_n$/50 hp)$^{0.6}$ x Cost of 50 hp pump (2009\$) (E.3.36)

Total capital cost of tank (2009\$) = Concrete wall costs + concrete slab costs + foundation costs + aerator costs+ centrifugal sludge pump costs (E.3.37)

Nitrification Clarifier

Concrete wall volume (CY) = [π x d$_{c1}$ x (0.91 yd/m) x d$_1$ x (0.91 yd/m) x T$_{w1}$ x 0.33 yd/ft)] + [π x d$_{c2}$ x (0.91 yd/m) x d$_2$ x (0.91 yd/m) x T$_{w2}$ x 0.33 yd/ft)] + ….+[π x d$_{cn}$ x (0.91 yd/m) x d$_n$ x (0.91 yd/m) x T$_{wn}$ x 0.33 yd/ft)] (E.3.38)

Concrete Slab Volume (CY) = [A$_{c1}$ x (0.84 yd^2/m^2) x T$_{s1}$ x 0.33 yd/ft)] + [A$_{c2}$ x (0.84yd^2/m^2) x T$_{s2}$ x 0.33 yd/ft)] + ….+[A$_{cn}$ x (0.84 yd^2/m^2) x T$_{sn}$ x 0.33 yd/ft)] (E.3.39)

Now, follow Eq. E.3.32 to E.3.34 to determine concrete structure costs (2009\$) for clarifier, but prior calculating total costs using Eq. E.3.37, determine

Scraper capital costs (2009\$) = [d$_{c1}$/(3.28ft/m)/30 ft]$^{0.6}$ x Cost of 30 ft scraper (2009\$) + …[d$_{cn}$/(3.28ft/m)/30 ft]$^{0.6}$ x Cost of 30 ft scraper (2009\$) (E.3.40)

Complete Nitrification Unit Costs

Cost of electricity (2009\$/yr.) = P$_{aerator,sludge,scraper}$ (kW/yr.) x Cost of Electricity (\$/kW) (E.3.41)

Total capital costs (2009\$) = Total capital cost of nitrifying tank + total capital cost of nitrifying clarifier (E.3.42)

Auxiliary equipment costs (2009\$/yr.) = 15% of total capital costs (2009\$/yr.) (E.3.43)

Total capital cost including contingency (2009\$) = [Total capital costs (2009\$) + Auxiliary equipment costs (2009\$) x (1 + % contingency) (E.3.44)

Labor and maintenance costs (2009\$/yr.) = 8% Total capital cost including contingency (2009\$/yr.) (E.3.45)

Operation and maintenance costs (2009\$/yr.) = Labor and maintenance costs (2009\$/yr.) + Cost of electricity (2009\$/yr.) (E.3.46)

Amortized capital costs (2009\$/yr.) = 12% Total capital cost including contingency (E.3.47)

Total costs/yr. (2009\$/yr.) = Amortized capital costs (2009\$/yr.) + Operation and maintenance costs (2009\$/yr.) (E.3.48)

Total capital costs (2009\$/kgal) = Amortized capital costs (2009\$/yr.) x 1000 gal/kgal/(Q (gal/day) x No. of days/yr.) (E.3.49)

Labor and maintenance costs (2009$/kgal) = Labor and maintenance costs (2009$/yr.) x 1000 gal/kgal/(Q (gal/day) x No. of days/yr.) (E.3.50)

Costs of Electricity (2009$/kgal) = Cost of Electricity (2009$/yr.) x 1000 gal/kgal/(Q (gal/day) x No. of days/yr.) (E.3.51)

Total costs (2009$/kgal) = Total capital costs (2009$/kgal) + Labor and maintenance costs (2009$/kgal) + Costs of electricity (2009$/kgal) (E.3.52)

E.3.2 Solids contact clarifier (S) for lime softening or phosphorous precipitation

Table E.3.2 provides the inputs and their default values considered for chemical precipitation of hardness using lime in the solids contact clarifier (S) unit. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual.

The flow splitting equations for the softening are the same as given in Eq. E.3.1 and E.3.2

Taking inputs from Table E.3.1,

Volume of inner hopper, $V_{in} = Q_e/24hr \times t_f/60min$ (E.3.53)

where,

Q_e = flowrate in each treatment unit = Q/n (m^3/day)

t_f = flocculation time (min)

Volume of outer hopper $V_{out} = Q_e/24hr \times t_s$ (E.3.54)

where,

t_s = sedimentation time (min)

Assuming the diameter of inner hopper = d_1,

Surface area of inner hopper, $A_{in} = \pi d_1^2/4$ (E.3.55)

Now, to determine area of zone 2 (surface area at bottom of inner hopper), we need to know theoretical volume,

$V_{in} = h_c/3(A_{in}+A_2+\sqrt{(A_{in} \times A_2)}$ (E.3.56)

where,

h_c = depth of contact clarifier (m)

A_2 = Surface area of zone 2

We determine A_2 from Eq. E.3.35 by trial and error, and from A_2 we determine d_2 i.e., diameter of bottom of inner hopper

Table E.3.2 Input parameters for design of solids contact clarifier

Design criteria	Values at 15°C	Units
Flocculation time	30	min
Sedimentation time	2	hours
Depth of water	5	m
Paddle area	10% of sweep area	
Length of ea. paddle	70% of radius of flocculator	
Shaft rotation speed	3	rpm
Coefficient of drag	1.8	
Fluid density	1000	kg/m^3
Dynamic viscosity	0.001	/s
No. of units to split flow	2	
Percent flow split between units	50	%
Diameter 1	4	(m)
Sulfuric acid		g/L
Influent Ca^{2+}	41.5	mg/L
	103.5	mg/L as $CaCO_3$
Influent Mg^{2+}	10.7	mg/L
	44.0	mg/L as $CaCO_3$
Bicarbonate alkalinity as HCO_3^-	0.00354	mol/L
Bicarbonate alkalinity as HCO_3^-	177	mg/L as CaCO3
Dissociation constant pKa (H2CO3)	6.3	
alpha	0.848	
H_2CO_3*	0.00071	mol/L
H_2CO_3*	35.32	mg/L as CaCO3
pH	7	
Lime density	2200.0	kg/m^3
Soda ash density (anhydrous)	2540.0	kg/m^3
Target effluent pH	7	
Excess lime dose added	30.0	mg/L as $CaCO_3$
% Solids in sludge	5	%
No. of operating days/yr.	365.0	days
Sludge pump power	2	hp
Scraper motor	2	hp
Truck haul load	13	tons/load
Hauling per hour	0.1	load/hr
No. of hours to drive to landfill	1	hr/load
No. of hours/day	24	hr

Now to determine the surface area of outer hopper, we use,

$$V_{out} = h_c/3(A_2+A_{out}+\sqrt{(A_2 \times A_{out})} \quad (E.3.57)$$

where,

A_{out} = surface area of outer hopper/settling zone.

We determine A_{out} from Eq. E.3.57 by trial and error, and from A_{out} we determine d_3 i.e., diameter of bottom of outter hopper

To calculate paddle details, divide the contact clarifier to 3 sections horizontally at depth = 1.5m, 3m and 4.5m. Since we assume top of inner hopper has d_1 = 4m and the inner hopper is inverted conical basin with surface area increasing towards the bottom of the hopper, the flocculator blade increases in diameter as we progress to deeper depth. We assume each section increases by 3.33 m diameter at their bottom respectively.

So now, diameter of flocculator at 1.5 m is,

$d_{f1} = d_1 + 2 \times (1.5\text{m/slope})$ (E.3.58)

Sweep area $A_{s1} = \pi\, d_{f1}^2/4$ (E.3.59)

Paddle area, $A_{p1} = 10\%\ A_{s1}$ (E.3.60)

Assuming there are four paddles and that length of each paddle = 70% of flocculator radius,

At depth =1.5 m,

Length of paddle, $L_{p1} = 0.7 \times d_{p1}/2$ (E.3.61)

Height of paddle, $H_{p1} = A_{p1}/\ (L_{p1} \times 4)$ (E.3.62)

Similarly, determine, d_{p2}, A_{s2}, A_{p2}, L_{p2}, H_{p2} and d_{p3}, A_{s3}, A_{p3}, L_{p3}, H_{p3} at 3m and 4.5 m respectively.

Given shaft rotation speed in rpm, at depth 1.5m, radius of paddle, $r_{p2} = d_{p1}/2$

Rotation speed of paddle = $2\pi\ r_{p2}$ x shaft rotation speed (E.3.63)

where,

d_{p1} (m) = diameter of paddle at depth 1.5 m

Similarly, determine rotation speed of paddle at 3m and 4.5m depth.

Relative to the water, the velocity of the paddle, v_{p1} = 0.75 x rotation speed of paddle (E.3.64)

Now,

Power required for flocculator, $P = ½\ C_D \Sigma A_p \rho\ (\Sigma v_p)^3$ (E.3.65)

where,

C_D = coefficient of drag of paddle

A_p = area of paddle at different depths (m^2)

ρ = fluid density (kg/m^3)

v_p = relative velocity of paddle with respect to water at different depths of contact clarifier (m/s)

Taking efficiency of motor = η, total power required = $P/\ \eta$

To justify contact clarifier is correctly designed; let's check the surface loading rate (SLR),

$SLR = Q_e/A_{out}$ (E.3.66)

where,

$A_{out} = (\pi d_3^2/4) - (\pi^*(d_{1+2xoutletlaunderwidth}{}^2)/4)$ (E.3.67)

Now, power required in the contact clarifier, Power required (kW/yr.) = 0.5 x Coeff or drag x (Σ_n paddle area) x ρ_w x (Σ_n relative speed of paddles) x No. of hrs/day/1000) (E.3.68)

For lime softening, the dose of lime to be added to for effective precipitation can be determined,

Total hardness = Influent Ca^{2+} x (MW $CaCO_3$/MW Ca^{2+}) + Influent Mg^{2+} x MW $CaCO_3$/MW Mg^{2+}) (E.3.69)

Total Lime dose required (mg/L) = [Influent Mg^{2+} (mg/Las $CaCO_3$) + Influent bicarbonate alkalinity (mg/Las $CaCO_3$) + $H_2CO_3^*$ (mg/Las $CaCO_3$) + Excess lime dose added (mg/Las $CaCO_3$)] x MW of lime/ MW $CaCO_3$) (E.3.70)

where,

$H_2CO_3^*$ (mol/L) = 10^{-pH} x Bicarbonate alkalinity (mol/L)/10^{-pka1} (E.3.71)

For *nitrified* effluent treatment only,

Total soda ash required (kg/yr) = soda ash dose (mg/L)*Q_e (m^3/day)*(365 days/yr)*(1 m^3/1000L) (E.3.72)

For lime sludge production calculation,

Total solids precipitated (kg/d) = [$H_2CO_3^*$ (mg/Las $CaCO_3$) + (2 x Influent Ca_2^+ (mg/L as $CaCO_3$) – Excess lime dose added (mg/Las $CaCO_3$)] x (1kg/1000000mg) x Q_e (L/d) (E.3.73)

Total solids precipitated (gal/hr) = Total solids precipitated (kg/day)/ [ρ_{sludge} (kg/L) x 3.784L/gal x 24 hrs/day] (E.3.74)

Sludge flowrate, Q_{sludge} (gal/min) = Total solids precipitated (gal/hr)/ [% solids in sludge x 60min/hr] (E.3.75)

No. of loads/day = Solids precipitated (kg/day)/[(1000 kg/ton)*Truck haul load (E.3.76)

No. of hours for hauling /day = (No. of loads/day)/(Hauling/hr) (E.3.77)

No. of hours required to haul and disposal= (No. of loads/day) x No. of hours to drive to landfill (E.3.78)

Power required by dewatering equipment (kW/yr) = [(Total solids precipitated (kg/day) x (2.205 lb/kg)/(No. of workings hrs/day)/438) x (density of sludge x 460 x No. of work hrs/day x 3 x (1 m^3/1000 L)] x No. of days/year (E.3.79)

Cost Estimation for Softening Unit

Concrete for wall for outer reactor (CY) = [π x $d_{3,1}$ x (0.91 yd/m) x ($h_{3,1}$+ 0.8) x (0.91 yd/m) x T_{w1} x 0.33 yd/ft)] + [π x $d_{3,2}$ x (0.91 yd/m) x $d_{3,2}$ x (0.91 yd/m) x T_{w2} x 0.33 yd/ft)] + ….[π x $d_{3,n}$ x (0.91 yd/m) x ($h_{3,n}$ + 0.8) x (0.91 yd/m) x T_{wn} x 0.33 yd/ft)] (E.3.80)

where,

$d_{3,1}$ = diameter of settling zone (d_3) in reactor 1 (m)

$h_{3,1}$ = depth of wall for settling zone for reactor 1 (m)

T_{w1} = thickness of concrete wall (ft)

n = no. of reactors or in other words, reactor no. 'n'

Concrete for inner reactor wall (CY)= [π x $d_{1,1}$ x (T_{w1}/(3.28ft/m)) x 0.84 yd^2/m^2)+(π x (($d_{1,1}$/2 + $d_{2,1}$/2) x (h_{s1}) x (T_{w1}/(3.28ft/m))+(0.76 yd^3/m^3) + [π x $d_{1,2}$ x (T_{w1}/(3.28ft/m)) x 0.84 yd^2/m^2)+(π x (($d_{1,2}$/2 + $d_{2,2}$/2) x (h_{s1}) x (T_{w1}/(3.28ft/m))+(0.76 yd^3/m^3) + ….. [π x $d_{1,n}$ x (T_{wn}/(3.28ft/m)) x 0.84 yd^2/m^2)+ (π x (($d_{1,n}$/2 + $d_{2,n}$/2) x (h_{s1}) x (T_{wn}/(3.28ft/m))+(0.76 yd^3/m^3) (E.3.81)

where,

$d_{1,1}$ = diameter at the top part of inner hopper (d_3) in reactor 1 (m)

$d_{2,1}$ = diameter at the bottom part of inner hopper (d_3) in reactor 1 (m)

h_{s1} = slant length of hopper for reactor 1 (m)

T_{w1} = thickness of concrete wall (ft)

n = no. of reactors or in other words, reactor no. 'n'

Concrete Slab Volume (CY) = [$A_{3,1}$ x (0.84 yd^2/m^2) x T_{s1} x 0.33 yd/ft)] + [$A_{3,2}$ x (0.84 yd^2/m^2) x T_{s2} x 0.33 yd/ft)] + ….[$A_{3,n}$ x (0.84 yd^2/m^2) x T_{sn} x 0.33 yd/ft) (E.3.82)

where,

$A_{3,1}$ = surface area for settling zone of reactor 1 (m)

T_{s1} = thickness of concrete slab for reactor 1 (ft)

Now, follow Eq. E.3.32 to E.3.34 to determine concrete structure costs (2009$) for clarifier, but prior calculating total costs using Eq. E.3.37, determine scraper costs, chemical metering and sludge pump costs as shown below,

Cost of Scraper with mixing equipment attached (2009$) = ($d_{3,1}$ x (3.28ft/m))/30ft)$^{0.6}$) x unit cost of 30ft scraper (2009$))+unit cost of mixer (2009$)$^{0.6}$) +….+ ($d_{3,n}$ x (3.28ft/m))/30ft)$^{0.6}$) x unit cost of 30ft scraper (2009$))+unit cost of mixer (2009$)$^{0.6}$) (E.3.83)

Cost of chemical metering pump (2009$) = (Total Lime required (gal/day)/ (24hr/day))/100 (gal/min))$^{0.6}$)*unit cost of (100 gal/min) pump) (E.3.84)

Cost of sludge pump (2009$) = (Sludge flowrate (gal/min)/10(gal/min))$^{0.6}$)*unit cost of (10 (gal/min) pump) (E.3.85)

Using Eq. E.3.41, the cost for electricity can be determined as

Cost of electricity = (Power for clarifier (kW/yr) + dewatering equipment (kW/yr)) x unit cost of electricity ($/kWh) (E.3.86)

Cost of Lime (2009$/yr.) = (Total Lime required (kg/day) x 2.205(lb/day)) x (Unit cost of 50 lb lime bag/50)) (E.3.87)

Cost of Electricity (2009$/yr.) = $P_{\text{mixer+scraper+sludge}}$ (kW/yr.) x Unit cost of electricity ($/kW) (E.3.88)

Cost of lime (2009$/kgal.) = Cost of Lime (2009$/yr.) x 1000 gal/kgal/(Q (gal/day) x No. of days/yr.) (E.3.89)

Eq. E.3.32 can be used to determine the auxiliary equipment costs for contact clarifier and Eq. 3.3.44 to 3.3.52 are used to calculated chemical, operation/labor and maintenance and total costs in 2009 USD/yr and 2009 USD/kgal.

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E.3.3 pH adjustment (pH)

Table E.3.3 provides the inputs and their default values considered for pH adjustment acidification (A) unit design. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual.

The flow splitting equations for the softening are the same as given in Eq. 3.3.1 and 3.3.2

Sizing the dimensions of the mixing tank,

Volume of chamber (m^3), V= Q_e (m^3/day) x contact time (min) x (60 min/hr) x (1 day/24 hr) (E.3.90)

Area of chamber, A = V (m^3) /Depth of tank (m) (E.3.91)

Width of chamber, w = √(A(m)/Length:Width ratio) (E.3.92)

Length of tank, l = width x Length:Width ratio (E.3.93)

Locating 2 baffle waffles parallel to length of tank, by dividing the tank into 3 equal sections along the width.

Now for dose of acid to be added, we need to determine the strength of Sulfuric acid, Sulfuric acid to be added (m^3/day) = (Acid to be added (100%) (mg/L)/1000000 mg/kg) x Q (m^3/day)/density of H_2SO_4 (kg/L)) (E.3.94)

Sulfuric acid to be added (kg/day) = Sulfuric acid to be added (m^3/day) x density of H_2SO_4 (kg/L) x (1000 L/m^3) (E.3.95)

Sulfuric acid (gal/day) = Sulfuric acid to be added (kg/day)/Strength of H_2SO_4 (kg/gal) (E.3.96)

Cost Estimation for Acidification Unit

Assuming that the baffles are equivalent to one parallel wall,

Concrete for concrete wall (CY)= (5 walls x ((l_1 x 0.91 yd/m) x ((h_1+2) x 0.33 yd/ft) x (T_{w1} x 0.33 yd/ft))+(2 walls x ((w_1 x 0.91 yd/m) x (h_1+2) x 0.33 yd/ft) x (T_1 x 0.33 yd/ft))+…. + (5 walls x ((l_n x 0.91 yd/m) x ((h_n+2) x 0.33 yd/ft) x (T_{wn} x 0.33 yd/ft))+(2 walls x ((w_n x 0.91 yd/m) x (h_n+2) x 0.33 yd/ft) x (T_{wn} x 0.33 yd/ft) (E.3.97)

where,

l_{w1} = length of wall for unit 1 (m)

w_{w1} = width of wall for unit 1 (m)

Concrete for concrete slab (CY) = (A_1 x 0.84 yd^2/m^2) x (T_{s1} x 0.33 ft/yd)+….+ (A_n x 0.84 yd^2/m^2) x (T_{sn} x 0.33 ft/yd) (E.3.98)

Now, follow Eq. E.3.32 to E.3.34 to determine concrete structure costs (2009$) for clarifier, but prior calculating total construction costs using Eq. E.3.37, determine chemical metering costs as shown below,

Cost of chemical metering pump (2009$) = (Total acid required (gal/day)/ (24hr/day))/100 $(gal/min))^{0.6}$)*unit cost of (100 gal/min) pump) (E.3.99)

Cost of sulfuric acid (2009$/yr.) = (Sulfuric acid added (kg/yr) x (2.205lb/kg)*(unit cost of sulfuric acid of 750 lb drum/750 lb) (E.3.100)

Eq. E.3.43 can be used to determine the auxiliary equipment costs for acidification unit and Eq. E.3.44 to E.3.52 are used to calculated chemical, operation/labor and maintenance and total costs in 2009 USD/yr and 2009 USD/kgal.

E.3.4 Filtration unit (F)

Table E.3.4 provides the inputs and their default values considered for mono-media filtration (F) unit design. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual.

Table E.3.3 Input parameters for acidification unit design

Typical Parameters	Values @ 15°C	Units
Contact time	5	mins
Side water depth	3	m
Length:width ratios	4	
Number of units	2	
Percent split between units	50	%
H_2SO_4 mol. wt.	98.08	g/mol
Volume conversion	3.79	L/Gal
Acid to be added (100% H_2SO_4)	0.05	Moles/L
Acid to be added (100% H_2SO_4)	122	mg/L
Moles of 93% H_2SO_4	17.50	moles/L
No. of operation hours	24	hrs
No. of working days in year	365	days
Density of 93% H_2SO_4	1.711	kg/m^3
Strength of the H_2SO_4	1716.4	g/L

Table E.3.4 Input parameters for filtration unit design

Typical Parameters	Values @ 15°C	Units
Type of filter operation	Semicontinuous	
Filtration rate	5	$gal/min.ft^2$
Type of filter	Conventional	
Filter bed	Mono-medium	
Filter media	Sand	
Flow direction	Downward	
Backwash operation	air-scour	
Backwash time	5	min
Filter rise rate	24	in/min
Free board above filter bed	24	in
Backwash rate	62.6	$gal/min.ft^2$
Depth of sand (in)	48	in
Effective size of sand (mm)	2.5	mm
Uniformity coefficient	≤1.5	
Length:width ratios	1	
Backwash air scour	5	hp
Depth of backwash reservoir	4	ft
No of backwash reservoir	2	
1 hp	0.746	kW
No. of operation hours	24	hrs
No. of working days in year	365	days
No. of back wash pumps	2	

Dimensioning the filters,

No. of filters, n_f = 2.7 x √(Q(MGD) (E.3.101)

Flow passing thro' each filter, Q_e = Q (gal/day)/ n_f (E.3.102)

Surface area of ea. filter, A_f (ft^2) = Q_e (gal/day) x (1440 min/day) x Filtration rate (gal/min.ft^2) (E.3.103)

Width of filter, w_f (ft) = √(A(m)/Length:width ratio) (E.3.104)

Length of filter, l_f (ft) = width x Length:width ratio (E.3.105)

Total filter area, A_t (ft^2) = A_f x n_f (E.3.106)

Since, clearwells run parallel to filters, the length and width of clearwells = length and width of filters

Depth of clearwell (ft) = [Free board above filter bed (in)+ filter rise level (in)+ Depth of sand (in)] x (1ft/12in) (E.3.107)

Total volume of sand, V_s (ft^3) = Depth of sand (in) x (1ft/12in) x l_f (ft) x w_f (ft) (E.3.108)

Backwash water needed, V_b (gal) = Backwash rate (gal/min.ft^2) x Backwash time (min) x A_f (ft^2) x n_f /no. of reservoirs. (E.3.109)

Surface area of ea. reservoir, A_b = [V_b(gal) x (3.784L/gal) x (1m^3/1000L) x (1ft^3/$3.28^3 m^3$)]/depth of backwash reservoir (E.3.110)

Width of reservoir, w_b (ft) = √(A_b(m)/Length:width ratio) (E.3.111)

Length of reservoir, l_b (ft) = w_b x Length:width ratio (E.3.112)

Power consumed, P (kW/yr.) = backwash air-scour pump (hp) x (0.746kW/hp) x (24 hrs/day) x (360days/yr.) x no of backwash pumps (E.3.113)

Cost Estimation for Filtration Unit

Concrete for concrete wall (CY)= (3 walls x ((l_f (ft) x h_f (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd)) + (2 x (((2 x w_f (ft)) x h_f (ft) 0.11yd^2/ft^2)* (T_{wf} (ft) x 0.33 ft/yd)) x n_f + (((2 x (l_b(ft) x h_b (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd))+(2 x (w_b(ft) x h_b (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd)) x n_b) (E.3.114)

where,

l_f = length of filter wall (ft)

w_f = width of filter wall (ft)

h_f = depth of filter wall (ft)

T_{wf} = thickness of filter wall (ft)

n_f = no. of filters

Concrete for concrete slab (CY) = (2 x A_f (ft^2) x 0.11 yd^2/ft^2) x (T_{ws} x 0.33 ft/yd) x n_f + ((A_b(ft^2) x 0.11 yd^2/ft^2) x (T_{ws} x 0.33 ft/yd) x n_b (E.3.115)

where,

A_f = surface area of filter (ft^2)

A_b = surface area of backwash water reservoir (ft^2)

T_{ws} = thickness of filter floor slab (ft)

n_b = no. of backwash reservoirs.

Now, follow Eq. E.3.32 to E.3.34 to determine concrete structure costs (2009$) for clarifier, but prior calculating total construction costs using Eq. E.3.37, determine filter underdrains, media and backwash pump as shown below,

Cost of filter underdrains (2009$) = $A_f^{0.6}$ (ft^2) x n_f x unit cost of underdrains ($/ft²) (E.3.116)

Cost of filter sand (2009$) = Volume of sand media (ft^3) x unit cost of sand ($/ft³) (E.3.117)

Cost of backwash pump (2009$/yr.) = [$Q_e$ (gal/day)/(A_f(ft^2) x no. of operating hrs. x 60 min/hr) x n_f) /(backwash rate (gal/ft^2.min))$^{0.6}$] x unit cost of backwash pump (2009$) x n_b (E.3.118)

Cost of Electricity ($/yr.) = power consumed (hp/yr.) x (0.746 kW/hp) x unit cost of electricity ($/kW) (E.3.119)

Eq. E.3.43 can be used to determine the auxiliary equipment costs for filtration unit and Eq. E.3.44to E.3.52 are used to calculated operation/labor and maintenance and total costs in 2009 USD/yr and 2009 USD/kgal.

E.3.5 Granular activated carbon contactor (G)

Table E.3.5 provides the input or design criteria and the default values used used for the design and costing of granular activated carbon filtration for total organic carbon reduction. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual.

First, a GAC pilot scale column to treat 75 gallons/day (0.052 gpm) shall be designed, which then shall be scaled to a 7.75 MGD flow of MWW_NF water with influent TOC = 11.5 – 13.5 mg/L for field experiments. This design is mainly based on flow characteristics and not the type of organic matter present.

For the bench scale experiments conducted in the University of Pittsburgh lab, Filtrasorb 300 GAC was used. From the Freundlich isotherm (Fig.C.5.1in Appendix C) equation obtained from batch studies in the laboratory,

q_e = 0.005 x (Influent TOC)$^{5.24}$ (E.3.120)

where,

q_e = adsorbent phase concentration after equilibrium, mg adsorbate/g adsorbent

From Metcalf and Eddy, 2003

q_e design = 25% of the q_e obtained from the experimental analysis (E.3.121)

Volume of GAC in the unit (m^3) = EBCT (hr)/24 hrs/day x Q (m^3/day) (E.3.122)

where,

EBCT = Volume of contactor occupied by GAC, m^3/Volumetric flowrate, m^3/ (E.3.123)

EBCT value for the experimental column is used in Eq. E.3.91 to determine volume of GAC in contactor by varying the volumetric flowrate.

Determine mass of carbon used by using the bench scale column dimensions and apparent density of carbon.

Mass of activated carbon in contactor (lb) = (2.205 lb/kg x Volume of GAC in ea. unit (m^3) x apparent density of carbon (g/m^3))/ (1000 g/kg) (E.3.124)

Based on number of contactors required, the mass or volume of activated carbon can be split as follows,

Volume of carbon in ea. GAC contactor = mass of activated carbon (lb) / (apparent density (lb/ft^3) x No. of contactors) (E.3.125)

Surface area of the contactor (ft^2) = Volume of GAC in ea. contactor (ft^3)/ depth of the filters and clearwell (ft) (E.3.126)

Assuming a length:width ratio to be 1:2,

Length of the contactor/filter (ft) = √surface area/2 (E.3.127)

Width of the filter (ft) = length of the filter x 2 (E.3.128)

Vol. of water treated in ea. contactor = Filtration rate in gpm/ft^2 x 1440 min/day x surface are of the filter (ft^2) /(1000000 gal/million gal) (E.3.129)

Since, clearwells run parallel to filters, the length and width of clearwells = length and width of filters

Depth of clearwell (ft) = [Free board above filter bed (in)+ filter rise level (in)+ Depth of GAC (in)] x (1ft/12in) (E.3.130)

Backwash water needed, V_b (gal) = Backwash rate (gal/min.ft^2) x Backwash time (min) x A_f (ft^2) x n_f /no. of reservoirs. (E.3.131)

Surface area of ea. reservoir, A_b = [V_b(gal) x (3.784L/gal) x (1m^3/1000L) x (1ft^3/3.28^3m^3)]/depth of backwash reservoir (E.3.132)

Width of reservoir, w_b (ft) = √(A_b(m)/Length:width ratio) (E.3.133)

Length of reservoir, l_b (ft) = w_b x Length:width ratio (E.3.134)

Power consumed, P (kW/yr.) = backwash air-scour pump (hp) x (0.746kW/hp) x (24 hrs/day) x (360days/yr.) x no of backwash pumps (E.3.135)

From bench scale experimental run results and Freundlich isotherm equation,

Carbon Usage Rate, CUR = C_o-C_e/q_e

(B-136)

where,

C_o and C_e = influent and effluent TOC concentration (mg/L)

Volume of water treated = m_{GAC}/CUR (E.3.137)

Bed Life, d = Volume of water treated for given EBCT/ Q (E.3.138)

Cost Estimation for GAC Filtration Unit

Activated carbon costs (2009$) = mass of activated carbon (kg) x cost of activated carbon ($/kg) x No. of contactors (E.3.139)

Regeneration costs (2009$/yr) = mass of activated carbon (kg) x cost of regenerated carbon ($/kg) x (No. of days/yr /bed life of carbon (days)) (E.3.140)

Now, for construction of infrastructure,

Concrete for concrete wall (CY)= (3 walls x ((l_f (ft) x h_f (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd)) + (2 x (((2 x w_f (ft)) x h_f (ft) 0.11yd^2/ft^2)* (T_{wf} (ft) x 0.33 ft/yd)) x n_f + (((2 x (l_b(ft) x h_b (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd))+(2 x (w_b(ft) x h_b (ft) x 0.11yd^2/ft^2) x (T_{wf} (ft) x 0.33 ft/yd)) x n_b) (E.3.141)

where,

l_f = length of filter wall (ft)

w_f = width of filter wall (ft)

h_f = depth of filter wall (ft)

T_{wf} = thickness of filter wall (ft)

n_f = no. of filters

Concrete for concrete slab (CY) = (2 x A_f (ft^2) x 0.11 yd^2/ft^2) x (T_{ws} x 0.33 ft/yd) x n_f + ((A_b(ft^2) x 0.11 yd^2/ft^2) x (T_{ws} x 0.33 ft/yd) x n_b (E.3.142)

where,

A_f = surface area of filter (ft^2)

A_b = surface area of backwash water reservoir (ft^2)

T_{ws} = thickness of filter floor slab (ft)

n_b = no. of backwash reservoirs.

Now, follow Eq. E.3.32 to E.3.34 to determine concrete structure costs (2009$) for clarifier, but prior calculating total construction costs using Eq. E.3.37, determine filter underdrains, backwash pump as shown below,

Cost of filter underdrains (2009$) = $A_f^{0.6}$ (ft^2) x n_f x unit cost of underdrains (\$/ft^2) (E.3.143)

Cost of backwash pump (2009\$/yr.) = [Q_e (gal/day)/(A_f(ft^2) x no. of operating hrs. x 60 min/hr) x n_f) /(backwash rate (gal/ft^2.min))$^{0.6}$] x unit cost of backwash pump (2009$) x n_b (E.3.144)

Cost of natural gas used to activate carbon = (Natural gas required to activate carbon (Nm^3/kg) /(1000 x (3.28 ft/m)^3 x cost of natural gas/1000 ft^3 x Mass of GAC (lb) x (No. of working days/yr/Bed life (days)) (E.3.145)

Cost of natural gas used to regenerate carbon = (Natural gas required to regenerate carbon (Nm^3/kg) /(1000 x (3.28 ft/m)^3 x cost of natural gas/1000 ft^3 x Mass of GAC (lb) x (No. of working days/yr/Bed life (days)) (E.3.146)

Cost of Electricity (\$/yr.) = power consumed (hp/yr.) x (0.746 kW/hp) x unit cost of electricity (\$/kW) (E.3.147)

Eq. E.3.43 can be used to determine the auxiliary equipment costs for filtration unit and Eq. E.3.44 to E.3.52 are used to calculated operation/labor and maintenance and total costs in 2009 USD/yr and 2009 USD/kgal.

E.3.6 Chemical addition unit (C)

Table E.3.6 provides the inputs and their default values considered for chemical conditioning (C) unit design. The doses of chemical given in Table E.3.6 are for the default case of direct addition to secondary treated municipal wastewater with filtration (MWW_F). Dosages for other alternatives are available in the input sheet of the model. Additional symbols or abbreviations and conversion of units used are also listed at the end of this procedure manual. Procedure followed to design the chemical addition unit is similar to acidification unit, except for the chemical dosing calculations.

The flow splitting equations for the softening are the same as given in Eq. E.3.1 and E.3.2

Table E.3.5 Typical input parameters for granular activated carbon design

Typical Parameters	Values @ 15°C	Units
Design adsorption isotherm const	25	%
Dia of experimental column	5.5	cm
Height of experimental column	23	cm
Volumetric flowrate in column	1.1	L/hr
Density of GAC	0.53	g/cm^3
No. of working days/yr.	365	yr
Natural gas required to activate carbon	4.9	Nm^3/kg of activated carbon
Natural gas required to regenerate carbon	2.7	Nm^3/kg of activated carbon
Backwash time	5	min
Backwash air scour	5	hp
1 hp	0.746	kW
No. of operation hours	24	Hrs
No. of working days in year	365	Days
No. of contactors required	1	
Tertiary treated influent TOC	12.5	mg/L
GAC treated effluent TOC	3	mg/L

Table E.3.6 Input parameters for chemical addition unit design

Parameters	Values	Unit
Average influent total coliform	10000	#/100 mL
Effluent total coliform	23	MPN/100 mL
Contact time ,t	15	mins
Side water depth	3	m
No. of units	2	
Percent split between units	50	%
MCA dose	64	mg/L
TTA dose	1	mg/L
PMA dose	1.25	mg/L
ClO_2 dose	0	mg/L
Length:Width	1.25	
Density of MCA	0.6	kg/L
Density of TTA	1.24	kg/L
Density of PMA	1.2	kg/L
Density of ClO_2	1.6	kg/L
No. of operating hours/day	24	
No. of days in year	365	

Sizing the dimensions of the mixing tank,

Volume of the tank, V= Q_e (m^3/day) x Contact time (min) x (60 min/hr) x (1 day/24 hr) (E.3.148)

Area of the tank, A = V (m^3) /Depth of tank (m) (E.3.149)

Width of tank, w = √(A(m)/Length:width ratio) (E.3.150)

Length of tank, l = width x Length:width ratio (E.3.151)

Locating 2 baffle waffles parallel to length of tank, by dividing the tank into 3 equal sections along the width.

Determining the quantity of MCA to be added,

MCA dosage (kg/day) = MCA dose (mg/L) x Q (gal/day) x (3.784 L/gal/1000000)/No. of units (E.3.152)

MCA dosage (gal/day) = MCA dosage (kg/day)/(density of MCA (kg/L) x (3.784 L/gal (E.3.153)

MCA dosage (gal/hr) = MCA dosage (gal/day)/ No. of hrs/day (E.3.154)

TTA dosage (kg/day) =TTA dose (mg/L) x Q (gal/day) x (3.784 L/gal/1000000)/No. of units (E.3.155)

TTA dosage (gal/day) = TTA dosage (kg/day)/(density of TTA (kg/L) x (3.784 L/gal) (E.3.156)

TTA dosage (gal/hr) = TTA dosage (gal/day)/ No. of hrs/day (E.3.157)

PMA dosage (kg/day) =PMA dose (mg/L) x Q (gal/day) x (3.784 L/gal/1000000)/No. of units (E.3.158)

PMA dosage (gal/day) = PMA dosage (kg/day)/(density of PMA (kg/L) x (3.784 L/gal) (E.3.159)

PMA dosage (gal/hr) = PMA dosage (gal/day)/ No. of hrs/day (E.3.160)

ClO_2 dosage (kg/day) =ClO_2 dose (mg/L) x Q (gal/day) x (3.784 L/gal/1000000)/No. of units (E.3.161)

ClO_2 dosage (gal/day) = PMA dosage (kg/day)/(density of ClO_2 (kg/L) x (3.784 L/gal) (E.3.162)

ClO_2 dosage (gal/hr) = PMA dosage (gal/day)/ No. of hrs/day (E.3.163)

Cost Estimation for Chemical Addition Unit

Concrete for concrete wall (CY)= (5 walls x ((l_1 x 0.91 yd/m) x ((h_1+2) x 0.33 yd/ft) x (T_{w1} x 0.33 yd/ft))+(2 walls x ((w_1 x 0.91 yd/m) x (h_1+2) x 0.33 yd/ft) x (T_{w1} x 0.33 yd/ft))+…. + (5 walls x ((l_n x 0.91 yd/m) x ((h_n+2) x 0.33 yd/ft) x (T_{wn} x 0.33 yd/ft))+(2 walls x ((w_n x 0.91 yd/m) x (h_n+2) x 0.33 yd/ft) x (T_{wn} x 0.33 yd/ft) (E.3.164)

where,

l_1 = length of wall for unit 1 (m)

w_1 = width of wall for unit 1 (m)

Concrete for concrete slab (CY) = (A_1 x 0.84 yd^2/m^2) x (T_{s1} x 0.33 ft/yd)+….+ (A_n x 0.84 yd^2/m^2) x (T_{sn} x 0.33 ft/yd) (E.3.165)

Cost of chemical metering pump (2009$) = (Sum of all chemicals required (gal/day)/ (24hr/day))/100 (gal/min))$^{0.6}$)*unit cost of (100 gal/min) pump) (E.3.166)

Cost of MCA (2009$/yr.) = (MCA Dose added (kg/yr.) x (2.205lb/kg)*(unit cost of MCA/lb) (E.3.167)

Cost of TTA (2009$/yr.) = (TTA Dose added (kg/yr.) x (2.205lb/kg)*(unit cost of TTA/lb) (E.3.168)

Cost of PMA (2009$/yr.) = (PMA Dose added (kg/yr.) x (2.205lb/kg)*(unit cost of PMA/lb) (E.3.169)

Cost of ClO_2 (2009$/yr.) = (ClO2 Dose added (kg/yr.) x (2.205lb/kg)*(unit cost of ClO_2/lb) (E.3.170)

Eq. E.3.43 can be used to determine the auxiliary equipment costs for filtration unit and Eq. E.3.44 to E.3.52 are used to calculated operation/labor and maintenance and total costs in 2009 USD/yr and 2009 USD/kgal.

E.3.7 Tertiary treated water supply system (WS)

Table E.3.7 provides the inputs and their default values considered for treated water supply system (WS) designed to deliver treated water from wastewater treatment plant to thermo-electric power plants. In previous study analysis, it was determined that sufficient treated municipal wastewater is available to satisfy cooling needs of most existing and proposed cooling power plants within 10-25 miles (Vidic et al., 2009).

Pipe dimensions were decided based on pipeline design 'rules of thumb',

Headloss due to friction loss (no elevation head is considered and complete 10 miles is assumed to be at constant head.

Flowrate in pipe, Q (ft^3/s) = Flowrate (m^3/day) x (3.28 ft/m)^3/ (No. of hr/day x No. of min/hr x No. of min/sec) (E.3.171)

Surface area of pipe (ft^2) = Flowrate in pipe, Q (ft^3/s)/velocity of flow (ft/s) (E.3.172)

where,

V (ft/s) = rule of thumb average velocity

Diameter of pipe,d (ft) = √(Surface area of pipe x 4/π) (E.3.173)

Reynold's Number = $\rho Vd/\mu$ (E.3.174)

where,

ρ (slugs/ft^3) = density of treated water

μ (lb/s.ft^2) = kinematic viscosity of water

Headloss due to friction loss, H_L (ft/1000ft) = 1000 [V/0.115C (d)$^{0.63}$]$^{1.852}$ (ft/1000 ft.) (E.3.175)

where,

C = Hazen William's constant = 140 for ductile iron pipe

Width of trench to lay pipe (ft) = (d+ 2 x Trench Width) x (1 ft/12 in) (E.3.176)

Cost Estimation for Treated Water Supply System

Treated water supply system consists of excavation, backfill, installation of pipeline, pipe, pump and pumping costs.

Cubic yard of trench excavation (CY) = Delivery distance (ft) x Width of trench (ft) x depth of trench (ft)/(3ft/yd)^3) + (Delivery distance x 2 x depth of trench (ft) x (0.104 ft/yd)^2) (E.3.177)

Pipe and installation (2009 \$/mi) = Cost of'd' inch pipe (\$/18 ft) x 5280ft/18 ft (E.3.178)

Excavation and fill (2009 \$/mi) = Cubic yard of trench to be excavated or filled x Cost per cubic yard (E.3.179)

Bedding (2009 \$/ mi) = Cost per linear ft. (\$/ft) x length of pipe (ft) (E.3.180)

Pump Cost (2009\$) = ((Q gpm/(No. of hr/day x No. of min/hr))/5000 gpm)$^{0.6}$ x Unit cost of pump with 5000 gpm capacity (E.3.181)

Pumping cost for 1000 ft. (elec.) = 1.65 x H_L x 5347 x (Cost of electricity,α (\$/kW)/(η/100) (E.3.182)

where,

η = efficiency of the pump (%)

Pumping cost for 10 miles = delivery distance (ft) x pumping cost (\$/yr/1000 ft)/1000 ft (E.3.183)

Total capital cost (2009\$/mi) = (Pipe and installation excavation + fill + bedding) x 1.35 (E.3.184)

Amortized capital cost = 12% of Total capital cost (E.3.185)

Repair and maintenance (2009\$/yr/mi) = 10% of Amortized capital cost (2009\$/yr./mi) (E.3.186)

Total O&M cost (2009\$/yr./mi) = Pumping+ Repair and maintenance (E.3.187)

Total Annual Cost (2009\$/yr) = Amortized capital cost + Total O&M (E.3.188)

Eq. E.3.49 to E.3.52 are used to calculated operation/labor and maintenance and total costs in 2009 USD/kgal.

E.3.8 River water withdrawal pumping

Table E.3.8 provides the inputs and their default values considered for river water pumping (RWP) designed to pump fresh water from source to thermo-electric power plants.

Flowrate in the withdrawal pipe is determined using Eq. E.3.171

Pressure difference between inlet and outlet (kg/m.s^2) = density of water (kg/m^3) x density at which water is being pumped from (ΔZ) (m) x acceleration due to gravity (m/s^2) (E.3.189)

Power required to lift water from reservoir (kgm^2/s^3) = Q (m^3/s) x pressure difference (kg/m.s^2)/(η/100) (E.3.190)

Power required to lift water from reservoir (kW) = Power required to lift water from reservoir (kgm^2/s^3)/1000 (E.3.191)

Power required to lift water from reservoir (kW/day) = Power required to lift water from reservoir (kW) x No. of hrs/day (E.3.192)

Power required to lift water from reservoir (kW/yr) = Power required to lift water from reservoir (kW/day) x No. of days/yr (E.3.193)

Cost Estimation for River Water Pumping System

Pump cost for withdrawal of s. water = ((Q (gal/day)/(No. of hrs/day x 60))/3500)$^{0.6}$ x unit cost of pump ($/3500 gpm) (E.3.194)

Pumping cost for surface water withdrawal (elec.) = Power required to lift water from reservoir (kW/yr) x unit cost of electricity ($/kW) (E.3.195)

Eq. E.3.43 can be used to determine the auxiliary equipment costs for river water pumping unit and Eq. E.3.44 to E.3.52 are used to calculated total costs in 2009 USD/yr and 2009 USD/kgal, but to determine

Repair and maintenance cost (2009$/yr) = 8% of Amortized capital cost (E.3.196)

Table E.3.7 Input parameters for treated water supply system design

Parameters	Values	Units
Velocity of flow	6	(ft/sec)
Delivery distance, L	10	(miles)
	54448	(ft)
Density of water	1.94	(slugs/ft^3)
	1000	kg/m^3
Acceleration due to gravity constant, g	9.81	m/s^2
Kinematic viscosity of water	2.34E-05	(lb/s/ft^2)
Hazen William's constant, C	140	
Material of pipe	Ductile Iron	
Trench width = 2 times dia of pipe	12	(inch)
No. of days in a year	365	
Operating hours	24	

Table E.3.8 Input parameters for river water withdrawal pumping design

Design Criteria	Values	Units
Density of water	1000	kg/m^3
Depth at which water is being pumped from, ΔZ	8	m
Acceleration due to gravity constant, g	9.81	m/s^2
Efficiency of pump (%)	90	%
No. of operation hours	24	hr
No. of working days in a year	365	days

Table E.3.9 Notations for design and costing variables

Variable Name	Symbol or Abbreviation	Units
Total Flowrate	Q	gal/day
Flowrate in ea. unit	Q_e	m^3/day
No. of units	n (with respective prefixes)	
Nitrification Unit (N)		
Solids/Sludge Retention Time (SRT)	θ_x	days
Hydraulic Retention Time	θ	hrs.
Influent total khjedahl nitrogen	TKN	mg NH_4^+-N/L
Effluent ammonia concentration	$S_{eff-NH4+}$	mg NH_4^+-N/L
Effluent nitrite concentration	$S_{eff-NO2}^-$	mg NO_2^--N/L
Chemical oxidation demand	COD	mg/L
Biochemical oxygen demand	BOD	mg/L
Influent Volatile suspended solids	VSS_{inf}	mg/L
Active Volatile suspended solids	VSS_a	kg VSS_a/day
Inert Volatile suspended solids	VSS_i	kg VSS_i/day
Residual Volatile suspended solids	VSS_r	kg VSS_r/day
Amount of solids decayed	VSS_d	kg VSS_A/day
Mixed Liquor Volatile Suspended Solids	MLVSS	mg/L
Food:microorganisms ratio	f_d	
Volume of reactor	V	m^3
Volume of clarifier	V_c	m^3
Area of reactor	A	m^2
Area of clarifier	A_c	m^2
Diameter of reactor	d	m
Diameter of clarifier	d_c	m
Sludge flowrate	Q_{sludge}	gal/min
Oxygen Demand for ammonia oxidation	OD1	kg OD/day
Oxygen Demand for nitrite oxidation	OD2	kg OD/day
Oxygen entering tank	O_{2IN}	kg OD/day
Oxygen exiting tank	O_{2OUT}	kg OD/day
Oxygen consumed	$O_{2CONSUMED}$	kg OD/day
Oxygen flowrate into tank	Q_O	kg/min
Recycle ratio	r	
Recycle Flowrate	Q_r	m^3/day
Depth of reactor	h	m or ft
Surface Overflow/Overloading rate	SOR	m^3/m^2.hr

Variable Name	Symbol or Abbreviation	Units
Power consumed/day	P	kW/yr.
Efficiency of pump	η	%
Solids Contact Clarifier (S)		
Volume of Inner Hopper	V_{in}	m^3
Volume of Outter Hopper	V_{out}	m^3
Surface area of inner hopper	A_{in}	m^2
Surface area of zone 2	A_2	m^2
Surface area of Outter Hopper	A_{out}	m^2
Diameter of Inner Hopper at top part (zone 1)	d_1	m
Diameter of Inner Hopper at bottom part (zone 2)	d_2	m
Diameter of Outter Hopper (zone 3)	d_3	m
Slant Height of Inner Hopper	h_s	m
Flocculation time	t_f	min
Sedimentation/Settling time	t_s	hr
Surface Loading Rate	SLR	m/hr
Depth of contact clarifier	h_c	m
Sweep area of paddle	A_s	m^2
Paddle area	A_p	m^2
Radius of paddle	r_p	m
Diameter of paddle	d_p	m
Length of paddle	L_p	m
Height of paddle	H_p	m
Coefficient of Drag for paddle	C_D	
Fluid (water) density	ρ	kg/m^3
Relative velocity of paddle	v_p	m/s
Acidification Unit and Chemical Addition (A and C)		
Width of tank	w	m
Length of tank	l	m
Chlorine residual	C_r	mg/L
Influent coliform count	C_o	#/100 mL
Effluent coliform coutn	C	MPN/100 mL
Filtration Unit (F)		
No. of filters	n_f	
Surface area of each filter	A_f	ft^2
Width of filter	w_f	ft
Length of filter	l_f	ft
Total Surface area of filter	A_t	ft^2
Volume of sand	V_s	ft^3
Volume of backwash in ea. Reservoir	V_b	gal
Surface area of each reservoir	A_b	ft^3

Variable Name	**Symbol or Abbreviation**	**Units**
Width of backwash reservoir	w_b	ft
Length of backwash reservoir	l_b	ft
Tertiary Treated Water Supply System (TTWS)		
Kinematic viscosity of water	μ	$lb/s/ft^2$
Hazen William’s Constant	C	
Headloss	H_L	ft/1000 ft
Unit cost of electricity	a	$/kWh
Efficiency of pump	E	%
Pumping Cost	PC	$/yr/ft
Cost Estimation		
Capital Recovery Factor	crf	%
Thickness of concrete Wall	T_w	ft
Thickness of concrete slab	T_s	ft

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APPENDIX F

Inventory Input Data for LCI Analysis and Supplementary Results of Toxic Release Inventory Emissions to Air, Surface Water and Land

F.1 Inventory Input Data for LCI Analysis

The inventory inputs used to obtain the emissions factors from the EIO-LCA and process-based models, the emissions and the characterization factors used to determine the emissions and impacts, respectively are provided in this appendix. Table F.1.1 provides the annual costs for respective tertiary treatment alternative, but only the infrastructure costs are used as inputs to the EIO model. Table F.1.2 provides the quantity of chemicals used for treatment and conditioning and Table F.1.3 is the national energy grid mix obtained from the U.S. Energy Information Administration (USEIA). Table F.1.4 is the simulated output of the transportation miles obtained from the EIO-LCA transportation sector for one million dollars of chemicals produced. Table F.1.5 was derived from literature for the inputs required to produce and regenerate activated carbon.

Table F.1.1 Infrastructure and O&M costs as inputs in 2009MM$/yr

Construction or O&M Parameters	RW	MWW	MWW _SF	MWW _NF	MWW _pH	MWW_ NSF	MWW_ NFG
Construction of treatment infrastructure	$0.27	$0.27	$0.51	$0.66	$0.33	$0.90	$1.02
Labor & maintenance	$0.18	$0.18	$0.44	$0.44	$0.22	$0.68	$0.68
Electricity for operation	$0.03	$0.01	$0.03	$0.14	$0.01	$0.16	$0.14
Chemical transport and manufacture for treatment	$0.00	$0.00	$0.34	$0.00	$0.71	$0.31	$0.00
Chemical transport and manufacture for conditioning	$0.72	$0.93	$0.63	$0.72	$0.93	$0.50	$0.60
Activated carbon regeneration	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.29
Water supply infrastructure and pumping		$0.67	$0.67	$0.67	$0.67	$0.67	$0.67
Raw source water	$0.85	$0.51	$0.51	$0.51	$0.51	$0.51	$0.51

Note: The rows highlighted in dark grey are not accounted for LCA analysis. The row highlighted in light grey are used as input to the EIO model

Table F.1.2 Type and dose of chemicals added during treatment and conditioning phases with reference to effluent quality

Chemical Type	Tertiary Treatment Alternative						
	RW	MWW	MWW _NF	MWW _SF	MWW _pH	MWW _NSF	MWW _NFG
Sulfuric Acid (kg/yr)					1.31E+06		
Hydrated Lime (kg/yr)				2.27E+06		8.00E+05	
Soda Ash (kg/yr)						1.31E+06	
Monochloramine (kg/yr)	6.85E+05	5.57E+05	5.57E+05	5.57E+05	6.85E+05	5.57E+05	6.85E+05
Tolytriazole (kg/yr)	8.11E-02	6.49E-02	6.49E-02	3.25E-02	8.11E-02	8.11E-02	8.11E-02
Polymaleic Acid (kg/yr)	1.35E+04	1.35E+04	N/A	N/A	1.35E+04	N/A	1.35E+04

Note: Blank cells in the table means that those chemicals are not used in the particular treatment alternative

Table F.1.3 U.S. National Electricity Mix (USEIA, 2009)

National Electricity Mix by Fuel Type	Percent By Composition
Electricity, coal	49.6
Electricity, Gas	18.8
Electricity, Hydropower	6.5
Electricity, Nuclear	19.3
Electricity, Oil	3.0
Electricity, Biomass	1.3
Others (Renewable and Unspecified)	1.5

Table F.1.4 Chemical transportation inventory data obtained from EIO-LCA for one million 2002 USD

Transported via	Mass transported for a given distance and U.S. dollar value (in ton-km/MM 2002 USD)		
	Basin inorganic	**Basic organic**	**All other chemical**
Road	1.37 x 10^6	0.75 x 10^6	0.64 x 10^6
Freight Rail	2.23 x 10^6	2.16 x 10^6	0.83 x 10^6
Water	0.5 x 10^6	0.6 x 10^6	0.2 x 10^6

Table F.1.5 Inventory for production and regeneration of 1 kg granular activated carbon

Material	Quantity	
	Activated Carbon	Regenerated Carbon
Crude Coal (kg)	2	0 (but 10% of coal combusted needs to be replaced with new activated carbon)
Natural Gas (Nm^3)	4.9	2.7
Steam (kg)	3	0.3
Hydrochloric acid (0.04)	0.04	0

Source: Adapted from Meier, 1997, taken from Ortiz, 2006

F.2 Toxic Release Inventory Emissions to Air, Water and Land

F.2.1 BTEX

Figure F.2.1 to F.2.3 present the estimated annual BTEX emissions (kg/yr) to air, surface water and land, respectively. Chemical manufacture for conditioning is a dominant contributor to air emissions and also releases moderately high emissions to both surface water and land. GAC manufacture and regeneration dominates the surface water emissions. Construction of infrastructure and electricity generation also releases BTEX emissions to land, even though at very low concentrations.

All tertiary treatment alternatives emit large amounts of BTEX to air, but MWW_pH and MWW alternatives have the higher comparative emissions since higher doses of conditioning chemicals are applied in these processes to make them suitable for reuse in recirculating cooling systems. MWW_NSF doesn't necessarily use higher dose of conditioners, but requires two types of chemicals for treatment plus conditioning chemicals are added and so the combined emissions makes it the least attractive option based on Figure F.2.1.

The coking process of coal extraction, especially for new coal produced, emits potential amounts of benzene to air, surface water and land, due to which GAC manufacture has the highest emissions especially to water and land as in Figure F.2.2 and F.2.3. Construction of infrastructure is seen to emit relatively high BTEX to land.

F.2.2 Lead and its compounds

Emissions of lead and its compounds to air, water and land are as shown in Figure F.2.4. Chemical conditioner manufacture was seen to be the main contributor of lead emissions to air and water; whereas construction of infrastructure emits lead to land/soil in larger quantities. MWW_NSF and MWW_NFG, which have greater infrastructure requirements, emit the highest amounts of lead to land, followed by MWW_SF and MWW_NF. The processes RW, MWW and MWW_pH emit lower quantities of lead to land, but have relatively high emissions to water. WSI&D is estimated to emit lead to land alone due to energy generation for piping water and for the manufacture of infrastructure for pipe distribution system.

F.2.3 Other TRI results

The remaining TRI emissions are shown in this section in kg/yr. Figure F.2.5, F.2.6 to F.2.8, F.2.9, F.2.10, F.2.11 to F.2.13 and F.2.14 show the Cadmium, Chloroform, Chromium, Cyanide, Mercury and Nickel emissions to air, surface water and land, respectively.

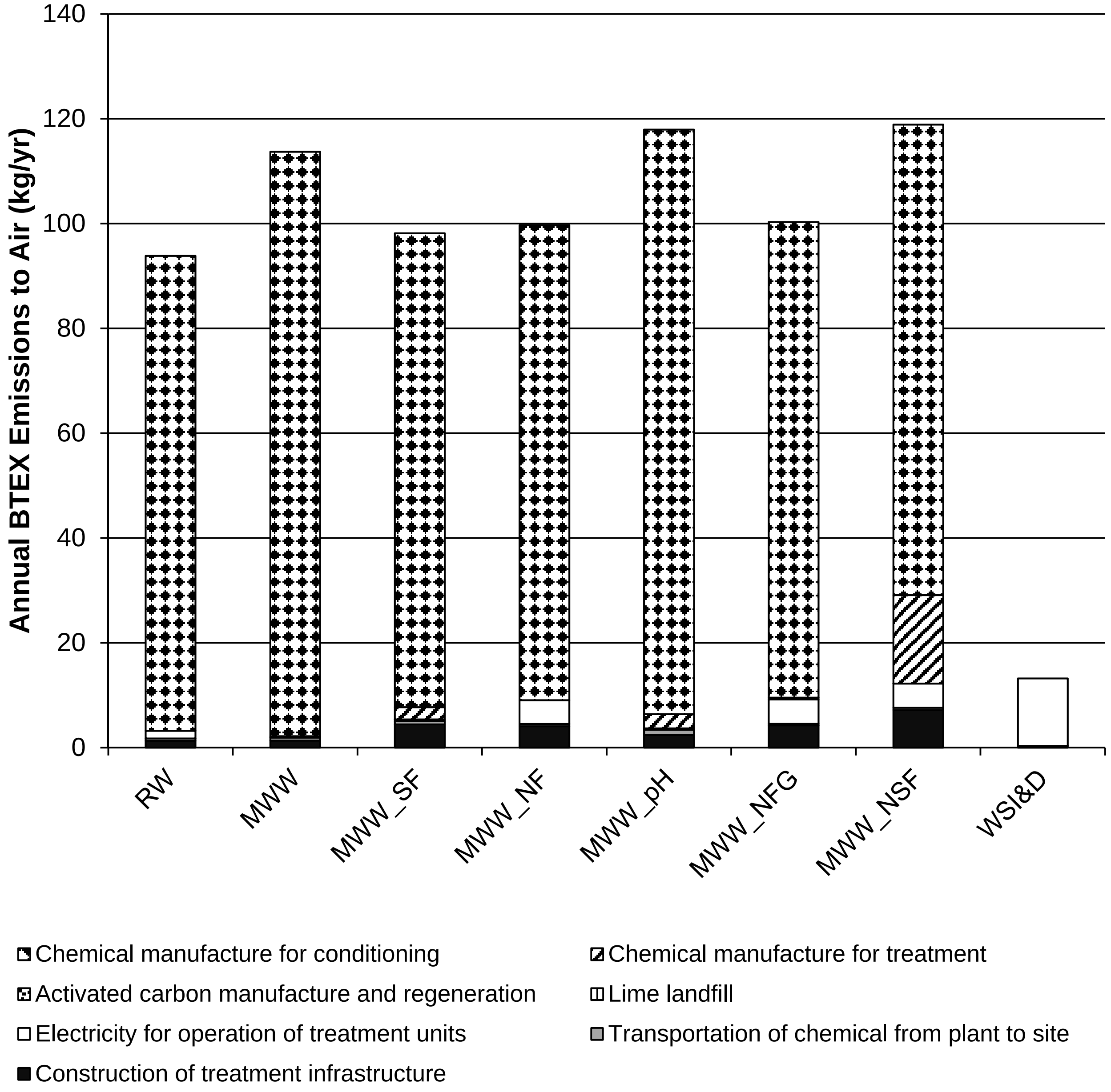

Figure F.2.1 BTEX (Benzene, Toulene, Ethylbenzene and Xylene) emissions to air (in kg/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

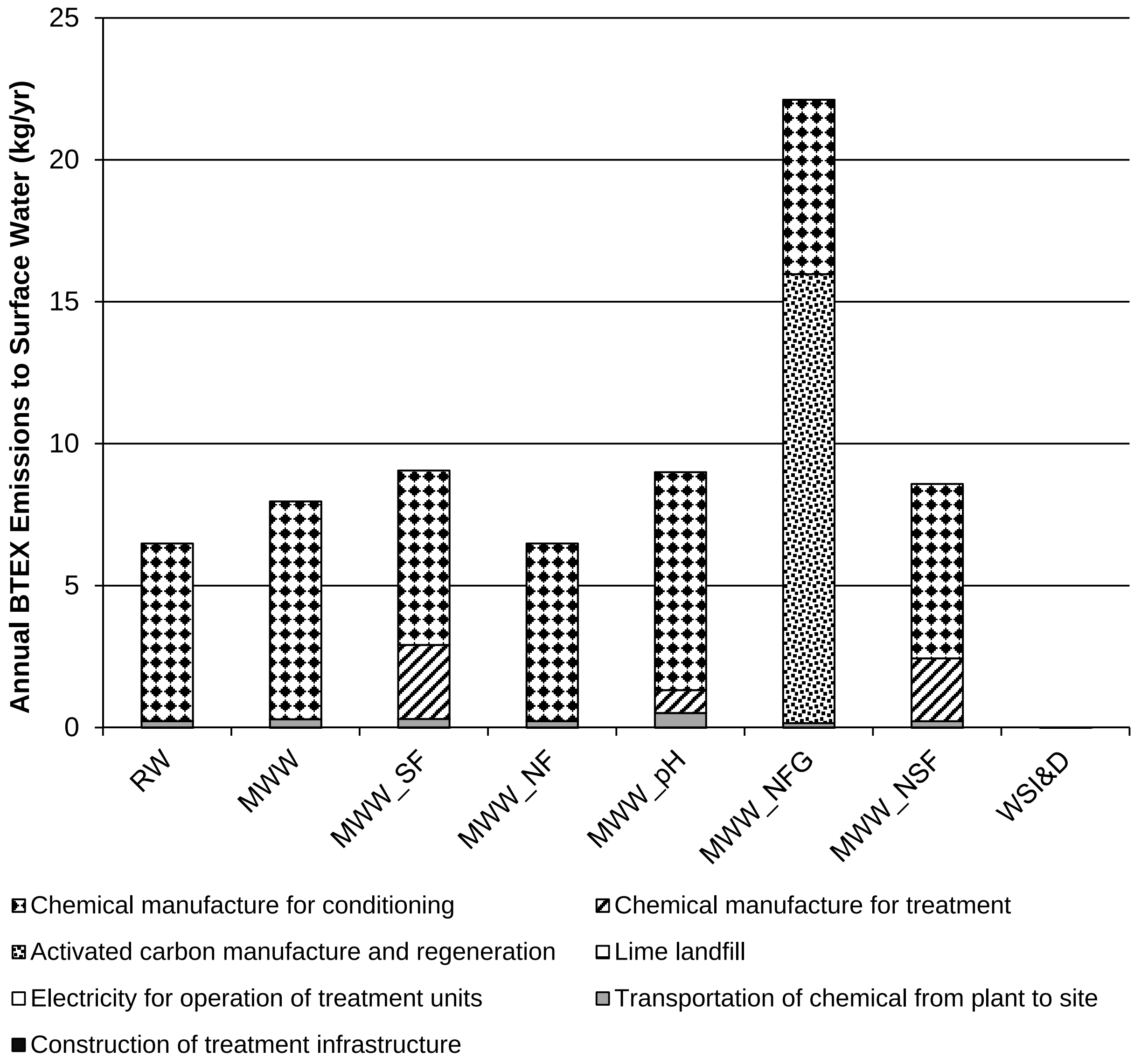

Figure F.2.2 BTEX (Benzene, Toulene, Ethylbenzene and Xylene) emissions to surface water (in kg/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

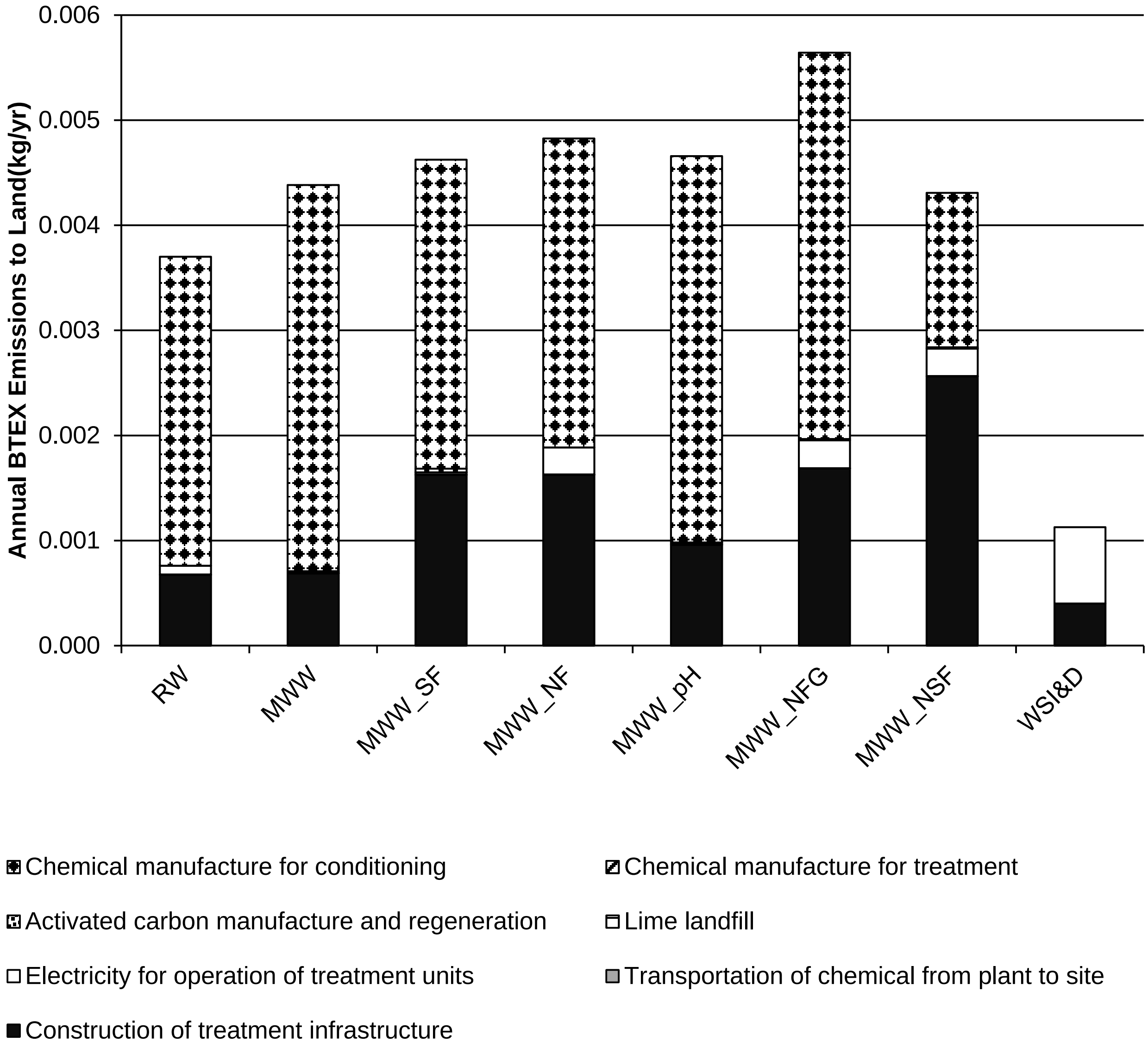

Figure F.2.3 BTEX (Benzene, Toulene, Ethylbenzene and Xylene) emissions to land/soil (in kg/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

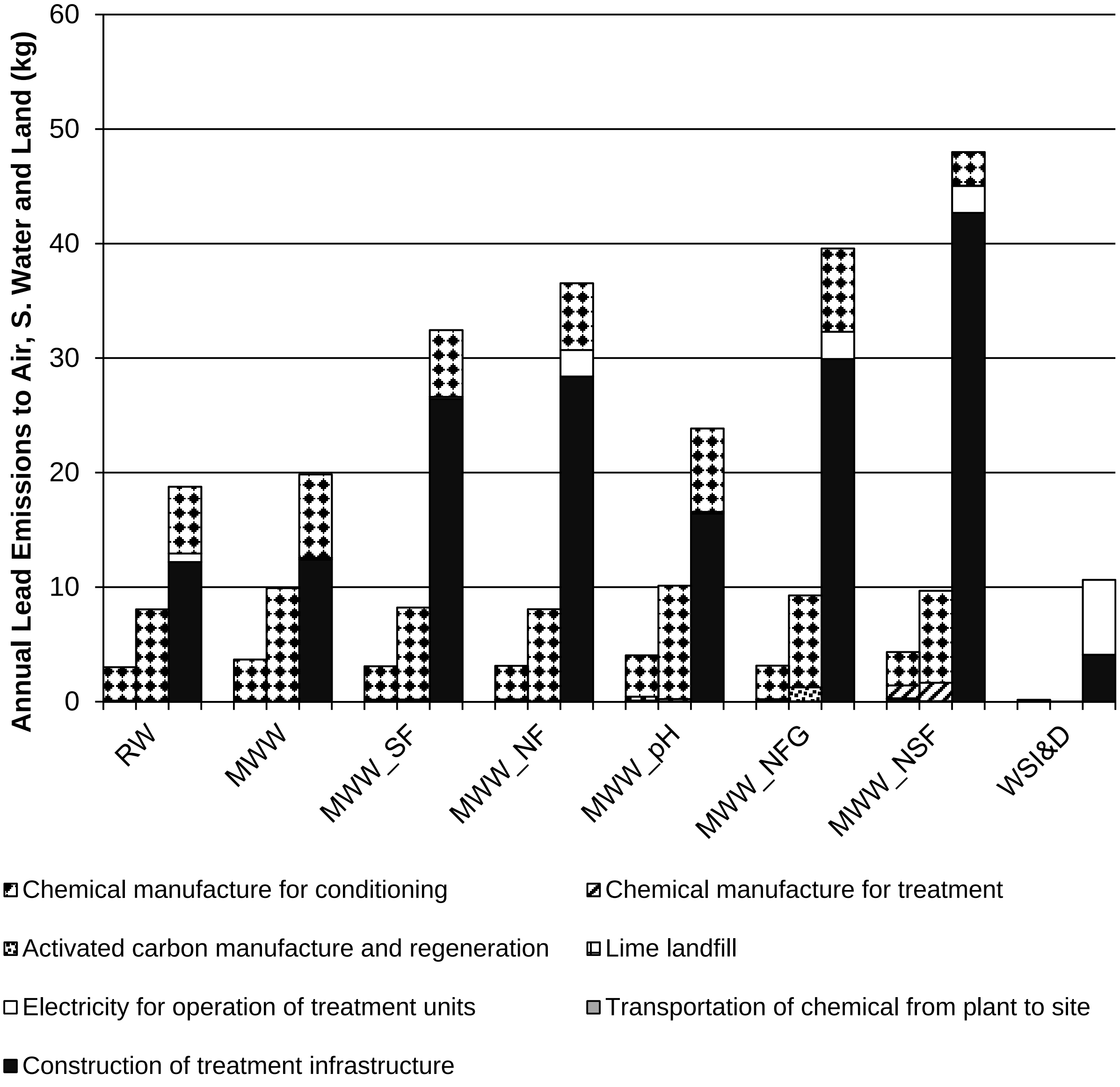

Figure F.2.4 Lead and its compounds emitted to air, surface water and land (in kg/yr) during construction and operation of various tertiary treatment processes for MWW reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

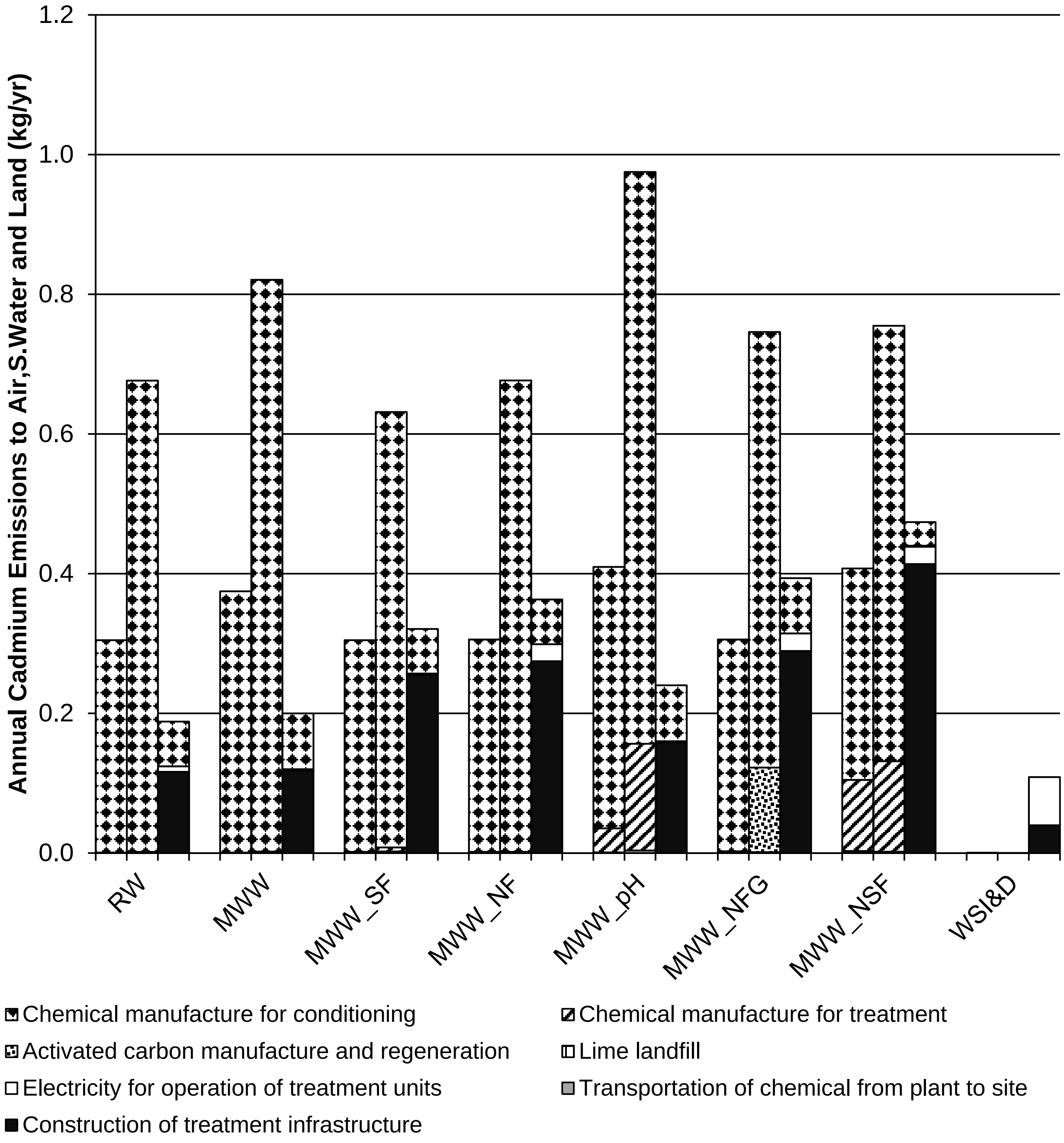

Figure F.2.5 Cadmium and its compounds emitted to air, surface water and land (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

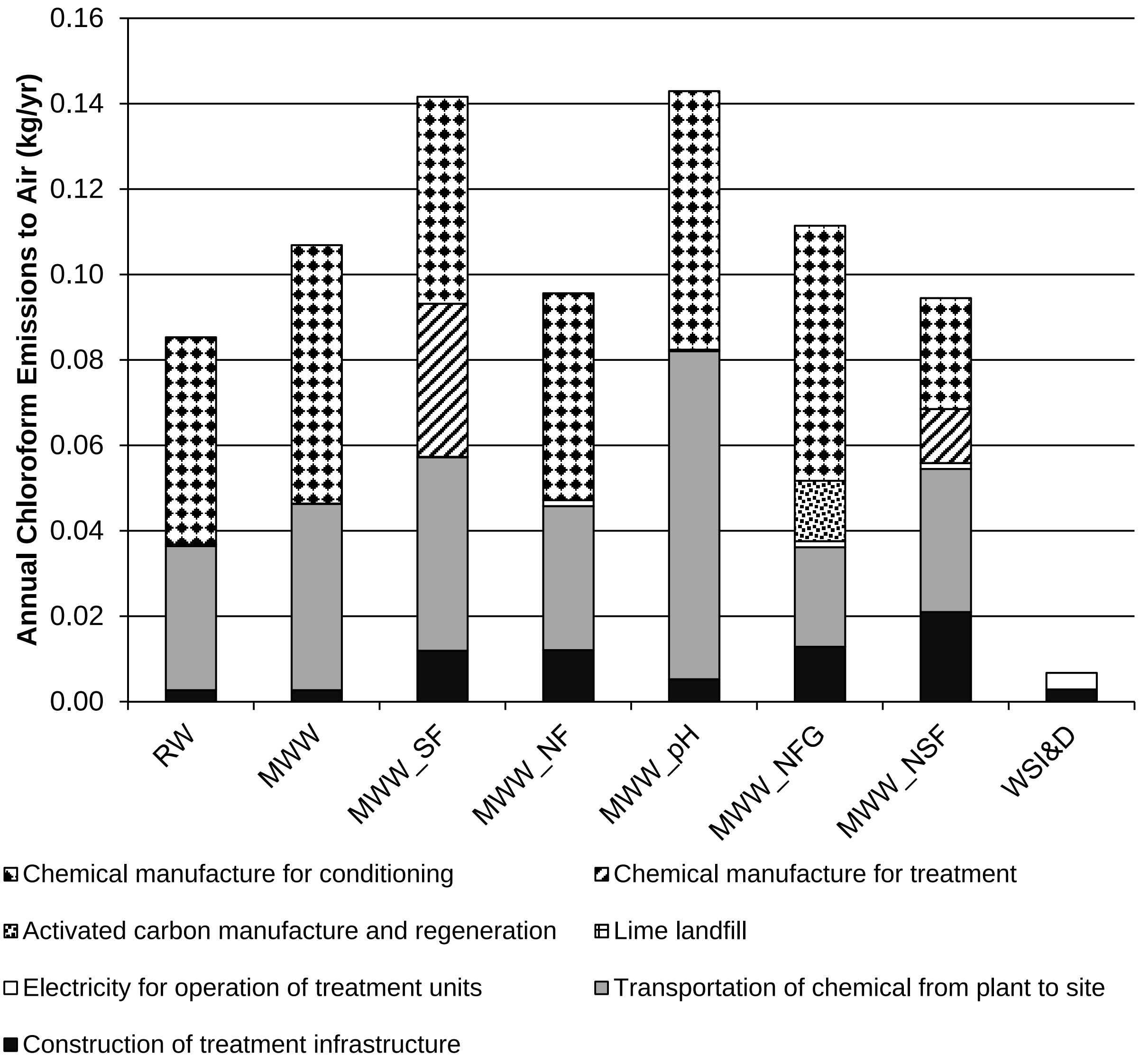

Figure F.2.6 Chloroform emissions to air (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

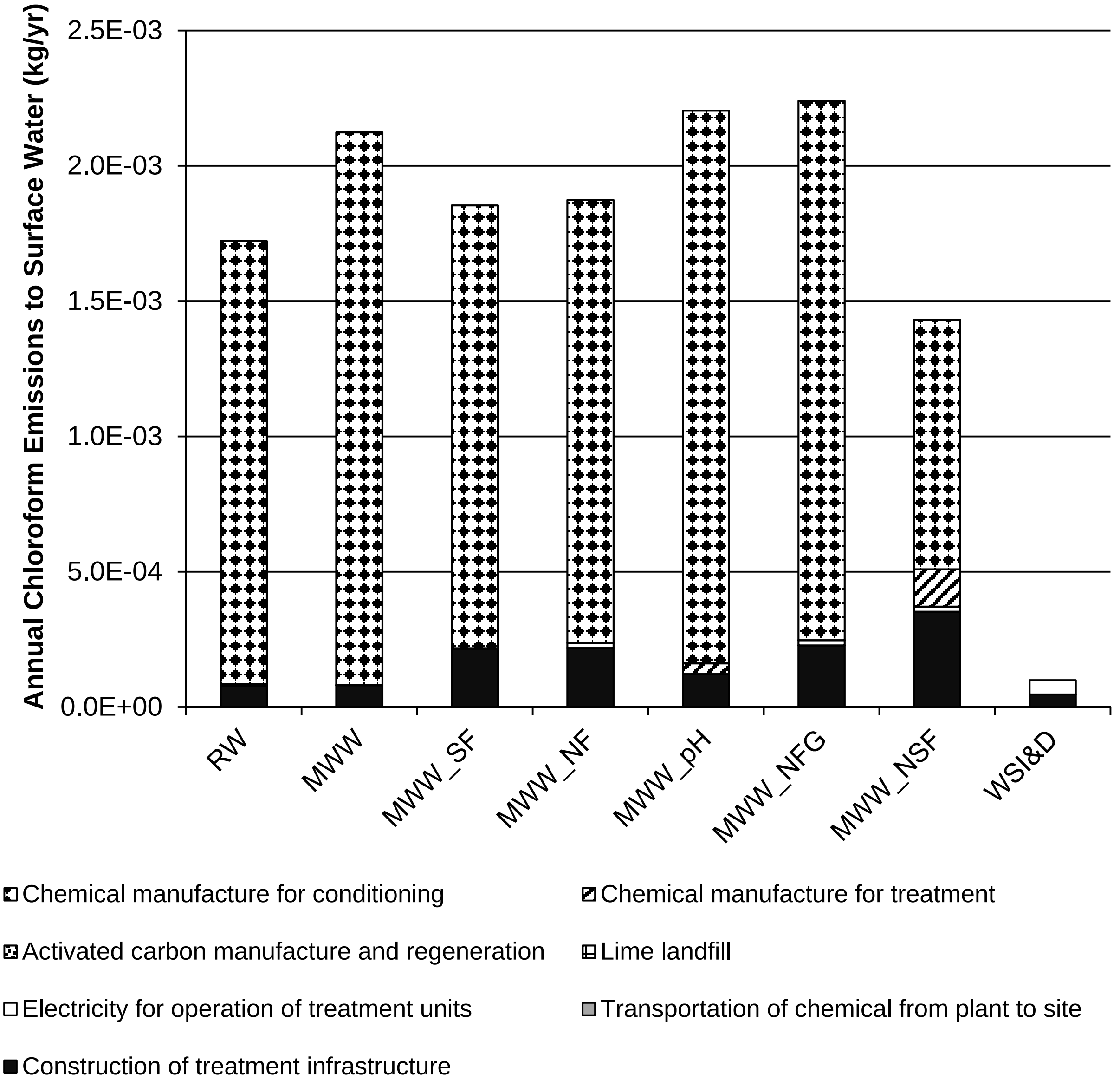

Figure F.2.7 Chloroform emissions to surface water (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

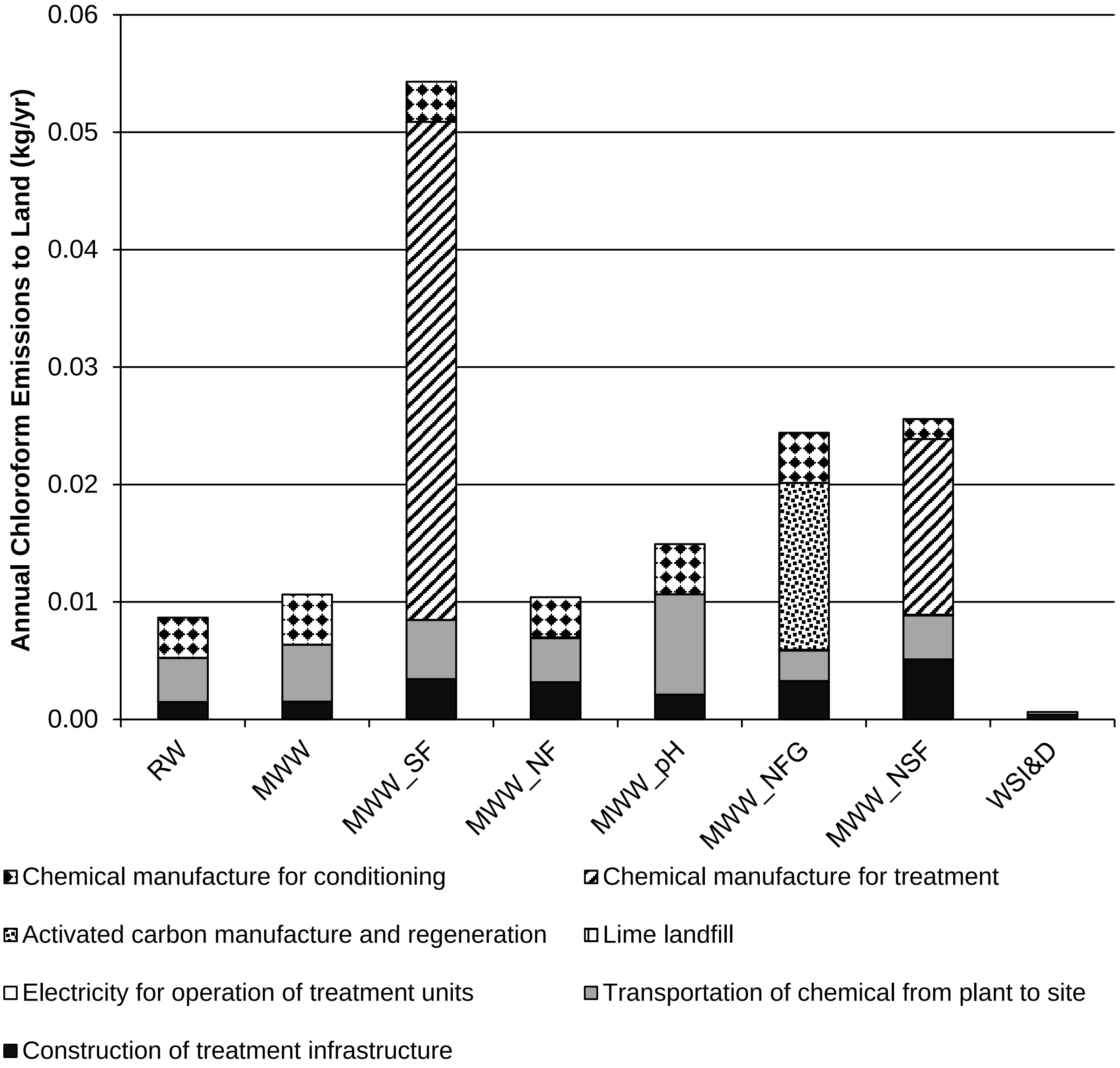

Figure F.2.8 Chloroform emissions to land/soil (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

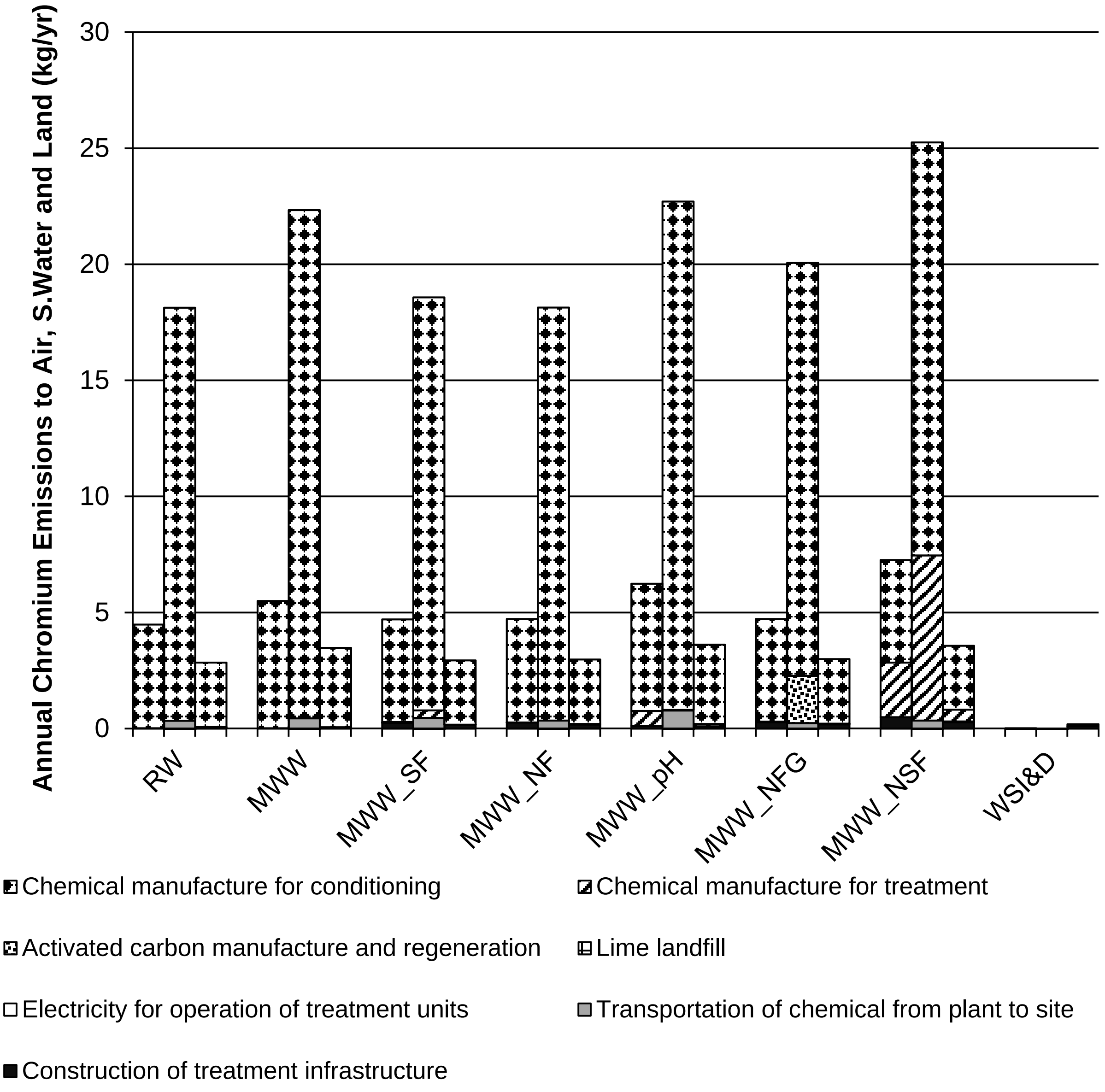

Figure F.2.9 Chromium and its compounds emitted to air, surface water and land (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

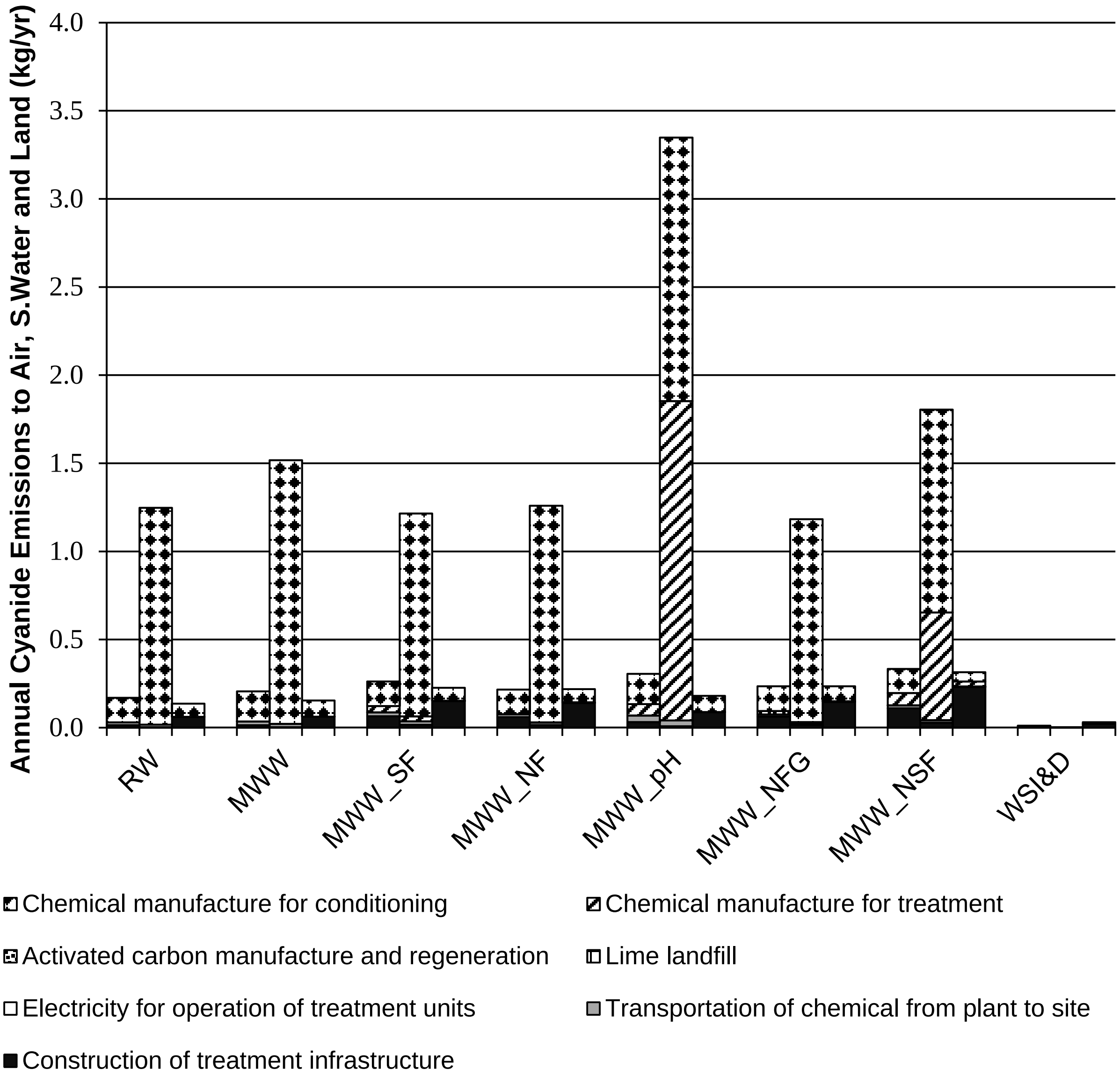

Figure F.2.10 Cyanide compounds emitted to air, surface water and land (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

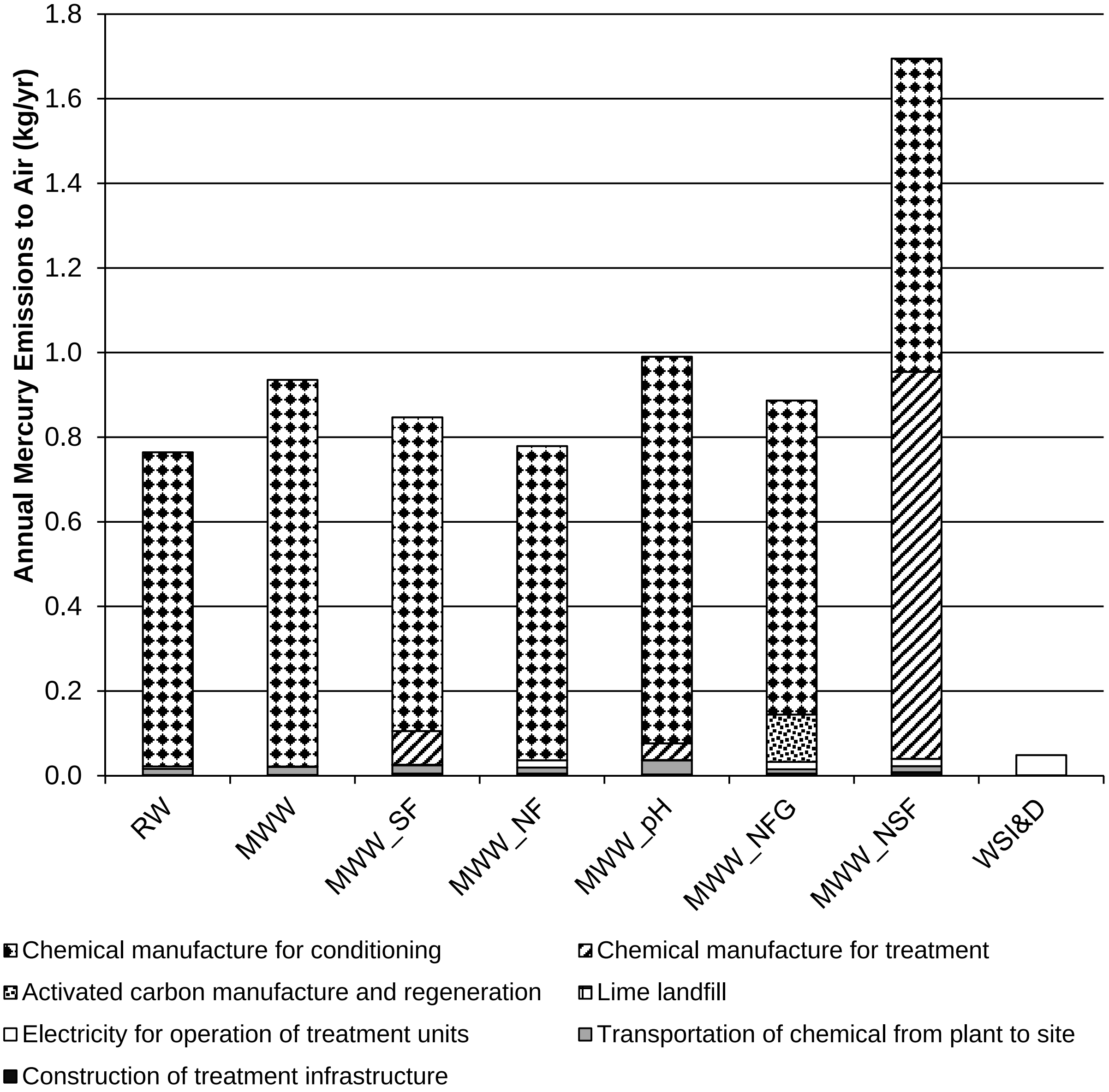

Figure F.2.11 Mercury emissions to air (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

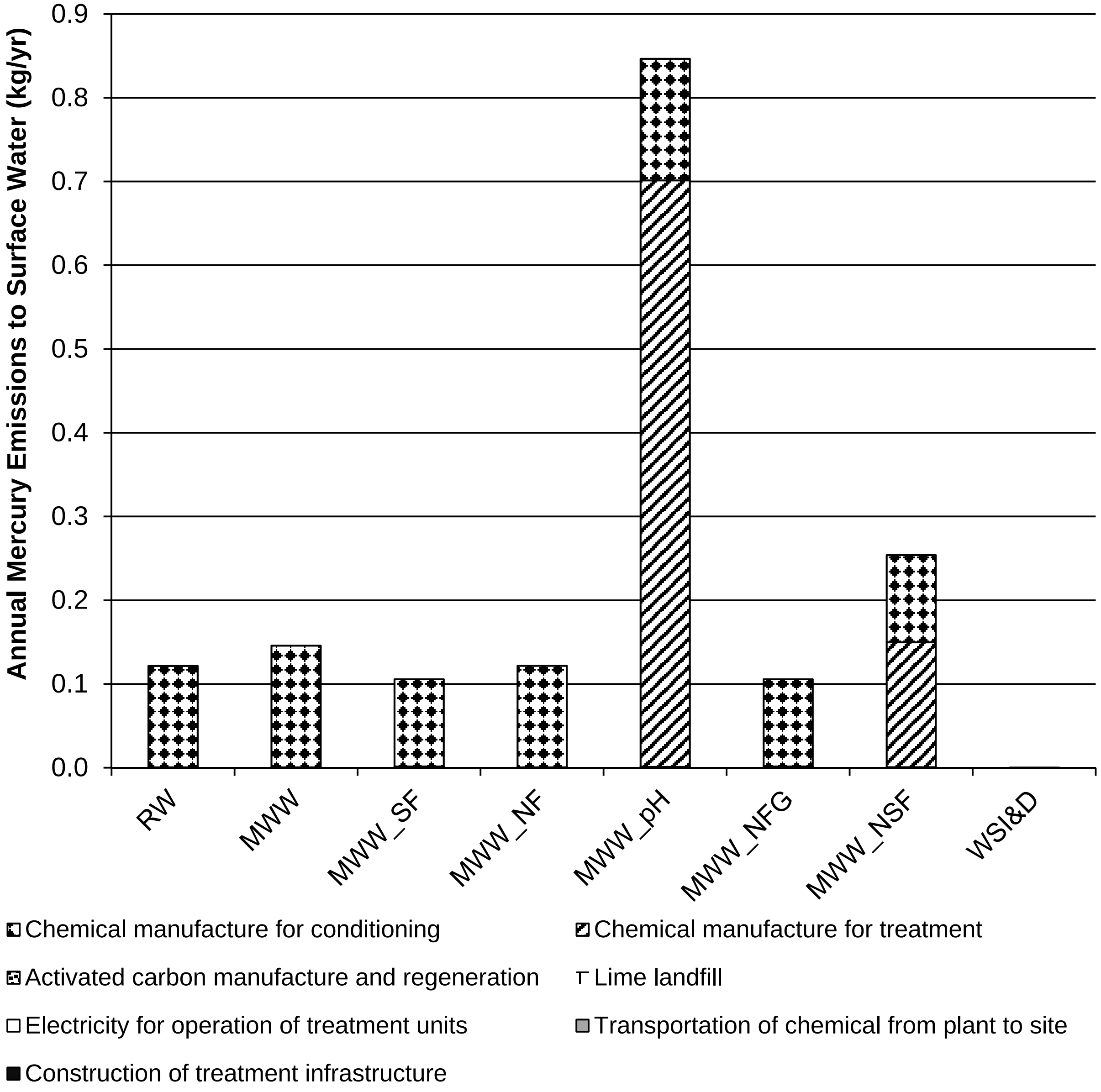

Figure F.2.12 Mercury emissions to surface water (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

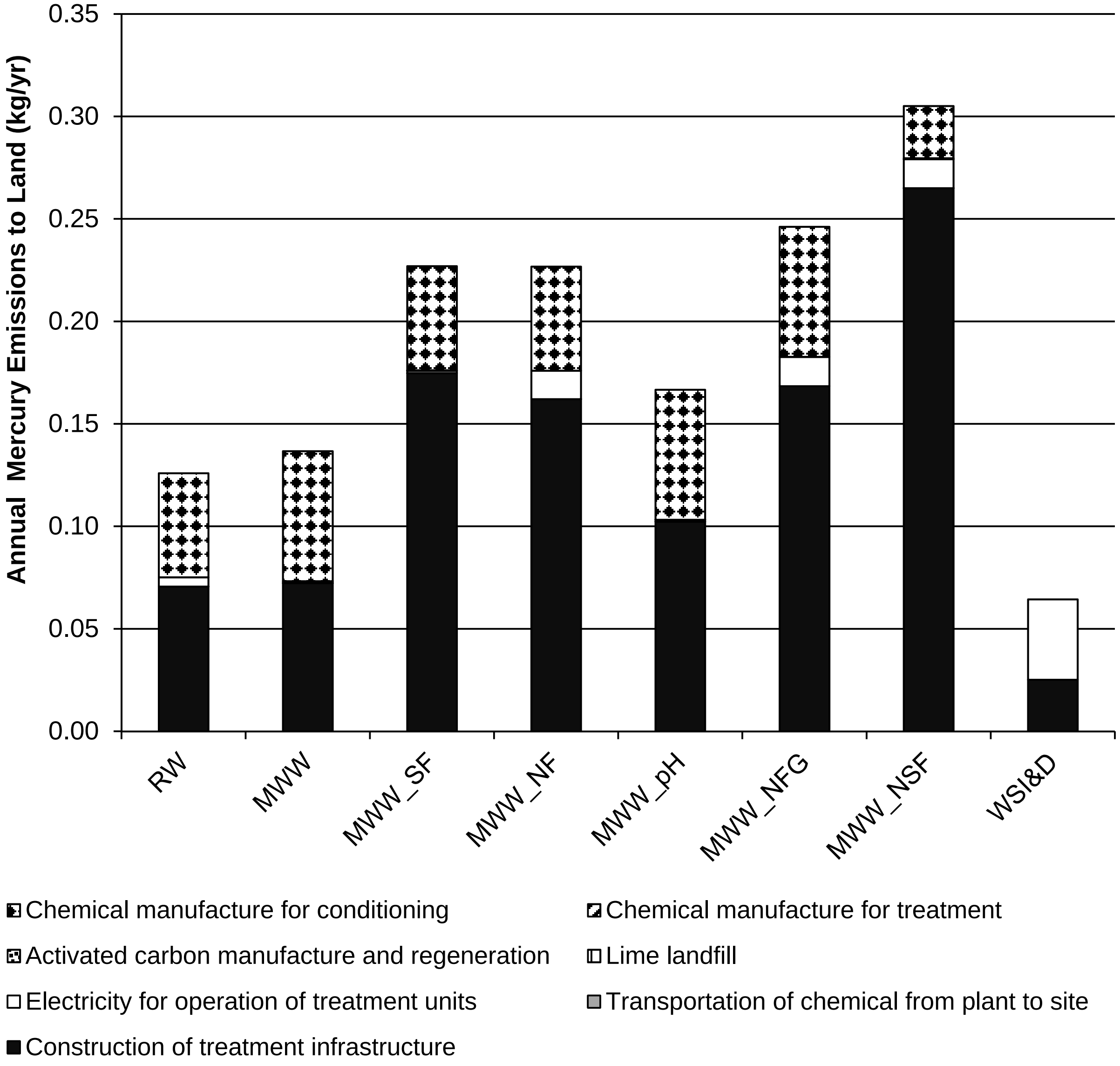

Figure F.2.13 Mercury emissions to land (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

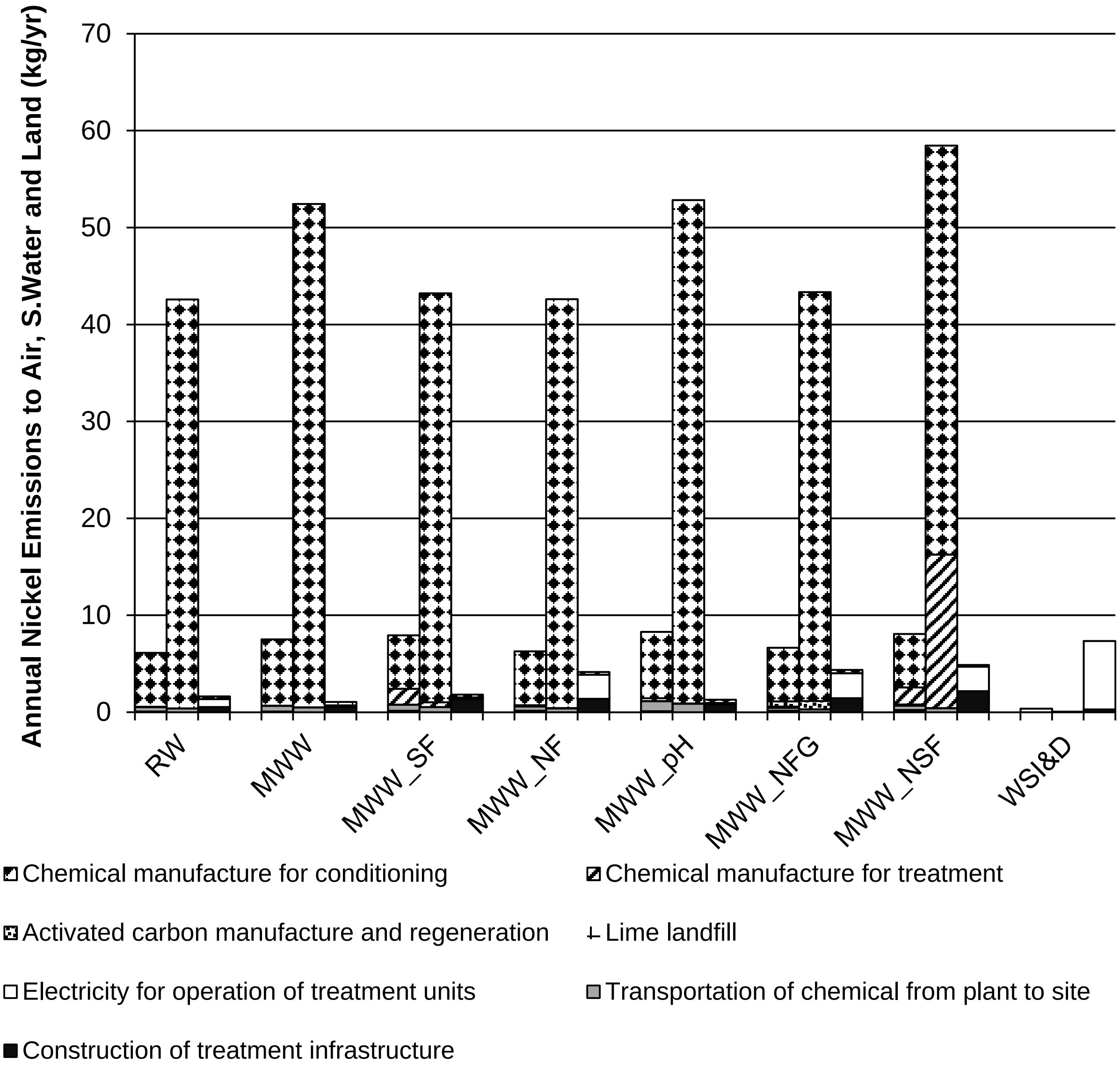

Figure F.2.14 Nickel and its compounds emitted to air, surface water and land (in kg/yr) during construction and operation of tertiary treatment of secondary municipal wastewater (MWW) for reuse in cooling systems; functional unit or treatment plant capacity is 7.75 MGD.

F.3 External Cost Assessments for Urban, Rural and All Counties in the U.S. Using APEEP County Level Social-Damage Factors

Since the location of the wastewater treatment plant is hypothetical in the analyses presented herein, and could be anywhere within the U.S., the uncertainty range of the external cost is higher because average social cost factors represented in APEEP model for approximately 3000 counties in the U. S. were used to calculate the damage-based costs. To account for the spatial variability of the social factors and reduce the existent uncertainty, three scenarios were studied which include average social cost factors for urban-only, rural-only and all counties included scenarios. The average cost factors were used for urban, rural and any national location and external costs are shown in graphs in Figures F.3.1, F.3.2 and F.3.3 represent the national average direct costs and urban, rural and all counties included external costs, respectively for all tertiary treatment alternatives chosen in study. The error bars on the figures represent the 5th and 95th percentile values for the costs

Figure F.3.4 is same as Figure 7.2.7 in Chapter 7 with additional error bars calculated using ±35% contingency for construction, labor and maintenance, chemical transport and manufacture, electricity for operation, water supply via pipe, maximum and minimum rate paid for raw source water purchase. The 5th and 95th values of ‘all counties included’ scenario external costs are also shown. From Figure F.3.4 it is seen that raw source water purchase cost introduces the highest uncertainty followed by the external costs, since both of these are dependent on spatial location of the wastewater treatment plant, and availability of reusable/freshwater.

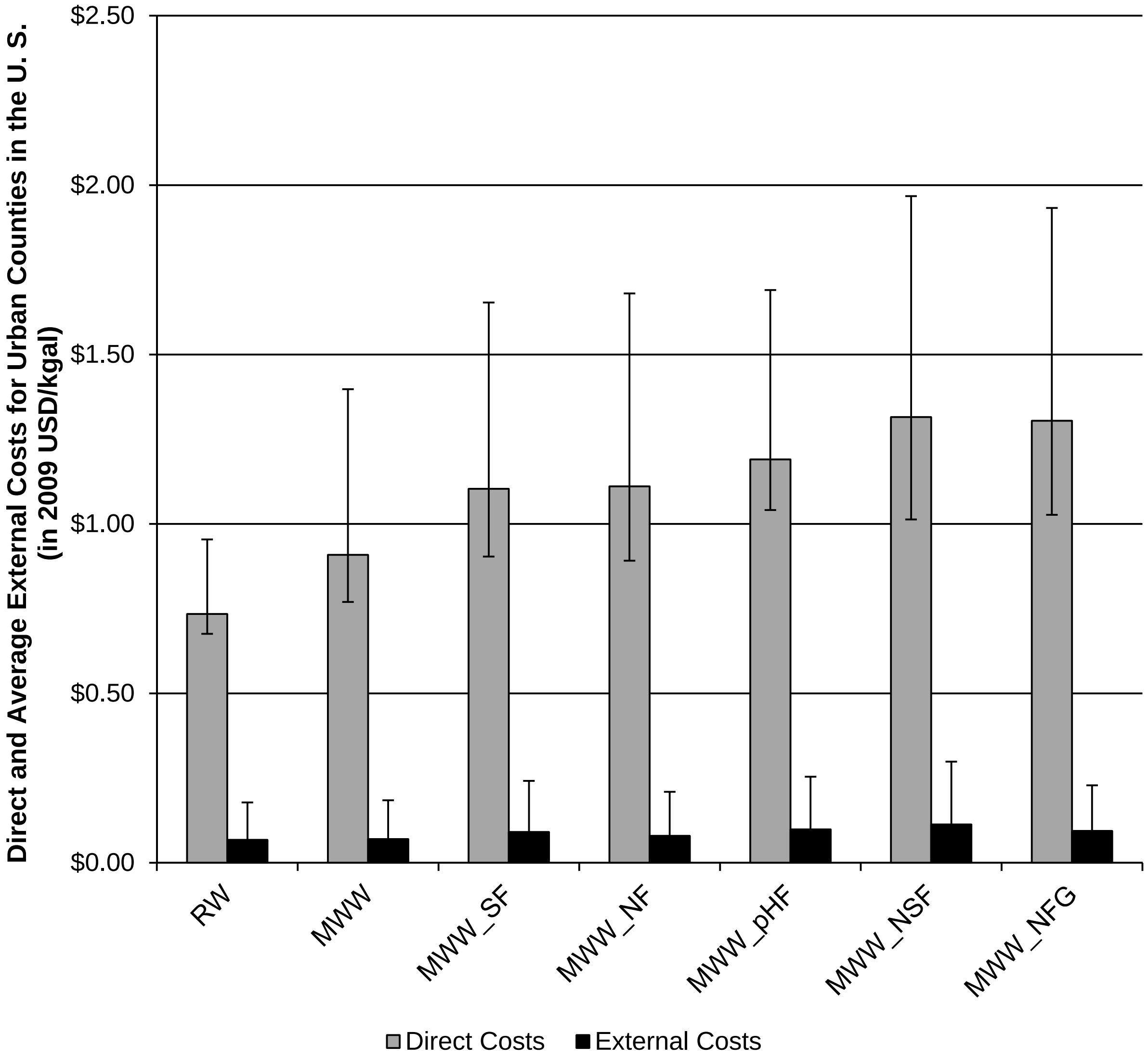

Figure 7.3.1 National average direct costs and external costs considering the hypothetical release of emissions to be in urban area of the U.S. for different tertiary treatment alternatives chosen in study. Error bars were calculated using ±35% contingency for construction, labor and maintenance, chemical transport and manufacture, electricity for operation, water supply via pipe, maximum and minimum rate paid for raw source water purchase; and 5th and 95th value of all urban counties included scenario external costs.

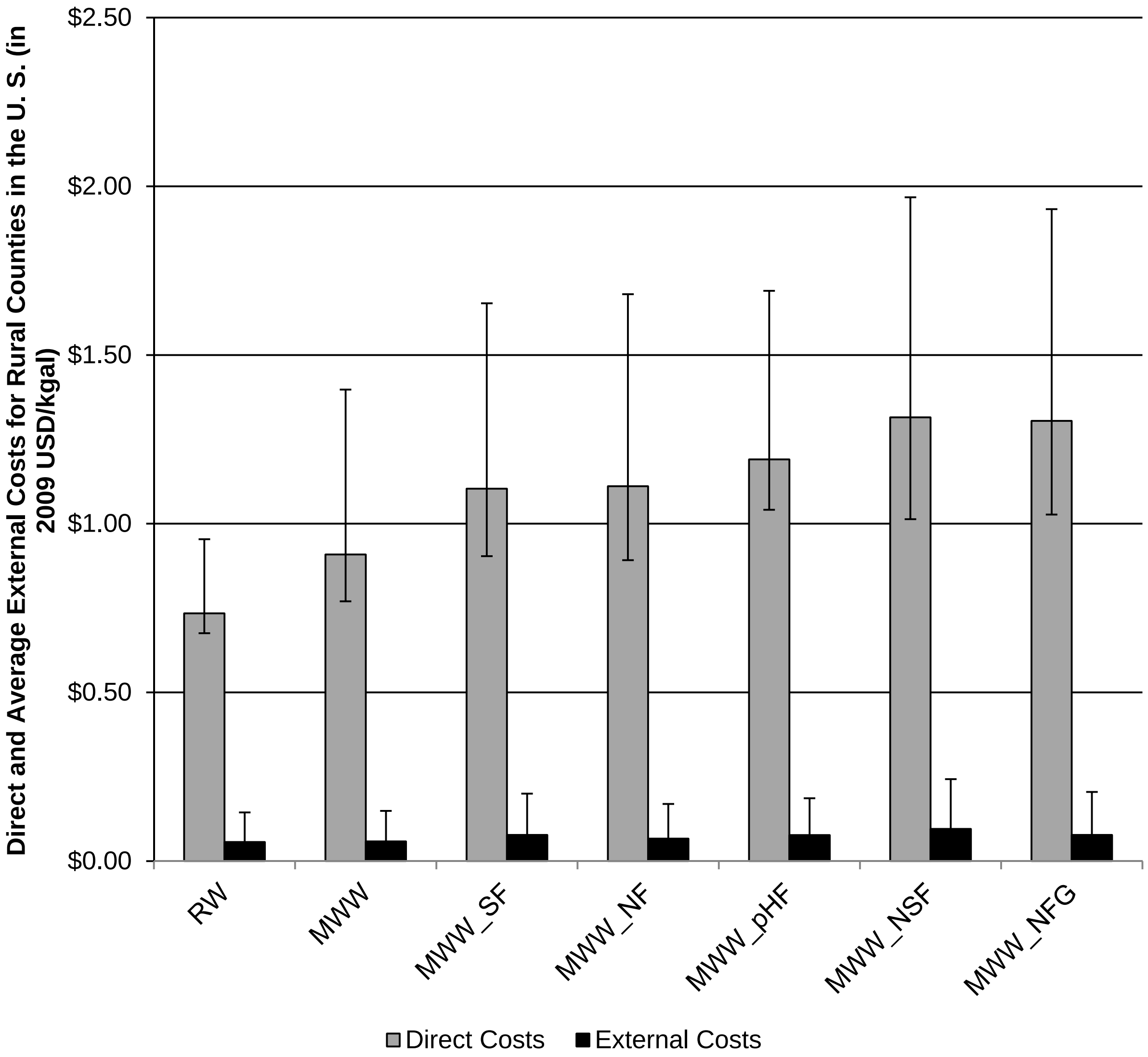

Figure 7.3.2. National average direct costs and external costs considering the hypothetical release of emissions to be in rural area of the U.S. for different tertiary treatment alternatives chosen in study. Error bars were calculated using ±35% contingency for construction, labor and maintenance, chemical transport and manufacture, electricity for operation, water supply via pipe, maximum and minimum rate paid for raw source water purchase; and 5th and 95th value of all rural counties included scenario external costs.

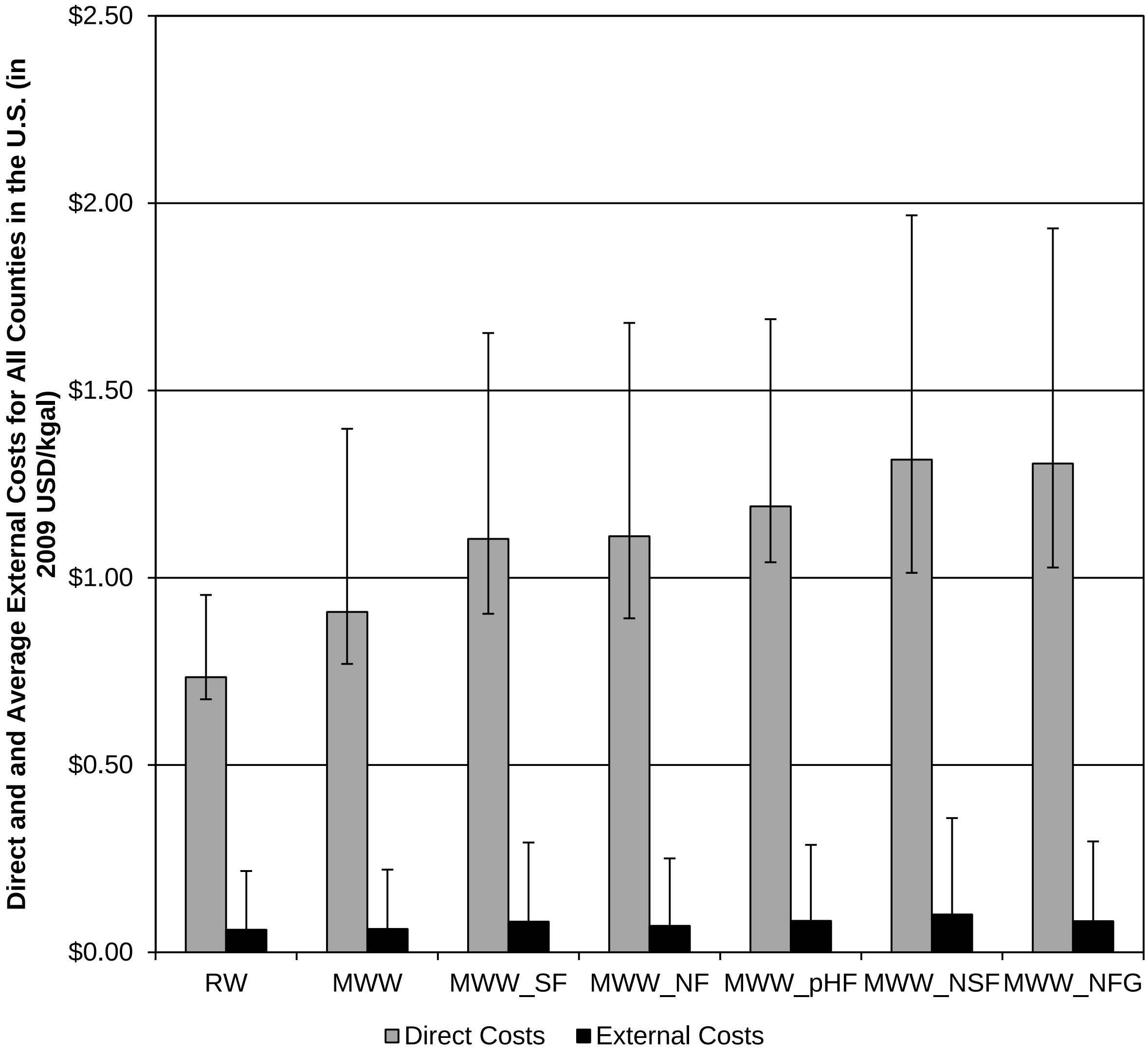

Figure 7.3.3 National average direct costs and external costs considering average cost factors across all the counties of the U.S. for different tertiary treatment alternatives chosen in study. Error bars were calculated using ±35% contingency for construction, labor and maintenance, chemical transport and manufacture, electricity for operation, water supply via pipe, maximum and minimum rate paid for raw source water purchase; and 5th and 95th value of all counties across the U.S. scenario external costs.

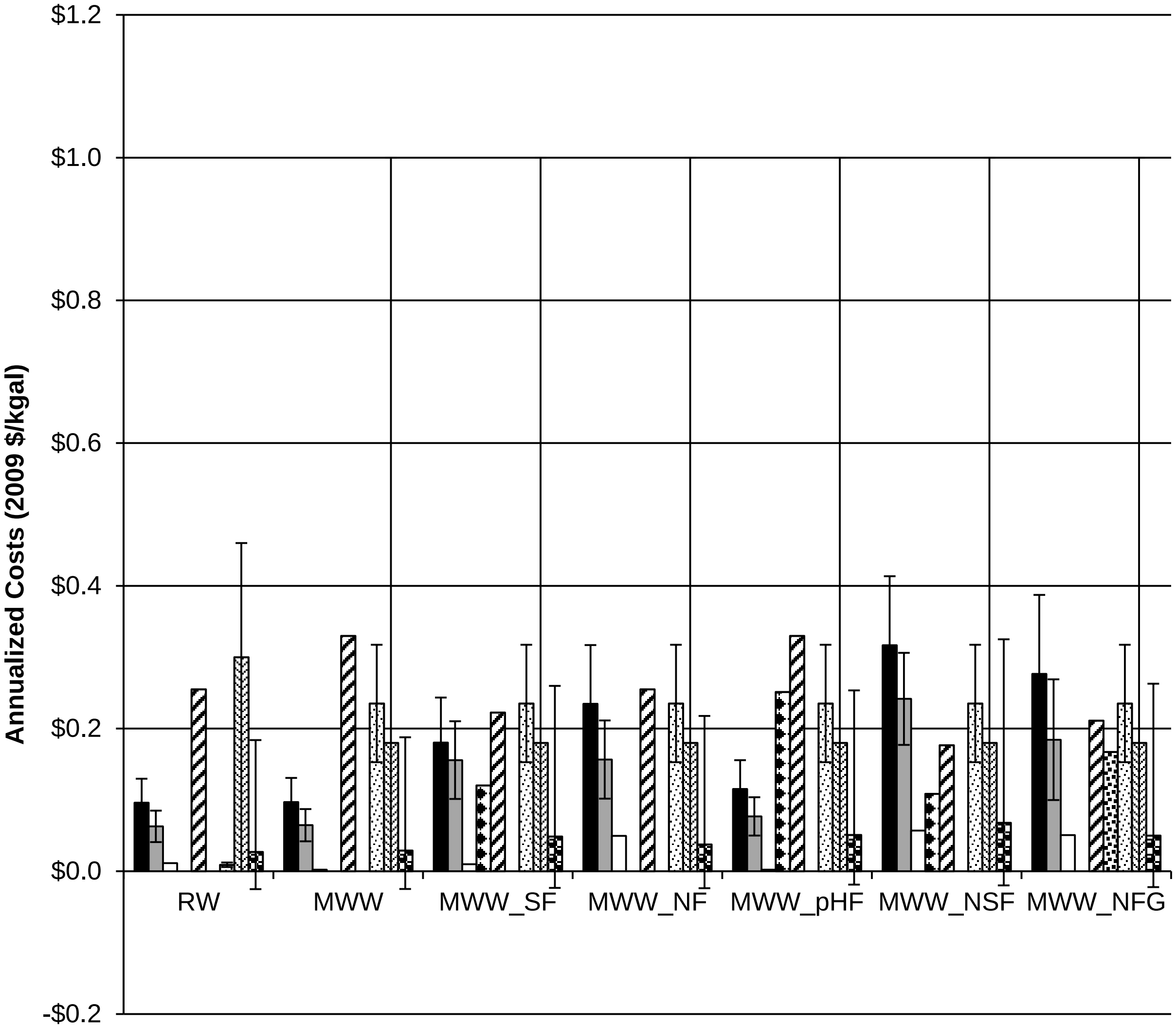

Figure 7.3.4 Classified construction, operation and maintenance and external costs (in 2009$/kgal) of tertiary treatment alternatives,. Average external damage costs per t CO_2 eq. = $30, per kg SO_2 = $5.71, per kg NO_x =$1.51 and per kg $PM_{2.5}$ =$12 were used for estimation of impact costs. Error bars were calculated using ±35% contingency for construction, labor and maintenance, chemical transport and manufacture, electricity for operation, water supply via pipe, maximum and minimum rate paid for raw source water purchase; and 5th and 95th value of all counties across the U.S. scenario external costs.

F.4 References

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