
# LONG TERM 2 ENHANCED SURFACE WATER TREATMENT RULE TOOLBOX GUIDANCE MANUAL

Office of Water (4606)
EPA 815-R-09-016
April 2010
www.epa.gov/safewater

**Purpose:**

The purpose of this guidance manual is solely to provide technical information on applying the "Toolbox" of *Cryptosporidium* treatment and management strategies that are part of the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). This guidance is not a substitute for applicable legal requirements, nor is it a regulation itself. Thus, it does not impose legally-binding requirements on any party, including EPA, states, or the regulated community. Interested parties are free to raise questions and objections to the guidance and the appropriateness of using it in a particular situation. Although this manual covers many aspects of implementing Toolbox options, the guidance presented here may not be appropriate for all situations, and alternative approaches may provide satisfactory performance. The mention of trade names or commercial products does not constitute endorsement or recommendation for use.

**Authorship:**

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## ACRONYMS

| | |
|---|---|
| AFO | Animal Feeding Operation |
| AOC | Assimilable Organic Carbon |
| AOP | Advanced Oxidation Process |
| ASTM | American Society for Testing and Materials |
| AWOP | Area Wide Optimization Program |
| AWWA | American Water Works Association |
| BMP | Best Management Practice |
| CAFO | Concentrated Animal Feeding Operation |
| CCP | Composite Correction Program |
| CFC | Chlorofluorocarbon |
| CFE | Combined Filter Effluent |
| CMOM | Capacity, Management, Operation, and Maintenance |
| CPE | Comprehensive Performance Evaluation |
| CSO | Combined Sewer Overflow |
| CSTR | Continuously Stirred Tank Reactor |
| CT | Contact Time |
| CTE | Comprehensive Technical Assistance |
| CWA | Clean Water Act |
| CWSRF | Clean Water State Revolving Fund |
| DBP | Disinfection Byproduct |
| DBPR | Disinfectants and Disinfection Byproducts Rule |
| DEM | Digital Elevation Model |
| DNA | Deoxyribonucleic Acid |
| DOP | Demonstration Of Performance |
| DWSRF | Drinking Water State Revolving Fund |
| EDTA | Ethylenediamine Tetra-acetic Acid |
| EM | Electromagnetic |
| EPA | United State Environmental Protection Agency |
| FBR | Filter Backwash Rule |
| FEMA | Federal Emergency Management Agency |
| GAC | Granular Activated Carbon |
| GIS | Geographic Information System |
| GPM | Gallons Per Minute |
| GPR | Ground Penetrating Radar |
| GROW | Geotechnical, Rock and Water Resources Library |
| GWUDI | Ground Water Under the Direct Influence of surface water |
| HAA | Haloacetic Acids |
| HDT | Hydraulic Detention Time |
| HEC-RAS | Hydrologic Engineering Centers River Analysis System |
| HUC | Hydrologic Unit Code |
| IDSE | Initial Distribution System Evaluation |

| | |
|---|---|
| IESWTR | Interim Enhanced Surface Water Treatment Rule |
| IFE | Individual Filter Effluent |
| IP | Induced Polarization |
| LRAA | Locational Running Annual Average |
| LRV | Log Removal Value |
| LT1ESWTR | Long Term 1 Enhanced Surface Water Treatment Rule |
| LT2ESWTR | Long Term 2 Enhanced Surface Water Treatment Rule |
| MCF | Membrane Cartridge Filter |
| MCL | Maximum Contaminant Level |
| MDBP | Microbial and Disinfection Byproduct |
| MF | Microfiltration |
| MGD | Million Gallons per Day |
| MPA | Microscopic Particulate Analysis |
| MRDL | Maximum Residual Disinfectant Level |
| MS4 | Municipal Separate Storm Sewer System |
| MST | Microbial Source Tracking |
| MTBE | Methyltertiarybutylether |
| NFF | National Flood Frequency |
| NGWA | National Groundwater Association |
| NHDES | New Hampshire Department of Environmental Services |
| NMP | Nutrient Management Plan |
| NOM | Natural Organic Matter |
| NPDES | National Pollutant Discharge Elimination System |
| NRCS | Natural Resource Conservation Service |
| NSF | National Science Foundation |
| NTNCWS | Nontransient Noncommunity Water Systems |
| NTU | Nephelometric Turbidity Unit |
| ORP | Oxidation-Reduction Potential |
| OSHA | Occupational Safety and Health Administration |
| PCR | Polymerase Chain Reaction |
| PEL | Permissible Exposure Limit |
| POTW | Publically Owned Treatment Works |
| PWS | Public Water System |
| QA | Quality Assurance |
| QC | Quality Control |
| RAA | Running Annual Average |
| RNA | Ribonucleic Acid |
| RO | Reverse Osmosis |
| SCD | Streaming Current Detector |
| SDWA | Safe Drinking Water Act |
| SOC | Synthetic Organic Compound |
| SOP | Standard Operating Procedure |
| SP | Self Potential |
| SPDES | State Pollutant Discharge Elimination System |

| | |
|---|---|
| SRF | State Revolving Fund |
| SSO | Sanitary Sewer Overflow |
| SSRC | Spores of Sulfide Reducing Clostridia |
| SSS | Specific System Study |
| SWAP | Source Water Assessment Program |
| SWP | Source Water Protection |
| SWTR | Surface Water Treatment Rule |
| TCE | Trichloroethylene |
| TEM | Transient Electromagnetic |
| THM | Trihalomethane |
| TMDL | Total Maximum Daily Load |
| TNCWS | Transient Noncommunity Water Systems |
| TNTC | Too Numerous To Count |
| TOC | Total Organic Carbon |
| TTHM | Total Trihalomethanes |
| UF | Ultrafiltration |
| UFRV | Unit Filter Run Volume |
| USDA | United States Department of Agriculture |
| USGS | United States Geological Survey |
| UV | Ultraviolet light |
| WCP | Watershed Control Program |
| WSC | Watershed Control |
| WTP | Water Treatment Plant |

# 1. Introduction

In establishing drinking water regulations for microbial and disinfection byproduct (M-DBP) control, the U.S. Environmental Protection Agency (EPA) is promoting a multi-barrier approach for treating drinking water. A multi-barrier treatment process provides a number of protective "layers" against contamination by using more than one method of prevention and treatment to remove or inactivate microorganisms and minimize disinfection byproducts (DBPs). To that end, EPA is publishing this guidance to help public water systems (PWSs) choose appropriate combinations of treatment processes for compliance with the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR).

The LT2ESWTR focuses on improved control of microbial contamination, specifically the protozoan parasite *Cryptosporidium*. Differing from previous drinking water microbial regulations, the LT2ESWTR requirements for each system are based on the PWS's vulnerability to contamination, as measured by the occurrence of *Cryptosporidium* in the source water. This "Microbial Framework" strategy stems from a recognition that only some systems may need to provide additional protection against *Cryptosporidium* and that such decisions should be made on a system-specific basis.

With this approach, systems serving 10,000 people or more initially conduct source water monitoring to determine average *Cryptosporidium* concentrations (small filtered systems serving less than 10,000 people can first monitor for *E. coli* to determine if *Cryptosporidium* monitoring is required unless the state notifies them otherwise). Based on their monitoring results, systems are classified into different categories (or bins). The bins indicate the additional *Cryptosporidium* treatment requirements, if any, that must be met to comply with the rule. Systems required to provide additional treatment will choose from a "toolbox" of options consisting of treatment technologies, process optimization techniques, and management techniques to meet the requirements. Thus, this approach requires enhanced *Cryptosporidium* treatment for systems with higher vulnerability to *Cryptosporidium* contamination and provides several options for those systems to achieve compliance. These options are described in this manual.

## 1.1 Guidance Manual Objectives

The primary objectives of this manual are to provide guidance to PWSs for selecting appropriate microbial toolbox options and achieving compliance for each option. To accomplish these objectives, this manual will describe each toolbox option in terms of achieving *Cryptosporidium* treatment credit(s) and discuss design and operational issues.

## 1.2 Guidance Manual Organization

This manual consists of fourteen chapters and five appendices:

| | |
|---|---|
| Chapter 1 | Introduction - The remainder of this chapter provides a regulatory history and then summarizes key provisions of the LT2ESWTR including minimum requirements for each toolbox option. |
| Chapters 2 – 14 | Toolbox Options - These chapters describe each toolbox option and how systems can implement these options to achieve the associated *Cryptosporidium* treatment credit. Where applicable, basic design criteria are recommended to achieve a given log removal. Each chapter contains its own list of references. |
| Appendix A | Site Specific Determination of Contact Time for Chlorine Dioxide and Ozone - describes the different elements of a site specific study to generate a set of chlorine dioxide or ozone CT values for that site and discusses some of the issues involved in the statistical analysis of the results. |
| Appendix B | Ozone CT Methods - describes the Segmented Flow Analysis and Extended-CSTR methods to calculate the CT inactivation credits with ozone. |
| Appendix C | Measuring Ozone Residual - discusses ozone residual sample collection, measurement, and online ozone residual analyzer calibration. |
| Appendix D | Derivation of Extended CSTR Equations - provides the derivation of the equation used to calculate k*. |
| Appendix E | Tracer Test Data Development and Analysis – describes how to conduct and analyze the results of tracer tests to determine the contact time for CT calculations. |
| Appendix F | Watershed Control Best Management Practices (BMPs) and Case Studies - provides a list of programmatic resources and guidance available to assist systems in building partnerships and implementing watershed protection activities. |
| Appendix G | Review Criteria for Use By States When Reviewing Watershed Control (WSC) Program Plans – provides a list of assessment criteria for use by states when reviewing WCP plans. |

## 1.3 Regulatory History

The following sections describe the predecessors to the LT2ESWTR. Section 1.3.5 summarizes key requirements of the Stage 2 Disinfectants and Disinfection Byproducts Rule

(DBPR), which was promulgated simultaneously with the LT2ESWTR to balance the risks between DBPs and microbial pathogens.

### 1.3.1 Surface Water Treatment Rule

Under the 1989 Surface Water Treatment Rule (SWTR) (54 FR 27486 June 29, 1989), EPA established treatment requirements for all PWSs using surface water or ground water under the direct influence of surface water (GWUDI) as a source. The requirements are intended to protect against the adverse health effects associated with *Giardia lamblia*, viruses, and *Legionella* and include the following:

- Maintenance of a disinfectant residual in water entering and within the distribution system.
- Removal/inactivation of at least 99.9 percent (3-log) of *Giardia* and 99.99 percent (4-log) of viruses.
- Filtration, unless systems meet specified avoidance criteria.
- For filtered systems, a turbidity limit for the combined filter effluent of 5 nephelometric turbidity units (NTUs) at any time and a limit of 0.5 NTU in 95 percent of measurements each month for treatment plants using conventional treatment or direct filtration (with separate standards for other filtration technologies). These requirements were superseded by the 1998 IESWTR and the 2002 LT1ESWTR.
- Watershed control programs and water quality requirements for unfiltered systems.

### 1.3.2 Interim Enhanced Surface Water Treatment Rule

The Interim Enhanced Surface Water Treatment Rule (IESWTR) (63 FR 69478 December 16, 1998) applies to PWSs serving at least 10,000 people and using surface water or GWUDI as a source. These systems were to comply with the IESWTR by January 2002. The requirements and guidelines include:

- Removal of 99 percent (2-log) of *Cryptosporidium* for systems that provide filtration.
- For treatment plants using conventional treatment or direct filtration, a turbidity performance standard for the combined effluent of filters of 1 NTU as a maximum and 0.3 NTU as a maximum in 95 percent of monthly measurements, based on 4-hour monitoring (these limits supersede the SWTR turbidity limits).
- Continuous monitoring of individual filter effluent turbidity in conventional and direct filtration plants and recording of turbidity readings every 15 minutes.

- A disinfection benchmark to assess the level of microbial protection provided before facilities change their disinfection practices to meet the requirements of the Stage 1 DBPR.

- Inclusion of *Cryptosporidium* in the definition of GWUDI and in the watershed control requirements for unfiltered PWSs.

- All new finished water reservoirs must be covered.

### 1.3.3 Stage 1 Disinfectants and Disinfection Byproducts Rule

Pursuant to requirements under the Safe Drinking Water Act (SDWA), EPA developed interrelated regulations to control microbial pathogens and disinfectants/DBPs in drinking water. These rules, collectively known as the M-DBP rules, are intended to address complex risk trade-offs between the two different types of contaminants. EPA promulgated the IESWTR concurrently with the Stage 1 DBPR so that systems could coordinate their responses to the risks posed by DBPs and microbial pathogens.

The 1998 Stage 1 DBPR (63 FR 69390 December 16, 1998) applies to all community water systems (CWSs) and nontransient noncommunity water systems (NTNCWSs) that add a chemical disinfectant to their water. Certain requirements in the rule also apply to transient noncommunity water systems (TNCWSs). Surface water and GWUDI systems serving at least 10,000 people were required to comply with the rule by January 2002. All other systems (including ground water systems) were required to comply by January 2004.

The Stage 1 DBPR sets maximum residual disinfectant levels (MRDLs) for chlorine, chloramines, and chlorine dioxide; and maximum contaminant levels (MCLs) for total trihalomethanes (TTHM), haloacetic acids (HAA5), bromate, and chlorite. The MRDLs and MCLs, except those for chlorite and chlorine dioxide, are calculated as running annual averages (RAAs).

### 1.3.4 Long Term 1 Enhanced Surface Water Treatment Rule

The Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) (67 FR 1811 January 14, 2002) was promulgated in 2002 and extends most of the requirements of the IESWTR to surface water and GWUDI systems serving fewer than 10,000 people.

### 1.3.5 Stage 2 Disinfectant and Disinfection Byproduct Rule

The requirements of the Stage 2 DBPR (71 FR 388 January 4, 2006) apply to all CWSs and NTNCWSs that add a disinfectant other than ultraviolet light (UV), or that deliver water that has been treated with a disinfectant other than UV. The Stage 2 DBPR builds on the 1998 Stage 1 DBPR by requiring reduced levels of DBPs in distribution systems. Major components of the rule are described below.

*Initial Distribution System Evaluations*

For many systems, compliance monitoring will be preceded by an Initial Distribution System Evaluation (IDSE) to identify Stage 2 DBPR compliance monitoring locations that represent distribution system sites with high TTHM and HAA5 levels. The IDSE consists of either standard monitoring or a system specific study (SSS). NTNCWSs serving fewer than 10,000 people are not required to perform an IDSE, and other systems may receive waivers from the IDSE requirement.

*Compliance with Stage 2 DBPR MCLs*

The numerical MCLs for the Stage 2 DBPR are the same as for the Stage 1 DBPR MCLs: 0.080 milligrams per liter (mg/L) for TTHM, and 0.060 mg/L for HAA5. The Stage 2 DBPR is designed to reduce high TTHM and HAA5 in the distribution system by changing compliance monitoring and calculation requirements. Compliance determination for the Stage 2 DBPR is based on a locational running annual average (LRAA) (i.e., compliance must be met at *each* monitoring location) instead of the system-wide RAA used under the Stage 1 DBPR.

*Routine Monitoring Requirements*

EPA has adopted a population-based monitoring approach for the Stage 2 DBPR, where compliance and IDSE monitoring requirements are based only on source water type and retail population served. This is a change from the plant-based approach used in the 1979 TTHM rule and the Stage 1 DBPR.

*Operational Evaluations*

Because Stage 2 DBPR MCL compliance for some systems is based on individual DBP measurements at a location averaged over a four-quarter period, a system could find higher TTHM or HAA5 levels than the MCL values, while at the same time maintaining compliance with the Stage 2 DBPR. This is because the high concentration could be averaged with lower concentrations at a given location. For this reason, the Stage 2 DBPR includes a provision for "operational evaluations" as follows:

A system has exceeded an operational evaluation level at any monitoring location when the sum of the two previous quarters' compliance monitoring results plus twice the current quarters result, divided by 4, exceeds 0.080 mg/L for TTHM or 0.060 mg/L for HAA5.

If an operational evaluation level is exceeded, the system must conduct an "operational evaluation" and submit a written report of the evaluation to the state.

## 1.4 Overview of the Long Term 2 Enhanced Surface Water Treatment Rule

The LT2ESWTR (71 FR 654 January 5, 2006) applies to all PWSs that use surface water or GWUDI (referred to collectively as "surface water systems" in this manual). It builds on the SWTR, IESWTR, and the LT1ESWTR by improving control of microbial pathogens, specifically *Cryptosporidium*. It requires filtered systems to monitor their source water for *Cryptosporidium*, and based on the results, to meet one of four levels of treatment for *Cryptosporidium* (with the first level requiring no additional treatment). Treatment requirements will be reassessed in the future based on a second round of source water monitoring under the current rule. For those systems that do not already provide filtration, the LT2ESWTR has specific requirements to inactivate two or three logs of *Cryptosporidium*, depending on source water monitoring results. It also requires systems with uncovered finished water reservoirs either to cover the reservoirs or to provide additional treatment to the reservoir effluent.

The next several sections provide a summary of PWS requirements for the LT2ESWTR, including a summary of microbial toolbox options in Section 1.4.3. Section 1.5 provides the implementation timeline for the rule.

### 1.4.1 Monitoring and Treatment Requirements for Filtered Systems

The LT2ESWTR requires most filtered PWSs to conduct source water monitoring to determine average *Cryptosporidium* concentrations. Based on the monitoring results, filtered PWSs must calculate an initial *Cryptosporidium* bin concentration for each plant for which monitoring was required [40 CFR 141.710]. Detailed requirements and guidance on how to determine source water *Cryptosporidium* bin concentrations are provided in the *Source Water Monitoring Guidance Manual for Public Water Systems for the Final Long Term 2 Enhanced Surface Water Treatment Rule*, finalized in 2006 and available online at http://www.epa.gov/safewater/disinfection/lt2/compliance.html

Exhibit 1.1 presents the bin classifications and their corresponding additional treatment requirements for all filtered systems. The treatment requirements are based on a determination that conventional, slow sand, and diatomaceous earth filtration plants in compliance with the IESWTR or LT1ESWTR achieve an average of 3-log removal of *Cryptosporidium*. EPA has determined that direct filtration plants achieve an average 2.5-log removal of *Cryptosporidium* (their removal is less than in conventional filtration because they lack a sedimentation process).

## Exhibit 1.1 Bin Classification and Additional Treatment Requirements for Filtered Systems[1]

| If your *Cryptosporidium* concentration (oocysts/L) is... | Your bin classification is... | And if you use the following filtration treatment in full compliance with existing regulations, then your *additional* treatment requirements are... | | | |
|---|---|---|---|---|---|
| | | Conventional Filtration | Direct Filtration | Slow Sand or Diatomaceous Earth Filtration | Alternative Filtration Technologies |
| < 0.075 | 1 | No additional treatment | No additional treatment | No additional treatment | No additional treatment |
| ≥ 0.075 and < 1.0 | 2 | 1-log treatment[2] | 1.5-log treatment[2] | 1-log treatment[2] | As determined by the state[2,4] |
| ≥ 1.0 and < 3.0 | 3 | 2-log treatment[3] | 2.5-log treatment[3] | 2-log treatment[3] | As determined by the state[3,5] |
| ≥ 3.0 | 4 | 2.5-log treatment[3] | 3-log treatment[3] | 2.5-log treatment[3] | As determined by the state[3,6] |

[1]40 CFR 141.710 and 40 CFR 141.711.

[2]Systems may use any technology or combination of technologies from the microbial toolbox.

[3]Systems must achieve at least 1-log of the required treatment using ozone, chlorine dioxide, UV, membranes, bag/cartridge filters, or bank filtration.

[4]Total *Cryptosporidium* treatment must be at least 4.0-log.

[5]Total *Cryptosporidium* treatment must be at least 5.0-log.

[6]Total *Cryptosporidium* treatment must be at least 5.5-log.

The LT2ESWTR requires systems to comply with additional treatment requirements by using one or more management or treatment techniques from the microbial toolbox of options. A description of the microbial toolbox options and basic requirements for achieving inactivation credit for each are provided in Section 1.4.3.

### 1.4.2 Monitoring and Treatment Requirements for Unfiltered Systems

All existing requirements for unfiltered PWSs remain in effect, including disinfection to achieve at least 3-log inactivation of *Giardia* and 4-log inactivation of viruses and to maintain a disinfectant residual in the distribution system. The LT2ESWTR requires 2- or 3- log inactivation of *Cryptosporidium*, depending on the source water concentrations of *Cryptosporidium* as shown in Exhibit 1.2. Detailed requirements and guidance on how to determine source water *Cryptosporidium* concentrations are provided in the *Source Water Monitoring Guidance Manual for Public Water Systems for the Final Long Term 2 Enhanced*

*Surface Water Treatment Rule*, finalized in 2006 and available online at http://www.epa.gov/safewater/disinfection/lt2/compliance.html

### Exhibit 1.2 LT2ESWTR Treatment Requirements for Unfiltered Systems[1]

| Average *Cryptosporidium* Concentration (oocysts/liter) | Additional *Cryptosporidium* Inactivation Requirements |
|---|---|
| ≤ 0.01 | 2-log[2] |
| > 0.01 | 3-log[2] |

[1]40 CFR 141.712.

[2]Overall disinfection requirements must be met with a minimum of two disinfectants.

Unfiltered systems must use chlorine dioxide, ozone, or UV to meet the *Cryptosporidium* inactivation requirements in Exhibit 1.2 and must meet overall disinfection requirements (i.e., *Cryptosporidium*, *Giardia*, and virus inactivation) with a minimum of two disinfectants [40 CFR 141.712 (d)]. Each of the two disinfectants must achieve by itself the total inactivation required for one of the three pathogen types.

### 1.4.3 Summary of Microbial Toolbox Options

Systems receive LT2ESWTR treatment credits by meeting conditions for the microbial toolbox options presented in Exhibit 1.3 [40 CFR 141.715]. Systems may use a combination of toolbox options to achieve the required log treatment. The intent of the toolbox is to provide systems with flexibility in selecting cost-effective LT2ESWTR compliance strategies. Unfiltered as well as filtered systems are eligible for treatment credits for the microbial toolbox options unless otherwise indicated in the table. Unfiltered systems must use one of the inactivation/disinfection tools in the toolbox.

## Exhibit 1.3 Microbial Toolbox Options with Available Log Credits[1]

| Toolbox Option | *Cryptosporidium* Treatment Credit with Design and Implementation Criteria |
|---|---|
| **Source Toolbox Components** | |
| **Watershed control program** | 0.5-log credit for state approved program comprising required elements, annual program status report to the state, and regular watershed survey. Unfiltered systems are not eligible for credit. See 40 CFR 141.716 (a) and Chapter 2 of this manual for specific criteria. |
| **Alternative source/ intake management** | No presumptive credit. Systems may conduct simultaneous monitoring for treatment bin classification at alternative intake locations or under alternative intake management strategies. See 40 CFR 141.716(b) and Chapter 3 of this manual for specific criteria. |
| **Pre-Filtration Toolbox Components** | |
| **Presedimentation basin with coagulation** | 0.5-log credit during any month that presedimentation basins achieve a monthly mean reduction of 0.5-log or greater in turbidity or alternative state-approved performance criteria. To be eligible, basins must be operated continuously with coagulant addition and all plant flow must pass through the basin. See 40 CFR 141.717(a) and Chapter 5 of this manual for specific criteria. |
| **Two-stage lime softening** | 0.5-log credit for two-stage softening where chemical additional and hardness precipitation occur in both stages. All plant flow must pass through both stages. Single-stage softening is credited as equivalent to conventional treatment. See 40 CFR 141.717(b) and Chapter 6 of this manual for specific criteria. |
| **Bank filtration** | 0.5-log credit for 25-foot setback; 1.0-log credit for 50-foot setback; aquifer must be unconsolidated sand containing at least 10 percent fines; average turbidity in wells must be less than 1 NTU. Systems using wells followed by filtration when conducting source water monitoring must sample the well to determine bin classification and are not eligible for additional credit. See 40 CFR 141.717(c) and Chapter 4 of this manual for specific criteria |
| **Treatment Performance Toolbox Components** | |
| **Combined filter performance** | 0.5-log credit for combined filter effluent turbidity less than or equal to 0.15 NTU in at least 95 percent of measurements each month. See 40 CFR 141.718 (a) and Chapter 7 of this manual for specific criteria. |
| **Individual filter performance** | 0.5-log credit (in addition to 0.5-log combined filter performance credit) if individual filter effluent turbidity is less than or equal to 0.15 NTU in at least 95 percent of samples each month in each filter and is never greater than 0.3 NTU in two consecutive measurements in any filter. See 141.718 (b) and Chapter 7 of this manual for specific criteria. |
| **Demonstration of performance** | Credit awarded to unit process or treatment train based on a demonstration to the state with a state-approved protocol. See 40 CFR 141.718 (c) and Chapter 12 of this manual for specific criteria. |

| Additional Filtration Toolbox Options | |
|---|---|
| **Bag or cartridge filters (individual filters)** | Up to 2-log credit based on the removal efficiency demonstrated during challenge testing with a 1.0-log factor of safety. See 40 CFR 141.719(a) and Chapter 8 of this manual for specific criteria. |
| **Bag or cartridge filters (in series)** | Up to 2.5-log credit based on the removal efficiency demonstrated during challenge testing with a 0.5-log factor of safety. See 40 CFR 141.719(a) and Chapter 8 of this manual for specific criteria. |
| **Membrane filtration** | Log credit equivalent to removal efficiency demonstrated in challenge test for device if supported by direct integrity testing. See 40 CFR 141.719(b) and Chapter 14 of this manual for specific criteria. |
| **Second stage filtration** | 0.5-log credit for second separate granular media filtration stage if treatment train includes coagulation prior to first filter. See 40 CFR 141.719 (c) and Chapter 9 of this manual for specific criteria. |
| **Slow sand filters** | 2.5-log credit as a secondary filtration step; 3.0-log credit as a primary filtration process. No prior chlorination for either option. See 40 CFR 141.719(d) and Chapter 9 of this manual for specific criteria. |
| **Inactivation Toolbox Components** | |
| **Chlorine dioxide** | Log credit based on measured CT in relation to CT table. See 40 CFR 141.720 (b) and Chapter 10 of this manual for specific criteria. |
| **Ozone** | Log credit based on measured CT in relation to CT table. See 40 CFR 141.720 (b) and Chapter 11 of this manual for specific criteria. |
| **UV** | Log credit based on measured CT in relation to CT table. See 40 CFR 141.720 (d) and Chapter 13 of this manual for specific criteria. |

[1] 40 CFR 141.715.

### 1.4.4 Requirements for PWSs with Uncovered Finished Water Reservoirs

The LT2ESWTR requires PWSs with uncovered finished water storage facilities to either cover the storage facility or treat the discharge of the storage facility that is distributed to consumers to achieve inactivation and/or removal of 4-log virus, 3-log *Giardia*, and 2-log *Cryptosporidium* [40 CFR 141.714].

### 1.4.5 Disinfection Profiling and Benchmarking Requirements

The LT2ESWTR includes a disinfection profile and benchmark requirement to ensure that any significant change in disinfection, whether for byproduct control under the Stage 2 DBPR, improved *Cryptosporidium* control under the LT2ESWTR, or both, does not significantly compromise existing *Giardia* and virus protection. A disinfection profile is a graphical representation of a system's level of *Giardia* and viral inactivation measured during the course of one or more year(s). A benchmark is the lowest monthly average of microbial inactivation during the disinfection profile period.

The profiling and benchmarking requirements under the LT2ESWTR are similar to those promulgated under the IESWTR and LT1ESWTR and are applicable to systems making a significant change to their disinfection practice. The LT2ESWTR defines significant change as follows:

- Changes to the point of disinfection.
- Changes to the disinfectant used at the treatment plant.
- Changes to the disinfection process.
- Any other modification identified by the state as a significant change to the disinfection practice. [40 CFR 141.708(b)].

Prior to changing the disinfection practice, the systems must notify the state and include the following information:

- A completed disinfection profile and disinfection benchmark for *Giardia lamblia* and viruses as described in 40 CFR 141.709.
- A description of the proposed change in disinfection practice.
- An analysis of how the proposed change will affect the current level of disinfection. [40 CFR 141.708(a)].

Detailed guidance for conducting a disinfection profile and calculating a benchmark is provided in the *IESWTR Disinfection Profiling and Benchmarking Guidance Manual* for systems serving at least 10,000 people and the *LT1ESWTR Disinfection Profiling and Benchmarking Technical Guidance Manual* for systems serving less than 10,000 people. Both manuals are available on-line at http://www.epa.gov/safewater/mdbp/implement.html and http://www.epa.gov/safewater/mdbp/lt1eswtr.html, respectively.

## 1.5 LT2ESWTR Implementation Schedule

The LT2 Rule defines four compliance schedules, which are based on the population served by systems as summarized in Exhibit 1.4. Wholesale PWSs must comply with Stage 2 DBPR and LT2ESWTR requirements based on the population of the largest PWS in the combined distribution system. This approach will ensure that PWSs have the same compliance schedule under both the LT2ESWTR and Stage 2 DBPR. Although consecutive systems without their own source are not required to conduct source water monitoring, they do need to cover any uncovered reservoirs or treat the discharge, and meet disinfection profiling and benchmarking requirements, if applicable.

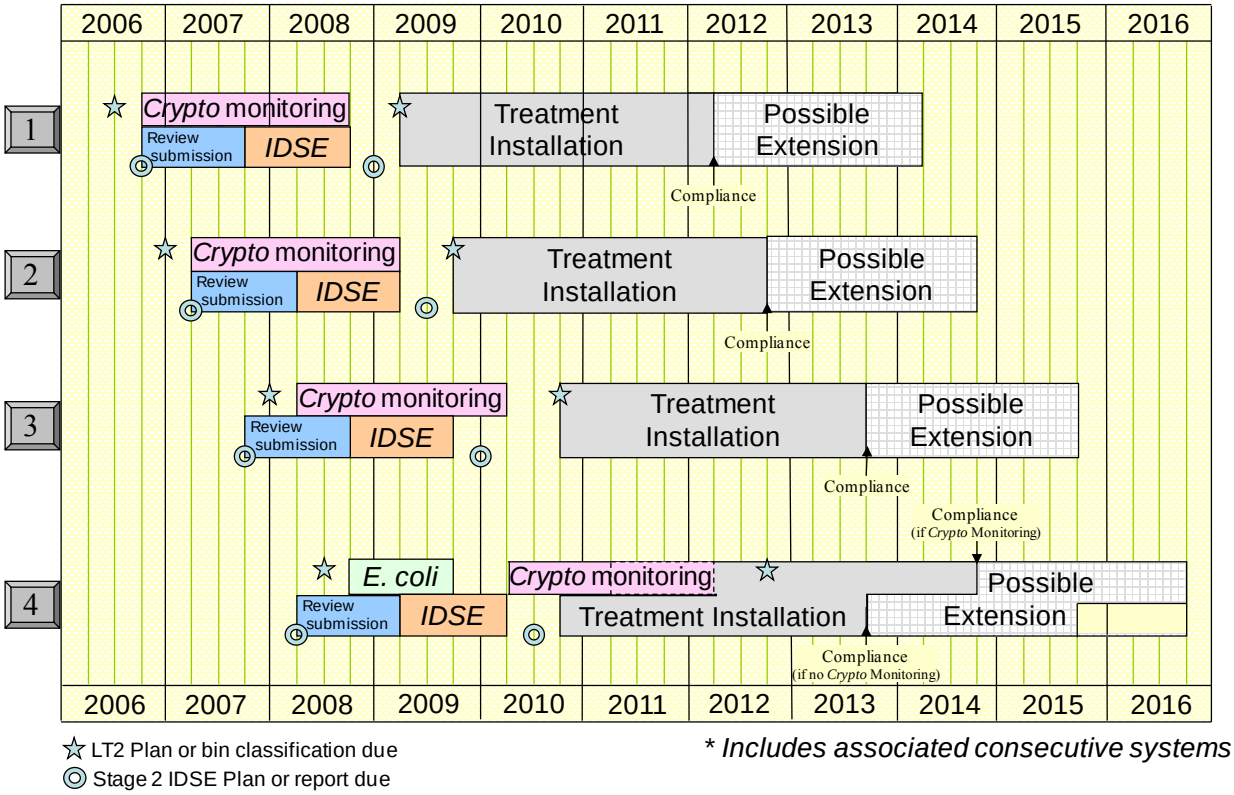
## Exhibit 1.4 Compliance Schedules

| If you have a Subpart H source and are this kind of system: | You are on schedule number |
|---|---|
| System serving 100,000 or more people OR a wholesale system producing surface water in a combined distribution system that contains a system serving 100,000 or more people | 1 |
| System serving 50,000 to 99,999 people OR a wholesale system producing surface water in a combined distribution system with the largest system serving 50,000 to 99,999 | 2 |
| System serving 10,000 to 49,999 people OR a wholesale system producing surface water in a combined distribution system with the largest system serving 10,000 to 49,999 | 3 |
| System serving fewer than 10,000 people. | 4 |

Source: U.S. EPA 2007. *The LT2ESWTR Implementation Guidance*. EPA 816-R-07-006, U.S. Environmental Protection Agency, Office of Groundwater and Drinking Water, Washington DC.

Exhibit 1.5 presents monitoring and treatment deadlines for the LT2ESWTR for systems on each of the four schedules defined in Exhibit 1.4. The compliance dates are designed to allow systems to comply simultaneously with the Stage 2 DBPR and the LT2ESWTR in order to balance risks associated with DBPs with risks associated with microbial pathogens. Compliance deadlines for individual microbial toolbox options are presented in Chapters 2 through 14 of this manual.

## Exhibit 1.5 Implementation Timeline for the LT2ESWTR




Source: U. S. EPA 2007. *The LT2ESWTR Implementation Guidance*. EPA 816-R-07-006, U.S. Environmental Protection Agency, Office of Groundwater and Drinking Water, Washington DC.

Notes:

For systems on Schedules 1 through 4 (see Exhibit 1.4).
Unfiltered systems must monitor for *Cryptosporidium*, regardless of size.

# 2. Watershed Control Program

## 2.1 Introduction

The watershed control program (WCP) credit provides the opportunity for public water systems (PWSs) with surface water sources employing filtration to obtain a 0.5-log credit from the Microbial Toolbox by developing and implementing a state-approved WCP plan. The elements of a state-approved WCP plan include identification of potential *Cryptosporidium* sources, prioritization of the identified sources, development of control measures to address the prioritized sources, and continuation of these efforts in the future. Systems with existing source water protection (SWP) efforts that meet these requirements can incorporate them into their state-approved WCP plan, while systems without existing programs can receive the same credit if they develop and implement similar SWP efforts as part of a WCP.

PWSs in the same watershed typically need to evaluate and control the same *Cryptosporidium* sources. Consequently, in order to pool resources and reduce duplication of efforts, in many cases the state and the PWSs in the watershed should work together to develop a single joint WCP plan that will allow the state to approve a 0.5-log credit for each PWS that participates in the implementation of the plan. This may not be practical or achievable in all cases, and in other cases a PWS may have a simpler and smaller watershed that does not include upstream PWSs. These systems are still encouraged to work with any downstream PWSs to develop joint WCP plans, but PWSs that develop and implement an individual WCP plan approved by the state are eligible for the WCP credit.

The remainder of this chapter discusses the following in more detail:

- Required elements for the WCP plan, and the process associated with obtaining and maintaining the WCP credit (Section 2.2).
- Benefits and advantages of the WCP (Section 2.3).
- Guidance and tools to help develop the WCP (Section 2.4).

### 2.1.1 Credits Available

Filtered systems that develop a state-approved WCP designed to reduce the level of *Cryptosporidium* in the watershed can receive a 0.5-log credit towards the *Cryptosporidium* treatment requirements under the LT2ESWTR (40 CFR 141.722). The WCP credit can be added to the credit awarded for any other toolbox component.

## 2.2 Application Process for the WCP Credit (PWS and State Responsibilities)

The following discussion describes PWS efforts necessary to apply for, implement, and maintain the WCP credit. Associated with the PWS efforts are complementary efforts by the state or other primacy agency to review and approve the initial award of the credit, and subsequent efforts to continue the credit as long as the PWS meets all of their commitments. There are six steps associated with obtaining and continuing the WCP credit, some to be performed by the PWS to gain the credit and some by the state to approve the credit, including the following (also see checklist described in Section 2.2.4):

- PWS notification to the state indicating that the PWS will be submitting a WCP plan.
- Development and submittal of a WCP plan by the PWS for review by the state.
- State review and approval of the WCP plan submitted by the PWS.
- Implementation of the state-approved WCP plan by the PWS.
- Continued maintenance of the activities outlined in the WCP plan by the PWS.
- Periodic review of progress by the state (annual report prepared by PWS, watershed sanitary survey every three years using state guidelines and state-approved personnel).

A PWS intending to utilize the WCP credit must have a WCP plan approved and in place within three years after the *Cryptosporidium* sampling and bin assignment are complete. Exhibit 2.1 outlines the deadlines for key compliance events associated with implementing and maintaining the WCP credit. Depending on size, different PWSs will have different deadlines for when bin assignment is completed. One year after this deadline (or earlier), the PWS must notify its state of its intent to apply for the watershed credit. One year after this deadline, a plan for WCP implementation must be prepared by the PWS and submitted to the state. All PWSs requiring credits under the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) must have these credits in place within three years after the bin assignment deadline. Consequently, the state must either approve, conditionally approve, or reject the PWS WCP plan by this date. If the state does not respond by this time, the credit will be assumed approved as long as all other requirements are met (i.e., WCP implementation and maintenance). In either case, if the state determines that the PWS is not implementing or maintaining the activities outlined in the approved WCP plan, the state may later withdraw the credit.

As discussed in the section on the annual WCP status report, if a PWS determines that a significant change is needed for a state-approved WCP, the PWS must notify the state, either separately or in the body of the annual status report, prior to making any of these changes. The notification must list actions the PWS will take in order to mitigate any "likely" (40 CFR 141.716 (a)(5)(i)) reduction in SWP that might result from the proposed change.

When a PWS develops a WCP plan for their own system they may choose to consult with other water systems to see if they are interested in cooperating with the water system and others to develop joint activities or even a common WCP plan. Such collaborative efforts can help increase the effectiveness of plan activities both in terms of effectively managing fiscal and technical resources but also with respect to the size of the watershed area brought into the plan. Describing a multi-PWS watershed control strategy to the state includes the same components as describing a single PWS effort. However, it is important to be particularly clear when describing implementing mechanisms, lines of authority, responsibilities, and other components that bear on inter-agency coordination. Credit may be available for all participants in a joint state-approved WCP plan, contingent on all participants following through on their designated roles during implementation and maintenance of the approved WCP plan.

### Exhibit 2.1 Checklist of PWS and State activities during preparation, implementation, and maintenance of WCP plan and associated 0.5-log LT2ESWTR treatment credit

| Compliance Event | Compliance Date for Systems of Different Sizes (by population served) | | |
|---|---|---|---|
| | >100,000 | 50,000 to 99,999 | 10,000 to 49,999 |
| Bin assignment deadline | April 2009 | October 2009 | October 2010 |
| Notification to state of PWS intent to prepare a WCP plan | April 2010† | October 2010† | October 2011† |
| PWS submit WCP plan | April 2011† | October 2011† | October 2012† |
| PWS implement state approved WCP plan‡ | April 2012† | October 2012† | October 2013† |
| First progress report (annually thereafter) | April 2013† | October 2013† | October 2014† |
| First sanitary survey report (every three years thereafter) | April 2015† | October 2015† | October 2016† |
| Second round of *Cryptosporidium* sampling | April 2015 | October 2015 | October 2015 |

† Can be completed earlier, pending completion of prerequisite events.

‡ If a PWS submits a WCP plan with all required elements by the required due date, and then state does not respond by the date in this table, the credit is considered approved and the credit will continue indefinitely into the future as long as PWS properly implements the plan and submits required annual progress reports and watershed sanitary surveys required every three years.

### 2.2.1 Notifying the State of Intention to Participate

A system must notify its state of its intention to implement a WCP at least two years prior to the applicable compliance date. (40 CFR 141.716(a)(1)). For example, as shown in Exhibit 2.1, a system serving 10,000 people has an October 2013 deadline for implementing the WCP plan. Therefore, it must inform the state that it intends to develop a WCP plan by October 2011. The application and plan must be submitted for approval at least one year prior to the applicable

compliance date (i.e., one year after informing the state of the intent to implement a WCP plan). (40 CFR 141.716(a)(2)).

### 2.2.2 Preparation of Watershed Control Program Plan

After notifying the state that the PWS intends to prepare a WCP plan, this plan and supporting documentation must be submitted to the state for review and approval no later than the date indicated in Exhibit 2.1, two years after the bin assignment date. The WCP plan must include delineation of the "area of influence," identification of potential and actual *Cryptosporidium* sources with this delineated area, an analysis of the proposed control measures, and identification of an action plan to attempt to reduce *Cryptosporidium* source water levels (40 CFR 141.716(a)(2)). Requirements for systems incorporating new SWP efforts into their WCP plan are identical to requirements for systems incorporating existing SWP efforts (40 CFR 141.716(a)(3)). The WCP plan must: a) explain how actions are expected to contribute to specified goals; b) identify watershed partners and their roles; c) identify resource requirements and commitments; and d) outline a schedule, with deadlines, for plan implementation and maintenance (40 CFR 141.716(a)(2)(iv)). Each of the activities in the WCP should have a timetable for implementation, a budget, and details about how the activity will be implemented

More information on development of the WCP plan is described later in Section 2.4 and characteristics of some key elements in the plan are briefly outlined below.

#### 2.2.2.1 Delineation of Area of Influence

An essential element for the WCP plan is the identification of the "area of influence," outside of which there is not a significant likelihood of *Cryptosporidium* or fecal contamination that affects the treatment plant intake. Identification of *Cryptosporidium* sources, associated control measures, and future watershed surveys (see Section 2.4.1) will be targeted within this area.

Methods to be used to establish the boundaries of the area of influence are at the discretion of the PWS, as long as the state considers it sufficient to approve the area delineated. Some methods that could be used include: a) characterization of watershed hydrology, b) modeling of *Cryptosporidium* travel time, or c) when sufficient data exists it can be useful to include factors such as fate and/or die-off/inactivation times in natural waters. A PWS could use one or more of these methods, or it could use methods that do not include any of the above as long as the state considers the results sufficient to adequately establish the boundaries of the area of influence.

More information on delineation of the area of influence is described later in Section 2.4.1.

#### 2.2.2.2 Identification of *Cryptosporidium* Sources

Potential as well as actual sources of *Cryptosporidium* contamination within the delineated area of influence must be identified and the relative impact on source water quality assessed (40 CFR 141.716(a)(2)(ii)). More information on watershed *Cryptosporidium* sources is included later in Section 2.4.2. Examples of "potential" sources include various land uses or facilities (e.g., publicly owned treatment works (POTWs), concentrated animal feeding operations (CAFOs), etc.) for which the PWS lacks specific data on *Cryptosporidium* occurrence in effluent or runoff, but there is a high likelihood of oocysts being present based on published research.

#### 2.2.2.3 Analysis of Control Measures

*Cryptosporidium* control measures included in watershed protection plans may include such diverse activities as structural best management practices (BMPs), land use control regulations, and public education. The application must present an analysis of control measures that address the sources of *Cryptosporidium* contamination identified for the water treatment plant source water. The analysis of control measures must discuss the effectiveness and feasibility of each measure in reducing *Cryptosporidium* loading in the source water (40 CFR 141.716(a)(2)(iii)).

More information on *Cryptosporidium* control measures is described later in Section 2.4.3.

#### 2.2.2.4 Partnerships for Source Water Protection

PWSs in the same watershed typically need to evaluate and control the same *Cryptosporidium* sources. Consequently, in order to pool resources and reduce duplication of efforts, in many cases the state and the PWSs in the watershed should work together to develop a single joint WCP plan that will allow the state to approve a 0.5-log credit for each PWS that participates in the implementation of the plan. This may not be practical or achievable in all cases, and in other cases a PWS may have a simpler and smaller watershed that does not include upstream PWSs. These systems are still encouraged to work with any downstream PWSs to develop joint WCP plans, but PWSs that develop and implement an individual WCP plan approved by the state will get the identical credit as PWSs involved in a state -approved joint WCP plan.

### 2.2.3 Approval and Continuation of the WCP Credit

#### 2.2.3.1 Initial Approval of the WCP Plan

The state must review each system's proposed WCP plan and either approve, reject, or conditionally approve the plan. See Appendix G for review criteria for use by states when

reviewing WCP plans (both required and recommended elements of a WCP are presented in Appendix G). If the plan is approved, or if the system agrees to implement the state's conditions for approval, the system will be awarded 0.5-log *Cryptosporidium* removal credit to apply toward the LT2ESWTR treatment requirements. The PWS will need a decision from the state within three years after bin assignment as outlined in Exhibit 2.1 and Figure 2-1 in order to fulfill the treatment requirements of the LT2ESWTR. If the state does not respond to a WCP plan by the required date, the WCP plan shall be considered "state-approved" and the 0.5-log WCP credit shall be awarded to the water system as long as the submitted WCP plan includes all required elements (40 CFR 141.716(a)(4)). Under any circumstances, the state can later withdraw an approved WCP credit if the state determines that the PWS has not implemented and maintained the activities outlined in the approved WCP plan (40 CFR 141.716(a)(5) and (6)).

The initial approval will be valid as long as the PWS continues to implement and maintain the approved WCP plan, as described in more detail below.

#### 2.2.3.2 Maintenance of the WCP Credit

Systems that have obtained state approval of their watershed control programs are eligible for the 0.5-log WCP credit as long as they continue to implement and maintain the efforts outlined in their state-approved WCP plan, as well as satisfactorily completing the following:

- Submit an annual WCP status report to the state (40 CFR 141.716(a)(5)(i)).

- Conduct a watershed sanitary survey every three years for community water systems (five years for non-community systems) using state guidelines and personnel approved by the state for this work and submit the survey report to the state (40 CFR 141.716(a)(5)(ii)).

After approval of the WCP plan, if the PWS determines that a change in the plan is needed, the PWS must notify the state prior to making the change and must outline any measures proposed to mitigate any reduction in SWP that is likely to result from this change (40 CFR 141.716 (a)(5)(i)). The description of this change must also be included in the next annual status report.

The annual status reports, WCP plan, and watershed sanitary surveys that are conducted every three years must be made available to the public upon request. These documents must be in plain language format and include criteria by which to evaluate the success of the program in achieving plan goals. The state may withhold portions of the annual status report, WCP plan, and watershed sanitary survey as requested by the PWS based on security considerations (40 CFR 141.716(a)(5)(iii)). To assist the state in this regard, the PWS should clearly indicate the specific information that should be held confidential. The system can identify those items for the state, or provide parallel "vetted documents" for dissemination to the public.

Once awarded the 0.5-log WCP credit, water systems will continue to receive the credit as long as they continue to implement and maintain the activities outlined in their state-approved WCP plan, including preparation and submittal of annual progress reports and sanitary survey reports every three years, as required. States may withdraw the credit if they determine that the

PWS is not carrying out the activities outlined in the state-approved WCP (40 CFR 141.716(a)(6)).

More details on preparation and review of the required reports by the PWS (or collection of PWSs in a joint WCP) and the state are briefly outlined below.

#### 2.2.3.2.1 Annual Status Report

The annual WCP status report must be submitted by the date established by the state. The report must describe the PWS's implementation of the approved plan and assess the adequacy of the plan for meeting the system's goals. It also must explain how the system is addressing any shortcomings in plan implementation, including those previously identified by the state or by the system during a watershed survey. If the system made any substantial changes to its approved program, it must describe the changes and explain the reason for making them. If the change is likely to reduce the level of SWP, the system must explain what actions it will take to mitigate the effects (40 CFR 141.716(a)(5)(i)).

The annual status report must describe progress being made implementing individual control measures (40 CFR 141.716(a)(5)(i)). Progress should be compared with the original timetable provided in the WCP plan. Implementation delays should be explained, and actions to prevent further delays should be described.

The original watershed program plans must include an analysis of the effectiveness and feasibility of control measures that could reduce *Cryptosporidium* loadings from sources of contamination to the system's source water. Annual status reports should provide updates on the control measures as they are implemented. The report should address progress being made on priority activities and, to the extent possible, evaluate whether projects are achieving their objectives. The report should also identify emerging issues and incorporate them into the watershed protection program. Since the annual status reports must be made available to the public on request, reports must be written in plain language format (40 CFR 141.716(a)(5)(iii)), though portions of the report can be withheld for PWS security considerations. To assist the state in this regard, the PWS should clearly indicate information in the status report that should be held confidential. The PWS can identify those items for the state, or provide a parallel "vetted report" for dissemination to the public.

#### 2.2.3.2.2 Watershed Sanitary Survey Report

A state-approved watershed survey must be conducted once every three years for community water systems (five years for non-community water systems), with the first report due three years after the WCP approval date (see Exhibit 2.1). The survey must be conducted according to state guidelines by persons approved by the state to conduct watershed surveys. A report on the results of the survey must be submitted to the state. The survey must meet the following criteria (40 CFR 141.716(a)(5)(ii)(A)):

- Cover the area of the watershed that was identified in the approved WCP plan as the area of influence.

- Assess the implementation of actions to reduce source water *Cryptosporidium* levels.

- Identify new sources of *Cryptosporidium.*

If the state determines that significant changes may have occurred in the watershed since the previous watershed sanitary survey, systems must undergo another watershed sanitary survey by a date required by the state, which may be earlier than the regularly scheduled survey (40 CFR 141.716(a)(5)(ii)(B)). In such an instance, the next survey and subsequent surveys will be required three years from this new date.

States developing a watershed sanitary survey program may wish to use the watershed sanitary survey manual developed by the California Department of Health Services and the California/Nevada Section of American Water Works Association (AWWA). PWSs are required to use state-designated persons for the sanitary survey. Conducting a useful watershed survey relies in large part on the competence of the individuals responsible for its execution. It is expected that the state will designate appropriately trained individuals, including civil or environmental engineers, or sanitarians with experience in the operation of water systems and a sound understanding of public health principles and waterborne diseases. Although other people may be performing the work for these individuals, the people designated by the state will supervise and direct the activities conducted for the survey. Efforts performed during the survey could include activities such as the following:

- Review of relevant National Pollutant Discharge Elimination System (NPDES) permits and discharge records.

- Review of pertinent databases (e.g., county geographic information system (GIS) systems, etc.).

- Review of most recent available aerial photography.

- Interviews with Natural Resources Conservation Service (NRCS), soil conservation service, local county planning agencies, regional planning organizations, and other organizations as applicable.

A final survey report must be submitted to the state for approval (40 CFR 141.716(a)(5)(ii)). The report should be completed as soon as possible after the survey is conducted. The length of the report will depend on the findings of the survey and the size and complexity of the watershed. The survey report should include: 1) the date of the survey; 2) who was present during the survey; 3) survey findings; 4) recommended improvements to the identified problems; and 5) the dates for completion of any improvements.

The watershed survey reports must be written in a plain language format. Survey results must be made available to the public upon request. The state may withhold portions of the survey report based on security considerations (40 CFR 141.716(a)(5)(iii)). To assist the state in this regard, the PWS should clearly indicate the specific information in the report that should be held confidential. The system can identify those items for the state, or provide parallel "vetted documents" for dissemination to the public.

#### 2.2.3.3 State Review and Continuation of the WCP Credit

Once water systems are awarded the 0.5-log WCP credit, they will continue to receive the credit as long as they implement and maintain the efforts outlined in their state-approved WCP plan. After the WCP plan is approved, ongoing reviews are the annual status report and the report from the sanitary surveys conducted and submitted once every three years.

The initial approval will be valid as long as the PWS continues to implement and maintain the approved WCP plan. After approval of the WCP plan, if the PWS determines that a change in the plan is needed, the PWS must notify the state prior to making the change and must outline any measures proposed to mitigate any reduction in SWP that is likely to result from this change (40 CFR 141.716 (a)(5)(i)). The description should include the impact of that change on the protection of the watershed so the state and water system will both understand whether the assumptions made during the "verbal approval" stage are holding true.

### 2.2.4 PWS and State Checklist for Preparation, Implementation, and Maintenance of the WCP Plan and Associated Credit

Exhibit 2.2 includes a summary of all activities associated with preparing the WCP plan by the PWS, review and approval of the plan by the state, implementation and maintenance of the plan by the PWS, and assessment of PWS activities by the state in order to allow continuation of the credit.

## Exhibit 2.2 Checklist of PWS and State activities during preparation, implementation, and maintenance of WCP plan and associated 0.5-log LT2ESWTR treatment credit

| Description of Task | Code of Federal Regulations Citation (40 CFR) | Section(s) of interest in this Guidance Manual |
|---|---|---|
| Notification Period<br>(due no later than one year following bin determination, see Exhibit 2.1) | | |
| The PWS must notify the state one year after bin assignment if they intend to later submit a WCP plan | 141.716(a)(1) | 2.2.1 |
| WCP Plan Preparation Period<br>(due no later than two years following bin assignment date, see Exhibit 2.1) | | |
| The PWS must submit a report containing the WCP plan including the following required elements: | 141. 716 (a)(2) | 2.2,<br>2.2.2 |
| Identification of area of influence | 141. 716 (a)(2)(i) | 2.2.2.1<br>2.4.1 |
| Identification of potential and actual sources of *Cryptosporidium* contamination in area of influence and assessment of relative impact of these sources on source water quality | 141. 716 (a)(2)(ii) | 2.2.2.2<br>2.4.2<br>2.4.2.1<br>2.4.2.2 |
| Analysis of the effectiveness and feasibility of control measures that could reduce *Cryptosporidium* loading from sources identified within area of influence | 141. 716 (a)(2)(iii) | 2.2.2.3<br>2.4.3<br>2.4.3.1 through 5 |
| Statement of goals and actions to undertake as part of the WCP plan implementation and maintenance efforts to reduce source water *Cryptosporidium* levels, including an explanation of how these actions are expected to contribute to achievement of stated goals. | 141. 716 (a)(2)(iv) | 2.2.2 |
| Identification of watershed partners and their roles | 141. 716 (a)(2)(iv) | 2.2.2.4 |
| Identification of resource requirements and commitments | 141. 716 (a)(2)(iv) | 2.2.2 |
| Development of an implementation schedule, including deadlines for completing specific actions identified in the WCP plan | 141. 716 (a)(2)(iv) | 2.2.2 |
| Systems can use existing watershed control programs to meet requirements of the rule, as long as the entire WCP plan includes all the same elements required for all systems | 141. 716 (a)(3) | 2.2.2<br>2.3.3 |

**Exhibit 2.2 (continued)**

| Description of Task | Code of Federal Regulations Citation (40 CFR) | Section(s) of interest in this Guidance Manual |
|---|---|---|
| State Review of the WCP Plan<br>(due no later than three years after bin assignment date, the same time as other treatment credits, see Exhibit 2.1) | | |
| The state is expected to approve, reject, or conditionally approve the WCP plan if it is submitted before the due date (see Exhibit 2.1) | 141. 716 (a)(2) | 2.2<br>2.2.3.1<br>2.2.3.3 |
| If a WCP plan containing all required elements (see "WCP Plan Preparation Period" items listed above) is submitted by the required date, but is not formally acted upon by the state prior to the required deadline (see Exhibit 2.1), then the WCP plan is considered approved by the state and the 0.5-log WCP credit is allowed. See discussion below regarding withdrawal of this or any other approval of the credit. | 141. 716 (a)(4) | 2.2<br>2.2.3.1<br>2.2.3.3 |
| Implementation and Maintenance of the 0.5-log WCP Credit<br>(occurs after credit is approved by state) | | |
| If the PWS or PWSs awarded a 0.5-log WCP credit do not carry out the actions outlined in their state-approved WCP, the state may withdraw the credit | 141. 716 (a)(6) | 2.2.3<br>2.2.3.2 |
| In order to maintain the 0.5-log WCP credit the PWS or PWSs associated with a state-approved WCP plan must carry out the actions outlined in the plan, assessed by evaluating the annual status report and report from the sanitary survey conducted every three years | 141. 716 (a)(5) | 2.2.3<br>2.2.3.2 |
| The annual status report must describe the PWS's implementation of an approved WCP plan and must assess the adequacy of the plan to continue to meet its goals | 141. 716 (a)(5)(i) | 2.2.3<br>2.2.3.2<br>2.2.3.2.1 |
| The annual status report must explain how the PWS or PWSs associated with the WCP plan are addressing any shortcomings in the implementation of the plan, including those previously identified by the state after the three-year sanitary surveys. | 141. 716 (a)(5)(i) | 2.2.3.2.1 |
| The annual status report must describe any significant changes that have occurred in the watershed since the last sanitary survey. | 141. 716 (a)(5)(i) | 2.2.3.2.1 |
| The PWS must notify and receive verbal approval from the state before implementing any changes to a state-approved WCP plan. The PWSs must propose actions they will undertake to mitigate any changes that appear likely to reduce the level of SWP. These changes must be described in the next annual progress report. | 141. 716 (a)(5)(i) | 2.2.3<br>2.2.3.2,<br>2.2.3.2.1 |

**Exhibit 2.2 (continued)**

| Description of Task | Code of Federal Regulations Citation (40 CFR) | Section(s) of interest in this Guidance Manual |
|---|---|---|
| A state-approved watershed survey must be conducted once every three years for community water systems (five years for non-community water systems), with first report due three years after the WCP plan is approved by the state. The survey must be conducted according to state guidelines by persons approved by the state to conduct watershed surveys. A report on the results of the survey must be submitted to the state once every three years. The survey must meet the criteria listed below: | 141. 716 (a)(5)(ii) | 2.2.3<br>2.2.3.2,<br>2.2.3.2.2 |
| Encompass the area of influence defined in the state-approved WCP plan | 141. 716 (a)(5)(ii)(A) | 2.2.3.2.2 |
| Assess actions implemented to reduce *Cryptosporidium* levels within the area of influence | 141. 716 (a)(5)(ii)(A) | 2.2.3.2.2 |
| Identify any significant new sources of *Cryptosporidium* in the area of influence | 141. 716 (a)(5)(ii)(A) | 2.2.3.2.2 |
| If the state determines that significant changes may have occurred in the watershed since the previous watershed sanitary survey, systems must undergo another watershed sanitary survey by a date required by the state, which may be earlier than the regularly scheduled survey. In such an instance, the next survey and subsequent surveys will be required three years from this new date. | 141. 716 (a)(5)(ii)(B) | 2.2.3.2.2 |
| The WCP plan, annual status reports, and watershed sanitary survey reports must be written in plain language format. All of these documents must be made available by the state to public, upon request. The state can withhold portions of these documents identified by the PWS due to security considerations. To assist the state in this regard, the PWS should clearly indicate the information in the status report that should be held confidential. | 141. 716 (a)(5)(iii) | 2.2.2<br>2.3.3<br>2.3.3.2,<br>2.2.3.2.1<br>2.2.3.2.2 |

## 2.3 Benefits and Other Characteristics of the WCP Credit and Related Activities

### 2.3.1 Benefits to the PWS and Watershed from a Successful WCP

A well-designed WCP can result in a reduction of overall microbial risk. The risk reduction is associated with the implementation of practices that reduce *Cryptosporidium* as well as other pathogens. Further, knowledge of the watershed and factors affecting microbial risk, including sources of pathogens, fate and transport of pathogens, and hydrology, can also help a system reduce microbial risk.

There are many potential sources of *Cryptosporidium* in watersheds, including sewage discharges and nonpoint sources associated with animal feces. The feasibility, effectiveness, and sustainability of control measures to reduce *Cryptosporidium* contamination of water sources will be site-specific. Consequently, the WCP credit centers on systems working with stakeholders in the watershed to develop a site-specific program, and state review and approval are required. This section is intended to assist water systems in developing their watershed control programs and states in assessing and approving these programs.

PWSs with existing SWP programs, watershed information, partnerships, etc., are encouraged to incorporate these into their WCP plans. Whether as a continuation of existing efforts or as a result of new efforts specifically initiated for WCP credit, SWP activities for identification, prioritization, and control of *Cryptosporidium* sources are important. These activities also provide proactive, preventive steps for reduction of *Cryptosporidium* risks for drinking water consumers. Efforts to identify, prioritize, and control *Cryptosporidium* sources in the watershed offer opportunities for cooperation and collaboration between PWSs who will benefit from joint efforts to reduce *Cryptosporidium* sources in their common watershed(s). Consequently, a WCP for one or more PWS can create the potential opportunity to extend watershed protection activities until they become watershed-wide, basin-wide, regional, state-wide, or even multi-state-wide if enough water systems can cooperate together to make it happen.

A WCP targeting *Cryptosporidium* reduction is the most advantageous when it is also the component of a larger comprehensive SWP program that addresses other chemical and/or microbial contaminant threats. For PWSs in many states, much of the background information and preparation needed to develop a WCP and comprehensive SWP program are already available as a result of the source water assessments required under the 1996 Amendments to the Safe Drinking Water Act (SDWA). Section 1453 of the Act required states to conduct source water assessments for all public water systems, including delineating the "boundaries of the areas providing source waters for PWSs and identifying the origins of regulated and certain unregulated contaminants in the delineated area to determine the susceptibility of the PWSs to such contaminants."

Information resulting from these assessments should be available from the states. Information may also be available for systems where watershed sanitary surveys have been performed. These surveys are required as part of the Interim Enhanced Surface Water Treatment Rule (IESWTR), and some states have required them for years. The completeness of the information contained in these existing resources may need to be supplemented by collecting additional background information, particularly information bearing on *Cryptosporidium* occurrence.

### 2.3.2 Advantages and Disadvantages of a Watershed Control Program

Section 2.3.2.1, and 2.3.2.2 explain the advantages and disadvantages of a WCP (respectively). Topics covered include the side impacts on public health and ecological goals, the incorporation of a multiple barrier strategy, the availability of analytical methods to track water quality progress, the level of commitment required of PWSs, and the potential costs and payoffs of implementation efforts.

#### 2.3.2.1 Advantages

Measures to control prioritized *Cryptosporidium* sources in the watershed, as required for the WCP credit, will in most instances control other contaminants of concern to PWSs. The reduction and prevention of source water contamination by microbial pathogens and other contaminants may also serve other public health and ecological goals, such as use of the water body for fishing and swimming, reduction of ground water contamination, and protection of aquatic habitats and the species that depend on such habitats for survival. The reduction in organic material in the source water will make treatment more efficient (and less expensive) and reduce the incidence of disinfection byproducts (DBPs).

While WCPs may be a cost-effective Microbial Toolbox option, the PWS commitment needed to initiate and maintain SWP efforts may be substantial. SWP efforts often require many years to start seeing measurable results. Furthermore, these efforts must continue to be maintained in order for initial improvement to persist. However, the potential payoff of these efforts are significant both for the water system and the community they serve. For example, one PWS estimates that its current SWP effort has helped the PWS avoid $100 million in capital costs and $10 million/yr in operating costs. This savings was accomplished with approximately $5 million expended to date on SWP efforts, of which $3 million was recovered through funding from grants (Crockett 2005).

Although the costs associated with implementing a particular Microbial Toolbox option are system-specific, a WCP can cost less than other Microbial Toolbox options that require installation of additional technology. This is especially the case if other stakeholders contribute time and resources to the watershed control program. Stakeholders could include other utilities and municipalities, other agencies in the same municipality, county or state agencies, and concerned citizens. Though watershed control programs involving land acquisition or purchase of conservation easements may be initially more expensive than installing treatment, the long-term benefits (improved quality and stability of source water, reduction in treatment costs, etc.) are potentially significant, though prediction of these improvements beforehand and measurement afterwards may at times be problematic.

Much of the information required to implement a watershed program, such as a contaminant source inventory and delineation of the watershed, may already be available in some states as a result of the source water assessment conducted under the 1996 SDWA Amendments. Although source water assessment programs vary from state to state, they should provide much of the basic information required to prepare a plan for a watershed control program, allowing systems to incorporate existing information into their WCP plans at minimal cost.

Control of *Cryptosporidium* sources in the watershed can contribute significantly to integrated multiple-barrier treatment strategies. For example, reducing influent *Cryptosporidium* loadings will facilitate pre-treatment (e.g., riverbank filtration), conventional treatment (e.g., clarification followed by filtration), or post-treatment (e.g., ultraviolet light (UV) irradiation). Control of *Cryptosporidium* sources in the watershed can benefit more than a single water treatment facility. In even small watersheds, there may be multiple water intakes that may serve other PWSs. Consequently, watershed-based coordination can assist PWSs to develop background information and to implement *Cryptosporidium* control measures. For example, since much of the background information needed for the WCP application submitted to the state can be very similar for multiple PWSs in the same watershed, working together PWSs can be more efficient developing background information and identifying priority actions to manage *Cryptosporidium* sources.

The use of stakeholder partnerships during development and implementation of a WCP can be substantial. In addition to being able to share resources and work together to find funding sources, collaboration with other stakeholders can bring important information and resources to the PWS. For example, increased contact with upstream or downstream utilities can result in sharing of information about source water quality issues, the means that other utilities have used to respond to contaminants and water quality changes, and increases the likelihood that other utilities will share information about sudden water quality changes that may also affect the other utilities using that particular source water.

#### 2.3.2.2 Disadvantages

Most water systems who have developed and implemented SWP efforts like those needed for the state-approved WCP credit have found that these efforts are able to substantially improve source water quality (Ashendorff et al. 1997, Vaux 2000). However, seldom will watershed activities result in immediate realization of benefits. Many land use policies, wildlife management, and public education programs require significant implementation timeframes. This challenge is further complicated by the target organism in this rulemaking, *Cryptosporidium*. *Cryptosporidium* occurs in low concentrations and is difficult to detect using existing analytical methods, consequently, it can be hard to discern reductions in Cryptosporidium concentrations resulting from watershed control programs even if substantial changes are realized in the watershed.

The PWS commitment needed to initiate and maintain source water protection efforts is substantial. SWP efforts often require many years to start seeing measurable results. Furthermore, these efforts must continue to be maintained in order for initial improvements to persist. Furthermore, it may not be possible to discern the improvements in source water quality using these monitoring approaches due to natural environmental variability, the characteristics of the source water improvements, and the limitations of the current analytical methods for *Cryptosporidium* or other fecal indicators. However the potential payoff of these efforts are significant both for the water system and the community they serve, as noted in the "advantages" section.

As with other treatment options, cost is a significant factor in determining if a WCP will be viable for any individual water system. Some federal funding is available to implement some aspects of a WCP. For example, the Clean Water Act (CWA) authorizes state revolving fund loans to upgrade wastewater treatment plants and provides grants (under Section 319) for control of nonpoint source pollution. The Farm Bill of 2002 authorizes several billion dollars for management of agricultural pollution. Drinking Water State Revolving Funds (DWSRF) are also available to a limited extent for SWP. Each state may set aside as much as 15 percent of its grant each year to provide loans for SWP activities, including land or easement acquisition, implementation of incentive-based voluntary SWP programs, and implementation of wellhead protection programs. A review of potential funding sources is provided in Gullick et al. (2007).

Like other components of the multiple barrier strategy, the barrier provided by watershed protection must be maintained into the future. Consequently, while some SWP efforts can produce visible evidence of action in fairly short periods of time (Capacasa 2005, Crockett et al. 2005), the ongoing maintenance of land use systems and key changes in private actions in the watershed will be more difficult to demonstrate. This lack of visibility can lead to challenges for long-term fiscal stability and stakeholder engagement, which is compounded when there are other competing demands for resources.

A WCP may not be as successful in some circumstances as in others. Potential pitfalls are important to consider in deciding whether to undertake a watershed control program. Microbiological contaminants are frequently related to nonpoint sources, and control of these sources is often highly dependent on changing the behaviors of large groups of people. In a voluntary program (e.g., if the water system has no authority to regulate land use and is encouraging landowners to voluntarily take action), it is difficult to determine whether individuals are making the recommended changes necessary to control contaminants. Although the required three-year watershed survey will assist in evaluating progress, systems that implement watershed control programs may need to be creative in finding ways to gauge the success of their programs.

Urban growth and land development can affect the success of a watershed control program. The success or failure of a WCP that is based on land use controls will rest in large part on how committed the effected communities are to supporting the land use constraints identified in the WCP plan.

A successful WCP requires the cooperation of a variety of stakeholders; however, it may be difficult to get agreement or participation from these stakeholders. Alternatively, stakeholder groups may agree to perform certain activities, such as outreach; but could lose funding and be unable to follow through on their commitments. Systems that have concerns about the likelihood

of building strong relationships with their stakeholders may decide that a WCP is not appropriate for them. In some watersheds, depending on size of the WCP and the ability to share costs with others, significant PWS staff time may be required to oversee a program. These costs and staff commitments may be prohibitive for some systems.

### 2.3.3 Incorporation of New Versus Existing Source Water Protection Activities Into a Watershed Control Program

PWSs that already have SWP activities in place that are suitable for incorporation into a WCP are encouraged to consider the WCP credit as one of their Microbial Toolbox options since continuation of these existing efforts offer the best chance of producing improvements in the watershed the most quickly. These systems are also encouraged to cooperate with other utilities in their watershed so that information these utilities have in common can be shared, allowing all utilities involved to focus on prioritized control measures rather than duplicating these efforts. PWSs with existing SWP programs must incorporate into their WCP implementation plans all required elements described earlier in Section 2.2. Some PWSs may already have existing SWP programs encompassing all of these elements, while others will need to develop and implement a combination of existing and newly designed efforts to meet all of the requirements. PWSs without existing SWP programs suitable for the WCP credit are encouraged to develop and implement appropriate SWP programs via the WCP credit mechanism.

## 2.4 Tools to Help PWSs Develop the Watershed Control Program Plan

The following subsections discuss the factors PWSs should consider in developing a WCP plan to improve *Cryptosporidium* reductions in source water, along with descriptions of BMPs and other control measures. However, each watershed is different and therefore, each WCP plan needs to be tailored for the specific circumstances.

Systems may be able to use the results of the source water assessments conducted under the SDWA Amendments of 1996 to support this effort to develop a WCP plan. These assessments establish a foundation for the WCP because they delineate the watershed, providing a starting point for defining the area of influence, and they inventory and rank the susceptibility of the water supply to actual and potential contamination sources. The assessments covered all priority contaminants in a watershed, including *Cryptosporidium* (U.S. EPA 1997). In some cases, if sufficiently detailed, the source water assessments may fully satisfy the analytical needs of the WCP plan. However in some cases the information available from the source water assessments is quite limited or outdated, and utilities will need to look for or develop other sources for this information.

Other source water and watershed information may be available from sanitary surveys conducted for the IESWTR and the Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) (these rules require sanitary surveys at least every three years for community water systems and at least every five years for noncommunity water systems). Guidance is available at www.epa.gov/safewater/mdbp/pdf/sansurv/sansurv.pdf. The California-Nevada section of the AWWA and the California Department of Health Services Division of Drinking Water and Environmental Management also have developed guidance specifically for watershed sanitary

surveys. Coordination of SWP efforts with those of CWA programs such as Total Maximum Daily Loads (TMDLs) is beneficial and encouraged.

Guidance for SWP activities is available from a wide variety of sources, including the following:

- U.S. EPA Source Water Protection webpage (www.epa.gov/safewater/protect.html) contains links to a variety of guidance materials.
- U.S. EPA draft Handbook for Developing Watershed Plans to Restore and Protect our Waters (U.S. EPA 2005d).
- Introduction to EPA's Drinking Water Source Protection Programs (U.S. EPA, 2003b).
- Source Water Protection: Best Management Practices and Other Measures for Protecting Drinking Water Supplies (U.S. EPA 2002d).
- State Source Water Assessment and Protection Programs Guidance: Final Guidance (U.S. EPA 1997).
- Protecting and Restoring America's Watersheds: Status, Trends, and Initiatives in Watershed Management (U.S. EPA 2001g).
- Getting in Step: A Guide for Conducting Watershed Outreach Campaigns (Tetra Tech, Inc. 2003).
- U.S. EPA Watershed Academy On-Line Training Modules (available at www.epa.gov/watertrain/, e.g., www.epa.gov/watertrain/pdf/ swp.pdf (January 2003) and www.epa.gov/watertrain/pdf/swpbmp.pdf (August 2002)).
- U.S. EPA National Agriculture Compliance Assistance Center website - information on animal production practices and BMPs (http://www.epa.gov/agriculture).
- U.S. EPA Animal Feeding Operations (AFO) Virtual Information Center website (http://cfpub.epa.gov/npdes/afo/virtualcenter.cfm).
- Source Water Protection for Concentrated Animal Feeding Operations: A Guide for Drinking Water Utilities (Gullick et al. 2007).
- AwwaRF (AWWA Research Foundation) Source Water Protection Reference Manual (CDM 2002).
- Source Water Protection: Effective Tools and Techniques You Can Use (1999 Participant Manual) (AWWA 1999).
- Effective Watershed Management for Surface Water Supplies (AwwaRF 1991).

- Guidance to Utilities on Building Alliances with Watershed Stakeholders (AwwaRF 2001).

- Protecting the Source: Land Conservation and the Future of America's Drinking Water (Ernst 2004).

- Path to Protection: Ten Strategies for Successful Source Water Protection (Ernst and Hart 2005).

- Source Protection Handbook: Using Conservation to Protect Drinking Water Supplies (Hopper and Ernst 2005).

- Source Protection: A National Guidance Manual for Surface Water Supplies (NEIWPCC 2000).

- The Center for Watershed Protection provides basic templates to help with design of watershed protection programs; see www.stormwatercenter.net.

- From Source to Tap: Guidance on the Multi-Barrier Approach to Safe Drinking Water (CCME 2002).

In addition, compilations of successful SWP program case studies are available from the following sources:

- Case Studies of Source Water Protection (U.S. EPA 2005a; http://cfpub.epa.gov/safewater/sourcewater/sourcewater.cfm?action=Case_Studies.)

- Section 319 Nonpoint Success Stories (U.S. EPA 2005b; www.epa.gov/owow/nps/Success319).

- Watershed Success Stories – Applying the Principles and Spirit of the Clean Water Action Plan (U.S. EPA 2000d; http://www.blueprintjordanriver.slco.org/docToPdf/WatershedSuccessStor.pdf

- Protecting Sources of Drinking Water: Selected Case Studies in Watershed Management (U.S. EPA 1999a, http://www.epa.gov/safewater/sourcewater/pubs/swpcases.pdf)

- Source Water Collaborative; see www.protectdrinkingwater.org.

### 2.4.1 Identification of the Area of Influence

An essential element for the WCP plan is the identification of the "area of influence." The area of influence is the area outside of which the likelihood of *Cryptosporidium* or fecal contamination affecting the treatment plant is not significant. Identification of *Cryptosporidium* sources, associated control measures, and future watershed surveys (see Section 2.2.2.1) will be targeted within this area. Methods to be used to establish the boundaries of the area of influence

are at the discretion of the PWS, as long as the state considers it sufficient to approve the area delineated.

*Delineation*

Systems may develop their own watershed delineation. The starting point for such a delineation may be the delineation developed by the state as part of the source water assessment program (SWAP). In referencing the delineation prepared in the SWAP process, systems should be careful to understand how the initial delineation was prepared. Different states employed different delineation approaches. To delineate watersheds, some states started with watersheds as catalogued by the U.S. Geological Survey (USGS). The USGS has assigned each watershed and its subwatersheds in the United States a hydrologic unit code (HUC). Because the HUC subwatersheds can be quite large, and a PWS's source may come from only a section of the watershed, or portion of a hydrologic unit, sometimes only the part of the watershed upstream of the PWS's intake was mapped. Sometimes watersheds were further segmented into "critical areas" within which more detailed assessments were performed. Some states delineated critical areas based on setbacks from the edge of the source water and all tributaries feeding into the source water. Others defined critical areas based on a fixed distance or time-of-travel from the intake (upstream of the intake or in all directions from the intake).

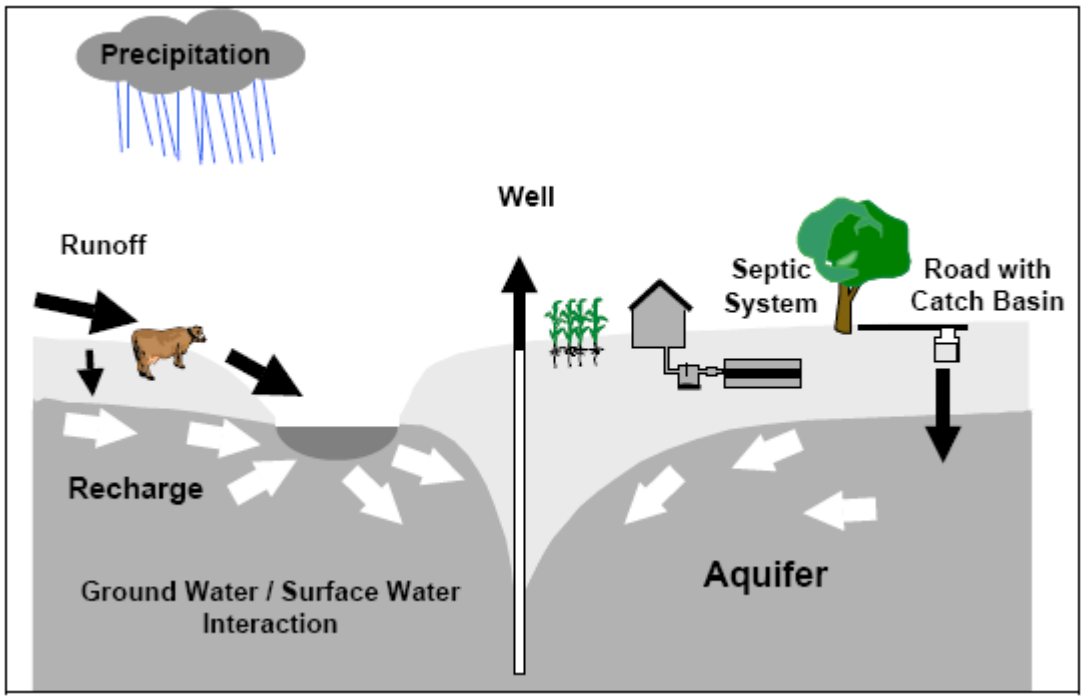
Systems that need to delineate their watersheds or subwatersheds for the first time and do not have GIS available can do so with topographic maps. The first step is to find the source (including tributaries) and the water treatment plant intake on the map. Each of the contour lines (which is actually not a line but a closed shape) around the source connects points of equal elevation. Upstream, the elevation indicated by each contour line increases with distance from the source. All precipitation falling within a zone of increasing elevation around the source will flow towards the source. Where the contour elevations stop increasing and begin decreasing is the break point. On the other side of the break point, water is flowing into a different watershed. The area delineated by connecting the break points is the watershed (AWWA 1999). See http://www.nh.nrcs.usda.gov/technical/Publications/Topowatershed.pdf for an illustrated fact sheet on delineation. If the intake is not at the downstream end of the watershed, it is only necessary to delineate the area upstream of the intake. Systems with GIS can follow the same process using digital elevation model (DEM) data rather than contour lines.

PWSs using ground water under the direct influence of surface water (GWUDI) as a source can delineate an area of influence by combining a delineation of the watershed influencing the ground water source with a delineation of the wellhead protection area.

*Watershed Hydrology*

Once the watershed has been delineated, PWSs should examine the hydrology of their watersheds to help determine the area of influence. The analysis submitted to the state must contain information on the watershed's hydrology. Stream discharge can affect the transport of sediment and *Cryptosporidium* oocysts, especially during and after storms. When more rain falls than can be absorbed immediately by the soil, soil cover, or impervious surface, water will pond on the surface. With increasing rainfall, the water will flow to a lower level on the surface, to a river, lake, or reservoir, as shown in Exhibit 2.3. As water travels, it may pick up contaminants on the soil surface (e.g., *Cryptosporidium* oocysts from deposited fecal matter). These particles are then suspended in the runoff and can be transported to surface water supplies. The microorganisms (including parasitic protozoa) associated with the soil can be transported as individual organisms, aggregates of organisms, or within an aggregate of soil particles and organisms.

## Exhibit 2.3 Ground Water/Surface Water Interaction




Ground water that is considered to be under the direct influence of surface water is usually immediately adjacent to surface water or to the discharge point of a spring. These ground water supplies are considered vulnerable to contamination by microbial contaminants like *Cryptosporidium* (consequently, GWUDI sources are treated like surface water sources under the SWTR, IESWTR, and LT2ESWTR). GWUDI may be contaminated through direct contamination (e.g., an inadequately protected spring), direct infiltration of oocysts from the surface as a result of rain, and as a result of the action of pumping wells (see Exhibit 2.3). Given sufficiently high pumping rates, wells can locally reverse the direction of ground water flow. In such cases, surface water is induced to flow from a river, lake, or reservoir into the adjacent ground water, where it may be extracted by pumping wells. If the surface water is contaminated with microbial contaminants, the adjacent ground water may also become contaminated.

Water quality flow models analyze specific hydrologic, geographic, and water quality parameters to estimate the travel time needed for contaminants to reach a drinking water intake and the amount of contamination at that intake. Surface runoff models may also be used to assess the potential impact of individual *Cryptosporidium* sources, and to identify watershed areas with the greatest potential impact on source water quality. Models should always be validated for the settings in which they are used.

PWSs should also consider topography and soil type, which can affect hydrology. Areas with steep slopes may experience a higher percentage of overland flow or runoff (as opposed to infiltration and subsurface flow) and have faster overland flow rates during rainfall than flat areas. *Cryptosporidium* may be more likely to be transported to water bodies in such areas, although if the topography is very steep, livestock that carry *Cryptosporidium* may not be present. Impermeable or compacted soil, impervious surfaces, unvegetated areas, and a high water table

can also affect overland flow. Riparian zones can be considered sensitive areas simply due to their proximity to streams that feed into source waters. They are also subject to erosion. PWSs should also factor soil types into their decisions; areas with high clay content may be more impermeable or more subject to erosion and can contribute to high turbidity.

### 2.4.2 Potential and Existing Sources of *Cryptosporidium*

All potential and actual *Cryptosporidium* sources in the area of influence must be identified and reported in the WCP plan (40 CFR 141.716(a)(2)(ii)) and should be evaluated for suitable control measures in the plan. Systems may be able to use source inventory data collected as part of the source water assessment program. Many states asked systems to assist with identifying significant potential contaminant sources, either through field verification or through review of inventory databases or other information. Therefore, some PWSs should already have this information available. States will also be assessing the risk of each source or category of sources, primarily through numerical ranking systems and matrices; systems will have this information at their disposal as well. It is possible that the inventory and ranking of potential sources may not be detailed enough for a *Cryptosporidium* watershed control program, but they should provide a good starting point.

After noting sensitive areas based on topography and geology, it is possible to determine whether these areas coincide with land uses that are likely to contribute microbiological contamination to the water supply. Reviewing land use and zoning maps can be used to identify areas for investigation or for prediction of potential future sources and loadings. Where existing mapping does not reflect available databases, investigation of local data sources, such as health department data on septic systems and recent sanitary survey results can provide additional detailed information. Data on point sources such as wastewater treatment plants that require EPA or state permits, e.g., NPDES are also readily available through State and federal data systems. NPDES information (also called water discharge permit or PCS data) is available on EPA's Envirofacts website at http://www.epa.gov/enviro/index_java.html. Local, state and federal data sets are useful for identifying potential sources of contaminants, but these data systems can be out of date; actual field surveys may be necessary to confirm the status of existing point and nonpoint sources.

The paragraphs below summarize existing research on *Cryptosporidium* sources and associated land use in watersheds. Because most studies of *Cryptosporidium* occurrence involve sampling at water system intakes, little information is available about occurrence of *Cryptosporidium* within watersheds and transport of oocysts to surface water supplies. When possible, developing site-specific relationships can facilitate how to more effectively affect oocyst levels released to surface water in the watershed.

*Land Use*

Many land uses in a watershed have the potential to introduce *Cryptosporidium* into water supplies. These include point sources—combined sewer overflows (CSOs), wastewater treatment plants, and CAFOs—and nonpoint sources, including livestock, wildlife, pets, stormwater runoff, and septic systems.

Kaplan et al. (2002) generated a database on pathogen occurrence in watersheds ranging from forested to highly urbanized. Corsi et al. (2003) studied the source(s) and magnitude of *Cryptosporidium* by broad characterization of rural, suburban, and urban land uses. In this case the primary sources were attributed to urban stormwater and wastewater treatment plant outflows during baseflow and stormwater events.

The character (topography, plant cover) and land uses (urban, farming) within a watershed also influence the occurrence or concentration of *Cryptosporidium* in surface water (Hansen and Ongerth 1991). Oocyst concentrations can be as much as 10 times higher in urban and agricultural watersheds (Hansen and Ongerth 1991, Stern 1996) than in undeveloped ones. However, such differences may be site-specific—in streams in an agricultural watershed in southern Ontario, no connection was found between *Cryptosporidium* concentration and sources or land use such as wastewater treatment plants, CSOs, livestock, crops, houses, wildlife, and campgrounds (Fleming et al. 1999). Davies et al. (2005) quantifies information on prevalence and viability of *E. coli* and *Cryptosporidium* in livestock. Crockett (N.d.) compares prevalence of *Cryptosporidium* in wastewater versus wildlife and livestock, including adult versus neonatal calves. Santin et al. (2004) studied genotype and speciation of *Cryptosporidium* in feces from 1,000 cattle on 15 farms in 7 states. Results indicate that *Cryptosporidium parvum*, which are potentially infectious to humans, were detected in feces from pre-weaned calves but nearly all *Cryptosporidium* in older cattle were other non-infectious species or genotypes.

Point and nonpoint sources of *Cryptosporidium* are described below.

*Point Sources*

Point sources such as CSOoutfalls, which are common in older municipalities, can be a significant source of oocysts, depending on the weather and the endemic rate of cryptosporidiosis. CSOs contain raw sewage diluted by stormwater. In one study, *Cryptosporidium* concentrations at CSO outfalls on the Allegheny River in Pittsburgh during storms ranged from 0 to 3,000 oocysts/100 L, with a geometric mean of 2,013 oocysts/100 L (States et al. 1997).

Wastewater treatment plants may also contribute to oocyst loads, depending on the amount of treatment provided. Primary treatment can remove as little as 27 percent of oocysts from effluent (Payment et al. 2001); most plants in the United States provide secondary treatment, so removal should be better. In the Netherlands, it is estimated that 85 percent of *Cryptosporidium* oocysts occurring in surface water are discharged in wastewater treatment plant effluent (Medema and Schijven 2001). In one study in Pittsburgh, oocysts were detected in 33 percent of samples with a geometric mean concentration of 924 oocysts/100 L over 24 months of sampling (States et al. 1997). In another study near Philadelphia, concentrations ranged from 33 to 2,490 oocysts per 100 L (67 percent of samples were positive); downstream from the plant, concentrations ranged from 325 to 825 oocysts per 100 L (Crockett and Haas 1997). Results from more recent research summarized by Crockett (N.d.) indicate that effluent wastewater *Cryptosporidium* concentrations ranges from 0.1 to 1,000 oocysts/L.

CAFOs can be a significant source of animal waste, which can contaminate source water in two ways. If not properly managed, waste can leak or overflow from waste storage lagoons, feedlots, or other facilities. In addition, waste applied as fertilizer to fields can run off into drinking water sources or source tributaries, especially if over applied. Gullick et al (N.d.) provide

a reference guide for drinking water utilities to implement source water protection activities related to CAFOs. Finstein (2004) describes impact of ammonia and temperature on *Cryptosporidium* oocyst survival during storage of livestock manure, including capital and management intensive treatment processes like anaerobic digestion and less capital intensive processes such as aerobic composting.

*Nonpoint Sources*

Agriculture can also be a nonpoint source of *Cryptosporidium.* On a stream running through a small dairy farm before feeding into the Allegheny River, *Cryptosporidium* was detected in 82 percent of samples (States et al. 1997), with a geometric mean concentration of 42 oocysts/100 L. Twice during the 24-month study, concentrations of more than 1,000 oocysts/100 L were observed. In an agricultural area in Canada, drain tiles contained average concentrations of 771 oocysts/100 L. Concentrations were high even in tiles on farms without barns (these farms were assumed not to have livestock present). Oocysts were also present in some samples in liquid swine manure storage lagoons (Fleming et al. 1999).

Cattle are thought to be significant sources of oocysts. *Cryptosporidium* infection rates in cattle depend on animal age. Calves, particularly those less than one or two months old, have the highest rates (infection rates in different studies range from 2 to 39 percent of calves) (Wade et al. 2000, Sischo et al. 2000, Huetink et al. 2001).

*Cryptosporidium* may directly enter surface water via waterfowl. Oocysts have been found in Canada goose feces collected in the environment (Graczyk et al. 1998). Canada geese, some of which no longer migrate, could cause considerable contamination of surface water sources and uncovered finished water reservoirs.

Other wildlife may also be a source of *Cryptosporidium,* though the impact on source water may not be as direct. Deer, muskrat, and other small mammals were shown to carry *Cryptosporidium* in upstate New York (Perz and Le Blancq 2001). In one study of California ground squirrels, 16 percent of squirrels sampled were found to shed an average of 50,000 oocysts per gram of feces (Atwill et al. 2001). The infection rate in each species and the species present in each watershed will vary, so the contribution from wildlife will also differ from watershed to watershed.

Although little research has been performed on the overall prevalence of *Cryptosporidium* in pets, *Cryptosporidium* has been detected in dogs and cats, although pets usually carry strains that are rarely detected in humans. Several studies have shown dogs to be significant carriers of *Giardia,* fecal coliform, and other bacteria (Schueler 1999), and these microbes have been found in stormwater, suggesting that *Cryptosporidium* may also be present in urban watersheds and stormwater runoff.

Low levels of *Cryptosporidium* may also enter surface water through septic systems and subsequent subsurface transport (Lipp et al. 2001).

"Microbial source tracking" (MST) is a rapidly expanding science designed to identify human or animal sources of fecal pollution in the environment. These methods can evaluate *E. coli, Giardia*, *Cryptosporidium*, and viruses, and identify whether the source is from human,

cattle, swine, bird, or other origin. Fecal source tracking using DNA "fingerprinting" can be used by water utilities to identify whether or not particular a potential fecal source in their watershed, such as an AFO or CAFO, is an actual contributor to wastes identified in the source water.

*Cryptosporidium* sources can be identified through polymerase chain reaction (PCR) analysis of *Cryptosporidium* DNA. PCR can be used to determine the species or genotype of *Cryptosporidium;* many genotypes or species are typically, although not exclusively, found in specific hosts, such as cattle, dogs, and humans. In mixed-use watersheds, this information can help determine whether *Cryptosporidium* in the source water could have come from agricultural runoff, CSOs, or stormwater runoff.

Numerous references are available that summarize the capabilities and state-of-the-science of MST, including the following:

- U.S. EPA Microbial Source Tracking Guide Document (U.S. EPA 2005c) http://www.ces.purdue.edu/waterquality/resources/MSTGuide.pdf.

- Microbial Source Tracking: Current Methodology and Future Directions (Scott et al. 2002) http://aem.asm.org/cgi/content/full/68/12/5796.

- Microbial Indicators of Fecal Contamination: Application to Microbial Source Tracking (Bitton 2005) http://www.florida-stormwater.org/pdfs/FSAMicrobialSourceTrackingReport.pdf.

- Microbial Source Tracking and Detection Techniques (USGS website) http://water.usgs.gov/owq/microbial.html.

#### 2.4.2.1 How Do Fate and Transport Affect the Way *Cryptosporidium* Impacts My Water Supply?

Transport of oocysts in surface water and ground water and survival of oocysts all affect the potential impact of *Cryptosporidium* on water supplies. A critical review of available research on transport of pathogens in watersheds was conducted by Ferguson et al. (2003) and Davies et al. (2005). Pyke et al. (2003) summarized the occurrence, sources, and fate of *Cryptosporidium* in agricultural environments. Approaches for reducing overall pathogen loading within an agricultural watershed are discussed by Rosen et al. (2001). A summary of the fate of *Cryptosporidium* and the effectiveness for *Cryptosporidium* inactivation via different agricultural BMPs (e.g., composting, anaerobic digestion, manure slurry storage) was provided by Finstein (2004). Yeghiazarian et al. (2004) developed a pathogen transport model. Crockett (N.d.) discusses the role of wastewater treatment in protecting water supplies against emerging pathogens such as *Cryptosporidium*.

The fate and transport of oocysts in the environment is briefly described below.

*Influence of Precipitation*

Sixty-eight percent of waterborne disease outbreaks between 1948 and 1994 were shown to be associated with heavy precipitation (Curriero et al. 2001). *Cryptosporidium* occurrence may also be related to seasonal variations in infection among livestock, but any correlation is site-specific and depends on the source.

Crockett and Johnson (2000) noted that *Cryptosporidium* concentration increased by about a factor of 10 during storm events and frequency of detection doubled during these same storm events. Research at one New Jersey utility indicated that during storm events, large turbidity increases accompanied increases in *Cryptosporidium* occurrence in the raw source water (LeChevallier et al. 1998, Atherholt et al. 1998). Consequently, the utility monitors raw water turbidity and can shut down the intake for up to 24 hours when raw water turbidity exceeds a certain threshold (typically ~15 ntu) during storm events.

One study showed both *Cryptosporidium* detection and concentrations at six watersheds were highest between the months of October and April, with March experiencing a detection rate of more than 30 percent and oocyst concentration of about 0.038 oocysts/L (Sobrinho et al. 2001). Other studies have noted a connection between rainfall and "extreme runoff' events in tributaries to drinking water sources (Kistemann et al. 2002). One study noted a *decrease* in farm stream concentrations of *Cryptosporidium* with an increase in 5-day cumulative precipitation (probably because continued rainfall washed most of the *Cryptosporidium* downstream) (Sischo et al. 2000).

In a study in six watersheds, Sobrinho et al. (2001) found no substantial difference between *Cryptosporidium* detection rates during "event" (rainfall, high turbidity, melting snow and spring runoff) and "non-event" sampling when all data were taken together. However, for three of the watersheds, when examined individually, detection within each watershed was significantly higher during event sampling.

One recently completed research project (Sturdevant Rees et al., N.d.) investigated the variability of pathogen occurrence and transport through watersheds. Even small rainfall events (less than 0.25 inches) were capable of washing *Giardia* cysts and *Cryptosporidium* oocysts into streams, and saturated or near-saturated ground conditions, and events characterized by rain falling on snow, resulted in higher rates of detection. In addition, little correlation was found between rainfall event accumulation and *Cryptosporidium* or *Giardia* detection. Rather, antecedent rainfall and/or remotely sensed soil moisture data indicating saturated or near saturated conditions may be important for identifying rainfall events where sampling for *Cryptosporidium* and *Giardia* is warranted (Sturdevant Rees et al. N.d.).

*Transport in Surface Water*

The buoyancy of oocysts in water depends on their attachment to other particles. Oocysts that are not bound to particles have a tendency to float, even after being centrifuged (Swabby-Cahill et al. 1996). *Cryptosporidium* oocysts have a very low density (about 1.05 g/cm$^3$) and a very low settling rate (2 mm per hour or less) as noted by Gregory (1994). Oocysts attached to wastewater effluent particles may settle more quickly than those that are freely suspended and sedimentation velocity increases with particle size (Medema et al. 1998). In source waters, many

oocysts are likely to be adsorbed to organic or other suspended material and would probably settle more quickly than free-floating oocysts (Medema et al. 1998).

*Transport in Runoff*

*Cryptosporidium* is thought to be easily transported over land. Because oocysts are approximately the size of clay/silt particles, the amount of kinetic energy needed to entrain and suspend oocysts in overland flow may be quite small (Walker et al. 1998). Overland flow transport models for pathogens were reviewed as part of a U.S. EPA workshop on AFOs (GeoLogics Corp. 2004). The effects of land slopes, vegetation, and rainfall intensities on overland and near-surface transport of *Cryptosporidium parvum* oocysts were examined by Trask et al. (2004). Zhang and coworkers (2001) describe the development and implementation of a model for simulating removal of *Cryptosporidium parvum* oocysts from overland flow, including modeling transport of oocysts through vegetative filter strips.

*Transport in Ground Water*

Surface water sediments and the aquifer matrix material may play significant roles in minimizing oocyst transport to water supply wells; however, it is difficult to isolate the effect of these materials on transport. Fractures and dissolution conduits in an aquifer can allow ground water and oocysts to effectively bypass the protective action of most of the aquifer matrix. John and Rose (2005) provide a review of the factors affecting microbial survival in groundwater.

It is known that *Cryptosporidium* can be transported through soil and ground water (Mawdsley et al. 1996, Hurst 1997). Movement of C. *parvum* through soil and ground water is affected by sedimentation and filtration of the surrounding soil and aquifer matrix (Brush et al. 1999, Harter et al. 2000). Adsorption of oocysts to matrix particles also affects filtration. Adsorption depends on the electrical charge of the organism and of the surrounding matrix (Brush et al. 1998).

Factors other than adsorption and micropore size may influence the oocyst movement. C. *parvum* transport in one study was greater in a silty loam and a clay loam soil than in a loamy sand soil (Mawdsley et al. 1996); this contradicts other evidence suggesting that clay soils exhibit greater adsorption and smaller micropores than sandy soils. The authors used intact soil cores (rather than columns created in the laboratory) to maintain the natural soil structure and macropores, and they concluded that the rapid flow of water through macropores largely bypasses the filtering and adsorptive effects of the soil and increases the risk of *Cryptosporidium* transport to ground water (Mawdsley et al. 1996).

Amirtharajah et al. (2002) investigated the transport of a *Cryptosporidium* surrogate (polystyrene microspheres) through unsaturated soils at an undisturbed field site used for cattle production. Results showed that the vertical migration of polystyrene microspheres in column studies suggests that migration of *Cryptosporidium parvum* oocysts through fine-textured soils is likely to be minimal, and that a small number of these surrogate particles travel through preferential flow paths at field sites, especially after rainfall events (Amirtharajah et al. 2002).

*Survival in the Environment*

Several factors influence oocyst survival, including temperature and desiccation. Davies et al. (2005) presented results of a series of research experiments designed to identify the key factors controlling pathogen transport. Among the many findings summarized is that temperature is a very influential factor on the survival of *Cryptosporidium* oocysts. Both high heat and freezing temperatures can result in *Cryptosporidium* inactivation, while at more moderate temperatures (e.g., 4 to 25°C) inactivation rates are relatively slow (Davies et al., 2005, Finstein 2004). Freeze-thaw cycling is more effective than freezing, perhaps because of increased mechanical damage to the oocyst wall during the temperature fluctuations (Walker et al. 2001).

Before oocysts enter a water source, they may be vulnerable to desiccation. Robertson et al. (1992) reported that air drying an oocyst suspension at room temperature for 4 hours eliminated viability. Oocysts in fecal material, however, are protected from desiccation, so their viability in the environment is prolonged (Rose 1997). In addition, *Cryptosporidium* in liquid swine manure has been shown to remain viable despite the high ammonia content of the manure (Fleming et al. 1999). However, Olson et al. (1999) found that oocyst survival appears to be better in soil than in feces.

Once initial contamination has occurred, water can remain a source of viable oocysts for days (Heisz 1997, Lisle and Rose 1995). Lisle and Rose reported a duration of 176 days to produce die-off rates of 96 percent in tap water and 94 percent in river water under laboratory conditions. After 2 days, only 37 percent of the oocysts in tap water were nonviable, suggesting that oocysts that reach the distribution system might be viable.

Olson et al. (1999) compared oocyst survival at temperatures and in media likely to occur in the natural environment. They examined survival in -4°, 4°, and 25°C. Unlike *Giardia,* which died off quickly at low temperatures, *Cryptosporidium* oocyst survival was best at -4°C, with close to 50 percent of oocysts remaining viable for 12 weeks. Survival was lowest at 25°C, but oocysts were still viable at six weeks. Survival rates were best in water and worst in feces.

Soil texture also can significantly affect inactivation of *Cryptosporidium* oocysts and viruses (CRC 2004). Vegetative filter strips can be effective at removing *Cryptosporidium* oocysts from surface runoff, however, viruses (e.g., PRD1 bacteriophage) and to a lesser extent bacteria (*E. coli*) are more easily transported (Davies et al. 2005, CRC 2004).

*Loading*

Once you have gathered information about *Cryptosporidium* sources and the likelihood of the oocysts reaching your source water (based on watershed characteristics and fate and transport), you should determine the amount and proportion of oocysts that each source is expected to contribute to the overall *Cryptosporidium* load. Loading can be calculated fairly easily for constant point sources such as wastewater treatment plants but is more difficult for farms and urban runoff.

### 2.4.2.2 What Role Should Monitoring Play in the Evaluation of Potential and Existing Sources of *Cryptosporidium*?

Monitoring of *Cryptosporidium* is not required to develop the WCP plan or to implement it once approved by the state. It may take years to realize measurable improvements in water quality after initiating source water protection efforts. Furthermore, discerning improvements in source water quality using monitoring can be difficult due to natural environmental variability, the nature of the source water improvements, and the limitations of the current analytical methods for *Cryptosporidium* as well as other fecal indicators. However, PWSs that choose to employ this option, either separately or in combination with other approaches, may gain some benefit using this approach. For example, while the state and/or PWS may already have some knowledge of potential *Cryptosporidium* sources through land use information or discharge permit data, monitoring can help determine the extent to which these sources are impacting a source and can help target portions of the watershed for extra protection or BMPs implementation. Although not required for WCP plan development, implementation, or maintenance, monitoring throughout the watershed for *Cryptosporidium* (or indicators of fecal contamination) can be a useful tool in evaluating the success of WCP controls

New technologies for MST including deoxyribonucleic acid (DNA) fingerprinting, genotyping, and multiple antibiotic resistance may be helpful and more effective at overall pathogen source identification (see Section 2.4.2 for references). Presumptive approaches using modeling and literature reported in data can also be used to simulate loadings and prioritize areas for monitoring or detailed study.

Watershed monitoring can help narrow down the locations of some sources and determine the load contributed by each source. The Philadelphia Water Department, for example, planned a four-tier study to determine why there was such a large difference in protozoan levels at two plants using the same source (the Schuylkill River) but located 2.5 miles apart (Crockett and Haas 1997) (see sidebar).

Because *Cryptosporidium* occur in low concentrations and are difficult to detect, it may be helpful to monitor other parameters in addition to or instead of *Cryptosporidium.* While *E. coli* concentrations often do not correlate with *Cryptosporidium* levels, they are good indicators of fecal contamination. Fecal coliform bacteria have traditionally been used as water quality indicators, but *E. coli* is thought to be more closely linked to fecal contamination.

Turbidity does not always indicate fecal contamination; often, increased turbidity is simply a product of high sediment levels. However, turbidity may indicate the presence of a

water quality problem, where additional research is necessary to determine its cause. Use of a surrogate such as turbidity should be supported by evidence of a correlation for that source water. Precipitation (rainfall amount and intensity) are important in the release and detachment of pathogens from fecal matter, and consequent mobilization in downgradient surface water or groundwater. This precipitation also can mobilize and release turbidity.

A New Jersey utility has shown that for their source water increases in raw water turbidity are accompanied by increases in source water *Cryptosporidium* during storm events (LeChevallier et al. 1998, Atherholt et al. 1998). As a result of this research for their source water, this utility has established a standard operating procedure that when raw water turbidity exceeds a threshold of about 15 ntu, intake can be shutdown for up to 24 hours without interrupting service in order to let storm related source water flow bypass the treatment plant.

Monitoring, when implemented, should be conducted regularly. Because nonpoint sources of microbiological contamination discharge primarily during wet weather flows monitoring during or soon after these events is also important. When combined with stream discharge data, rates of storm-related *Cryptosporidium* transport and loading can be calculated. The monitoring frequency should be such that seasonal variability in *Cryptosporidium* levels is observable.

There are two types of watershed monitoring for stream networks. First, basinwide monitoring involves monitoring just upstream of the confluence of two streams (AwwaRF 1991). Conducted at stream junctions throughout the watershed, basinwide monitoring helps give a general picture of the water quality and helps isolate the stream reaches contributing to contamination. Second, site-specific monitoring involves monitoring just upstream and downstream of a suspected or known point or nonpoint source, as the Philadelphia Water Department did (Crockett and Haas 1997). Such monitoring is appropriate where impacted stream reaches have already been identified. The results of any monitoring should enable the system to compare the relative contribution of various sources to the overall *Cryptosporidium* occurrence in the watershed and their effect on water quality.

**Monitoring to Locate *Cryptosporidium* Sources**

*To determine the source of Cryptosporidium contamination in the Schuylkill River, the Philadelphia Water Department decided to focus on a creek feeding into the Schuylkill just upstream of the Queen Lane plant (Crockett and Haas 1997). This creek has several wastewater treatment plants in its upper reach and farms and parks along its lower reach. In the first phase, the water department tested the Queen Lane intake during dry flow. It then sampled a site along the creek downstream of the wastewater treatment plants and one downstream of the farms during various weather conditions. In the third phase of sampling, the department sampled wastewater effluent and additional sites up- and downstream of some of the wastewater treatment plants during different weather events. Lastly, the department planned to focus on the prevalence of Cryptosporidium and Giardia in livestock and wildlife along the creek.*

Monitoring in a reservoir or lake, if applicable, can help systems determine the fate of *Cryptosporidium* once it flows from a stream into the lake, or once it enters the lake directly from land immediately adjacent to the lake. Sampling patterns should depend on the shape and depth of the lake. A round lake should be sampled at several locations and depths near the center of the lake; a long lake should be sampled in a transect along its long axis (AwwaRF 1991). More specific monitoring may be needed to answer more detailed questions on fate and transport. For instance, does *Cryptosporidium* concentration decrease due to sedimentation or dilution? How long does it take for *Cryptosporidium* to flow from one end of the reservoir to the intake?

PWSs may find it helpful to use a GIS to analyze their water quality and contaminant source data. For systems that have ArcView software, BASINS 3.0, a software and GIS package developed by EPA can assist systems with integrating local data and nationally available pre-formatted spatial data (e.g., watershed HUCs, DEM data, roads, NPDES permit data, and Clean Water Needs Survey data on wastewater treatment plants). BASINS also includes a model for determining nonpoint source loading and other models for loading and transport, as well as tools for assessing contamination from various sources.

### 2.4.3 Analysis of Control Measures

The analysis of control measures submitted with the WCP plan must address the relative effectiveness of each measure at reducing *Cryptosporidium* loading to the source water, along with the feasibility of each measure (40 CFR 141.716(a)(2)(iii)). This analysis can be based on either site-specific experience or information from the peer-reviewed literature. Numerous references are available that describe control measures and other BMPs, as listed previously in this Section 2.4 of this guidance, and presented below for select topic areas.

Control measures may include 1) the elimination, reduction, or treatment of wastewater or stormwater discharges, 2) treatment of *Cryptosporidium* contamination at the sites of the waste generation or storage, 3) prevention of *Cryptosporidium* migration from sources, or 4) any other measures that are effective, sustainable, and likely to reduce *Cryptosporidium* contamination of source water.

#### 2.4.3.1 Available Regulatory and Management Strategies

For systems in watersheds where most of the land is privately owned, land use regulations may be the best way to control pollution, especially in heavily developed or growing areas. Examples of possible regulations include septic system requirements, zoning ordinances specifying minimum lot sizes or low-impact development, limits on discharge from wastewater treatment plants and other facilities, pet waste cleanup ordinances, and requirements for permits for certain land uses. Your ability to regulate land use will depend on the authority granted to your municipality by the state, the ownership of your system (public or private), and the support of your local government and the public. Regulatory authority, steps for designing a regulation that can withstand lawsuits, and types of land use regulations are described in the paragraphs below.

*Determining Authority to Regulate*

The ability of a municipality to pass a land use ordinance or other law to help reduce contamination may depend on the authority the state grants to the local government in the state constitution or through legislation, although states normally do not interfere with the actual land use and zoning rules (AwwaRF 1991). Privately owned water systems may need to ask the cooperation of the local government to get source water regulations passed. Publicly owned PWSs face less of a hurdle, although winning support of the local government may still be difficult.

If a PWS does not own or otherwise have authority over the *Cryptosporidium* sources in the watershed, the analysis will need to reflect developing and maintaining partnerships to assure adequate control is in place. This could include coordination with other municipal governments, farmers, wastewater treatment plant operators, regional planning agencies, and others.

If the area of influence on water quality extends throughout several municipalities, it can be difficult to standardize watershed control practices throughout the watershed. The legal framework used will depend on who has jurisdiction over land use in the watershed and on the authority of the water system (AwwaRF 1991). For example, some states may create agencies authorized to promulgate and enforce watershed protection regulations, or interstate agencies may be created to regulate watersheds where watersheds cross state boundaries. County governments in some states may have some zoning authority and may be able to assist with enforcement of some regulations affecting source water (e.g., septic systems).

Where PWSs do not have regulatory or enforcement authority, they should work with other local governments' PWSs and agencies in their watersheds to sign memoranda of agreement or understanding, in which each entity agrees to meet certain standards or implement certain practices.

*Zoning*

Early zoning laws simply prohibited certain land uses that would be considered nuisances in certain areas. Later, zoning ordinances became more specific; further restrictions were imposed on the permitted uses, such as limits on building or population density, percentage of impervious surface area, building height, and minimum distance of buildings from property boundaries. Most zoning ordinances have grandfather clauses that allow nonconforming uses to continue. Ordinances may also allow the zoning authority to grant variances if the topography or size of a lot make it difficult to comply with a zoning requirement.

To make sure a zoning law can withstand a legal challenge, it is important to make sure the appropriate procedures are followed and that the law has sufficient scientific basis (AWWA 1999). First, be sure you have the authority to regulate. Make sure the rule is specific enough. Comply with all administrative procedure requirements; failure to do so is the most common reason for rules being revoked. The ordinance should conform to the objectives of the WCP plan, which should contain enough data to illustrate how the ordinance will affect water quality.

Ordinances should also be designed to withstand a takings lawsuit (AWWA 1999). The fifth amendment to the U.S. Constitution states that private property may not be taken for public

use without just compensation. Any physical invasion without consent is always considered a taking, even if the landowner retains ownership of the land. Installation of a monitoring well or stream gauge without consent is an example of a taking.

To prevent takings claims, the municipality should show the need for the regulation and a connection between the ordinance and the expected result (AWWA 1999). This proof should be based on a scientific analysis beginning with an accurate delineation of the watershed or wellhead protection area/recharge area.

Following the delineation, determine the impact the regulation will have by mapping current and projected development under current zoning requirements. Then map current and projected development for the proposed ordinance and determine the potential pollutant load under each scenario (AWWA 1999). Local groups or universities may be able to provide pollutant data and assist with modeling. This "buildout analysis" will help you show that your proposed ordinance advances a legitimate government interest and how the effect of the ordinance is proportional to the impact of land use in your watershed.

*Types of Ordinances*

Watershed ordinances usually apply within an "overlay district," which may be the area of influence you determined for your WCP plan. All existing zoning or land use regulations apply within that area, but additional requirements apply within the overlay district. Within your watershed, particularly within the area of influence, there are many different kinds of regulatory controls you may wish to consider:

- Large-lot or low-density zoning.
- Limits on certain types of land use except by special permit.
- Impact fees.
- Submission and approval of a watershed protection plan or impact study as a condition for development of a subdivision or apartment complex.
- Performance standards, which permit development but limit the impact of the development.

More detail on each of these types of ordinances is found in Appendix E. Examples of source water protection ordinances can be found on EPA's website at http://www.epa.gov/owow/nps/ordinance/osm7.htm.

*Land Acquisition/Conservation Easements*

Acquisition of watershed land by the PWS or its affiliated jurisdiction is often the most effective approach to protecting the water source. EPA's Drinking Water State Revolving Fund allows a percentage of the fund to be set aside for land acquisition associated with watershed protection.

Land trusts and conservancies can help systems purchase land to protect drinking water quality, especially when government agencies are unable to move quickly enough to buy land when it becomes available. Trusts can buy and hold land from multiple landowners on behalf of a water system until the system can assemble funding to purchase it from the trust. The Trust for Public Land (http://www.tpl.org) can provide more information.

Trusts also can work with landowners to buy or have landowners donate conservation easements. An easement is a legal document that permanently limits the development of a piece of land, even after the land is sold or otherwise changes ownership. See http://www.landtrust.org/ProtectingLand/EasementInfo.htm for frequently asked questions about easements and for an example of a model easement for use in the state of Michigan. The Land Trust Alliance (http://www.lta.org), a trade organization for land trusts, has published handbooks on designing and managing conservation easement programs.

Other government agencies, such as the U.S. Forest Service or state natural resource departments, may be able to buy parcels in your watershed if you are unable to afford to purchase all the land that needs to be protected.

#### 2.4.3.2 Partnerships in Watershed Control Plans

Many watershed management practices cannot be implemented by water systems alone. For example, agricultural BMPs must be implemented by farmers; stormwater BMPs are implemented by developers, manufacturers, and government agencies. Parts of your watershed may be in different municipalities. Therefore, partnerships with local government and landowners is often central to effectively implementing a watershed control program. The WCP can reflect a variety of different types of partnerships:

- Memoranda of agreement or other formalized arrangements with government agencies.
- Education through technical assistance providers such as cooperative extension agents or association representatives.
- Data collection through local university programs.
- Agreements to hold conservation easements with state agencies or non-governmental organizations.
- Private agreements with individual property owners.

Stakeholder participation can be a useful tool in WCP planning. Dialogue with stakeholders can be used to identify win-win solutions for both the water supplier and its partners. The book *Guidance to Utilities on Building Alliances with Watershed Stakeholders* (AwwaRF 2001) explains how to present issues to stakeholders, how to target stakeholders, and how to structure your partnership with stakeholders. In addition, developing alliances between water utilities and agricultural interests is discussed by Fletcher et al. (2004).

An important potential partnership opportunity that many water systems should consider when pursuing the WCP credit is to develop a cooperative relationship with other PWSs in the same watershed. These PWSs in the same watershed will normally have overlapping areas of influence and consequently will have some of the same priority *Cryptosporidium* sources. Consequently, each of these PWSs will have an interest in developing control measures for these shared *Cryptosporidium* sources. By cooperating together the PWSs can reduce duplication of efforts and thereby collectively focus their energies on prioritized activities. PWSs that cannot cooperate together should not pursue joint efforts, because this will create difficulties during implementation and maintenance of the WCP plan. However, if they can develop a plan to work together to prepare the WCP plan, and can provide the staff and resources to complete their designated tasks during implementation and maintenance of the state-approved WCP plan, the resulting joint effort can potentially produce a greater benefit (reduction in source water *Cryptosporidium* ) at lower cost than the if each PWS worked separately.

Watershed control plans must identify watershed partners and their roles (40 CFR 141.716(a)(2)(iv)). Plans should document the efforts to be made to establish voluntary local partnerships, including solicitation of private individuals living within the defined area of influence who are likely to be affected by decisions made as part of the watershed protection program, and whose participation is important for the success of the program. Plans should also document how members of municipal or other local governments or political subdivisions of the state that have jurisdiction over the area of influence will participate in the watershed protection effort. Watershed protection plans should include descriptions of how the proposed local partnership has or will identify and account for any voluntary or other activities already underway in the area of influence that may reduce or eliminate the likelihood that *Cryptosporidium* will occur in drinking water.

#### 2.4.3.3 Addressing Point Sources

Changes in farming practices and in wastewater treatment technologies in the past decade have resulted in new management strategies for agricultural and urban point sources. The following sections briefly describe solutions for agricultural, wastewater, and stormwater point sources; detailed descriptions are provided in Appendix E. As part of your application for WCP approval, you must submit an analysis of control measures that can mitigate sources of *Cryptosporidium* such as these (40 CFR 141.716(a)(2)(iii)). Loans from the Clean Water State Revolving Fund (CWSRF) can be used to fund projects associated with wastewater treatment and watershed and estuary management. See www.epa.gov/owm/cwfinance/cwsrf/index.htm for more information.

*Concentrated Animal Feeding Operations*

AFOs are facilities where animals are confined for 45 days or more a year and where no vegetation grows in the area used for confinement. This includes farms where animals graze the majority of the year but are confined and fed during the winter for at least 45 days. Some AFOs are also CAFOs (see Appendix E). EPA recently issued a rule that changed the requirements on CAFOs that must apply for NPDES permits (U.S. EPA 2008). The new CAFO rule requires CAFOs to implement nutrient management plans that affect manure handling, storage, and land application. These plans will include BMPs primarily designed to reduce nitrate and phosphorus

contamination but which will at the same time reduce pathogen contamination. Elements of this plan may include limiting the manure land application rate, instituting buffer zones where manure is applied, ensuring adequate manure and wastewater storage, and others. Gullick et al. (2007 provide a reference guide for drinking water utilities to implement source water protection activities related to CAFOs.

*Wastewater Treatment Plants*

All wastewater treatment plants in the United States are required to provide secondary treatment (U.S. EPA 2001e). Most plants are also required to disinfect their effluent before discharging. However, conventional chlorine disinfection in wastewater plants is ineffective against *Cryptosporidium.* Some wastewater treatment facilities are beginning to implement treatment similar to that used for drinking water treatment (e.g., filtration, advanced treatment such as UV disinfection). PWSs should identify all wastewater treatment plants in their watersheds and determine what their permit effluent limits are and whether the limits are being met.

*Combined Sewer Overflows (CSOs)*

Combined sewers carry both sewage and stormwater to wastewater treatment plants. During storms, the volume of water in combined sewers may become too great for wastewater plants to treat. As a result, the excess sewage and stormwater are released untreated into surface water through CSOs. CSOs are most common in older cities in the northeastern and midwestern United States and can be a significant contributor of *Cryptosporidium* to urban watersheds.

There are three major structural solutions to the problem of CSOs:

- Separate combined sewers into sanitary and storm sewers, where sanitary sewers flow to the wastewater treatment plant and storm sewers release to surface water.
- Increase the capacity of the wastewater treatment plant so that it is able to treat combined sewage from most storms.
- Build aboveground covered retention basins or construct underground storage facilities for combined sewage to hold the sewage until the storm has passed and can be treated without overloading the plant.

Although CSOs are not regulated directly under their own program, EPA has a CSO control policy (U.S. EPA 1994) which encourages minor improvements to optimize CSO operation, and CSO management may be written into NPDES or State Pollution Discharge Elimination System (SPDES) permits. Minor improvements include maximizing in-line storage within the sewer system, reducing inflow, and treatment of CSO outfalls. Stormwater BMPs can also reduce the impact of CSOs.

*Sanitary Sewer Overflows*

Watersheds with separate sanitary and storm sewer systems may still have water quality problems. Sanitary sewer overflows (SSOs) occur when untreated and mostly undiluted

sewage backs up into basements, streets, and surface water. SSOs discharging to surface water are prohibited under the CWA. Insufficient maintenance and capacity and illegal connections are some of the primary causes of SSOs.

SSOs can be reduced by cleaning and maintaining the sewer system; reducing inflow and infiltration by repairing leaking or broken service lines; increasing sewer, pumping, and/or wastewater treatment plant capacity; and constructing storage for excess wastewater (U.S. EPA 2001f).

*Municipal Separate Storm Sewer Systems*

Municipal separate storm sewer systems (MS4s) in areas with populations of more than 100,000 are also required to obtain NPDES permits. Information on storm sewer outfall locations, volume discharged, conventional pollutant loads, and existence of illicit discharges is submitted as part of the permit application process (U.S. EPA 1996). In addition, these MS4s must develop management plans addressing items such as outfall monitoring, structural and nonstructural BMPs to be implemented, and identification and elimination of illicit discharges. Illicit discharges to MS4s include any non-stormwater discharges, such as discharges that should be connected to sanitary sewers (e.g., water from sinks, floor drains, and occasionally toilets), illegal dumping of sewage from recreational vehicles, sanitary sewer overflow backing up through manhole covers into storm drains, effluent from failing septic systems, water from sump pumps, etc.

Small MS4s (serving areas with populations of less than 100,000), with some exceptions, are subject to NPDES permit requirements if they are located in "urbanized areas" as determined by the Bureau of the Census. Those MS4s subject to NPDES permits must implement "control measures" in six areas, including a plan for eliminating illicit discharges (U.S. EPA 2000b).

PWSs should work with all MS4 utilities in the area of influence to gather existing information about stormwater contamination. MS4 utilities may need to install or retrofit structural BMPs, such as retention ponds, to reduce contamination.

#### 2.4.3.4 Addressing Nonpoint Sources

The following sections briefly describe BMPs for agricultural, forestry, and urban sources of *Cryptosporidium;* more detailed descriptions are provided in Appendix E. Your WCP plan must discuss how these or any other BMPs you choose will be implemented in the area of influence. EPA Section 319 grants and Clean Water State Revolving Fund loans can be used for nonpoint sources and watershed management purposes.

*Agricultural BMPs*

Management Programs

The U.S. Department of Agriculture recommends the following "control points" for controlling pathogens (USDA 2000):

- Preventing initial infection by controlling pathogen import to the farm.
- Controlling the reproduction and spread of the pathogen throughout the farm.
- Managing waste.
- Controlling pathogen export from the farm.

PWSs should work with local soil conservation districts or cooperative extensions for technical assistance with BMPs.

BMPs that can reduce pathogen loading include the following:

- Composting.
- Waste management (manure storage and land application).
- Grazing management.
- Feedlot runoff diversion.
- Buffer or filter strips.

*Composting*

- Can effectively reduce pathogen concentrations.
- Entire waste mass should be uniformly treated and there should be no cold spots.

*Buffer Strips*

- Provide buffer between area of manure application or grazing and adjacent streams or lakes.
- USDA (2000) recommends that buffer and filter strips be considered secondary practices for pathogen control and be used in conjunction with control measures.

*Grazing Management*

- Managed grazing can be cheaper and less environmentally damaging than confined feeding and unmanaged grazing. It decreases feed, herbicide, equipment, and fertilizer costs, while reducing erosion and increasing runoff infiltration and manure decomposition rates (Ohio State University Extension, undated).

- In managed, or rotational, grazing, a sustainable number of cattle or other livestock graze for a limited time (usually 2-3 days) on each pasture before being rotated to the next pasture.

*Manure Storage*

- Manure storage facilities allow farmers to wait until field conditions are more suitable for land application.

- Manure storage facilities should be designed to prevent discharge through leaching or runoff. They should be lined, and if possible, covered. Facilities that are not covered should be designed to contain precipitation and runoff from a 25-year 24-hour storm.

*Land Application of Manure*

- Several precautions taken in manure application can prevent runoff from entering surface water, reducing the likelihood of *Cryptosporidium* contamination.

- Manure should not be applied to frozen or saturated ground or before predicted rainfall, or near tile drains or dry wells or to land subject to flooding.

- For pastures to be used for grazing, waste should be stored for at least 60 days and then applied at least 30 days before the scheduled grazing period to avoid infection of the animals.

*Feedlot Runoff Diversion*

- Diverting clean water before it drains into the feedlot can significantly reduce the amount of wastewater that needs to be managed.

- All roofs that could contribute to feedlot runoff should have gutters, downspouts, and outlets that discharge away from the feedlot.

*Forestry BMPs*

- Logging can cause increased erosion, leading to increased runoff and making it more likely that *Cryptosporidium* present in wildlife will reach the source water. Logging can also cause elevated sediment levels, resulting in high turbidity, which affects water treatment efficiency. Examples of forestry BMPs are listed below:

  - filter strips
  - streamside or riparian management zones
  - logging roads should be constructed to minimize runoff
  - road runoff should be diverted away from streams and prevented from channelizing
  - loggers should minimize soil disturbance and compaction on skid trails

*Urban/Suburban BMPs*

See http://www.epa.gov/owm/mtb/mtbfact.htm for fact sheets on technologies and BMPs municipalities can use to reduce contamination from wastewater and stormwater.

*Buffer Zones*

- For watersheds in urban areas, buffer zones help to protect development on the floodplain from being damaged when the water is high, as well as protect the stream from the effects of the development.
- The extent to which buffer zones reduce *Cryptosporidium* loading is not well understood; therefore, they should be used to augment, rather than replace, other watershed management practices.

*Dry Detention Basins*

- Dry detention basins temporarily store stormwater runoff and release the water slowly to allow for settling of particulates and the reduction of peak flows.

*Infiltration Devices*

- Infiltration devices remove pathogens and particles by adsorption onto soil particles and filtration as the water moves through the soil to the ground water. Infiltration devices include (NALMS 2000):

  - infiltration basins
  - infiltration trenches
  - dry wells

*Sand Filters*

- Sand filters can be used to treat stormwater runoff from large buildings and parking lots.

*Wet Retention Ponds*

- Ponds can effectively reduce suspended particles and, presumably, some pathogens, by settling and biological decomposition.

- There is concern, however, that ponds attract wildlife that may contribute additional fecal pollution to the water, rather than reducing contamination.

*Constructed Wetlands*

- Constructed subsurface flow wetlands (where wetland plants are not submerged) can reduce *Cryptosporidium* and bacteria concentrations in wastewater (Thurston et al. 2001).

- Wetlands may also be useful for treating stormwater or other polluted water.

*Runoff Diversion*

- Structures can be installed in urban settings to divert clean water flow before it reaches a contamination source. Structures that channel runoff away from contamination sources include stormwater conveyances, such as:

  - swales

  - gutters

  - channels

  - drains

  - sewers

*Pet Waste Management*

- Municipalities can implement pet waste management programs to encourage pet owners to properly collect and dispose of their animals' waste.

*Water Conservation*

- Can help preserve the amount of water available for use, especially during times of drought.
- Can also decrease the amount of wastewater and stormwater generated, thereby protecting the quality of the water supply (U.S. EPA 2002d).
- The following are examples of water conservation methods:
  - low-flow toilets and showerheads
  - reducing lawn watering

*Low Impact Development*

- Low impact development tries to reduce the amount of impervious cover, increase natural lands set aside for conservation, and use pervious areas for more effective stormwater treatment of residential and commercial developments.

*Septic Systems*

- Failing septic systems can result in clogging and overflow of waste onto land or into surface water.
- Water systems should work closely with the local regulatory authority to ensure that septic system codes are being properly enforced and to strengthen codes where necessary.
- PWSs should encourage residents with septic systems in the watershed to understand their systems and the proper maintenance that their systems require. Cooperative extensions can work with residents on this issue.

*Wildlife BMPs*

- Steps taken to prevent wildlife from contaminating source water vary with the source and type of wildlife. The following are examples of wildlife BMPs:
  - boats with noisemakers to scare seagulls and geese away
  - fences on the water's edge to keep out larger land animals and humans

#### 2.4.3.5 Is Purchase/Ownership of All or Part of the Watershed a Viable Option?

PWSs will have the maximum opportunity to realize their watershed protection goals if they have complete ownership of the watershed. Where feasible, PWSs should include in their watershed protection plan a description of efforts that will be made to obtain ownership, such as any special programs or budget. Since complete ownership of the watershed or area of influence is

not practical in almost every instance, the system should explain any efforts the PWS will make to gain ownership of some critical elements, such as reservoir or stream shoreline and access areas.

Where ownership of land is not possible, PWSs should describe plans to obtain written agreements that recognize the watershed as part of a public water supply. As much as possible, maximum flexibility should be given to the PWS to control land uses which could have an adverse effect on the water quality. PWSs should include with these descriptions an explanation of how they will ensure that landowners will comply with the agreements.

Utilities can also facilitate the prioritization and purchase of parcels by third parties in upstream communities that are already looking to preserve, own, and maintain land. This can be done by partnering on grants or other efforts.

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# 3. Alternative Source/Intake

## 3.1 Introduction

Changing the water source or intake location can improve source water quality and reduce treatment requirements for the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). The rule states that systems may be classified in a bin based on monitoring of an alternative source or intake location or monitoring using an alternative procedure for managing the timing or level of withdrawal. This monitoring must be conducted in addition to and concurrently with monitoring conducted using the existing intake or withdrawal practice. Since the LT2ESWTR requires that alternative monitoring be conducted concurrently with source water monitoring, this toolbox option needs to be evaluated ***prior* to the start of source water monitoring**. After monitoring and with state approval, a system would then choose which source, intake location, or intake procedure it will use based on bin classification results. (40 CFR 141.716(b))

This chapter discusses the concurrent monitoring options of changing sources, moving the plant intake, and managing the timing or level of withdrawal and is organized as follows:

3.2 Changing Sources - discusses factors to be considered in changing sources, including advantages and disadvantages and influence of source water characteristics on existing treatment requirements.

3.3 Changing Intake Locations - discusses the applicability of changing the intake locations and variables affecting *Cryptosporidium* concentrations in reservoirs, lakes, streams, and rivers.

3.4 Changing Timing of Withdrawals - describes different approaches, and advantages and disadvantages to changing the timing of withdrawals.

## 3.2 Changing Sources

In order to be able to relocate an intake to a different source, a system would need to identify an unallocated source within a reasonable distance of its treatment plant. It is recommended that both drinking water programs and state water resource agencies be contacted regarding putting new sources into service. The new source would require sufficient unallocated flow to meet the system's needs, including those for peak flow and future growth. The effect of the different water quality on the existing treatment process should also be considered.

### 3.2.1 Advantages and Disadvantages

The main advantage of changing sources is avoiding the addition of a new treatment process. The capital expense of a new well or new intake may be less than the expenses associated with installing and operating a new treatment technology. In addition to having a lower *Cryptosporidium* concentration, the new source may also have better water quality that could reduce treatment costs. Systems should assess any potential new source to ensure its integrity, quantity, and quality. In addition, switching to a new source often requires approval by the state.

A disadvantage associated with changing sources is that the different source water may respond differently to the treatment train already existing at the plant. This may require changes in plant operating procedures, such as changing the type and amount of coagulant added, the length of filtration runs, and the dose of disinfectant added. Another disadvantage is that the source may be lower in *Cryptosporidium* concentration but have higher concentrations of other contaminants. There may also be legal and environmental issues associated with tapping a new source. Plant standard operating procedures (SOPs) should be updated if a new source is added. Finally, the cost of installing a new intake and transmission line should be considered; depending on the location of the source or intake in relation to the plant or to existing transmission lines, a new source/intake could be more expensive than other toolbox options.

### 3.2.2 Evaluation of Source Water Characteristics for Existing Treatment Requirements

If a new source is to be introduced to an existing treatment plant, the treatability of the new water by the existing process should be considered. For example, in a conventional treatment train consisting of coagulation, sedimentation, and dual media filtration, each source water will have unique coagulation properties depending on its characteristics. Organic content, alkalinity, and pH all affect the coagulation process. Consequently, water quality parameters including pH, alkalinity, total organic carbon (TOC), UV254, turbidity, and iron and manganese concentrations should be measured and evaluated against the existing water and the treatment train. If coagulation is used as a part of the treatment process, jar tests should be conducted to determine the coagulation and settling properties of the new water and to aid in calculating the required dose of coagulant. (See American Water Works Association (AWWA) Manual M37, Operational Control of Coagulation and Filtration Processes for more information on jar testing.) Pilot plant studies can also help determine the treatability of a proposed new source.

## 3.3 Changing Intake Locations

Another method for reducing *Cryptosporidium* source concentrations is to move the intake within the same source. This could involve relocating an intake within a source or changing the depth from which the intake draws.

### 3.3.1 Applicability

Relocating an intake can be a good strategy if an obvious source of *Cryptosporidium* is present which can easily be avoided by moving the location of the intake. One example of such a situation is if an intake could be moved upstream of a municipal wastewater discharge in a river, where it had previously been located downstream of the discharge.

#### 3.3.1.1 Advantages and Disadvantages

One advantage of moving the location of an intake is its potentially low relative cost, if the distance the intake must be moved is relatively short. This option could be particularly attractive if an existing intake structure can be used to withdraw water from a different depth, resulting in decreased *Cryptosporidium* concentrations.

Disadvantages could include significant amounts of excavation and piping, as well as additional pumping if the intake must be relocated a considerable distance. Also, altering the intake may not bring the desired reduction or provide any additional protection against future increases or spikes in *Cryptosporidium* concentration.

### 3.3.2 Reservoirs and Lakes

Several variables can affect the concentration of *Cryptosporidium* at a particular location in a reservoir or lake, including the intake depth, the way in which the lake mixes, the thermal properties of the lake, and the proximity of the intake to streams and other discharges. It is recommended that a water system develop an SOP for water withdrawal based on the specific conditions of the waterbody being used as the source.

#### 3.3.2.1 Depth

The intake depth can significantly change the quality of the water being drawn and used. In general, shallow intakes are more likely to draw water exposed to recreational activity and surface water runoff. Deeper intake locations are often more protected from sources of *Cryptosporidium*, unless an intake location is so deep that it draws water containing re-suspended material from the lake or reservoir bottom. Water systems are often well-advised to draw water from intermediate depths, where they can avoid higher oocyst concentrations that may exist near the lake or reservoir surface, and also avoid particles that may be stirred up near the bottom.

#### 3.3.2.2 Stratification and Mixing

Another factor that can affect the depth profile of *Cryptosporidium* in a lake or reservoir is the amount of stratification or mixing present. Larger lakes and reservoirs often stratify, especially in the summer months, forming a hypolimnion (a cold lower layer) and an epilimnion (a warm upper layer) separated by a thermocline. There is very little mixing between these layers when a lake is strongly stratified. Particles may settle through the layers, but there is little other mixing. The epilimnion is often well mixed because of the mixing action of wind. Therefore, it is likely that *Cryptosporidium* may be present at uniform concentrations throughout the epilimnion. *Cryptosporidium* oocysts that have attached to particles and settled will have a concentration gradient in the hypolimnion. The shape of any concentration gradient will depend on local conditions such as temperature, stream inflows, and particle settling rates. Lakes or reservoirs that are strongly stratified and have a high input of organics can often develop anoxia in the hypolimnion. Therefore, all water quality parameters should be considered before determining the depth from which to draw the water. Extremely high withdrawal rates may provide enough energy to overcome stratification and draw from the layer outside of where the intake is located.

#### 3.3.2.3 Proximity to Inflows

The proximity of the intake to stream inflows may affect the quality of the intake. Streams carrying agricultural or urban runoff can cause water quality degradation if located too close to a source water intake. States et al. (1998) reported an increase in *Cryptosporidium* concentrations with wet weather events, particularly as the sampling location became closer to the contamination source. Kortmann (2000) reported a system substantially reduced coliform bacteria in their source water by moving their intake further away from a stream which drained an agricultural area and by installing an artificial partition in the reservoir to limit the exchange of water between the stream input and the rest of the lake.

### 3.3.3 Streams and Rivers

There are several factors to consider when deciding where to locate an intake on a river including depth, flow hydraulics, seasonal effects, and upstream sources of contamination.

#### 3.3.3.1 Depth

Depth is not as likely to affect *Cryptosporidium* concentrations in small rivers and streams as it is in lakes and reservoirs. Fast moving or shallow streams are likely to be fairly mixed across all depths. In contrast, deeper and slower moving rivers may be less mixed and may show some concentration gradient of *Cryptosporidium* with unattached oocysts being greater near the surface and oocysts attached to particles being greater near the bottom. In rivers and streams, intakes located near the bottom are more likely to draw sediment and other particles resuspended from the bottom.

#### 3.3.3.2 Flow and River Hydraulics

Hydraulics of the river and the flow around the intake are extremely important in determining the quality of water that enters the system. In general, portions of a stream or river with lower velocities and less turbulence will contain less sediment and possibly less *Cryptosporidium* oocysts. Care should also be taken to make sure that the design of the intake does not cause turbulence which might stir up sediments.

#### 3.3.3.3 Upstream Sources of Contamination

Any potential sources of contamination upstream of a new intake should be identified and their impact considered with respect to both biological and chemical contamination. Contaminant sources of particular concern for *Cryptosporidium* include animal feeding operations and sewage outfalls. If an intake cannot be located upstream of such a source, then locating it as far as possible downstream to allow time for particles to settle may be the next best alternative. Analyses of the vulnerability of a stream source should be made on a regular basis. Any changes in the vulnerability of a source to *Cryptosporidium* or other contaminants should be reported to the primacy agency.

#### 3.3.3.4 Seasonal Effects

*Cryptosporidium* concentrations tend to be higher during runoff events, particularly in the spring. Although it is probably not feasible to cease withdrawals during such incidents, it may be possible to alter flow rates and coagulant doses to offset the effect of such events.

### 3.4 Changing Timing of Withdrawals

The LT2ESWTR allows the option of changing the timing of withdrawals to obtain a lower source water concentration of *Cryptosporidium* for bin assignment (40 CFR 141.716(b)(1)). If the system calculates its bin assignment based on this alternative timing, then after the compliance deadline, it must continue to draw its source water in the same manner (40 CFR 141.716(b)(4)). The operating conditions under which the samples were collected for the LT2ESWTR must be reported and submitted to the state with the monitoring results (40 CFR 141.716(b)(3)).

#### 3.4.1 Toolbox Selection Considerations

As stated above, the change in timing must be consistent during *Cryptosporidium* monitoring and during routine operation after monitoring. Additionally, the LT2ESWTR does not allow source water monitoring to deviate from a predetermined schedule by more than 2

days, unless extreme conditions or situations arise that prevent sampling (40 CFR 141.702(b) and (c)). Given these limitations, the following provides examples of approaches that are recommended and others that are not recommended.

*Recommended Approaches*

Changing the timing of withdrawal on a daily basis (e.g., from the afternoon to morning to avoid suspended material stirred up by recreational water use).

Use a water quality indicator to avoid short-term increases in *Cryptosporidium* due to short-term weather or source water contamination events. For example, if a system routinely experiences a spike in turbidity and subsequently, *Cryptosporidium*, for a 12-24 hour period following a storm event, then the system may choose to set up a monitoring plan that delays withdrawal for a 24 hour period when detecting a spike in turbidity.

*Approaches Not Recommended*

Limiting withdrawal in response to seasonal effects or weather effects lasting on the order of days. This would be a difficult monitoring strategy to follow and stay in compliance with the 2 day sampling window.

#### 3.4.1.1 Advantages and Disadvantages

The advantage of changing the timing of withdrawals is it requires no treatment changes, only a change in operations. For systems with multiple sources it also allows the greatest flexibility in meeting water quality goals.

A disadvantage of relying on changing withdrawals to lower *Cryptosporidium* concentrations is that it may result in decreased flexibility, since systems must follow the same withdrawal practices they did during *Cryptosporidium* source water monitoring. If electing to practice a withdrawal approach that defers withdrawal during likely *Cryptosporidium* events, then a system may need some raw water storage capacity.

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# 4. Bank Filtration

## 4.1 Introduction

Bank filtration is a surface water pretreatment process that uses the bed or bank of a river (or lake) and the adjacent aquifer as a natural filter. The natural filter performs most efficiently when the surface water passes slowly through unconsolidated granular material. In such locations and under normal ground water flow conditions, bank filtration is suitable for accomplishing sufficient *Cryptosporidium* removal to partially meet the requirements of the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). To accomplish this, a pumping well located in the adjacent aquifer induces surface water infiltration through the bed and bank.

Bank filtration differs significantly from artificial recharge and from aquifer storage and recovery, both of which rely on engineering works to move water into specially constructed and maintained recharge basins or wells for infiltration into or replenishment of the aquifer. Although microorganism removal can occur in such engineered systems, they are not bank filtration. This is because bank filtration relies solely on the natural properties of the surface water bed and aquifer, unmodified by engineered works or activity, except for the recovery of ground water via a pumping well. Sites with artificial recharge and aquifer storage and recovery operations may also receive bank filtration *Cryptosporidium* removal credit after a suitable site-specific study but are not eligible for automatic credit. Slow sand filtration also relies on engineered materials as the filter medium and so is not bank filtration.

A significant proportion of microorganisms and other contaminants are removed by contact with the aquifer material as the water travels to the well through the subsurface. Flow to the well may be horizontal or vertical, but more typically will take a variable path with both horizontal and vertical components. The water which has been induced to infiltrate through the river's bed and bank is known as "bank filtrate." It will be mixed with ambient ground water that has taken a different and typically longer path to the well. The ambient ground water may have originated as bed or bank infiltration from an upstream portion of the river or from a lake. It may have originated from infiltrating precipitation. Regardless, ambient ground water is likely to contain different contaminants and contaminant concentrations than bank filtrate because its origin and flow pathways differ significantly. Ambient ground water should not be assumed to be uncontaminated.

Aquifers suitable for bank filtration are composed of unconsolidated, granular material (i.e., grains) and have open, interconnected pores that allow ground water to flow. Pathogen removal is enhanced when fine-grained sediment is present along the flow path. Geologic units consisting primarily of fine-grained (e.g., clay-sized) materials will have higher removal but will be incapable of yielding economically significant water flow rates. In aquifers containing both sand-sized and finer grains, the presence of fine grains increases the possibility that pathogens will encounter a grain surface. This is because flow is slower and flow paths are longer than they would be in aquifers without such fine grains. Microorganisms will be removed from flow as they contact and attach to grain surfaces. Although microorganism (e.g., *Cryptosporidium*) detachment can occur, it usually does so at slow rates (Harter et al. 2000). When little or no

detachment occurs or when detachment is slow, microorganisms can become non-viable while attached to grain surfaces. Thus, bank filtration provides physical removal, and in some cases, inactivation, to remove pathogens from water supplies.

The purposes of this chapter are: 1) to clarify the requirements of the LT2ESWTR related to receiving *Cryptosporidium* removal credit for the use of bank filtration systems; 2) to present the current state-of-the-science, advantages and disadvantages of *Cryptosporidum* removal by bank filtration; 3) to explain how local geologic and hydrologic conditions affect the functioning and effectiveness of bank filtration systems; 4) to provide suggestions for optimal operation of bank filtration systems; and 5) to discuss necessary and sufficient elements of a field and laboratory investigation as part of a demonstration of performance (DOP) at a bank filtration site (with or without engineered systems) to qualify for additional *Cryptosporidium* removal credits.

This chapter is organized as follows:

4.2 LT2ESWTR Compliance Requirements - describes requirements for receiving automatic *Cryptosporidium* removal credits related to the proposed installation of bank filtration wells.

4.3 Toolbox Selection Considerations - describes the advantages and disadvantages of using bank filtration as a pretreatment technology.

4.4 Site Selection and Aquifer Requirements - characterizes surface water and aquifer types that are suitable for bank filtration.

4.5 Design and Construction - describes the types of wells eligible for bank filtration credits and the locations at which such wells are best placed.

4.6 Operational Considerations - describes issues relevant to the optimal operation of bank filtration systems in order to protect public health.

4.7 Demonstration of Performance – describes the recommendations for receiving additional *Cryptosporidium* removal credits after a site-specific field and laboratory investigation.

## 4.2 LT2ESWTR Compliance Requirements

Systems that propose to install bank filtration wells to meet any additional treatment requirements imposed by the LT2ESWTR may be eligible for 0.5 or 1.0 log *Cryptosporidium* removal credit (40 CFR 141.717(c)). Systems meeting all regulatory requirements (e.g., systems with conventional or direct filtration that meet the well siting and monitoring requirements of LT2ESWTR) may receive *Cryptosporidium* log removal credit. For those systems which already use bank filtration as a component of their treatment process and which also have existing conventional or direct filtration treatment, the LT2ESWTR requires source water monitoring of

produced water from the bank filtration well. This will determine the initial bin classification for these systems. Because their source water monitoring accounts for any bank filtration treatment, these systems are not eligible for subsequent additional bank filtration credits (40 CFR 141.703(d)(1)).

Systems using ground water under the direct influence of surface water (GWUDI) or bank filtered water without additional filtration must take source water samples in the surface water to determine bin classification (40 CFR 141.703(d)). This applies to systems using an alternative filtration demonstration to meet the *Cryptosporidium* removal requirements of the Interim Enhanced Surface Water Treatment Rule (IESWTR) or Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) (40 CFR 141.173(b) and 141.552(a)). The requirements and guidance provided in this chapter do not apply to existing primacy agency actions providing alternative filtration *Cryptosporidium* removal credit for IESWTR or LT1ESWTR compliance.

Alternatively, PWSs may apply to the state for *Cryptosporidium* treatment credit using a DOP (see Chapter 4.7). States may award greater than 1.0-log *Cryptosporidium* treatment credit for bank filtration based on a site-specific demonstration. States may also award DOP *Cryptosporidium* treatment credit based on a site-specific study to systems that are unable to qualify for the 0.5 or 1.0-log removal credit as described in Chapter 4.2.1. For a bank filtration DOP study, the following criteria must be met:

- The study must follow a state-approved protocol and must involve the collection of data on the removal of *Cryptosporidium* or a surrogate for *Cryptosporidium* and related hydrogeologic and water quality parameters during the full range of operating conditions.

- The study must include sampling both from the production well(s) and from monitoring wells that are screened and located along the shortest flow path between the surface water source(s) and the production well(s).

### 4.2.1 Credits

The LT2ESWTR specifies the following design requirements for systems to receive log removal credit for bank filtration (40 CFR 141.717(c)):

- Wells must draw from granular aquifers that are comprised of clay, silt, sand, or pebbles or larger particles. Minor cement may be present.

- Only horizontal and vertical wells are eligible for bank filtration log removal credit.

- Other ground water collection devices such as infiltration galleries and spring boxes are ineligible.

- Systems using horizontal or vertical wells located at least 25 feet from the surface water source are eligible for a 0.5 log removal credit and those located at least 50 feet from the surface water source are eligible for a 1.0 log removal credit.

  - Systems with vertical wells must identify the distance to surface water using the floodway boundary or 100 year flood elevation boundary as delineated on Federal Emergency Management Agency (FEMA) Flood Insurance Rate maps.

  - Systems with horizontal wells must measure the distance from the normal flow stream bed to the closest horizontal well lateral.

- Systems must characterize the aquifer at the proposed production well site to determine aquifer properties.

  - At a minimum, the aquifer characterization should include the collection of relatively undisturbed continuous core samples from the surface to a depth at least equal to the projected bottom of the well screen for the proposed production well.

  - The recovered core length must be at least 90 percent of the total depth to the projected bottom of the well screen and each sampled interval should be a composite of no more than 2 feet in length.

  - Each composite sample must be examined to determine if at least 10 percent of the grains in that interval are less than 1.0 millimeter (mm) in diameter.

### 4.2.2 Monitoring Requirements

The LT2ESWTR requires systems to monitor turbidity in bank filtration wells to provide assurance that the assigned log removal credit is appropriate. The LT2ESWTR specifically requires the following monitoring (40 CFR 141.717(c)(5)):

- Turbidity measurements must be performed on representative water samples from each wellhead every four hours that the bank filtration system is in operation or more frequently if required by the state.

- Continuous turbidity monitoring at each wellhead may be used.

- If the monthly average of daily maximum turbidity values at any well exceeds 1 nephelometric turbidity units (NTU), the system must report this finding to the state within 30 days. In addition, within 30 days of the exceedance the system must conduct an assessment to determine the cause of the high turbidity levels and submit that assessment to the state for a determination of whether any previously allowed credit is still appropriate.

## 4.3 Toolbox Selection Considerations

Bank filtration is best suited to systems that are located adjacent to rivers with consistent surface water quality and that plan to use bank filtration as one component of their treatment process. For systems that can meet the aquifer requirements (section 4.4) and the design criteria (section 4.5), bank filtration can be an efficient, cost-effective pretreatment option to improve water quality (Berger 2002). Medema et al. (2000), Medema and Stuyfzand 2002, and Wang et al (2000, 2002) documented high removal of *Cryptosporidium* surrogate organisms at production well sites in The Netherlands and in Louisville, Kentucky. There was very little occurrence of *Cryptosporidium* in river water at the Kentucky site and no *Cryptosporidium* was found in the well water at either site. The amount of *Cryptosporidium* removal at either site is unknown.

The efficient removal of indicator organisms at the Netherlands site was likely due to the relatively impermeable, fine-grained layer of river sediment present, as well as the effect of pyrite oxidizing to ironhydroxides for oxidized ground water. Ironhydroxides (at a pH below 7.0) may enhance the attachment of microorganisms to riverbed sediments (Medema et al. 2000; Medema and Stuyfzand 2002). In Louisville, Kentucky, an alluvial aquifer was chosen for the bank filtration site. Wang et al (2000, 2002) found that removal of biological particles increased with filtration distance of the riverbank filtration process, although most of the removal occurred at the surface of the riverbed, within the first two feet of filtration. Wang et al (2002) attributed the removal in their bank filtration system to a combination of mechanical filtering and biological activity (e.g., biofiltering) at the surface of the riverbed.

As will be discussed in section 4.4, only certain sites are suitable for bank filtration. It is important to understand the type of bed and aquifer material present, the dynamics of groundwater flow, and the potential for scouring of riverbed materials at a potential bank filtration site. The degree to which the bed and banks of surface water bodies may effectively filter *Cryptosporidium* may vary not only from site to site, but also at a single site over time. A site-specific DOP study requires not only a good understanding of past ground water flow and *Cryptosporidium* surrogate removal efficiency but also ongoing monitoring to identify and to take preventive action during poor removal periods.

### 4.3.1 Advantages and Disadvantages

#### 4.3.1.1 Removal of Additional Contaminants

The two research sites with published data (Medema et al. 2000, Medema and Stuyfzand 2002, Wang et al. 2000, Wang et al. 2002, Berger 2002) have shown that bank filtration is effective at removing *Cryptosporidium*. Bank filtration has also been shown at some sites to be an effective technology for attenuating a variety of additional microorganisms as well as particulates, ammonia, nitrate, pesticides (e.g., atrazine), heavy metals, ethylenediamine tetra-acetic acid (EDTA), alkylated and chlorinated benzenes and other organic contaminants, and disinfection byproducts (DBPs) precursors in the form of Natural Organic Matter (NOM) (Schijven et al. 2003, Tufenkji et al. 2002, Ray et al. 2002, Kuehn and Mueller 2000). Bank

filtration achieves the removal of these diverse contaminants by facilitating or enhancing physical and chemical filtering, sorption, reduction/oxidation, precipitation, ion exchange, and biodegradation (Schijven et al. 2003, Ray et al. 2002, Tufenkji et al. 2002). Bank filtration further reduces contaminant concentrations and especially shock contaminant loads from spills and intentional acts by providing for the multidimensional dispersion and dilution of contaminants (Ray et al. 2002).

The degree to which any particular contaminant will be removed via bank filtration depends on site-specific conditions. For example, under aerobic conditions, ammonia is often completely transformed to nitrate and nitrite, whereas such removal may not occur under more reducing conditions. Oxygen is usually significantly depleted within 5-15 feet of the riverbed, due to microbial activity in this zone. As infiltrating water becomes increasingly depleted of organic matter due to degradation, microbial activity diminishes, and the aquifer may be reaerated at a certain distance from the riverbed (Tufenkji et al. 2002). The anaerobic part of the aquifer was observed to remove up to 99 percent of polar organic contaminants at a site in central Germany (Juttner 1995). Miettinen et al (1994) found that almost 90 percent of the high molecular weight fraction of NOM had been removed at a bank filtration site in Finland.

Bank filtration can reduce treatment costs by reducing the need for more expensive treatment technologies. Particle and microorganism removal during bank filtration allows for more efficient filtration, use of membranes, and disinfection during subsequent treatment steps. For example, decreasing the concentration of dissolved organic carbon during bank filtration can reduce the amount of dissolved organic carbon that needs to be removed in a downstream treatment process such as activated carbon filtration. The removal of ammonia means that the additional treatment step of oxidizing ammonia with chlorine may be unnecessary. The removal of nitrate when water is induced to flow through anaerobic areas may eliminate the need for expensive ion exchange or reverse osmosis treatment processes (Kuehn and Mueller 2000). Finally, because it is effective at biodegrading many contaminants, including trace organic contaminants, bank filtration reduces the need for adding large quantities of flocculants to drinking water (Kuehn and Mueller 2000).

Another advantage of bank filtration as a pretreatment technology is that it acts to equalize fluctuations in contaminant concentrations observed in surface waters. This is due to the effects of dilution and dispersion which serve to spread peaks in contaminant concentrations over space and time by the time they reach wells. Contaminant concentration peaks may be due to variations in river water levels, seasonal effects, and runoff, in addition to spills, terrorist acts and emissions by municipal and industrial institutions (Kuehn and Mueller 2000). Bank filtration is continuously active, and the decreased amplitude of the contaminant peak by the time it reaches a well (an inherent result of subsurface transport through porous material) allows for easier and less expensive treatment by utilities with limited capabilities. In addition, the time lag between contamination of surface water and arrival of contaminant at a well would give utilities more of an opportunity to respond to a threat or an accidental spill. Kuehn and Mueller (2000) estimate that in many modern bank filtration systems bank filtrate spends anywhere from 5 to 15 days in the subsurface before reaching supply wells. At one site in the Netherlands, bank filtrate was estimated to spend 45-65 days in the subsurface before reaching the supply well (Medema, et al.

2000). Residence time depends on site-specific hydrogeology as well as bank filtration system design. Bank filtration also smoothes out fluctuations in water temperature.

The removal of NOM during bank filtration is useful because NOM occurrence can result in the production of harmful DBPs, as discussed above. In addition, moderate to high concentrations of NOM in drinking water can result in unpleasant taste and odor. Finally, NOM removal via bank filtration can also aid in the removal of a large variety of additional organic and inorganic contaminants. These contaminants are sometimes made more mobile in surface and ground waters due to a partitioning process whereby they are attached to NOM, which is relatively mobile, and thereby carried along a flow path. The removal of NOM and associated contaminants prior to above-ground treatment is likely to lessen the overall cost of water treatment at a given facility.

#### 4.3.1.2 Clogging of Pores

Clogging of the surface water - ground water interface by physical, chemical, and biological processes has the potential to be a problem with any riverbank filtration system. Partial clogging during riverbank filtration system operation is likely to be unavoidable (Wang et al. 2001, Goldschneider et al. 2007); however, its effects are not always deleterious. The disadvantage of clogging is that it can reduce hydraulic conductivity of the local riverbed and the aquifer, thereby temporarily or permanently reducing well yields. On the other hand, a limited accumulation of fine-grained sediments and the accompanying development of a biologically active zone can enhance pathogen removal. Indeed, this enhanced removal is a basic principle behind riverbank filtration as a water treatment technology. An optimal amount of clogging is beneficial because it can reduce the size of large pores or reduce entrances to pores in a stream bed or aquifer. Pore size reduction and decreased hydraulic conductivity also result in longer travel times which can result in additional pathogen inactivation. Transport of fewer pathogens is also likely because there are more opportunities for pathogen contact with aquifer grain surfaces.

Physical clogging of the surface water - ground water interface results from the deposition of fine-grained, suspended sediment at the interface and in the near surface pores. The deposition and growth of microorganisms also contribute to physical clogging. This clogging may be exacerbated during periods of low surface water discharge, and is most apparent near the river's edge where flow velocities are generally lower than at the center of the river. Chemical clogging can result from precipitation of dissolved surface water constituents and may occur near the interface or anywhere along the flowpath. This is due to the change in geochemical conditions as infiltrating water enters the riverbed and aquifer. Factors to be considered when evaluating the potential for chemical clogging include electrolyte concentration, pH, redox potential, presence of dissolved or colloidal organic matter, and the mineralogy and surface characteristics of stream bed and aquifer solids.

Lastly, biological or microbial clogging can result from the accumulation of bacterial cells in pore spaces, the production of extra-cellular polymers, the release of gaseous byproducts from denitrifying bacteria and methanogens, and the microbially mediated accumulation of

insoluble precipitates (Vandevivere et al. 1995, Baveye et al. 1998). Biogenic gas bubbles have the effect of blocking or partially blocking water flow through pores in much the same way that solid particles do (Orlob and Radhakrishna 1958, Oberdorfer and Peterson 1985, Sanchez de Lozada et al. 1994). Insoluble sulfide salts can cause clogging due to the activity of sulfate reducing bacteria, whereas iron hydroxide and manganese oxide deposition can be brought on by bacterial iron metabolism (Vandevivere et al. 1995, Baveye et al. 1998). Biological clogging is most likely to occur near the surface water - ground water interface where nutrients are most available.

Some or all of these processes may act at a particular site to lower hydraulic conductivity and thus decrease flow velocities. For example, several months of pumping from a new riverbank filtration well in Louisville, Kentucky resulted in a significant decline in well production, presumably due to a 70 percent reduction in leakance from the river to the adjacent aquifer. The reduced well yields were attributed to the physical clogging of riverbed sediments (Schafer, 2000). The disadvantage of reduced well yields accompanies the advantages of increased microbial inactivation rates due to lower flow velocities (and thus longer residence times in the aquifer) as well as increased removal of pathogens due to smaller pores.

#### 4.3.1.3 Scour

Both the positive and negative effects of clogging on riverbank filtration system performance may be diminished following periodic flooding. Scour refers to the erosion of the river's bed and banks. The extent of erosion depends on both flood conditions and the resistance of the bed and bank material that has been deposited at a particular site. During flooding, the river channel may be scoured and fine sediments at the surface water - ground water interface mobilized.

Much of the removal of the contaminants and microbes discussed previously occurs during the first few centimeters of the flow path due to the significant filtering and sorptive capabilities of sediments in the riverbed. These sediments are often organic-rich, highly biologically active, and fine-grained. The effectiveness of bank filtration, however, may be temporarily threatened during high flows if this active layer is washed away or scoured. EPA suggests that the potential for stream channel scour be evaluated during riverbank filtration site selection (section 4.4). Section 4.5 provides further discussion of scour and its implications for riverbank filtration system operation.

#### 4.3.1.4 Additional Treatment Steps

In addition to clogging and scour, there are several disadvantages to bank filtration which utilities may wish to consider and balance against the advantages and cost savings described in section 4.3.1. One disadvantage is that an additional aeration step may be required due to the possible depletion of oxygen from biological activity during riverbank filtration pretreatment (Kuehn et al. 2000). This oxygen depletion may lead to extremely anaerobic conditions over a

portion of the flow path, which may result in the release of iron and manganese from the bank sediment into the flowing water. This process occurs due to a redox reaction which reduces iron and manganese to their water-soluble forms. This condition may necessitate the removal of these metals during subsequent treatment steps (Kuehn et al. 2000, Tufenkji et al. 2002).

On the other hand, if the flow path between the riverbank and the well is long enough, iron and manganese may precipitate onto the sediments in the subsurface before ever reaching the well (Tufenkji et al. 2002). The aquifer becomes reaerated with increasing distance from the riverbed. This is one reason for locating riverbank filtration wells greater than 25 or 50 feet from the river, as discussed in section 4.5.2.2. Even though most contaminant removal occurs during the first few centimeters of subsurface transport, the reaeration and associated precipitation reactions in the aquifer may significantly improve water quality before it reaches the well (Tufenkji et al. 2002). The location of the aerated and anaerobic portions of the aquifer vary seasonally due to variable recharge, precipitation infiltration, microbial activity and changing pumping rates.

Lastly, riverbank filtration is ineffective at removing a few persistent compounds, primarily non-polar organic compounds and highly soluble chemical contaminants such as methyltertiarybutylether (MTBE) and trichloroethylene (TCE). These would need to be addressed during subsequent treatment steps. In addition, when bank filtration is used to induce infiltration of highly contaminated surface water, it may be important to include additional adsorption steps during later treatment (Kuehn et al. 2000).

## 4.4 Site Selection and Aquifer Requirements

Unconsolidated, granular aquifers with sufficient amounts of fine-grained material (see section 4.4.2) are eligible for *Cryptosporidium* removal credits under the LT2ESWTR. Partially consolidated, granular aquifers may also be eligible for removal credits. Each granular aquifer proposed as a bank filtration site should be evaluated on a case-by-case basis with regard to its grain size distribution and degree of cementation. For example, a partially consolidated, granular aquifer may be too cemented, and thus perhaps too fractured, to provide adequate pathogen removal. Geophysical methods, discussed in section 4.5.2.2, may be helpful in determining the degree of fracturing of such aquifers.

This section characterizes river and aquifer types that may be suitable for bank filtration surface water treatment. A list of selected sites in the United States and Europe which have used bank filtration is provided for reference. No information is available for these sites, however, regarding whether they would meet the siting criteria in the LT2ESWTR. Some common aquifer types that are clearly not appropriate for this technology are described as well. Finally, site-specific aquifer criteria which shall be met in order for systems to receive *Cryptosporidium* removal credits are outlined in section 4.4.3.

### 4.4.1 Selected Bank Filtration Sites

## Exhibit 4.1 Selected Bank Filtration Systems in Europe and the United States

| Site Location | Well Type* | Number of Wells | Maximum Capacity *mgd ($m^3/s$)* | River System |
|---|---|---|---|---|
| Europe | | | | |
| Torgau, Germany | V | 42 | 39.7 (1.737) | Elbe |
| Mockritz, Germany | V | 74 | 28.8 (1.260) | Elbe |
| United States | | | | |
| Cincinnati, OH | V | 10 | 40.0 (1.750) | Great Miami |
| Columbus, OH | H | 4 | 40.0 (1.750) | Scioto/Big Walnut |
| Louisville, KY | H | 1+ | 20.0 (0.875) | Ohio |
| Terra Haute, IN | H | 1 | 12.0 (0.525) | Wabash |
| Jacksonville, IL | H | 1 | 8.0 (0.350) | Illinois |
| Galesburg, IL | H | 1 | 10.0 (0.438) | Mississippi |
| Henry, IL | V | 1 | 0.7 (0.030) | Illinois |
| Mt. Carmel, IL | V | 1 | 1.0 (0.044) | Wabash |
| Quincy, IL | H | 1+ | 10.0 (0.438) | Mississippi |
| Sacramento, CA | H | 1 | 10.0 (0.438) | Sacramento |
| Sonoma County, CA | H, V | 5 (H) + 7 (V) | 85.0 (3.727) | Russian |
| Independence, MO | H † | 1 | 15.0 (0.656) | Missouri |
| Lincoln, NB | H, V | 2 (H) + 44 (V) | 35.0 (H) (1.530) | Platte |
| Kennewick, WA | H | 1 | 3.0 (0.130) | Columbia |
| Kalama, WA | H | 1 | 2.6 (0.110) | Kalama |
| St. Helens, OR | H | 3 | 5.0 (0.219) | Columbia |
| Kansas City, KS | H | 1 | 40.0 (1.750) | Missouri |
| Sioux Falls, OK | H | 1+ | 40.0 (1.750) | Missouri |

* H–horizontal, V–vertical.
† Gravel-packed Laterals.


### 4.4.2 Aquifer Type

#### 4.4.2.1 Unconsolidated, Granular Aquifers

Unconsolidated, granular aquifers can be composed of a wide range of sediment sizes including clay, silt, sand, and larger particles. They may also exhibit minor cementation, but subsurface samples are typically friable (readily crumbled by hand). To be eligible for bank filtration credits under the LT2ESWTR, unconsolidated granular aquifers are expected to contain a sufficient amount of fine-grained sediments to achieve adequate pathogen removal and/or inactivation (section 4.4.3 prescribes the amount deemed sufficient). In aquifers with these characteristics, the flow path is tortuous at the micro-scale, providing many opportunities for removal of microorganisms by straining or by their attachment to grain surfaces.

Many alluvial aquifers contain significant amounts of well-sorted, fine-grained sediments. Alluvial aquifers are produced by fluvial depositional processes and are adjacent to modern streams. Aquifers formed in glacial deposits may also contain sufficient amounts of fine-grained material. These may be "till" deposits, which have a wide range of poorly sorted sediment sizes,

or glacial outwash deposits that are formed by meltwater and often contain well-sorted, sand-sized sediments. Any of these alluvial or till aquifers would likely be suitable for a bank filtration system. On the other hand, coarse gravel aquifers produced by the rapid drainage of glacial lakes, or in outwash environments that deposit little fine-grained material, may not be eligible for bank filtration credits unless sieve analysis shows sufficient fine-grained material as discussed in section 4.4.3.2.

Alluvial aquifers may be identified on detailed hydrogeologic maps simply as "Quaternary alluvium," indicating both their genesis and relative age. Glacial deposits are documented on surficial geology maps and, where aquifer-forming, may be identified on large-scale hydrogeologic maps.

#### 4.4.2.2 Karst, Consolidated Clastic, and Fractured Bedrock Aquifers

In karst, consolidated clastic, and fractured bedrock aquifers, ground water velocities are fast, and flow paths may be direct, allowing microbial contaminants to travel rapidly to a well with little removal or inactivation. Because they do not meet 40 CFR 141.717(c)(2), these aquifer types are not eligible for bank filtration treatment credits.

Karst may be broadly defined as a region where the dissolution of calcitic or other soluble bedrock, primarily limestone (calcium carbonate), produces a unique subsurface drainage network and associated surface landforms. Ground water movement in karst aquifers differs from that in porous, granular aquifers in that flow in the former occurs predominantly in conduits and dissolution-enlarged fractures. Consequently, there is little physical removal of microbes and other particles by filtration and few opportunities for microbes to come in contact with the surfaces of aquifer materials. Furthermore, rapid flow creates conditions where inactivation is less likely to occur before ground water reaches a well.

Although fractures have a role in ground water movement through any aquifer, fractures provide the dominant flow-path in fractured consolidated clastic and fractured bedrock aquifers. Most consolidated aquifers can be presumed to be fractured. Similar to solution conduits in karst aquifers, fractures in consolidated aquifers provide preferential flow paths that may transmit ground water at high velocities, and in a relatively direct flow path to a well, with little time or opportunity for inactivation or removal of microbial pathogens (e.g., Gaut et al. 2008). Wells located in these aquifers would not meet 40 CFR 141.717(c)(2), and therefore would not be eligible for bank filtration credit.

#### 4.4.2.3 Partially Consolidated, Granular Aquifers

Granular aquifers formed by marine processes earlier than Quaternary alluvial or glacial deposition may be partially consolidated by natural cement that fills pores, connects grains, and makes the aquifer material less friable. Partially consolidated, granular aquifers are present within the Atlantic Coastal Plain, Gulf Coast Lowland, Texas Coastal Upland, and Mississippi

Embayment aquifer systems (USGS 1998). When significant proportions of cement are present, fractures are more likely to exist. As in consolidated aquifers, fractures in partially consolidated, granular aquifers create direct paths for microbial contamination that minimize the natural filtration capabilities of the aquifer system. EPA suggests that partially consolidated aquifers be evaluated at the proposed well location to determine if they may be too cemented, and thus perhaps too fractured, to provide sufficient natural filtration.

The degree of cementation can be evaluated by a variety of methods. Geologic material collected from below the aquifer's weathered zone that is friable upon touch is likely to be adequate for bank filtration purposes. Another test for the degree of cementation includes the slaking test, which involves alternate wetting and drying of the sample in water, or in salt or alcohol solutions. A triaxial compression test can also be used to measure strain in three mutually perpendicular directions. Less cemented samples will be more deformable during such tests.

### 4.4.3 Aquifer Characterization

Systems seeking *Cryptosporidium* removal credit are required to characterize the aquifer properties between their surface water source and their well (40 CFR 141.717(c)(2)). The aquifer characterization will include, at a minimum, core sampling to determine grain size distribution. This data will establish whether enough fine-grained sediment is present to provide adequate filtration. The following procedure outlines the steps necessary under the LT2ESWTR and steps recommended by EPA to perform such a characterization, which will ultimately determine eligibility for bank filtration treatment credits under the LT2ESWTR.

The necessary steps are:

- Systems must **characterize the aquifer** at the proposed production well site to determine aquifer properties.
  - The recovered core length must be at least 90 percent of the total.
  - Each composite sample must be examined to determine if at least 10 percent of the grains in that interval are less than 1.0 mm in diameter.

EPA also recommends:

- The aquifer characterization should include the collection of relatively undisturbed continuous core samples from the surface to a depth at least equal to the projected bottom of the well screen for the proposed production well.
- Each sampled interval should be a composite of no more than 2 feet in length. If core recovery is insufficient, another well core should be obtained.

- Each 2 foot long composite sample of recovered core should be examined in a laboratory using sieve analysis to determine grain size distribution. Core intervals are typically 2 feet long for a conventional split-spoon sampler and 3 or 4 feet long for soil probes (e.g., a Giddings-type soil probe).

#### 4.4.3.1 Coring

The collection of relatively undisturbed cores in unconsolidated aquifers can be quite difficult, especially when gravel-sized clasts are present. The two most important criteria for successful test drilling to obtain a core are sample accuracy and drilling speed. Borehole stability is a major problem in drilling in an unconsolidated gravelly formation. Rotary core drilling is particularly suited to drilling in unconsolidated formations because the drilling fluid, which cools the drill bit and carries up the core, also acts to stabilize the borehole (Driscoll 1986).

Other drilling methods require the installation of a casing to stabilize the borehole, a process which slows down the speed of drilling. Rotary core drilling is the fastest method for drilling in an unconsolidated formation. One disadvantage to rotary core drilling is the separation of different sized core particles as they rise (smaller particles rise faster) and cross-contamination by overlying borehole material. An experienced driller can avoid cross contamination by using the dual-wall method of rotary core drilling. In the dual-wall method, the core is pushed up the inner pipe of the drill rather than traveling in the space between the drill and the borehole wall (Driscoll 1986). Shallow wells will have fewer particle size separation problems than deeper wells. The freeze-core method (Balcsak 1995) can be used to obtain in-situ cores from streambeds.

Auger drilling is another method for drilling test wells. In this method an earth auger is screwed into the earth by rotation. Auger drilling in an unconsolidated formation is slower than rotary core drilling, due to the necessary installation of casing to support the borehole. Sampling with augers can provide reliable samples from any depth. A split spoon sampler can be used wherein a split spoon is driven to the bottom of the hole. The depth to which an auger can drill is dependent on the size of the rig. The maximum drilling depth possible for a small drill rig is approximately 250 ft. (Driscoll 1986).

Information about drilling and finding a driller can be found through the National Groundwater Association (NGWA) website: http://www.ngwa.org/. In addition, the EnviroDirectory™ provides listings for laboratories and drillers in New England, the Mid-Atlantic, and the Great Lakes regions (www.envirodirectory.com).

#### 4.4.3.2 Sieve Analysis

The American Society for Testing and Materials (ASTM) has a published standard for conducting sieve analysis, the Standard Test Method for Sieve Analysis of Fine and Coarse Aggregates: Standard C 136-1 (ASTM 2003).

Sieve analysis is used to determine the particle size distribution of a sample of dry aggregate of known mass by passing the sample through a series of sieves with progressively smaller openings. Sieve analysis requires the following equipment:

- A balance, accurate to 0.1g or 0.1 percent of test load for fine aggregate, or accurate to 0.5g or 0.1 percent of test load for a mixture of fine and coarse aggregate.
- Stackable sieves.
- A mechanical sieve shaker (for sample sizes greater than 20kg).
- An oven capable of maintaining 110 ± 5°C (230 ± 9°F).

In the first step of sieve analysis the sample is dried using the oven. Once dry, its weight is measured and recorded. While the sample dries, sieves are selected with suitable openings to furnish the information required. For bank filtration related sieve analyses, it is only necessary to determine what percentage of the sample is less than 1.0mm; however, it is recommended that sieves covering a range of sizes be used so as to prevent the overloading of any one sieve. Once the sample is dry and the sieves are stacked in order of decreasing mesh size, the sample is placed in the top sieve and sieving either by machine or by hand begins. Sieving should be continued until no more than 1 percent by mass of the material retained on an individual sieve will pass through that sieve during 1 minute of continuous hand sieving. Lastly the mass on each sieve is weighed. The total mass of the material after sieving should correspond closely with the original mass of the sample. Using the mass for each size increment and the total mass of the sample, the size distribution of the sample can be determined (ASTM 2003).

Further information about sieve analysis can be found at the ASTM web site (www.astm.org). A multi-media sieve analysis demonstration can be found at Geotechnical, Rock and Water Resources Library (GROW) http://www.grow.arizona.edu/Grow--GrowResources.php?ResourceId=139.

ASTM also provides a search engine which allows the user to search for laboratories that perform sieve analyses (http://www.astm.org/LABS/search.html). The EnviroDirectory™ provides listings for laboratories and drillers in New England, the Mid-Atlantic, and the Great Lakes regions (http://www.envirodirectory.com).

### 4.4.4 Site Selection as it Relates to Scour

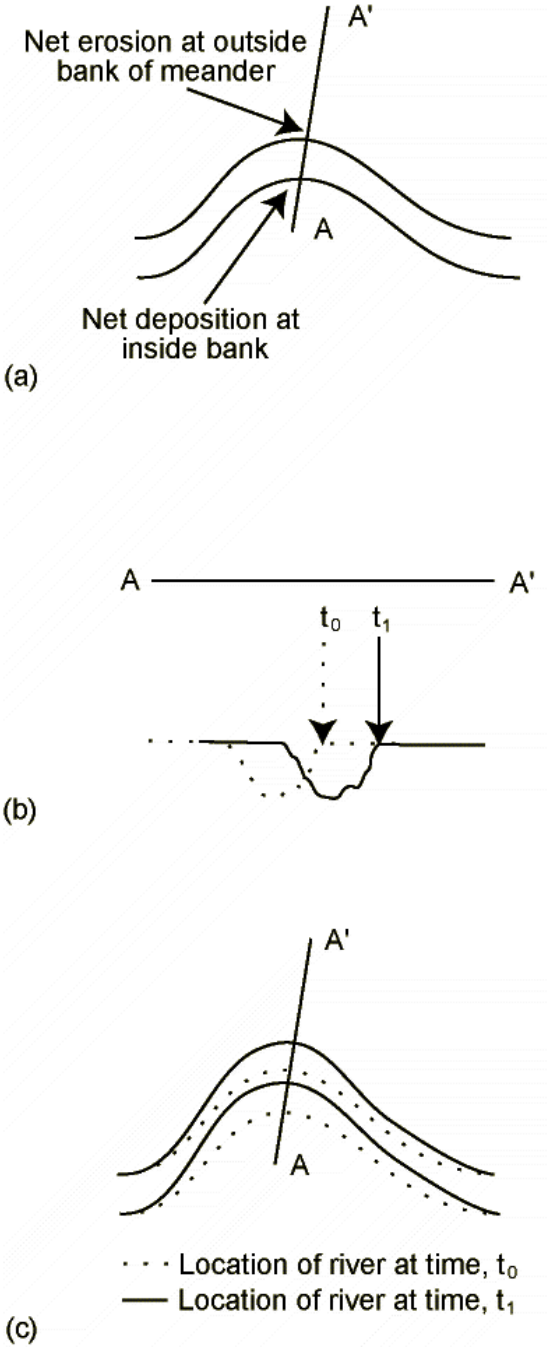
Stream channel scour may often be an important consideration in choosing sites that are suitable for riverbank filtration. This section discusses stream channel erosional processes in general, as well as reasons sites with certain characteristics may be unsuitable for riverbank filtration. Section 4.6 discusses the implications of periodic scour for riverbank filtration system operations. Detailed information on fluvial erosional processes can be obtained from any of a number of texts on fluvial geomorphology, hydrology, and river hydraulics (e.g., Leopold et al. 1964, Ritter et al. 1995, Chow 1964).

#### 4.4.4.1 Stream Channel Erosional Processes

This discussion focuses on the dominant erosional processes of alluvial rivers because, given the LT2ESWTR's aquifer requirements, such rivers may be among the most suitable for bank filtration credits. Although many lake banks are also suitable sites for bank filtration, lakes will not be discussed in detail in this section. Lake bank filtration settings typically do not change rapidly with time and climate. Their hydrologic properties are not highly variable and thus do not require the detailed evaluation discussed here for riverbank filtration settings.

The width, depth, and gradient of an undisturbed alluvial river has typically adjusted to prevailing discharge conditions and sediment loads such that no net erosion or deposition occurs over long time periods (Mackin 1948, Leopold and Maddock 1953). The dominant scouring process in alluvial rivers is lateral migration (Exhibit 4.2). This process is responsible for the stream meanders visible on many floodplains, and is accomplished by the progressive erosion of the outside bank of a river bend with concurrent deposition on the inside bank. Because erosion is generally matched by deposition in this process, channel dimensions do not change significantly over time, and the net result is migration of the channel across the floodplain. Stream channel meanders are characteristic of many alluvial rivers and are indicative of a graded stream.

## Exhibit 4.2 Generalized Depiction of Stream Channel Lateral Migration




(a) Map of a Stream Meander; (b) Cross-section of the Channel from A-A′ with Channel Positions at 2 Successive Times ($t_0$, and $t_1$); (c) Map of Stream Meander Showing Location After Migration.

Downcutting, another type of scour that can occur in fluvial environments, is the vertical erosion of the streambed. Downcutting is fairly uncommon in alluvial rivers except during floods or if the stream is not graded. The long-term dynamic equilibrium of a graded stream can be disrupted by a variety of changing hydrologic and geologic conditions and especially by anthropogenic activity. Human activities in a watershed or river channel may alter the conditions to which an alluvial river has become adjusted, initiating a period of readjustment marked by either progressive downcutting or aggradation (deposition).

Urbanization generally increases the proportion of impervious surface in a watershed, increasing flood volumes during precipitation events because less water is able to infiltrate the land surface and recharge ground water (Jacobsen et al. 2001). Increased flood volumes may cause higher water levels in a river channel, increasing the shear stress on the channel bed and causing scour (Booth 1990). Downcutting may continue until the channel gradient, and/or channel dimensions, become adjusted to the new flooding regime.

Impoundment is another activity that may disrupt the quasi-equilibrium state of a graded river and initiate readjustment of the channel. The sharp decrease in sediment supply, which commonly occurs subsequent to dam and reservoir construction, may initiate downcutting in the reach immediately downstream until the channel adjusts to the lightened sediment load. This has been observed downstream of many dams throughout the world. One of the most dramatic examples is the 7.5 meters of channel-bed degradation that occurred twelve kilometers downstream of the Hoover Dam after its completion in 1935 (Williams and Wolman 1984).

The construction of artificial levees (raised banks along a stream channel) also may result in flooding downstream. Levees allow greater quantities of water to be carried by the stream, thus decreasing the probability of flooding in the vicinity of the levee, but increasing flood hazards downstream (Montgomery 2000). Even if flooding downstream does not result, the high flows downstream may cause downcutting of the river, removal of fine-grained bed material, and thus a threat to the protectiveness of a riverbank filtration system. Another possible effect of levees is an increase in sedimentation in the channel. Sediment that would otherwise be deposited on the floodplain may be trapped within the channel. This can raise the channel bottom and thus raise stream stage or the elevation of the water surface in the channel (Montgomery 2000). The consequences of this for a riverbank filtration system are variable. Increased sedimentation may lead to clogging and/or decreased well yields. On the other hand, higher stream stages may result in flooding and scour along certain portions of the river as the channel adjusts to a new equilibrium condition. Understanding the impact of current or planned upstream activities can be an important part of site selection for a riverbank filtration system.

#### 4.4.4.2 Unsuitable Sites

Some sites may be unsuitable for bank filtration credit due to the type of aquifer adjacent to the river. For example, a stream reach adjacent to a wellfield might be dredged for gravel mining. A system may choose to evaluate such situations on a site-by-site basis, however, except as specified in the LT2ESWTR, EPA does not require such evaluations or any particular

decisions made on the basis of such evaluations. EPA recommends, however, that this information be considered in order to ensure that bank filtration systems are protective of public health.

Lower log removals are expected to occur during and shortly after floods because protective layers may be removed by flood scour. If such situations are expected to occur very frequently, and if a system cannot envision a way of managing the system so as to adequately protect its water supply during such events, sites on such rivers may be inappropriate for riverbank filtration. EPA recommends that the potential for scouring be considered during site selection. If a site that undergoes occasional scour is selected for riverbank filtration, the system may wish to locate its wells at greater than the required separation distance from the surface water body, as discussed in section 4.5.2.2. Such a solution helps to ensure the protection of public health.

The potential for scour can be evaluated first by examining the past frequency of high flow and flood events. Data on flood history and discharge is typically available from the U.S. Geological Survey, the Army Corps of Engineers, the U.S. Bureau of Reclamation and the Department of Homeland Security (formerly FEMA). State and county highway and transportation departments typically evaluate river scour to determine the safety of bridge supports. A more comprehensive evaluation of the potential for scour can be conducted when the effect of past and current human activities (as discussed in section 4.4.4.1) is considered in comparison to the history of flood events.

**Sources of high flow and flood data**

**United States Geological Survey (USGS)**

*Main Page:* http://water.usgs.gov

The National Flood Frequency (NFF) Program:
http://water.usgs.gov/pubs/wri/wri024168/pdf/entirereport.pdf
A computer program developed by the USGS for estimating the magnitude and frequency of floods for ungaged sites. Since 1993, updated equations have been developed by the USGS for various areas of the nation. These new equations have been incorporated into an updated version of the NFF Program.

USGS Fact Sheets (listed by state):
http://water.usgs.gov/wid/index-state.html
Includes NFF program methods for estimating flood magnitude and frequency (in rural and urban areas) for: AL, AZ, AR, CA, CN, HI, LA, MD, MO, NV, NM, NC, OK, SC, SD, TX UT, VT, VA, and WA. These fact sheets describe the application of the updated NFF Program to various waterways within the specific state. Includes maps of each of the above state's hydrologic regions, as well as regression equations and statistics.

*WaterWatch:*
http://water.usgs.gov/cgi-bin/dailyMainW?state=us&map_type=flood&web_type=map
Map of current flood and high flow conditions in the United States. The map shows the location of streamgages where the water level is currently at or above flood stage (⮝) or at high flow (•). The high flow conditions are expressed as percentiles that compare the current (i.e., within the past several hours) flow value to historical daily mean flow values for all days of the year. The real-time data used to produce the maps have not been evaluated or edited.

**Army Corps of Engineers**

*Main Page:* http://www.usace.army.mil/

Flood control and management pages. For example, river and reservoir reports including flood level data are available for the St. Louis district of Missouri (see example below) (http://mvs-wc.mvs.usace.army.mil/dresriv.html).

Mississippi River:

| River Mile | Gage Station | 6am Levels | 24-hr Change | National Weather Service River Forecast | | Flood Level | Gage Zero | Record Level | Record Date |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Next 3 Days | Crest Date | | | | |
| 309.0 | Hannibal | 16.9 | -0.2 | 16.6 16.1 15.6 | | 16.0 | 449.3 | 31.80 | 07/10/93 |
| 301.2 | Dam 22 tw | 15.9 | -0.3 | 15.8 15.3 14.7 | | 16.0 | 446.1 | 29.58 | 07/16/93 |

**U.S. Bureau of Reclamation**
Main Page: http://www.usbr.gov/main/
*Dams and Reservoirs Page:* http://www.usbr.gov/dataweb/html/dam_selection.html
The project DataWeb provides the most current information on the bureau's projects, facilities, and programs including dam and reservoir information for western states. This data can be obtained by selecting a dam or from the state and Region maps.

**The Department of Homeland Security (FEMA)**
*Main Page:* http://www.fema.gov/
*Flood Hazard Mapping:* http://www.fema.gov/fhm/
The flood maps describe where the flood risks are, based on local hydrology, topology, precipitation, flood protection measures such as levees, and other scientific data. Fee to obtain maps.

## 4.5 Design and Construction

This section describes the type of wells eligible for bank filtration credits under the LT2ESWTR. Because specific well construction requirements (e.g., casing depths) vary by state and with geologic conditions, this guidance will address these issues only briefly where appropriate. Readers are referred to the agency within their state that makes regulations or recommendations regarding well construction for details on issues such as casing depths, annular seals, drilling methods, filter packs, etc. Other good general references on well construction include Driscoll (1986) and U.S. EPA (1975).

### 4.5.1 Well Type

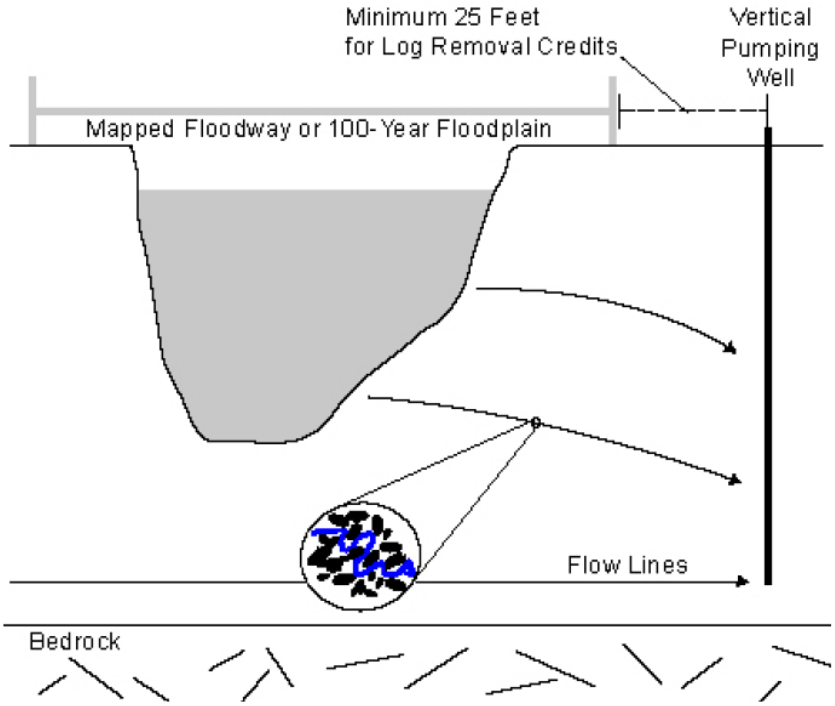
Only vertical and horizontal wells are eligible for bank filtration credits. Other types of ground water collection devices may not provide adequate filtration of pathogens. For example, a spring box is a ground water collection device located at the ground surface and is designed to contain spring outflow and protect it from surface contamination until the water is used. Spring boxes are found where local hydrogeologic conditions have focused ground water discharge into a smaller area (i.e., a spring) and at a faster volumetric flow rate than elsewhere. Often, localized fracturing or dissolution-enhanced channels are the cause of the focused discharge to the spring. As noted in section 4.4.2.2, fractures and dissolution channels have significant potential to transport microbial contaminants. Thus, spring boxes are not eligible for bank filtration credit (40 CFR 141.717(c)(6)).

Infiltration galleries (or filter cribs) are also not eligible for bank filtration credits. Infiltration galleries are designed to collect water infiltrating from the surface, or to intercept ground water flowing naturally toward surface water, using a slotted pipe installed horizontally in a trench and backfilled with granular material (Symons et al. 2000). An infiltration gallery is not

bank filtration even if the material overlying an infiltration gallery is engineered to optimize oocyst removal. Bank filtration systems are defined as relying solely on the natural properties of the system to remove microbial contaminants. At least one infiltration gallery is associated with an outbreak of cryptosporidiosis in the United States (also British Columbia, Canada and Japan). The Medford and Talent, Oregon outbreak resulted from treated (filtered) water taken from an infiltration gallery intake buried under Bear Creek, Talent, Oregon (Leland et al. 1993). An infiltration gallery may, however, be eligible for *Cryptosporidium* removal credit through a demonstration of performance under 40 CFR 141.718(c).

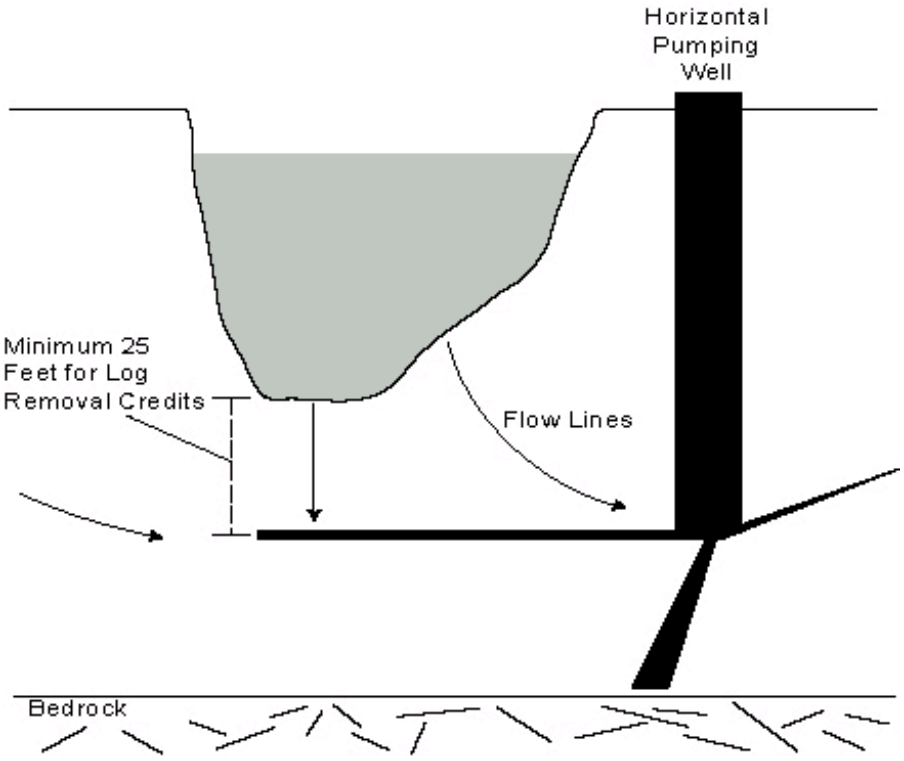
Horizontal and vertical wells are both eligible for bank filtration credits. They are distinguished from each other by the orientations of their well screens, and the important implications this has for their well hydraulics (Exhibit 4.3 and 4.4). Collector horizontal wells are constructed by the excavation of a central vertical caisson or pipe. One or more laterals (i.e., collector lateral well screens) extend horizontally from the caisson bottom and may be very long. Laterals may extend radially in all directions - resulting in a radial collector well- or primarily in the direction of the river (Driscoll, 1986; Ray, 2001a). The lateral well screens are often installed near the bottom of the formation, allowing a greater proportion of the saturated thickness of the aquifer to be used. A greater proportion of pathogens and other contaminants are removed when the distance between the surface water body and the laterals is increased (Ray 2001a). Section 4.5.2.2 contains a discussion of when it may be appropriate to locate wells at separation distances greater than those required by the LT2ESWTR. Laterals may extend underneath a surface water body in the United States. This is generally not how horizontal wells are placed in Europe (Ray 2001a) because in Europe such wells are required to meet a 55-60 day average travel time requirement. An example of a pump house for a horizontal collector well in Louisville, Kentucky, is shown in Exhibit 4.5. It is elevated to prevent flood waters from entering it.

## Exhibit 4.3 Schematic Showing Generalized Flow and Required Separation Distance to a Vertical Well




Note that the exhibit shows tortuous ground water flow at the micro-scale.

## Exhibit 4.4 Schematic Showing Generalized Flow and Required Separation Distance to a Horizontal Well With Three Laterals



## Exhibit 4.5 Taking a Water Level Reading

The pump house for the horizontal collector well caisson is in the background.

The choice between using a vertical or horizontal well for a bank filtration system depends on the site hydrogeology and the pumping requirements. For systems with large production requirements or for pumping in shallow alluvial aquifers, horizontal wells may be preferred because they are designed to capture large volumes of surface water recharge with little drawdown (Driscoll 1986). Vertical wells with large production requirements are not well suited to shallow alluvial aquifers because the necessary low drawdown cannot be sustained (Ray 2001a).

Finally, a comparison of construction expense with the costs of well maintenance may play a role in the choice of well type. Horizontal collector wells are substantially more costly than vertical wells (Driscoll 1986). However, moderately large utilities may need many smaller capacity vertical wells to match the capacity of a horizontal well. The maintenance of these vertical wells may require significant effort and expense (Ray 2001a). In such cases, horizontal collector wells may be preferred.

### 4.5.2 Filtrate Flow Path and Well Location

For systems to receive *Cryptosporidium* log removal credits, the ground water flow path length between the edge of the surface water body and the well should be sufficient for effective oocyst removal. This section discusses EPA's requirements for appropriate flow path lengths and associated well locations for the log removal credits available under the LT2ESWTR. The ground water flow path length necessary to receive credits is specified for both vertical and horizontal wells. A discussion of how to obtain information necessary to define the edge of the surface water body is also included.

#### 4.5.2.1 Required Separation Distance Between a Well and the Surface Water Source

*Cryptosporidium* oocyst removal may vary significantly throughout the year in many bank filtration systems. At most typical bank filtration locations, high log removal rates (e.g., 3.5 log removal over 13 m) may be expected with the surface water discharges that predominate during most of the year. During short flood periods, however, there may be substantially lower removal (e.g., 0.5 to 1.0 log removal over 13 m) due to scouring of the surface water–ground water interface. A number of factors may contribute to increased risk of *Cryptosporidium* reaching wells. These factors include the presence of coarse-grained aquifer or stream bed sediments, high river velocities, and frequent scouring of riverbeds. Given the need to protect water supplies during periods of high surface water discharge with their potentially lower log removal capabilities, the LT2ESWTR rule language (40 CFR 141.717(c)) provides 0.5 log removal credit for systems with bank filtration wells located greater than 25 feet from a surface water source and 1.0 log removal credit for wells located greater than 50 feet from a surface water source.

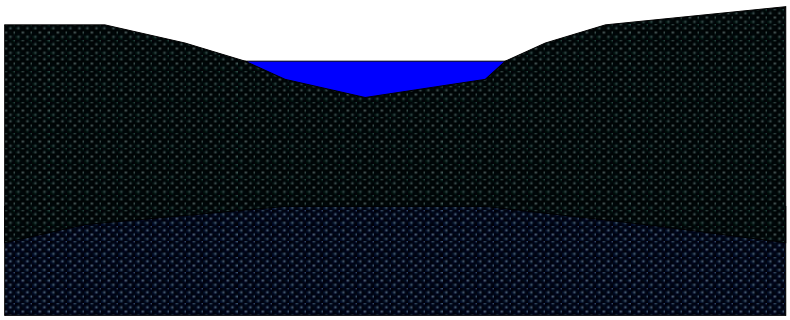
#### 4.5.2.2 Locating Wells at Greater than Required Distances from the Surface Water Source

Given the dynamic nature of riverbanks and aquifer systems, including scouring processes it may sometimes be advisable to place bank filtration wells at distances greater than 25 or 50 feet from a surface water source. This extra precaution may also be advisable when a system is uncertain as to whether the riverbed and bank contain sufficient fine-grained material to provide adequate removal of *Cryptosporidium* oocysts. That is, EPA is requiring the separation distances of 25 feet and 50 feet for the log removal credits discussed above, but greater separation distances may result in additional public health protection at some sites. As discussed in Section 4.3.1.4, longer flow paths may result in changes in the water oxygen content that may be advantageous for iron and manganese removal. The disadvantage of using greater separation distances between the surface water source and the bank filtration well is that water yields to the well will be decreased. When a system makes a decision to place wells at a greater distance from a surface water source than EPA requires, it will need to balance the sacrifice in well yield with the added public health protection.

The remainder of this section discusses geophysical methods which may be of used to construct a conceptual model of subsurface flow conditions in riverbank filtration systems. By obtaining hydrogeologic information through geophysical or other means (e.g., pre-existing hydrogeologic or geologic maps), systems can determine the degree to which local conditions may affect *Cryptosporidium* removal at the bank filtration site. For example, if mapping the bedrock-alluvial interface and the water table at a particular site indicates that the aquifer is fairly thin, it is unlikely that infiltrating river water will be diluted by much ambient ground water. In such a case it may be advisable to locate wells at greater than the required distance from the surface water source. On the other hand, if detailed hydrogeologic investigations indicate that the aquifer contains a large proportion of fine-grained sediments, it would not be advisable to locate the well at greater than the required distance from the surface water source, because the aquifer is already likely to be an efficient pathogen filter, and it would be inadvisable to further sacrifice well yields.

When the aquifer contains fine-grained material, it is possible that well over-pumping may break the hydraulic connection between ground water and surface water, yielding a variably saturated zone underneath a perched stream, as shown in Exhibit 4.6. Formation of such a variably saturated zone during periods of high pumping can greatly alter the existing ground water flow paths. New ground water flow paths could result in marked changes in water quality. For example, surface water infiltration could occur further upstream, resulting in a longer ground water flow path for infiltrating surface water flowing towards the well. The increase in flow path-length could improve water quality. Alternatively, over-pumping can decrease water quality. This may occur because the decreased thickness of the saturated aquifer - due to the formation of a large variably saturated zone - may cause faster ground water flow (assuming pumping rates remain constant). Faster ground water flow provides less time for contaminant attenuation within the aquifer. Finally, the variably saturated zone itself, to the extent that it transmits water, can improve water quality because contaminant attenuation is usually increased under variably saturated conditions. If possible, the potential for formation of a variably saturated zone should be investigated to provide additional information regarding the desirability of locating wells at greater than required distances from the surface water source.

**Exhibit 4.6 The Streambed of a Perched Stream Is Well above the Water Table**



Geophysical methods generally do not disturb subsurface materials. They are often less expensive than labor-intensive digging of trial pits or drilling of boreholes. Furthermore, the useful information gleaned by using geophysical methods can aid in choosing the best locations for wells (Reynolds 1997). Geophysical methods include gravity and magnetic methods, seismic methods, electrical methods, and ground penetrating radar (GPR).

Hydrogeophysical methods can be used in pre-existing boreholes, thereby providing high resolution data for a very localized area around the borehole. Alternatively, surface geophysical methods can be used to obtain more generalized information over a large area, including information on the depth to the water table, the depth to bedrock, and stratigraphy (Hubbard, 2003). The discussion below provides only a generalized overview of currently available geophysical methods. More detailed information can be obtained from texts by Hearst (2000), Reynolds (1997), Rubin and Hubbard (2003), Keys (1990) and Burger (1992).

Gravity surveying measures variations in the acceleration due to the Earth's gravitational field which are caused by density variations in subsurface rocks. Subsurface cavities can be detected with this technology; however, sites with such cavities would not be suitable for bank filtration. Reynolds (1997) states that gravity methods are fairly uncommon in hydrogeological work compared to electrical methods. On the other hand, in the Arizona district of the USGS, gravity methods have been in use for over 15 years to evaluate changes in water storage in aquifers. These methods can detect water table elevation changes of as little as a few inches (Callegary 2003). Thus, gravity methods may be useful at riverbank filtration sites for assessing the depth to water table, aquifer thickness, and seasonal effects on the dilution of infiltrating river water with ambient groundwater. Magnetic surveying or magnetic anomolies can also be used in hydrogeologic investigations. For example, clay infilling bedrock cavities can be detected due to slight changes in the magnetic susceptibility of clay and most bedrock (Reynolds 1997).

Seismic methods are widely used in hydrogeologic investigations. Applied seismology involves generating a signal through an explosion or other method at a specific time. The generated seismic waves travel through the subsurface, are reflected and refracted back to the surface, and the return signals are detected on monitoring instruments. The amount of time that elapses is the basis for determining the nature of subsurface layers/materials (Reynolds 1997). Reynolds (1997) provides a detailed example of the use of seismic refraction surveying for locating the bedrock/alluvial interface at one particular site.

Seismic methods can be used to:

- Estimate depth to bedrock (ideal for riverbank filtration applications).
- Determine the nature of bedrock (e.g., cavernous) or location of cavities. Note that karst buried by alluvium may contain unexpected ground water flowpaths.

- Determine the location of faults that may juxtapose bedrock against alluvial material.
- Determine stratigraphy (useful where sands and clays may be interlayered).
- Estimate porosity prior to coring.
- Determine ground water particle velocities (an important parameter for riverbank filtration systems).

Electrical resistivity methods are used extensively in downhole logging to identify hydrogeologic units that will produce high flow rates. Electrokinetic surveying makes use of electrodes implanted at the ground surface to identify the location of the water table. This may be useful at riverbank filtration sites, where water table layer and depth to bedrock can be used to determine aquifer thickness - an important parameter in determining how much dilution of bank filtrate with ambient groundwater is occurring. A more recent development is the use of electrokinetic methods to measure flowrates in boreholes (Reynolds 1997).

The spontaneous polarisation or self-potential (SP) method is conducted by measuring differences in ground electrical potential at different locations, but is still fairly uncommon. Another electrical method, the induced polarisation (IP) method can be used to detect ground water and water tables, however electromagnetic (EM) induction methods are generally considered more practical for these purposes in the field. Contaminated ground water within subsurface clays can also sometimes be detected with the IP method (Reynolds 1997).

EM methods have been used in groundwater investigations to delineate contaminant plumes, and thus can be useful in conceptualizing flow systems in a riverbank filtration context if there is a significant contrast between river water and ambient ground water. Pulse-transient EM (TEM) surveys (a type of EM method) may be useful in conceptualizing flow for riverbank filtration. It may also be useful in monitoring the quality of infiltrating water. When data is available from both borehole and surface instruments, EM and electrical methods can be used to map subsurface geology such as the locations of coarse-grained and fine-grained units.

GPR has been used as a surface method for contaminant plume mapping and monitoring pollutants in groundwater. To operate such a system, a signal generator, transmitting and receiving antennae, and a receiver should be used. Generated radiowaves travel in a broad beam at high speeds. Energy is lost or attenuated depending on the subsurface materials through which the waves travel. GPR has proven valuable in mapping sediment sequences, and can be used to investigate sediments through freshwater up to 27 m deep (Reynolds 1997). Thus, it may be of use in gaining information about the composition of riverbeds, and for monitoring the effects of scour on riverbed composition. GPR can also be used to locate water tables, delineate sedimentary structures which may contain pockets of coarse-grained alluvium, and determine the spatial extent and continuity of buried clay and peat layers within subsurface deposits. Borehole radar can also be used for hydrogeologic investigations.

Before choosing a specific geophysical method the following should be considered: desired level of resolution, area of coverage, site-specific conditions and their influence on the applicability of the method, possible non-uniqueness of the geophysical attribute, resources needed to interpret the geophysical data, and possible integration with direct measurements. In general, mapping the water table and finding the depth to bedrock are considered standard hydrogeophysical procedures. Other applications such as estimating permeabilities or porosities are at an earlier stage of development and may not yet be appropriate for routine use at riverbank filtration sites (Hubbard 2003).

#### 4.5.2.3 Delineating the Edge of the Surface Water Source

The flow paths due to induced infiltration to a vertical well have both vertical and horizontal components, and are tortuous at the micro-scale. Such flow will typically have a significant horizontal component, especially if the vertical well is screened in a shallow, unconsolidated, alluvial aquifer that is eligible for bank filtration credits. Therefore, for the purpose of receiving log removal credits, the flow path length to a vertical well is to be determined using the measured horizontal distance from the edge of the surface water body to the well screen. The edge of the surface water body is defined as the edge of either the 100-year floodplain or the floodway, discussed below. The 100-year floodplain is defined by its boundary - the flood elevation that has a one percent chance of being equaled or exceeded each year.

As a first step, utilities may use the online maps available at the following website to get a general idea of the mapped extent of the 100-year floodplain in their area: http://www.esri.com/hazards/makemap.html. In order to satisfy the requirements of the LT2ESWTR for the location of the wells of a bank filtration system, however, an official FEMA (now part of the Department of Homeland Security) flood hazard map must be used. Such maps can be ordered in either paper or digital formats from FEMA. The following web site can be used to order these maps: http://www.fema.gov/hazard/map/flood.shtm.

For many areas, the mapped extent of the floodway will also be drawn on the flood hazard map obtained by FEMA. The utility may choose to use the edge of the floodway rather than the edge of the 100-year floodplain for the purpose of determining the required separation distance between a river and a riverbank filtration well. If the mapped extent of the floodway is unavailable, the utility may opt to perform the mapping using one of a number of hydraulic models approved by FEMA. A list of these approved models is available at http://www.fema.gov/plan/prevent/fhm/en_hyala.shtm. EPA recommends using the US Army Corps of Engineers' Hydrologic Engineering Centers River Analysis System (HEC-RAS) model for mapping floodway limits. The HEC-RAS software is available for free downloading from http://www.hec.usace.army.mil/software/hec-ras/hecras-download.html. The user's manual, applications guide, and hydraulic reference manual are available at http://www.hec.usace.army.mil/publications/pub_download.html.

When a utility elects to determine the edge of the floodway and to model the floodway boundaries if they are not available from FEMA, the preferred encroachment method within

HEC-RAS is Method 4. Method 4 can be summarized as follows, according to FEMA's Map Assistance Center (2003):

> The Method 4 encroachment operates by analyzing the hydraulic conveyance for the unencroached one percent annual chance floodplain at each cross section, then equally reducing the conveyance from both overbank areas by moving toward the stream channel from the edge of the floodplain until the resulting water-surface elevation is one foot higher than the unencroached elevation, and the resulting encroached conveyance is approximately equal to the unencroached conveyance. The new left and right cross-section limits are assumed to be vertical walls. Finally, a backwater energy balance is calculated using the new cross sections, which results in the encroached or floodway water-surface profile. The floodway modeling process requires adjustments and rerunning of the model because the final calculation is the backwater energy balance between new cross sections. Many times the 1.0-foot target cannot be achieved exactly at each cross section because of energy balance considerations. Floodplain geometry, constrictions at culvert and bridge crossings, and constrictions from other man-made obstructions in the floodplain may require adjustments to the encroachment widths to stay at or below the 1.0-foot maximum water-surface increase. Chapter 10 of the HEC-RAS User's Manual includes a discussion of performing a floodway encroachment analysis.

In most areas, however, EPA expects that utilities will find it preferable and simpler to use the previously mapped limits of the 100-year floodplain to determine the edge of the river for riverbank filtration separation distances.

Although in some areas of the United States the mapped extent of the 100-year floodplain may be more easily accessible than the mapped extent of the floodway, some utilities may choose to use the edge of the floodway as a starting point for measuring separations distances to wells. This is because it typically allows wells to be placed slightly closer to the river and is thus a somewhat less restrictive requirement. The floodway is a regulatory concept, and is defined as that portion of the overbanks that must be kept free from encroachment to discharge the one percent annual chance

#### 4.5.2.4 Measuring Separation Distances for Horizontal Wells and Wells that are Neither Horizontal Nor Vertical

As noted in section 4.5.1, horizontal wells may have laterals that extend underneath a surface water body. The flow direction for induced infiltration to a horizontal well that extends under a surface water body is predominately downward. Therefore, the flow path length to a horizontal well is the measured vertical distance from the bed of the river under normal flow conditions to the closest horizontal well lateral's intake.

Some wells may be constructed so that the well is neither truly horizontal nor truly vertical. In these cases, there is greater uncertainty about the definition of separation distance

from surface water. For simplicity, if the well inclination is closer to being a vertical well than to being a horizontal well (i.e., the well is oriented at greater than a 45 degree angle to a horizontal line), the separation distance is defined for the purposes of this toolbox option to be the horizontal distance from the edge of the river to the closest (in terms of horizontal distance) intake on the well. Similarly, if the well is closer to being a horizontal well as opposed to a vertical well, separation distance is defined as the shortest possible vertical distance from the riverbed to an intake on the well.

## 4.6 Operational Considerations

### 4.6.1 High River Stage

When the river stage (i.e., the elevation of the water surface) is high, the increased head gradient between the river and the adjacent aquifer results in increased infiltration and increased ground water flow rates. This condition can be expected to occur periodically throughout the year at many sites, and will generally be associated with reduced log removals (Gollnitz 1999, Ray 2001b, Rohns et al. 2006). High river stage is often associated with scouring of riverbed sediments. Nevertheless, even when scour does not occur, the high ground water velocities associated with high river stage can be a significant threat to a riverbank filtration system.

One solution to this problem is that pumping rates can be temporarily decreased during periods of high river flow (Medema et al. 2000). Decreased pumping rates will in turn decrease the head gradient between the river and the well, thereby decreasing subsurface velocities, increasing residence times, and facilitating pathogen inactivation.

### 4.6.2 Implications of Scour for Bank Filtration System Operations

Periodic, short-term flood scour can have both negative and positive impacts on the performance of a bank filtration system. As noted previously lower log removals of oocysts are expected during floods because higher river shear velocities and associated increases in bedload transport mobilize fine sediments deposited when discharges were lower.

Removal of fine sediments opens large pore spaces, increasing the hydraulic conductivity across the surface water–ground water interface (Gollnitz 1999, Ray 2001a, Ray 2001b). Unfortunately, this potentially increases the number of pathogens transported. Furthermore, the microbial activity and unique geochemical environment of the riverbed, which serves to facilitate the removal of pathogens via sorption and other processes, may not be present for short periods following flood scour. Recent work in Germany (Baveye et al. 2003) suggests that the biologically active zone is re-established very quickly after scour, perhaps within 3 days, at least when measured in terms of the ability to degrade certain organic compounds. Limited scour can reduce clogging at the surface water–ground water interface and improve well yields (Wang et al. 2001).

When high river stages or high turbidity levels indicate that flood scour may be occurring and compromising the effectiveness of a bank filtration system, pumping rates can be decreased. This will lead to lower velocities and longer subsurface residence times, thereby increasing the protectiveness of the system (Medema et al. 2000, Juhasz-Holterman 2000).

### 4.6.3 Anticipating High Flow Events / Flooding

Many factors can affect the probability of flood events. Intense rainfall is the most apparent factor, however the geomorphology of a watershed is important in determining how quickly water will enter a stream system after a rainfall event, as well as how quickly water will enter a major river from smaller tributaries. Systems can anticipate that a high flow event will occur if a rapid spring thaw follows a winter of unusually heavy snowfall. It is also important to be aware of recent changes in vegetation due to wildfires or urbanization. When vegetation is removed or decreased, there are fewer barriers to rapid surface runoff, plant roots no longer keep soil loose and permeable (thus more compact soils will be less able to decrease surface runoff), and plants themselves will be unavailable to take in a certain proportion of precipitation (Montgomery 2000). Therefore, systems may wish to monitor for pathogens more frequently or change pumping regimes in riverbank filtration systems when high flows are anticipated.

### 4.6.4 Possible Responses to Spill Events and Poor Surface Water Quality

One response to a serious water quality threat is to stop pumping from all bank filtration production wells. Other pumping regime changes can also be implemented to reduce risks, including decreasing the number of hours the system is in operation each day. For systems that have a number of wells in operation, it may be advisable to increase pumping rates for wells further from the surface water source and decrease pumping rates for wells that are closer (Juhasz-Holterman 2000). Juhasz-Holterman (2000) recommended that this kind of change be implemented seasonally at a site in the Netherlands. Her study of the site's hydrology indicated that during the winter months, wells were more vulnerable to contamination due to "short-circuited" flow paths from the polluted river through the subsurface. Her solution involved both restricting extraction rates to a few hours a day (which was acceptable due to decreased demand during the winter months) as well as an altered pumping regime which relied more on wells located further from the river. In general, systems that receive water from multiple pumping wells should manage their well field so as to maximize the water residence time in the subsurface, to the extent possible, while meeting changing water quantity demand. Methods to evaluate subsurface residence time are discussed in Section 4.7, Demonstration of Performance.

### 4.6.5 Maintaining Required Separation Distances

Alluvial rivers that are experiencing active, progressive erosion as an adjustment to new flooding regimes or sediment loads, or in relation to natural lateral migration, may pose serious, longer-term challenges to bank filtration systems. For example, significant log removal

reductions may be more frequent in an urbanizing basin as a consequence of more frequent flooding and associated scouring. In extreme cases, long term degradation of the bed or banks may reduce the threshold separation distances between the surface water source and bank filtration well. Recall that these separation distances - 25 feet for 0.5 log removal credit and 50 feet for 1.0 log removal credit - are required to receive log removal credits under the LT2ESWTR.

Systems may wish to assess their sites for active, progressive erosion. Lateral migration rates can be calculated using sequential aerial photography and/or topographic maps, if available. Systems without such data may need to obtain the needed information by conducting sequential field surveys of the floodplain area proposed for the site. This will require a far more lengthy investigation period. Progressive downcutting could also be measured with sequential field surveys of the channel bed elevation over a period of years. Regardless of the method used, the threshold separation distances between the surface water source and the bank filtration well must be maintained.

## 4.7 Demonstration of Performance

PWSs using GWUDI as their source water may receive 0.5 or 1.0-log *Cryptosporidium* removal credit based on well siting and aquifer critera as described in section 4.2. Alternatively, PWSs may apply to the state for *Cryptosporidium* treatment credit using a DOP. States may award greater than 1.0-log *Cryptosporidium* treatment credit for bank filtration based on a site-specific demonstration. States may also award DOP *Cryptosporidium* treatment credit based on a site-specific study to systems that are unable to qualify for the 0.5 or 1.0-log removal credit as described in section 4.2.1.

PWSs using existing bank filtration as pretreatment to a filtration plant are not eligible to receive additional treatment credit for bank filtration. In these cases, the performance of the bank filtration process in reducing *Cryptosporidium* levels will be reflected in the monitoring results and bin classification under the LT2ESWTR.

For a bank filtration DOP study, the following criteria must be met:

- The study must follow a state-approved protocol and must involve the collection of data on the removal of *Cryptosporidium* or a surrogate for *Cryptosporidium* and related hydrogeologic and water quality parameters during the full range of operating conditions.

- The study must include sampling both from the production well(s) and from monitoring wells that are screened and located along the shortest flow path between the surface water source(s) and the production well(s).

The purpose of this section is to provide additional guidance on the design and conduct of a DOP study, as well as guidance on the interpretation of the study data and the award of

*Cryptosporidium* removal credits if warranted. Finally, this section describes the necessity for long term performance evaluation monitoring to determine if the log removal credit continues to be appropriate.

### 4.7.1 Identification of Collection Devices and Alternative Treatment Technologies at the Site

Prior to initiating a DOP study, it is necessary to 1) identify all of the treatment technologies and collection devices in use and 2) design a study tailored for each treatment technology and collection device as a separate unit process. Evaluating each unit process and device separately will enhance knowledge about *Cryptosporidium* removal at the site.

A PWS may operate one or more vertical wells, horizontal collector wells (caisson wells), infiltration galleries or spring boxes. Wellfield management operations may include surface water diversion into recharge basins or injection of treated or untreated surface water into the subsurface. Bank filtration relies upon the nature of the undisturbed (by humans) subsurface materials to provide natural filtration, and is relatively unmodified by application of engineered structures, flow control or engineering operations such as surface water basin recharge. When other technologies, such as artificial recharge, or differing collection devices, such as infiltration galleries are used, the treatment processes may vary spatially or temporally.

The physico-chemical removal processes utilized by the various well field management technologies and collection devices will differ. Depending upon the site characteristics and recharge operations, an artificial recharge technology may be influenced by intermittent recharge, pre-treatment, unsaturated flow and transport, biofiltration, chemical precipitation by oxidation and reduction, or other phenomena. The removal processes may operate with significantly different removal rates among the differing technologies or collection devices. For example, 1) oocyst attachment to the gas-water interface may be a significant removal process in an intermittent artificial recharge technology, but will be absent or insignificant in a bank filtration system where unsaturated conditions never occur; or 2) a shallow infiltration gallery might operate under aerobic conditions only, as compared with a deeper vertical well that might operate at times under anaerobic conditions. As discussed in section 4.3.1.4, oxidation and reduction reactions may govern *Cryptosporidum* or especially surrogate transport through the subsurface. Thus, each treatment unit process and collection device should be evaluated separately in the DOP study.

Wells, both horizontal and vertical, that are significantly influenced by surface water spreading operations may be considered to be artificial recharge alternative technology if a large component of their yield results from artificial recharge rather than from bank filtration. As part of the study design, a putative assignment of alternative treatment technology type should be made for each collection device. Some wells may have high uncertainty about which alternative treatment technology (bank filtration or artificial recharge) is appropriate. For these wells, the study design should include elements suitable to both technologies. For example, spiking studies may be appropriate for artificial recharge but not for bank filtration studies. Such spiking studies

should be planned for those wells for which there is high uncertainty about the appropriate alternative treatment technology.

### 4.7.2 Source Water Quality and Quantity

Water quality, flow, and flood data should be compiled to characterize possible worst case scenarios. Flooding can, for example, dilute pathogenic contamination in unsewered areas with minimal agricultural activity or can increase contamination where combined sewer overflows are present. As discussed earlier in this chapter, flood scour can have a deleterious effect by removing the protective fine-grained material in the riverbed.

The DOP study should identify and investigate the temporal variability of upstream point and non-point source dischargers and pathogen concentrations. The study should compile historical data and collect new data to determine the quality of the in-stream flow adjacent to the well field and elsewhere upstream, focusing primarily on cyst and oocyst concentrations and their most appropriate indicators in surface waters under a variety of surface water flow conditions.

Surface water samples should be composite samples representative of the water quality in differing stream tubes both laterally and vertically within the river (because river water tends to remain unmixed, river flow is idealized as flow in a composite of parallel tubes). River water samples should be proportionately representative of the actual flow conditions based on the historic record and should be collected during both low water and high water stages (if safety conditions permit) as well as under normal conditions.

### 4.7.3 Ground Water Travel and Residence Time Calculations and Ambient Ground Water Dilution

The study should determine the time lag due to travel between the surface water source and the wells or other collection devices. To accurately calculate pathogen removal, the approximate lag times are necessary because variable pathogen concentration in surface water will affect the removal calculations. Approximate lag times can be determined by collecting suitable site-specific parameter data such as surface water and ground water temperature, chloride and/or bromide concentrations, and then refined using the most appropriate site-specific parameters.

Environmental tracer data (isotopes, Chlorofluorocarbons (CFCs), pharmaceutical compounds, etc.) can be collected to verify lag times calculated using temperature, chloride and other parameters. Samples from collection devices will typically be mixtures of induced stream water and ambient ground water. The ambient ground water may have subsurface residence times of months, years or decades and therefore, may have very low concentrations of pathogens and indicators. The collection device sample concentrations should be corrected for ambient ground water dilution before determining removal by artificial recharge or bank filtration. Ground water flow models, water quality data and environmental tracers can be used to determine the amount

of ambient ground water dilution for each well or cluster of wells. Because ambient ground water dilution varies depending on the well pumping rate and/or schedule, the dilution should be calculated using several well pumping scenarios.

The DOP should recognize that it is important to document the ground water flow paths contributing to a collection device to improve confidence in the assigned pathogen removal credit for each device. For example, a well providing water which is typically 80 percent ambient groundwater and 20 percent naturally-filtered surface water may appear to have adequate pathogen removal when, in fact, dilution by uncontaminated groundwater is the major factor resulting in low pathogen concentrations in the produced water. If ambient ground water dilution changes significantly (e.g., 20 percent ambient ground water and 80 percent naturally-filtered surface water), this well could demonstrate markedly different pathogen removal efficiencies. For public health protection, it is essential to ensure that the well will provide 99 percent *Cryptosporidium* removal from the surface water through the alternative filtration technologies, without dilution by ambient ground water.

Ground water flow models with particle tracking capability can be used to calculate travel times, ground water residence times, lag periods and ambient ground water dilution (e.g., Abdel-Fattah et al. 2007). Ground water flow model calculations can be improved if site-specific data are collected on the hydraulic conductivity of the streambed using seepage meters or other technologies and by analysis of well core or cutting samples from the aquifer. Most recently, new models have been developed that can explicitly simulate ground water flow to a horizontal collector well (e.g., Bakker et al. 2005). If ground water flow models are used, then the study design should include the elements appropriate to standard ground water model quality assurance, including calibration, history matching, verification and sensitivity analysis. Although the dimensionality of the modeling is a site-specific decision, the groundwater flow models should have the capability to simulate mutually-interfering pumping wells with non-uniform surface recharge over the domain.

The concept of long and short flow paths to a well is illustrated by Gollnitz et al. (2005) for a site in Casper, WY, and was determined using a particle tracking ground water flow model. However, at the Casper site, the shortest flow path to a collection device is from the surface to the infiltration gallery, which is located at a depth of 15-20 feet below the bottom of the recharge basins (Gollnitz and Clancy 1994). Comparison of biological particle data among two 30 foot deep vertical wells, a 30 foot deep horizontal collector well and the infiltration gallery shows poorest removal in all samples collected from the infiltration gallery (Gollnitz et al. 2005, Table 2), suggesting that there is a correlation between flow path-length and removal efficiency.

### 4.7.4 Surface and Ground Water Data Collection, Methods and Sampling Locations

As required by the LT2ESWTR, the DOP study must include the collection of hydrogeologic and water quality parameter data during the full range of operating conditions. Thus, the paired surface water and ground water samples should be collected, at a minimum, monthly for eighteen months to capture both high flow and low flow events over a long time

period. Sampling for eighteen months insures that at least one wet season or one dry season is sampled twice. Unusual conditions associated with a wet or dry season are less likely to re-occur during the second sampling period.

EPA recognizes that not all wells in a well field are equally at risk from *Cryptosporidium*. Higher risk wells are those that receive surface water that has the shortest ground water residence time (or flow path-length). Sampling should be more intensive at higher risk wells (e.g., Gollnitz et al. 2005).

Data can be collected on an ongoing basis or it can be event based, with the intention of capturing the worst case possibilities due to floods. Safe sample collection procedures however, should be observed at all times. Ongoing data collection takes place over a longer period of time, but tends to provide an average characterization. Event based sample collection is a high frequency of monitoring over short periods of time to characterize the worst case scenario. The monitoring strategy should try to maximize the possibility of capturing infrequent events without sacrificing long-term characterization of average conditions. An appropriate data collection strategy could include periodic data that is collected on an ongoing basis as well as data collected during and after flood events (with consideration of appropriate lag time for ground water samples), with the intention of capturing the worst case possibilities due to floods. Thus, all data should be collected periodically during normal flows but at a higher monitoring frequency over short-term, high water periods to characterize the potentially worst case scenario. Late summer or drought low flow conditions should also be more intensively sampled if the low water levels represent a possible worst case scenario.

The study design should ensure that the number and location of river water samples collected are representative of high and low consumptive use (e.g., pumping for drinking water supply, and irrigation) periods. River water samples should be representative of the entire river volume, rather than consisting only of samples collected at the surface water intake for the treatment plant. If point sources discharge upstream and the stream is not well mixed, then the river samples should be proportionate in number and location to the volume of the highly concentrated plumes emanating from the point sources.

As discussed above, the study should be designed to determine proportions of each of the unit processes operating at the site (ambient ground water, artificial recharge water and bank filtrate water) and contributing to a collection device. Typically, identification of ground water sources is accomplished by compilation of historical ground water geochemical data and measurement of major and trace elements, isotopes and other environmental tracers, interpreted with the assistance of geochemical and ground water flow models. Suitable parameters measured could include, but are not limited to, organic carbon, chloride, bromide, total dissolved solids (TDS), hydrogen, oxygen, uranium, and other isotopes, and CFCs.

Data collection activities should be designed to ensure that the collected samples are representative and random. Data analyses should include quantitative assessment of the uncertainty associated with each conclusion. Study design should include sufficient sample

numbers so as to determine statistical significance for each conclusion to a pre-determined confidence level.

The study design should also include a quality assurance project plan, identifying 1) reference to the analytical method and laboratory, 2) a reasonable number and percent of blank, replicate and spiked samples, 3) detection limits, and 4) sample holding times.

The presence of multiple data collection wells can serve to increase confidence in the conclusions. Monitoring well data (preferably from multiple wells) collected along the flow path should show a decrease in indicator concentration with distance from surface water to improve overall confidence that the measured log removal results are meaningful.

The DOP should determine the capture zone of each collection device and/or conduct dye trace studies from local sources such as septic tank leach fields to ensure that indicator organisms are coming from the source water rather than from land-side septic tanks. Well-water counts of *E. coli* for example, that includes *E. coli* that originates at a nearby septic tank, rather than at the river, will yield lesser calculated removal efficiencies than the actual removal.

### 4.7.5 Monitoring Tools

The DOP study should consist of monitoring for *Cryptosporidium* or a suite of *Cryptosporidium* surrogate organisms at each collection device (or device type cluster) and the source river water. Pathogen monitoring could also include *Giardia* and perhaps members of the Microsporidia family (Brusseau et al. 2005). In the absence of *Cryptosporidum* oocyst removal data (calculated using measurable oocyst concentrations in the river ***and*** in the collection device), the DOP should use *Cryptosporidium* surrogate microorganisms and should demonstrate that removal of the recovered surrogate organism(s) would be similar to the removal of *Cryptosporidium* oocysts (to the extent possible using the scientific literature, laboratory and/or field studies).

Bank filtration efficiency can be meaningfully demonstrated and is permitted in a DOP only in porous media (similar in concept to slow sand filtration but without use of engineered materials, flow and flux control and active *schmutzdecke* management). Log removal calculations require counts per volume of the same organism in both surface water and nearby collection devices (wells). Comparison of the two values provides information on attenuation during subsurface passage. However it is important to calculate log removal only for microorganisms that are similar to *Cryptosporidium* oocysts. Log removal calculations for particles or organisms that significantly differ from oocysts in size, shape and porous media transport capability or have unknown original or final size and shape (and charge), such as turbidity, standard particle counts, and total algae, or larger organisms such as rotifers, crustaceans or fish are less meaningful and should not be used. Pumping wells generate turbidity in the aquifer as a result of pumping (van Beek et al 2010). Therefore, for groundwater, turbidity data are useful primarily for determining disinfectant treatment efficiency. It is not meaningful to count particles not known to originate in the surface water, as is the case for turbidity or standard particle counters.

*Cryptosporidium* and *Cryptosporidium* surrogate organism transport in porous media has been well studied in both laboratory and field experiments (e.g., Schijven et al. 2003) and will not be detailed here. In general, microorganism and porous media grain size and shape are important parameters that govern removal efficiency together with grain coatings and water chemistry. Predictions without field measurements are highly uncertain. Thus, paired samples from surface water and ground water are necessary.

When subsurface materials are coarse grained (e.g., gravel), ground water flow is relatively fast and bank filtration efficiency is significantly reduced. For example, in one study in a gravel aquifer (not a bank filtration site), aerobic spores traveled 90 m in about one day (Pang et al. 1998, Pang et al. 2005). For coarse grained aquifers, EPA recommends significant additional study (increased monitoring frequency of multiple microorganisms from multiple monitoring wells) to improve removal efficiency measurement at high ground water velocity.

Surrogate microorganisms are more likely to be recovered from collection devices when the concentration in the surface water is high, such as during a diatomaceous algal bloom (e.g., Kearney, Nebraska) (Berger et al. 2002) or during high water stage. Eckert and Irmschser (2006) report *E. coli* recovery in Dusseldorf bank filtration wells only following a flood event.

The choice of the appropriate suite of *Cryptosporidium* surrogate organisms is the most important element of a DOP study. Favorable surrogate organisms should be 1) equivalent in size and shape to *Cryptosporidium* oocysts (i.e., 4-6 µm and slightly oblate), 2) sufficiently numerous in both ground water and surface water so as to be suitable for log removal calculations (log removal calculations require counts per volume of the same organism in both surface water and nearby collection devices/wells); and 3) sufficiently long-lived in the subsurface (at least as long-lived as oocysts) so that inactivation during subsurface passage does not significantly affect the calculation.

The identification of *Cryptosporidum* oocyst surrogate organisms is based primarily on similarity in size and shape. Other factors such as total net charge or charge distribution on the outer surface of the microorganism are important elements governing *Cryptosporidium* transport in the subsurface. However, choice of surrogate organisms based on charge or factors other than microorganism size and shape is an important research topic (Tufenkji 2007, Tufenkji et al. 2006) but the available information is insufficient for inclusion in this guidance. Exhibit 4.7 lists the size ranges of common pathogenic protozoa and surrogate bacteria.

No single *Cryptosporidium* surrogate organism is best. Each organism has strengths and weaknesses. Multiple surrogates should be analyzed initially to ascertain which surrogate suite is best suited to the DOP at that site. Examples of surrogates include total aerobic bacterial spores (e.g., *Bacillus subtilus),* anaerobic bacterial spores (e.g., *Clostridium perfringens* and/or *Clostridium bifermentans),* total coliform, *E. coli,* enterococci bacteria, bacteriophage (e.g., *Bacteroides* phage), coliphage (male-specific and somatic), diatoms (Reilly et al. 2005) at the genus or species level, turbidity, particle counting and microscopic particulate analysis (MPA) (U.S. EPA 1992, AWWA 1990). EPA recommends monitoring for at least three or four surrogate

organisms using paired surface water and ground water samples to calculate log removal efficiency.

As with any monitoring program, there is a trade-off between monitoring frequency and information cost. The cost for each *Cryptosporidium* surrogate assay varies between $50 and $250. EPA recommends that the less expensive assays, such as total aerobic spores, total coliform, and enterococci be performed more frequently and the more expensive assays, such as MPA, be performed at a lesser frequency.

### Exhibit 4.7 Size of Pathogenic Protozoa and Surrogate Bacteria

| Protozoa | Size (µm) | Surrogate Bacteria | Size (µm) |
|---|---|---|---|
| *Cryptosporidium parvum* oocyst (Xiao et al., 2000) | 4.2-5.6 | Total Coliform (Holt, 1986) | ~0.5-6.0 |
| *Giardia lamblia* cyst (WHO, 2004) | 8-12 | *Escherichia coli* (vegetative cell form) (Foppen and Schijven, 2006) | 1.1-6.0 |
| *Cyclospora sp.* (Mota et al., 2000) | 8-10 | *Clostridium perfringens* (vegetative cell form) (Holt, 1986) | 2-19 |
| *Microsporidia* (Brusseau et al., 2005) | 1-5 | *Clostridium perfingens* spore (Lund and Peck, 1994) | 0.3-0.4 |
| | | *Clostridium bifermentans* (vegetative cell form) (Holt, 1986) | 1-11 |
| | | *Clostridium bifermentans* spore (Brock and Madigan, 1991) | 1.2 |
| | | *Bacillus subtilus* (vegetative cell form) (Holt, 1986) | 2-5 |
| | | *Bacillus subtilus* spore (Rice et al., 1996 and P. Payment, personal communication | 0.5-0.8 |

*Cryptosporidium* oocysts are slightly oblate with a length-to-width ratio that ranged, in one study, from 1.04 to 1.33 (Xiao, 2000). Aerobic spores are typically slightly oblate as well but smaller than oocysts, ranging from 0.5-0.8 microns in diameter as compared with 4-6 microns for oocysts. Bacterial vegetative cells of *E. coli* are slightly larger than aerobic spores but significantly differ in length to width ratios (2.0-6.0 µm × 1.1-1.5 µm, Foppen and Schijven 2006). Futhermore, vegetative cells produce extracellular polymers, particularly if these cells form biofilms, and these polymers may significantly alter passage characteristics in the subsurface. The bacterial spore form is significantly longer lived in the environment (and especially the subsurface) than the vegetative cell form.

EPA recently completed a laboratory study (12 laboratories) of the total aerobic spore method. The study used natural ground water from a deep confined aquifer in Montana and Ohio River surface water. The ground water was analyzed to insure that it was devoid of (but not

sterile) aerobic spore forming bacteria. Aerobic spores (from BioBall) with well-defined counts but subject to variability were spiked in the ground water samples. Split surface water samples of unknown variability were also prepared. One ground water and one surface water sample was sent to each laboratory for multiple assay.

Laboratory performance was evaluated by comparing mean assay values separately for ground water and surface water. Spiked ground water sample variability among the twelve laboratories using Youden's Laboratory Ranking Test (Youden 1969) did not identify any outlying laboratories. Surface water mean values showed two outlying laboratories (one high outlying value) but the range of mean values was about a factor of five different between the high (23,962 CFU/100 ml) and (low 4,370 CFU/100 ml) values. Spore removal in this guidance is typically an assessment of whether surface water spore counts are diminished by a factor of one hundred when measured at a well. A factor of five difference is acceptable variability, assuming other information does not contradict that assessment.

The laboratory study shows that the aerobic spore laboratory method can be reproducibly performed by different laboratories and also provides acceptable recoveries of spores from spiked water samples. Because aerobic spores are: 1) relatively cheap and easy to measure, 2) identifiable without unacceptable laboratory error, 3) long-lived in the environment, 4) similar in shape to oocysts (albeit slightly smaller), and 5) do not produce extracellular polymers, EPA recommends aerobic spores, if present in large numbers in the surface water at the DOP site, as the most useful *Cryptosporidium* surrogate organism. Aerobic spores have long been recognized as a useful measure of surface water influence on and hygienic quality of ground water (e.g., Schubert 1975). Aerobic spores are also commonly used to assess the performance of engineered filtration systems (e.g., Mazoua and Chauveheid 2005).

The MPA method counts spores but these are fungal spores and not bacterial spores. Bacterial spore assay requires, at present, a culture step that is not currently part of the MPA method. MPA simply concentrates particulates and counts them. An aerobic spore assay standard method is available (APHA 2004). EPA recommends that unused aerobic spore sample be refrigerated (4 degrees C.). These refrigerated samples may then be re-assayed up to 24 hours post-receipt of the sample at the laboratory so that additional and differing dilutions can be conducted to reanalyze samples that are reported as "Too Numerous to Count" (TNTC).

Aerobic spore data have been collected from several recent studies at potential bank filtration sites (e.g., Weiss et al. 2005, Vogel et al. 2005, Gollnitz et al. 2004, Gollnitz et al. 2005, Partinoudi and Collins 2007, Gollnitz et al. 2007). It is important to differentiate sites that may be described as riverbank filtration sites but are not recognized as GWUDI by the state. For example, both Lincoln, NE (Vogel et al. 2005) and Cincinnati, OH (Gollnitz et al. 2004) field sites were studied in great detail using very sophisticated methods despite not being regulated as GWUDI. Thus, high aerobic spore log removal at these sites is expected because they are regulated as ground water rather than as surface water. Finally, at least one laboratory counted aerobic spore colonies in ground water without use of a dissecting microscope (in contrast to APHA 2004) (Partinoudi and Collins 2007). Partinoudi and Collins (2007) did not use a microscope so they report a high aerobic spore detection limit (<30 CFU/100 ml). Based on

unpublished data from Casper, WY and results reported in Locas et al. (2008), Schubert (1975) and Rice et al (1999), the aerobic spore natural background concentration is about 10 CFU/100 ml or less. Based on these studies, values higher than 10 CFU/100 ml may be considered to have some surface water influence. Thus, a high detection limit makes it difficult to differentiate native and surface water-influenced ground water.

The ability to produce environmentally-resistant aerobic spores is fairly limited in the bacterial world. Current methodology recovers almost exclusively those aerobic spores from the genus *Bacillus*. There are many species of *Bacillus* and all produce aerobic spores. Members of this genus are found naturally in all environments, including surface waters and especially in soil. It is assumed that, when found in surface water, aerobic spores found naturally in soil are washed into surface water by natural processes. As soil bacteria, aerobic spore populations in surface water are expected to comprise a diverse bacterial population, representing all aerobic spore taxa within the surface water watershed. The aerobic spore population in a ground water sample may be 1) similarly as taxonomically diverse as the surface water population, 2) taxonomically less diverse than the surface water population because some spore taxa have favorable properties (e.g., charge) for subsurface passage while other taxa are more likely to attach to aquifer solids, or 3) taxonomically diverse or not but representative only of the spore population in the soil in the immediate vicinity of the wellhead.

Gollnitz et al. (2005) suggest that ground water aerobic spore samples exhibit "endospore monocultures" and also suggest that these "monocultures" explain the instances when collection devices exhibit negative (low) removal efficiency. (Given the high uncertainty in log removal calculations, the difference between negative and low removal efficiency is not significant.) "Monocultures" implies that all of the cells recovered on the growth medium are of the same strain and possible clonal, being genetically identical. This would ordinarily imply that the organisms were growing either in the groundwater itself or in the sample once it was collected. Some type of genetic profiling analysis would need to be run in order to document all of the cells recovered as clonal. To date no such data has been reported. A more likely explanation for a sample yielding *Bacillus* colonies that are morphologically similar is laboratory contamination. Thus, any suggestion that, in the absence of genetic profile or speciation data, colonies are "monocultures," should be recognized as premature and possibly incorrect. Finally, as discussed above, low taxa diversity in a ground water sample (when recognized by speciation data) provides no information on log removal by subsurface passage.

Anaerobic spores are also recommended as surrogate microorganisms because these microorganisms, like aerobic spores, are small (0.3-1.2 μm), spherical, and long-lived. Riverbank filtration studies in the Netherlands (e.g., Shijven et al. 2003, Medema and Stuyfzand 2002) used spores of sulfide-reducing Clostridia (SSRC) and *Clostridium bifermentans* spores as *Cryptosporidium* surrogates in studies of the Rhine and Meuse Rivers. However, anaerobic spores probably do not have a significant presence in surface water unless there are significant upstream sewage discharges to surface water, as there are in the Rhine and Meuse. Thus, the utility of anaerobic spores at a DOP site where surface water quality is typically very good is limited.

Total coliform bacteria are vegetative cells that, like *Bacillus sp.,* originate largely as soil bacteria, and are found at high density in surface water. Total coliform density in wells has been used for GWUDI determination in bank filtration settings. Price et al., 1999 show that higher total coliform density occurs in horizontal collector well #5 from January to April, during highest flow conditions in the river. Thus, well #5 is used primarily during summer months when water use demand is high and river flow is low. However, using measurement of aerobic spores (or carboxylated microspheres, Metge et al. 2007) in addition to measurement of total coliform vegetative cells might provide a differing assessment because aerobic spores should more efficiently passage through the subsurface and may be present in significant density before January or after April.

Diatoms are a specialized group of marine and freshwater algae that all produce a rigid cell wall (frustule) composed of silica. There are 58 freshwater diatom genera (AWWA 1995). Diatoms are counted separately from other algae in a MPA (U.S. EPA 1992). The MPA method only counts whole diatoms; diatom fragments are not considered. Exhibit 4.8 shows the size and shape of some common freshwater diatoms. Diatoms vary in size (e.g., from 4-10 microns to 60 or 70 microns). Smaller diatoms may be transported through sand and other porous media at rates similar to oocysts. Larger diatoms may, if they have large length to width ratios, orient themselves in the ground water flow field so that the long axis is parallel to the flow direction, which also may allow them to pass through sand and other porous media. Diatoms are photosynthesizing algae that require light to maintain their green chlorophyll. After about 6 months residence time in the subsurface, the green color will fade. (Susan Boutros, EPA GWUDI Determination Presentation, Denver CO, verbal communication on unpublished laboratory experiments with diatoms placed in a refrigerator).

### Exhibit 4.8 Size of Some Common Fresh Water Diatoms

| Diatom | Size (length x width) (µm) | Shape |
|---|---|---|
| Stephanodiscus hantzchii | 10 × 5-8 | Cylindrical (Hendricks et al., 2000) |
| Synedra acus | 60-70 × 3-4 | Needle (Hendricks et al., 2000) |
| Cyclotella meneghiniana | 5 × 3 | Cylindrical (Hendricks et al., 2000) |
| Cyclotella pseudostelligera | 4-10 | Centric (Reilly et al., 2005) |
| Fragilaria crotonensis | 40-170 × 2-4 | Pennate (Reilly et al., 2005) |
| Aulacoseira granulata | 4-30 | Centric (Reilly et al., 2005) |
| Asterionella formosa | 40-80 × 1.3-6 | Pennate (Reilly et al., 2005) |
| Nitzschiia palea | 15-70 × 2.5-5 | Pennate (Reilly et al., 2005) |

Because most diatoms are larger than *Cryptosporidium* oocysts (see section 4.7.2), diatom occurrence in a well signals that oocysts, like diatoms, could also be present in inadequately filtered drinking water from a well. Diatoms occurrence is subject to less uncertainty because the rigid frustule is not likely to be sufficiently deformable to pass through smaller pores, unlike most biological particles. Thus, one or more whole diatom tests, identified in well water, and counted using the MPA or another method, are particularly meaningful data. Some diatom species are also identifiable using immunoassay methods (Walker et al. 2005),

although the detection limit is high (500 cells per liter) and thus not well suited for porous media groundwater sites where the diatom count is expected to be significantly lower.

EPA recommends weekly or biweekly aerobic spore samples plotted on a graph together with monthly diatom data from MPA and river stage values to evaluate bank filtration efficiency. The spore data are a measure of bank filtration efficiency which should decrease with increasing river stage (i.e., high spore occurrence in a well at high river stage). For a regulated river (e.g., upstream dams and reservoirs) the correlation between aerobic spore recovery in wells and river stage might be muted or non-existent. Diatom data are used to validate the aerobic spore data; weeks or months with high spore recovery in wells should also have accompanying diatom occurrence in well water.

### 4.7.6 Tracer Tests and Use of Isotopes

It may be useful for the study design to include a pilot test at one or more collection devices taken off-line. The pilot test could consist of adding a known quantity of inactivated cysts or indicator organism (including microspheres) and conservative tracer ("spiking") to the source and collecting and analyzing samples from the collection device. This type of pilot test may assist in the assessment of actual removal through the alternative filtration system. If a correlation between cysts and indicator organism(s) can be established, this correlation could be used to focus and expedite monitoring. Spiking studies are best suited for artificial recharge studies but can also be used in bank filtration studies if an injection well is drilled to insert the spiked samples. If an injection well is used, it would be preferable to drill it as a slant well under the riverbed that bottoms just below the riverbed.

As a supplement to special spiking studies, the DOP could conduct pilot laboratory column or tank studies of relatively undisturbed natural or engineered materials to evaluate their performance when challenged with a cyst, oocyst, indicator(s) and conservative tracer spike. These studies are especially important for demonstrating that the porous media transport of the indicator(s) identified in field studies is similar to the transport of cysts and oocysts.

At least three studies (Coplen et al. 1999, Vogel et al. 2005, Hunt et al. 2005) have used stable isotopes in a North American bank filtration study, although isotopes are commonly used in less specialized ground water and surface water interaction studies in the United States and elsewhere. Coplen et al. showed that, for Portland, OR, municipal field well #1, increasing Columbia River contribution to well yield with pumping, with about half of the yield from surface water at day 7 and culminating at 82 percent surface water on day 23 of the pump test. Hunt et al. showed that the travel time of surface flood water to the municipal wells in La Crosse, WI was approximately 2 months as compared to inter-flood periods with about 9 month travel times. Age dating using $^{3}H$-$^{3}He$ and tracers such as CFCs and $SF_6$ were less useful at the site.

### 4.7.7 Monitoring Wells Located Along the Shortest Flow Path

The LT2ESWTR allows DOP credit for a site-specific study ONLY if one or more monitoring wells are screened and located along the shortest flow path between the surface water source(s) and the production device(s) (well). As discussed in section 4.7.3, particle-tracking ground water flow models are best suited for identifying the shortest flow path and the appropriate depth of the screened interval. However, optimally located, existing monitoring wells may also be used if they are directly located between the river and the well and are screened at a depth intermediate between the river channel bottom and the production well screen.

### 4.7.8 Post-decision Routine Monitoring and Sampling

Any DOP study should include a component to develop a routine sampling and monitoring program that would validate the continuation of any removal credit granted. EPA recommends continued post-decision bi-weekly or monthly aerobic spore monitoring to insure that any approved *Cryptosporidium* log removal credit is maintained over time. Post-decision routine monitoring is similar to filtration plant performance testing and will allow comparison to previously-collected data to determine if degradation of alternative filtration performance is occurring and to document improvements in alternative treatment removals based on improved wellfield operation and maintenance.

## 4.8 Reference

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# 5. Presedimentation

## 5.1 Introduction

Presedimentation is a preliminary treatment process used to remove gravel, sand, and other material from the raw water and dampen particle loading fluctuations to the rest of the treatment plant. A plant may receive credit for a presedimentation basin for any month the basin meets the requirements as described in 40 CFR 141.717(a).

Sedimentation processes are common in the water treatment process and much design and operational information is available. However, the use of an additional sedimentation basin in series, or a pre-sedimentation basin at the head of the treatment plant is not as common as the standard sedimentation basin, and little information is available. Therefore, the guidance provided in this chapter is based on the design and operational principles of sedimentation processes.

This chapter on presedimentation is organized as follows:

5.2 Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) Compliance Requirements - This section describes the criteria presedimentation basins must achieve in order to receive *Cryptosporidium* removal credit.

5.3 Toolbox Selection Considerations - This section assists systems in determining whether the presedimentation toolbox option is a viable and beneficial option for meeting the LT2ESWTR bin requirements.

5.4 Types of Presedimentation Basins - This section compares several sedimentation basins and clarifiers in terms of structure and factors affecting settling efficiency.

5.5 Design and Operating Issues - This section discusses typical design and operational issues including redundancy, short circuiting, sludge removal, and coagulant addition.

## 5.2 LT2ESWTR Compliance Requirements

### 5.2.1 Credits

Presedimentation basins with coagulant addition may receive 0.5 log *Cryptosporidium* removal credit under the LT2ESWTR if they meet the following criteria (40 CFR 141.717(a)):

- The presedimentation basin must be in continuous operation and must treat all of the flow taken from a surface water or ground water under the direct influence of surface water (GWUDI) source.

- A coagulant must be continuously added to the presedimentation basin while the plant is in operation.

- The presedimentation basin must achieve a monthly mean reduction of 0.5 log (68 percent) or greater in turbidity or alternative state-approved performance criteria that demonstrate at least 0.5 log mean removal of micron sized particulate material through the presedimentation process.

### 5.2.2 Monitoring Requirements

Systems must measure presedimentation basin influent and effluent turbidity at least once per day or meet state-specified performance criteria (40 CFR 141.717(a)). State-specified criteria could include aerobic spore removal (see Chapter 12, Section 12.4.2.1 of this guidance manual) or particle count reduction. Laboratory support would be needed for spore counts and grab sampling and dilution would be needed to assess particle count reduction.

### 5.2.3 Calculations

For compliance with the LT2ESWTR, the log turbidity reduction must be calculated as a monthly mean, from readings collected daily, according to the following equation (40 CFR 141.717(a)(3)(i)).

Log Reduction =

$\text{Log}_{10}$(Monthly Average Influent Turbidity) - $\text{Log}_{10}$(Monthly Average Effluent Turbidity)

Or if calculated as a percent,

Percent Reduction =

$$\frac{\text{(Monthly Average Influent Turbidity) - (Monthly Average Effluent Turbidity)}}{\text{(Monthly Average Influent Turbidity)}} \times 100$$

*Example Calculation*

Average influent turbidity = 16.3 nephelometric turbidity units (NTU)
Average effluent turbidity = 4.2 NTU

Log Reduction = $\text{Log}_{10}(16.3) - \text{Log}_{10}(4.2) = 0.59$

Percent Reduction = (16.3-4.2)/16.3 = 74.2%

## 5.3 Toolbox Selection Considerations

The purpose of this section is to assist systems in determining whether the presedimentation toolbox option is a viable and beneficial option for meeting the LT2ESWTR bin requirements. There are two general aspects for systems to evaluate when considering this toolbox option:

1) Can the turbidity removal requirements be met consistently over the expected range of raw water conditions?

2) What are the advantages and disadvantages of installing a presedimentation basin?

For presedimentation, the first question is driven by source water particle load and how much of that load a proposed sedimentation basin would remove. Before researching potential presedimentation designs, a system should determine if their source water has a high enough turbidity on a consistent basis. Section 5.3.1 discusses the source water characteristics necessary to meet the compliance requirements. Section 5.3.2 discusses the advantages and disadvantages of adding a presedimentation process to the treatment train.

### 5.3.1 Source Water Quality

To meet the 0.5 log turbidity removal requirement, the source water should have consistently high turbidity, and comply with state specified performance criteria. When source water turbidity is seasonally or consistently low, most presedimentation basins will have difficulty achieving 0.5 log reduction, and systems may need to use another tool in the toolbox to meet state-specified criteria such as aerobic spore removal or particle count reduction. For example, if a system has an average of 10 NTU source water turbidity for a few months of the year, the average effluent turbidity would have to be 3.2 NTU for those months, which could be difficult for some systems to achieve. Exhibit 5.1 lists influent and effluent turbidity values that yield 0.5 log reduction. These are example values to help when considering using the tool.

**Exhibit 5.1 Example Influent and Effluent Turbidity Values Resulting in 0.5 Log Reduction**

| Monthly Average Turbidity (NTU) | |
|---|---|
| **Influent** | **Effluent** |
| 2 | 0.6 |
| 10 | 3.2 |
| 30 | 9.5 |
| 50 | 15.8 |
| 70 | 22.1 |
| 80 | 25.3 |
| 100 | 31.6 |

### 5.3.2 Advantages and Disadvantages of Installing a Presedimentation Basin

The presedimentation process can reduce influent fluctuations in particle loading, flow, and other water quality parameters. An additional sedimentation process in series provides increased operational flexibility to handle rapid changes in influent turbidity. It also allows for enhanced performance of subsequent processes in the treatment plant.

As with the addition of many unit processes, the two major disadvantages are capital costs and land requirements. The requirement of coagulant addition may increase chemical costs, although the amount added in the next stage could be reduced. Whether these chemical costs offset each other is site-specific.

## 5.4 Types of Sedimentation Basins

There are several types of sedimentation basins (also called clarifiers) used for drinking water treatment. Selection of a basin for presedimentation should be based on turbidity removal capability and meeting the flow and space requirements of the facility. The focus of this chapter is on guidance for complying with the LT2ESWTR, therefore the discussion in this section is limited to factors affecting settling efficiency, as measured by turbidity removal. Further information on design can be found in the following literature:

- Water Quality and Treatment—A Handbook of Community Water Supplies, 5th ed. (AWWA 1999).

- Integrated Design and Operation of Water Treatment Facilities, 2nd ed. (Kawamura 2000).

Exhibit 5.2 provides a comparison of several sedimentation basins and clarifiers. It is likely that only horizontal clarifiers would be chosen for presedimentation, since they are less complex in operation compared to the others (i.e., upflow, high rate, reactor, and ballasted sand clarifiers). The table includes the additional types since some plants that choose to employ the presedimentation toolbox option may elect to use their current sedimentation basin for presedimentation and construct a new basin for primary sedimentation. The performance advantages and disadvantages listed in the table relate to settling efficiency or indications for potential process upset. These were derived from *Integrated Design and Operation of Water Treatment Facilities* (Kawamura 2000) and are characteristic of sedimentation processes, not specifically presedimentation processes. The remainder of this section provides short descriptions of different clarifier types.

## Exhibit 5.2 Comparison of Sedimentation and Clarifier Types

| Type | Performance Advantages | Performance Disadvantages |
|---|---|---|
| **Applicable for Presedimentation and Sedimentation** | | |
| **Horizontal Flow** (general) | -Easy to operate and maintain | |
| Rectangular Basin | -Tolerant to shock loads<br>-Good for handling large flows | -Subject to wind and density currents (causing short-circuiting)<br>-Designs with trays have shown poor settling efficiency |
| Circular Basin | -Easy sludge removal<br>-Can obtain high clarification efficiency | -Greater potential for hydraulic imbalance in comparison to rectangular basin (not good for removing alum flocs) |
| **Applicable for Sedimentation** | | |
| **Upflow Clarifier** (general) | -High clarification efficiency | -Need constant flow rate and water quality<br>-Limitations on size |
| Center Feed | -Easy sludge removal | -Short circuiting |
| Peripheral Feed | -Good for source water with high solids | -Potential short-circuiting |
| **High Rate Settlers** (horizontal flow or upflow) | -Increases the hydraulic load capability and settling efficiency of horizontal flow basins and clarifiers | -Can form scales (calcium carbonate) which clog flow<br>-Poor flocculation possible |
| **Reactor Clarifiers** (general) | -Good clarification due to seeding effect | -Need constant flow rate and water quality<br>-Requires greater operator skill |
| High recirculation and mechanical sludge plow | -Tolerant to shock loads | -Dependent on one drive unit<br>-Limitations on size |
| Sludge blanket zone and mechanical sludge plow | -Good turbidity removal | -Very sensitive to shock loads<br>-Requires 2-4 days to build sludge blanket |
| **Ballasted sand** | -Can handle higher flows with very low detention times (on the order of minutes)<br>-Can handle shock particle loads without increasing coagulant dose<br>-Quick process startup | -Short detention time means not much time for process adjustments |

Note: Adapted from "Integrated Design and Operation of Water Treatment Facilities." Kawamura (2000).

Sedimentation processes can be categorized in three general types: horizontal flow basins or clarifiers, upflow clarifiers, and reactor clarifiers. High rate settlers are modified horizontal or upflow clarifiers with plate or tube modules placed into the basin to increase the settling area. An

additional design described in this chapter that differs from the three general types is ballasted sand or high-rate microsand process (a proprietary design).

### 5.4.1 Horizontal Flow

#### 5.4.1.1 Rectangular

In rectangular sedimentation tanks the water flows in one end and ideally proceeds through the basin in a plug flow manner. A uniform distribution at the inlet is an important design factor. Rectangular basins can be susceptible to density currents that cause short circuiting. These basins are easy to operate, have low maintenance costs, offer predictable performance under most conditions, and are most tolerant to shock loads. High rate settlers can be easily installed to improve settling efficiency. Rectangular basins are particularly well suited for large systems compared to circular basins that require additional space and yard piping for equivalent flow.

#### 5.4.1.2 Circular

The flow in circular basins is commonly from a center feed well, radially outward to the peripheral weirs. In comparison to rectangular basins, circular basins will have more land between the basins and also require more yard piping. Circular basins have easy sludge removal, can obtain high clarification efficiency, and are adaptable to high rate settling modules. However, if flow distribution from the inlet is not uniform, the settling efficiency will be hindered. These basins are not as hydraulically stable as rectangular basins.

### 5.4.2 Upflow Clarifier

In upflow clarifiers the influent enters at the bottom and clarified water flows upward while the solids settle to the bottom. As with horizontal flow basins, upflow clarifiers can also be modified with high rate settling modules. Upflow clarifiers can provide higher clarification efficiency than horizontal flow; however, they are more sensitive to shock loads than horizontal flow basins.

### 5.4.3 Reactor Clarifier

Reactor clarifiers use the seeding concept to improve settling. The water flows through the sludge layer so particles can coalesce with already formed flocs. Two common designs of reactor clarifiers are slurry recirculation and sludge blanket clarifiers. Both operate on a center feed system with built-in flocculation zones. The process is more complex than traditional horizontal or upflow clarifiers. Reactor clarifiers can provide high clarification efficiency but at the cost of flexibility—the source water quality and hydraulic loads must be constant.

### 5.4.4 High Flow Rate Designs

High rate settlers are modules of inclined tubes or plates that are installed in horizontal flow (plates only) or simple upflow clarifiers. They provide increased surface area for particles to settle and reduce settling time. Kawamura (2000) noted poor performance occurred when flow distribution was uneven and flocculation was poor.

### 5.4.5 Ballasted Flocculation

Ballasted flocculation is a high-rate, physical-chemical clarification process that uses sand to improve the settling of flocculated particles. The floc attaches to the surface of a sand particle, which has a settling time 20 to 60 times faster than an alum floc (Kawamura 2002), thus creating a high-rate settling process. Because of the increased settling rate, the space required is much less than other clarifiers.

## 5.5 Design and Operational Issues

### 5.5.1 Redundancy

As stated earlier, for compliance with the LT2ESWTR, all flow must be treated by the presedimentation process to receive *Cryptosporidium* treatment credit (40 CFR 141.717(a)). Systems should consider the need for redundancy in the design of a presedimentation process. Smaller systems or systems with a demand much lower than the design capacity may be able to shut down the water treatment plant for presedimentation basin maintenance activities and, thus, not require additional basins for redundancy. However, systems that operate on a continuous basis do not have that flexibility and should have a plan for staying in compliance while a basin is shut down.

### 5.5.2 Short Circuiting

A common issue that must be considered in the design and operation of presedimentation basins is short-circuiting. If a portion of flow does not have the adequate detention time, then the effluent turbidity is likely to be higher than anticipated. Several factors affect short-circuiting including even distribution of flow at the inlet, density or temperature differentials between influent and basin water, surface currents, and basin cleaning and sludge removal.

A proper design of the inlet is one of the most important design factors. In addition to flow short-circuiting, a poorly designed inlet can lead to overall hydraulic instability in the settle

zone. Installation of perforated baffles is a simple and effective method for even flow distribution from the inlet to the basin.

Temperature differentials and high wind velocities could induce circular currents in the vertical direction of the basin. Influent water warmer than the basin water will rise to the surface and reach the outlet of the sedimentation basin much faster than the intended detention time of the basin. Influent water colder than the basin water will dive to the bottom of the basin and flow along the bottom of the basin and rise to the top of the basin at the outlet, thereby reaching the outlet of the sedimentation basin much faster than the intended detention time of the basin. Above ground tanks built of steel are more susceptible to temperature differentials from exposure to the sun and heat transfer.

The degradation of effluent water quality due to wind is more noticeable in circular or square sedimentation basins of diameters greater than 100 – 115 feet. When using long, shallow rectangular settling basins, effects of wind induced currents can be minimized by ensuring that the longitudinal axis of the basin is perpendicular to the prevailing wind direction. In addition to causing flow short-circuiting, currents can also scour settled solids, causing resuspension of settled solids and increasing effluent turbidity.

### 5.5.3 Sludge Removal

Sludge build-up in the tank decreases the volume of the sedimentation basin and reduces the settling time in the basin. Additionally, as sludge builds up, particles become more susceptible to resuspension during sludge removal, increasing the effluent turbidity. Sedimentation basins with high rate settlers accumulate sludge rapidly, and therefore require continuous sludge removal.

### 5.5.4 Coagulant Addition and Dose Ranges of Common Coagulants

Current operational practices of presedimention processes often focus on mitigating shock loads in the raw water supply (such as turbidity spikes due to precipitation in river source waters). However, during periods of low influent turbidity less attention may be given to the actual performance of the basin, resulting in less than 0.5 log turbidity reduction through the basin. To receive the credit, the presedimentation basin may need to be operated more stringently, including the addition of coagulant. The coagulant dose required to treat an influent stream depends on the chemical composition of the influent, the characteristics of the colloids and suspended matter in the influent, the addition of a coagulant aid, the water temperature, and mixing conditions. Coagulant dose and other water chemistry parameters of the coagulation and sedimentation processes are system-specific. Jar test procedures for evaluating the appropriate coagulants, dosages, and other chemical attributes for a treatment train are provided in AWWA's *Operational Control of Coagulation and Filtration Processes*.

## 5.6 References

AWWA. 2000. Operational Control of Coagulation and Filtration Processes, AWWA Manual M37, Second Edition, pp. 1-34.

Kawamura, Susumu. 2000. *Integrated Design and Operation of Water Treatment Facilities*. John Wiley & Sons, Inc.

U.S. EPA, 1998. Optimizing Water Treatment Plant Performance Using the Composite Correction Program, pp. 233-236.

# 6. Lime Softening

## 6.1 Introduction

Lime softening is a drinking water treatment process that uses chemical precipitation with lime and other chemicals to reduce hardness and to enhance clarification prior to filtration. Lime softening can be categorized into two general types: (1) single stage softening that is used to remove calcium hardness and (2) two-stage softening that is used to remove magnesium hardness and high levels of calcium hardness. A single stage softening plant includes a primary clarifier and filtration components. A two stage softening plant has an additional clarifier located between the primary clarifier and filter. Within these general categories there are several possible treatment schemes; however, describing each is beyond the scope of this chapter.

This toolbox option is practical for lime softening plants that either have a two stage process or could upgrade to a two stage process. The advantage of using this toolbox option to achieve compliance with the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) is that systems will have the treatment process in place or if an upgrade or modification is needed, it could benefit the treatment of other contaminants. A disadvantage for softening plants is a potential reduced flexibility in the treatment train since all water must be treated by both stages.

Since the water systems considering this toolbox option will most likely have a lime softening process in place, this section does not provide design or operational information. Instead, this section focuses on the requirements that lime softening systems must meet to receive *Cryptosporidium* removal credit and how those requirements can be met with general process modifications. The chapter is organized into two sections:

6.2 LT2ESWTR Compliance Requirements - describes the criteria that plants must meet in order to receive additional credit for *Cryptosporidum* removal, and reporting requirements to maintain compliance.

6.3 Split Flow Processes - addresses compliance issues for split flow processes.

## 6.2 LT2ESWTR Compliance Requirements

### 6.2.1 Credit for *Cryptosporidium* Removal

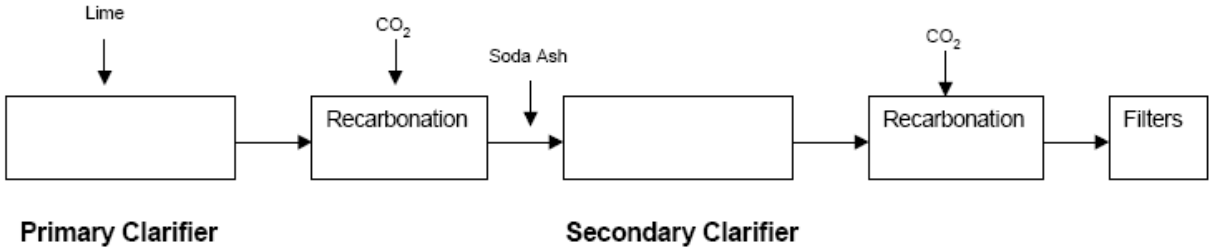
The LT2ESWTR requires plants to meet the following criteria in order to receive 0.5-log credit towards additional *Cryptosporidium* treatment requirements (40 CFR 141.717(b)):

- The plant must have a second clarification step between the primary clarifier[1] and filter which is operated continuously. For split treatment processes, only the portion of flow going through two clarification stages can receive credit. If a portion of flow bypasses one stage, additional treatment must be provided to the bypassed portion (see section 6.3).

- Chemical addition and hardness removal must occur in two separate and sequential stages.

Exhibit 6.1 shows a typical two stage lime softening process.

**Exhibit 6.1 Typical Two-Stage Lime Softening Process**




### 6.2.2 Reporting Requirements

The LT2ESWTR requires monthly verification and reporting of the following conditions for systems using the lime softening option (40 CFR 141.721):

- Chemical addition and hardness precipitation occurred in two separate and sequential softening stages prior to filtration.

- Both clarifiers treat 100 percent of the plant flow.

A schematic of the treatment processes, clearly identifying the two stages of clarification, will assist the state in evaluating the process for the purposes of LT2ESWTR removal credit. Monitoring of chemical doses in the secondary clarifier over the expected range of seasonal raw water quality and recording of minimum and average chemical concentration will assist the state in evaluating the process and the system in determining operating criteria.

[1] For purposes of compliance with the lime-softening toolbox option, "clarifier" is used as a general term for processes with settling and solids removal.

## 6.3 Split-Flow Processes

Split-flow processes divert a portion of the flow from either the first or second stage of the process and then blend the two streams together further downstream. Only the portion of flow that receives the two stages of treatment would be eligible for the 0.5 log credit. In these situations, systems would either have to: 1) eliminate the bypass and direct the entire flow through both stages, or 2) treat the bypassed portion with another toolbox option, such as chlorine dioxide, membranes, or ozone to receive *Cryptosporidium* inactivation/removal credit for that stream.

# 7. Combined and Individual Filter Performance

## 7.1 Introduction

Turbidity is an optical property that measures the amount of light scattered by suspended particles in a solution. It can detect a wide variety of particles in water (e.g., clay, silt, mineral particles, organic and inorganic matter, and microorganisms), but cannot provide specific information on particle type, number, or size. Therefore, the U.S. Environmental Protection Agency (EPA) recognizes that turbidity reduction is not a direct indication of pathogen removal, but is an effective indicator of process control.

The Surface Water Treatment Rule (SWTR), Interim Enhanced SWTR (IESWTR), and Long Term 1 Enhanced SWTR (LT1ESWTR) all motivate public water systems to achieve a certain level of finished water quality by requiring them to meet specified filtered water turbidity limits. Under the IESWTR and LT1ESWTR, combined filter effluent (CFE) turbidity in conventional and direct filtration plants must be less than or equal to 0.3 nephelometric turbidity units (NTU) in 95 percent of samples taken each month and must never exceed 1 NTU. These plants are also required to conduct continuous monitoring of turbidity for each individual filter, and provide an exceptions report to the state or regulating agency when certain criteria for individual filter effluent (IFE) turbidity are exceeded.

The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) awards additional *Cryptosporidium* treatment credit to certain plants that maintain finished water turbidity at levels significantly lower than currently required. This credit is not available to membrane, bag/cartridge, slow sand, or diatomaceous earth plants, due to the lack of documented correlation between effluent turbidity and *Cryptosporidium* removal in these processes.

The remainder of this chapter is organized as follows:

7.2 LT2ESWTR Compliance Requirements - describes the conditions for receiving *Cryptosporidium* removal credit and monitoring requirements for maintaining compliance.

7.3 Reporting Requirements - describes the routine reporting requirements that systems must follow to receive credit.

7.4 Process Control Techniques - discusses modifications or operational aspects that provide the tightened process control needed to meet the turbidity requirements for this toolbox option.

7.5 Process Management Techniques - describes standard operating procedures (SOPs), response plans for loss of chemical feed, adequate chemical storage, and voluntary programs that encourage full process control from administration to operation and maintenance.

## 7.2 LT2ESWTR Compliance Requirements

### 7.2.1 Treatment Credit

For systems using conventional or direct filtration treatment to obtain an additional 0.5 log *Cryptosporidium* removal credit, the LT2ESWTR requires the CFE turbidity measurements taken for any month at each plant are less than or equal to 0.15 NTU in at least 95 percent of the measurements (40 CFR 141.718(a)).

The LT2ESWTR also allows systems using conventional or direct filtration treatment to claim an additional 0.5 log *Cryptosporidium* removal credit for any month at each plant that meet both of the following IFE turbidity requirements (40 CFR 141.718(b)):

1) IFE turbidity must be less than 0.15 NTU in at least 95 percent of values recorded at each filter in each month, excluding the 15 minute period following return to service from a filter backwash.

   AND

2) No individual filter may have a measured turbidity greater than 0.3 NTU in two consecutive measurements taken 15 minutes apart.

Systems may claim credit for combined filter performance AND individual filter performance in the same month (40 CFR 141.718(b)) for 1.0 log total.

### 7.2.2 Monitoring Requirements

For both the CFE and IFE options, compliance with the LT2ESWTR is determined by sample measurements taken for the IESWTR and LT1ESWTR (40 CFR 141.718). In other words, the LT2ESWTR does not require any additional monitoring from the IESWTR and LT1ESWTR.

#### 7.2.2.1 Combined Filter Effluent

The monitoring frequency and compliance calculation requirements for the CFE option are that CFE turbidity must be measured at 4-hour intervals (or more frequently) and 95 percent of the measurements from each month must be less than or equal to 0.15 NTU (40 CFR 141.721).

#### 7.2.2.2 Individual Filter Effluent

The LT2ESWTR has specific reporting requirements. The monitoring frequency and compliance calculation requirements for the IFE option are that IFE turbidity must be measured every 15 minutes (excluding the 15 minute period following return to service from a filter backwash) and 95 percent of the measurements from each month must be less than or equal to 0.15 NTU (40 CFR 141.721).

The LT2ESWTR specifies no individual filter may have a measured turbidity greater than 0.3 NTU in two consecutive measurements taken 15 minutes apart (40 CFR 141.721). If the individual filter is not providing water which contributes to the CFE (i.e., it is not operating, is filtering to waste, or its filtrate is being recycled) the system does not need to report the turbidity for that specific filter.

### 7.2.3 Turbidity Monitors

An important aspect of awarding additional removal credit for lower finished water turbidity is the performance of turbidimeters in measuring turbidity below 0.3 NTU. EPA believes that currently available turbidity monitoring equipment is capable of reliably assessing turbidity at levels below 0.1 NTU, provided instruments are well calibrated and maintained. EPA strongly recommends systems that pursue additional treatment credit for lower finished water turbidity develop the procedures necessary to ensure accurate and reliable measurement of turbidity at levels of 0.1 NTU and less, and believes these procedures to be essential to maintain toolbox credit.

Turbidimeter maintenance should include frequent calibration by the manufacturer's methods as well as frequent verification, in order to measure accurately in the low turbidity ranges required for this toolbox option. Chapter 3 of the *LT1ESWTR Turbidity Provisions Guidance Manual* describes the sampling methods, operation, maintenance, and calibration for turbidimeters and discusses quality assurance and quality control measures. This section summarizes the information from that chapter, including the approved methods, commonly used turbidimeters, calibration standards, and important factors of maintaining turbidimeters. Systems are encouraged to review Chapter 3 of the *LT1ESWTR Turbidity Provisions Guidance Manual* to ensure their operation, maintenance, and calibration practices meet or exceed those recommended by EPA. For a full copy of this document see:

> The LT1ESWTR guidance manuals are available on EPA's website at:
>
> http://www.epa.gov/ogwdw000/mdbp/lt1eswtr/guidance_lt1_turb.pdf.

#### 7.2.3.1 Methods

Currently, EPA has approved four methods for the measure of turbidity (described in 40 CFR 141.74):

- EPA Method 180.1.
- Standard Method 2130B.
- Great Lakes Instrument Method 2.
- Hach Filter Trak.

#### 7.2.3.2 Maintenance and Calibration

Maintenance and calibration of both benchtop and on-line turbidimeters are fully described in the *LT1ESWTR Turbidity Provisions Guidance Manual*. It is very important to follow the manufactures procedures for maintenance and calibration of turbidimeters, as they vary between manufacturers. Exhibits 7.1 and 7.2 list several maintenance and calibration activities common among manufacturers for on-line and bench top turbidimeters. These activities should be conducted for all turbidimeters to ensure proper operation on a consistent basis.

### Exhibit 7.1 Maintenance and Calibration Activities for On-line Turbidimeters

| Activity | Recommended Frequency |
|---|---|
| Inspect for cleanliness | Weekly |
| Verify sample flow rate | Weekly |
| Verify calibration with primary standard, secondary standard or by comparison with bench-top | Weekly on CFE turbidimeter and monthly on all IFE turbidimeters[1] |
| Clean and calibrate with primary standard | Quarterly |
| Replace lamp | Annually |

[1]Clean and recalibrate with primary standard if verification indicates greater than a +/-10 percent deviation from secondary standard.

## Exhibit 7.2 Maintenance and Calibration Activities for Bench Top Turbidimeters

| Activity | Recommended Frequency |
|---|---|
| Inspect for cleanliness of bulbs and lenses | Regularly, such as monthly or quarterly |
| Verify calibration with secondary standard | Daily[1] |
| Clean and calibrate with primary standard | Quarterly |
| Replace lamp | Annually or according to manufacturer's recommendations |

[1] Instrument calibration should be verified on a daily basis (http://www.epa.gov/ogwdw000/mdbp/pdf/turbidity/chap_03.pdf). Clean and recalibrate with primary standard if verification indicates greater than a +/-10 percent deviation from secondary standard.

In addition to those activities listed in the tables, the following documentation or record keeping items should be developed and kept up to date:

- Log of turbidimeter maintenance and calibration.
- Quality assurance/quality control (QA/QC) plan for accuracy and consistency.
- SOPs

#### 7.2.3.3 Quality Assurance / Quality Control (QA/QC)

Systems should develop a QA/QC plan for measuring turbidity. This plan should include written SOPs to ensure that operation, maintenance, and calibration activities are carried out in a consistent manner, and that each activity is understood by all that are involved. At a minimum, systems should develop SOPs for cleaning turbidimeters, creating Formazin Standards, calibrating turbidimeters, and referencing index samples.

For bench top turbidimeters, measurement errors can be introduced by dirt, scratches, or condensation on the glassware, air bubbles in the sample, and particle settling. Operators should strictly follow manufactures procedures for sampling and maintenance.

## 7.3 Reporting Requirements

### 7.3.1 Combined Filter Performance

In order to receive the 0.5 log removal credit for the LT2ESWTR, a water system must submit monthly verification of CFE turbidity levels less than or equal to 0.15 NTU in at least 95 percent of the 4-hour CFE measurements taken each month (40 CFR 141.721).

### 7.3.2 Individual Filter Performance

For the 0.5 log IFE removal credit under the LT2ESWTR, a water system must report monthly verification of IFE turbidity levels less than or equal to 0.15 NTU in at least 95 percent of all maximum daily IFE measurements taken each month for each filter (excluding the 15 minute period following startup after backwash), and monthly verification that there were no IFE measurements greater than 0.3 NTU in two consecutive readings 15 minutes apart for any filter (40 CFR 141.721).

As requirements of the IESWTR and the LT1ESWTR, water systems must report monthly that they have conducted individual filter turbidity monitoring. Systems are required to report actual IFE measurements only if they have exceeded one of the IFE turbidity triggers. Systems that would apply successfully for the 0. 5 log *Cryptosporidium* IFE removal credit for LT2ESWTR compliance would not, by definition, be systems that were required to report IFE measurements under the earlier regulations. A system must, therefore, submit additional information about IFE turbidity measurements in order to receive the 0.5 log credit.

## 7.4 Process Control Techniques

To meet the lower finished water turbidity requirements, systems will need a high level of process control from the source water intake to the filters. The *Guidance Manual for Compliance with the IESWTR: Turbidity Provisions* (U.S. EPA 1999) discusses many design and operational aspects water systems should consider for achieving low effluent turbidity. Chapter 4 of that manual provides design and operational modifications systems can use to optimize their process for compliance with the LT2ESWTR toolbox requirements. This chapter of the Toolbox Guidance Manual builds on that information, by highlighting those modifications or operational aspects that provide the tightened process control needed to meet the turbidity requirements for this toolbox option. To meet the lower finished water turbidity requirements of the CFE or IFE performance standards, systems will need consistent process performance and the ability to maintain the high filtered water quality under sub-optimal conditions and changing water quality.

> The IESWTR guidance manuals are available on EPA's website at:
>
> http://www.epa.gov/OGWDW/mdbp/mdbptg.html.

Design and operational factors are not the only considerations for maintaining the high filtered water quality standards; all areas of a water system must be dedicated towards the process optimization goal, including administration and maintenance. This toolbox option will require continuing effort and commitment from management and operations staff. Exhibit 7.3 lists several factors in the areas of administration, design, operation, and maintenance that may

limit a system's ability to continually meet the LT2ESWTR lower finished water turbidity requirements. This table demonstrates the importance of considering the capabilities of the entire water system. This table was adapted from the Composite Correction Program (CCP), an EPA program for optimizing water treatment plant performance (discussed in section 7.5.4.2).

## Exhibit 7.3 Performance Limiting Factors (Adapted from the Composite Correction Program)

| | |
|---|---|
| **ADMINISTRATION** | |
| **Plant Administrators** | |
| Policies | Do existing policies or the lack of policies discourage staff members from making required operation, maintenance, and management decision to support plant performance and reliability? |
| Familiarity with Plant Needs | Do administrators lack first-hand knowledge of plant needs? |
| Supervision | Do management styles, organizational capabilities, budgeting skills, or communication practices at any management level adversely impact the plant to the extent that performance is affected? |
| Planning | Does the lack of long range planning for facility replacement or alternative source water quantity or quality adversely impact performance? |
| Complacency | Does the presence of consistent, high quality source water result in complacency within the water utility? |
| Reliability | Do inadequate facilities or equipment, or the depth of staff capability, present a potential weak link within the water utility to achieve and sustain optimized performance? |
| Source Water Protection | Does the water utility lack an active source water protection program? |
| **Plant Staff** | |
| Number | Does a limited number of staff have a detrimental effect on plant operations or maintenance? |
| Plant Coverage | Does the lack of plant coverage result in inadequate time to complete necessary operational activities? (Note: This factor could have significant impact if no alarm/shutdown capability exists - see design factors). |
| Personnel Turnover | Does high personnel turnover cause operation and maintenance problems that affect process performance or reliability? |
| Compensation | Does a low pay scale or benefit package discourage more highly qualified persons from applying for operator positions or cause operators to leave after they are trained? |
| Work Environment | Does a poor work environment create a condition for “sloppy work habits” and lower operator morale? |
| Certification | Does the lack of certified personnel result in poor O&M decisions? |
| **Financial** | |
| Operating Ratio | Does the utility have inadequate revenues to cover operation, maintenance, and replacement of necessary equipment (i.e., operating ratio less than 1.0)? |
| Coverage Ratio | Does the utility have inadequate net operating profit to cover debt service requirements (i.e., coverage ratio less than 1.25)? |
| Reserves | Does the utility have inadequate reserves to cover unexpected expenses or future facility replacement? |
| **DESIGN** | |
| **Source Water Quality** | |
| Microbial Contamination | Does the presence of microbial contamination sources in close proximity to the water treatment plant intake impact the plant’s ability to produce an adequate treatment barrier? |

| | |
|---|---|
| **Unit Process Adequacy** | |
| Intake Structure | Does the design of the intake structure result in excessive clogging of screens, build-up of silt, or passage of material that affects plant equipment? |
| Presedimentation Basin | Does the design of an existing presedimentation basin or the lack of a presedimentation basin contribute to degraded plant performance? |
| Raw Water Pumping | Does the use of constant speed pumps cause undesirable hydraulic loading on downstream unit processes? |
| Flow Measurement | Does the lack of flow measurement devices or their accuracy limit plant control or impact process control adjustments? |
| Chemical Storage and Feed Facilities | Do inadequate chemical storage and feed facilities limit process needs in a plant? |
| Flash Mix | Does an inadequate mixing result in excessive chemical use or insufficient coagulation to the extent that it impacts plant performance? |
| Flocculation | Does a lack of flocculation time, inadequate equipment, or lack of multiple flocculation stages result in poor floc formation and degrade plant performance? |
| Sedimentation | Does the sedimentation basin configuration or equipment cause inadequate solids removal that negatively impact filter performance? |
| Filtration | Do filter or filter media characteristics limit the filtration process performance? |
| Disinfection | Do the disinfection facilities have limitations, such as inadequate detention time, improper mixing, feed rates, proportional feeds, or baffling, that contribute to poor disinfection? |
| Sludge/Backwash Water Treatment and Disposal | Do inadequate sludge or backwash water treatment facilities negatively influence plant performance? |
| **Plant Operability** | |
| Process Flexibility | Does the lack of flexibility to feed chemicals at desired process locations or the lack of flexibility to operate equipment or processes in an optimized mode limit the plant's ability to achieve desired performance goals? |
| Process Controllability | Do existing process controls or lack of specific controls limit the adjustment and control of a process over the desired operating range? |
| Process Instrumentation /Automation | Does the lack of process instrumentation or automation cause excessive operator time for process control and monitoring? |
| Standby Units | Does the lack of standby units for key equipment cause degraded process performance during breakdown or during necessary preventive maintenance activities? |
| Flow Proportioning | Does inadequate flow splitting to parallel process units cause individual unit overloads that degrade process performance? |
| Alarm Systems | Does the absence or inadequacy of an alarm system for critical equipment or processes cause degraded process performance? |
| Alternate Power Source | Does the absence of an alternative power source cause reliability problems leading to degraded plant performance? |
| Laboratory Space and Equipment | Does the absence of an adequately equipped laboratory limit plant performance? |
| Sample Taps | Does the lack of sample taps on process flow streams prevent needed information from being obtained to optimized performance? |

| | |
|---|---|
| **OPERATION** | |
| **Testing** | |
| Process Control Testing | Does the absence or wrong type of process control testing cause improper operational control decisions to be made? |
| Representative Sampling | Do monitoring results inaccurately represent plant performance or are samples collected improperly? |
| **Process Control** | |
| Time on the Job | Does staff's short time on the job and associated unfamiliarity with process control and plant needs result in inadequate or improper control adjustments? |
| Water Treatment Understanding | Does the operator's lack of basic water treatment understanding contribute to improper operational decisions and poor plant performance or reliability? |
| Application of Concepts and Testing to Process Control | Is the staff deficient in the application of their knowledge of water treatment and interpretation of process control testing such that improper process control adjustments are made? |
| **Operational Resources** | |
| Training Program | Does inadequate training result in improper process control decisions by plant staff? |
| Technical Guidance | Does inappropriate information received from a technical resource (e.g., design engineer, equipment representative, regulator, peer) cause improper decision or priorities to be implemented? |
| Operational Guidelines/Procedures | Does the lack of plant-specific operating guidelines and procedures result in inconsistent operational decision that impact performance? |
| **MAINTENANCE** | |
| **Maintenance Program** | |
| Preventive | Does the absence or lack of an effective preventive maintenance program cause unnecessary equipment failures or excessive downtime that results in plant performance or reliability problems? |
| Corrective | Does the lack of corrective maintenance procedures affect the completion of emergency equipment maintenance? |
| Housekeeping | Does a lack of good housekeeping procedures detract from the professional image of the water treatment plant? |
| **Maintenance Resources** | |
| Materials and Equipment | Does the lack of necessary materials and tools delay the response time to correct plant equipment problems? |
| Skills or Contract Services | Do plant maintenance staff have inadequate skills to correct equipment problems or do the maintenance staff have limited access to contact maintenance services? |

### 7.4.1 Chemical Feed

There are two main considerations for the chemical application of a coagulation and flocculation treatment process:

- Are the chemicals and their dose optimum for the treatment process?
- Are they properly mixed or dispersed at the right point in the system?

#### 7.4.1.1 Type of Chemical and Dose

Optimizing the coagulation and flocculation for the range of water quality and demand experienced by the plant is a key factor in improving the overall treatment performance and ensuring process control. One method commonly used to evaluate the type and dose of coagulant and other chemical additives is the jar test (AWWA 2000a).

To provide the process control necessary for producing consistently low filter water turbidity, systems should establish SOPs for changing chemical additions when raw water quality changes significantly. The SOPs should list the appropriate chemicals to be added and the dose according to specified raw water conditions. Jar tests or other chemical evaluations should be conducted with raw water samples representing conditions from high water quality to the worst-case scenario and should reasonably represent the treatment process.

#### 7.4.1.2 Mixing

Adding coagulants at the proper location and providing the right amount of mixing is critical to the coagulation and flocculation processes.

- Metal salts such as alum and ferric chloride should be added at the point of highest mixing.
- Low weight polymers can be added with the metal salts or at a second stage mixing process.
- High weight polymers should be added at a point of gentle mixing.

The coagulation process occurs rapidly; therefore, it is important that the coagulant is well-dispersed and distributed across the width of the flow stream at the point of addition. Flash mixers are necessary for coagulants requiring instantaneous mixing. Systems with mechanical mixers for these types of coagulants should consider changing to a design that provides more uniform dispersion as studies have indicated that mechanical mixers experience short circuiting and frequent maintenance requirements (Kawamura 2000). Kawamura rated several flash mixer designs according to (in order of importance) effectiveness, reliability, minimal maintenance, and cost:

1) Diffusion mixing by pressured water jets.
2) In-line static mixing.
3) In-line mechanical mixing.
4) Hydraulic mixing.

5) Mechanical flash mixing.

6) Diffusion by pipe grid.

The mixing speed should be adjustable and changed with flow and raw water conditions as necessary. Cold water is more viscous and may require a higher mixing energy. Highly turbid or colored water may also require more mixing power to properly disperse the coagulant. For flash mixing, Kawamura (2000) recommends $G \times t$ values of 300 to 1600, where $G$ is the mixing energy (expressed in seconds-1) and $t$ is time (seconds).

#### 7.4.1.3 Streaming Current Detectors and Zeta Potential Monitors

The coagulation process should be monitored continuously, with real time output. Streaming current detectors (SCDs) can provide on-line coagulation control, by measuring the net surface charge of the particle and ionic species in a sample of water. Through jar testing or other coagulant studies, the charge measurement is correlated to the optimal coagulation conditions. The SCDs are typically located directly after coagulant addition to allow the operator time to adjust the dose of the coagulant before filtration. This quick response can prevent process upsets due to fluctuations in influent water quality. Zeta potential monitors also indicate particle surface charge and can be used in the same manner as SCDs.

Source waters high in iron or manganese concentrations and the use of treatment chemicals with iron salts or potassium permanganate can extensively increase maintenance requirements (AWWA 2000a). Additionally, use of powdered activated carbon can increase maintenance requirements. AWWA recommends comparing SCD and zeta potential monitoring results to jar tests on a regular basis (AWWA 2000a).

### 7.4.2 Flocculation

The purpose of the flocculation process is to aggregate the particles into larger groups of particles or "flocs" that will settle in the subsequent sedimentation process. Through gentle and prolonged agitation, the suspended particles collide with each other and form flocs. The mixing must be thorough enough to provide opportunities for the particles to collide but also gentle enough to prevent the flocculated particles from breaking apart. It is likely, however, that some floc breakup will occur. As aggregates grow in size, they are more likely to break up due to the shearing forces in the mixing chamber. In this situation the aggregation and breakup can occur simultaneously leading to a steady-state distribution of floc sizes.

The key factors of an effective flocculation process include: adequate mixing, low floc breakup, and plug flow conditions. The following guidance can help to achieve these conditions:

- Tapered mixing is most appropriate with variable G values ranging from 70 $sec^{-1}$ to 15 $sec^{-1}$.

- If flow is split between two flocculators, they should be mixing at the same speed. Coagulant dosages are most likely optimized to one speed.

- Basin inlet and outlet conditions should prevent floc breakup.

- Baffling should be adequate to provide plug flow conditions.

### 7.4.3 Sedimentation

The purpose of the sedimentation process is to enhance filtration by removing the flocculated particles. As with other unit processes, the sedimentation process should be optimized and provide a consistent settled water quality. The key factors of a good settling process include:

- Minimization of short circuiting.

- Sludge removal equipment should not re-suspend particles or produce currents in the water.

- Surface loading rate, or overflow rate, needs to provide enough settling time. If flocculated particles are not settling, it could be a function of particle density or the surface loading rate.

- Continuous or frequent turbidity monitoring of settled water.

To provide consistent, well-clarified water from the sedimentation basin, the operating parameters of the sedimentation basin may need to be adjusted with significant fluctuations in raw water quality. For example, if a runoff event causes a spike in turbidity, the particles may need more time to settle, and by decreasing the flow through the basin it is possible to achieve the desired level of clarification. Exhibit 7.4 lists sedimentation basin effluent turbidity goals for several state and industry optimization programs, such as the Area-Wide Optimization Program (AWOP). This is a multi-state effort whose goal is to help conventional surface water treatment plants optimize their existing particle removal and disinfection capabilities. For information on AWOP, including state contacts, please visit this site: http://www.asdwa.org/index.cfm?fuseaction=Page.viewPage&pageId=481&parentID=473&nodeID=1. Operators need knowledge and authority to modify the coagulation and flocculation processes or reduce the flow to the plant when settled water quality goals are not being met. For long-term process control, tracking seasonal raw water quality changes and their impacts on the

settling process can provide valuable information for optimizing the overall sedimentation process.

### Exhibit 7.4 Effluent Turbidity Goals for the Sedimentation Process

| Optimization Program | Sedimentation Basin or Clarifier Effluent Turbidity Goal |
|---|---|
| California - *Cryptosporidium* Action Plan | 1 to 2 NTU |
| Texas | < 2 NTU |
| Partnership for Safe Water / EPA Composite Correction Program (CCP) | 1 NTU for raw water conditions of < 10 NTU |
| | 2 NTU for raw water conditions of > 10 NTU |

Note: For information on the Partnership for Safe Water, please visit this site: http://www.awwa.org/Resources/PartnershipMain.cfm?ItemNumber=51227&navItemNumber=51231.

The sludge blanket level is also an important factor for optimum settling conditions. A water system should have SOPs for sludge draw-off that include routine checks of the sludge pumping lines. Sludge pumping lines can plug, causing disruption of the sludge blanket and consequently disrupting the settling process.

## 7.4.4 Filtration

Filtration is the last step in the particle removal process. Although filter performance is a function of the coagulation, flocculation, and sedimentation processes, proper filter operation is needed to provide the high quality finished water required for this toolbox option. The following factors should be considered when optimizing or evaluating filtration performance.

### 7.4.4.1 Flow Split

Systems should evaluate the flow distribution to the filters to ensure there is an even load across all filters under the range of expected operating conditions (e.g., filter out of service, backwash).

### 7.4.4.2 Filter Beds

The filters should be operated with a design capacity that considers at least one filter as a reserve. The reserve filter is put on-line to maintain flow stability to the filters; if this is not possible, flow to the filters should be reduced. This will allow consistent flow when one filter is backwashed or taken out of service for maintenance.

Media loss or disturbance can lead to particles passing through the filters. The filter should be inspected on a regular basis to detect changes in the media. Media should be inspected to ensure depths of media are proper, the media are evenly distributed, and the size distribution

of the media are still to specifications. Media samples can be taken with a coring device or by excavation for the inspection. If media are lost or damaged, they should be replaced.

#### 7.4.4.3 Backwashing

Backwashing is an integral part of the filtration process. Two important operating parameters for backwashing are the backwash flow rate and frequency of cycles. Other factors relating to backwash that affect filter effluent quality are hydraulic surges and filter start-up or "ripening."

*Flow rate*

Systems should determine the appropriate flow that will clean the filter and prevent mudball formation, but will not upset the filter media and subject the underdrain to sudden momentary pressure increases. Typical flow rates are 15 to 20 gpm/ft$^2$ which result in 15 to 30 percent bed expansion.

*Frequency*

Although the filter effluent turbidity is the indicator for pathogen control and the determining factor for compliance, other operating parameters should be used to determine when backwash is needed. Emelko et al. (2000) performed filtration studies where pathogen breakthrough occurred towards the end of the filter cycle before an increase in turbidity was detected. Their studies emphasize the need to evaluate and optimize backwashing cycles with respect to filter effluent water quality. Most systems use filtration time, headloss, effluent turbidity, or effluent particle counts to indicate when backwashing is needed. For improved process control, it may be beneficial to use all indicators.

Systems with multiple filters also should evaluate the hydraulic surges resulting from backwashing. The timing of individual filter backwash cycles should be considered with respect to the other filters, particularly adjacent filters. Consider the following two examples:

- If a large system with 50 filters backwashed 10 filters at the same time, this would cause a 20 percent increase in flow to the other filters. In this situation, the system could backwash fewer filters at one time or reduce the flow to the filters to avoid the filter overload.

- When one filter is backwashed, a hydraulic surge can be experienced by an adjacent filter.

*Improving filter effluent during start-up*

It is very important for systems to conduct a full evaluation of their backwashing process and operational variations to optimize the process. At the process optimization level, systems should minimize turbidity spikes in the filter effluent resulting from the backwashing process—it only takes a few high turbidity readings to cause non-compliance. The following operational practices may provide improved filter effluent during start-up:

- Ramping the backwash rate down in increments to allow better media gradation.
- Resting a filter after backwash for several minutes or up to several hours before putting the filter in service.
- Adding a polymer to the backwash water.
- Slowly increasing the hydraulic load on the filter as it is brought back on line.

#### 7.4.4.4 Filter to Waste

During the beginning of a filter cycle the filter is "ripening" and the effluent turbidity is usually higher. To avoid sending this poorer quality water to the CFE stream, the filter effluent produced during the first few minutes of a filter cycle can be sent to waste (filter to waste) or recycled to the head of the plant. Some systems filter to waste or recycle until the filter effluent reaches the desired level of turbidity. Practicing filter to waste produces an overall higher quality water and may be necessary to maintain a CFE or IFE below 0.15 NTU.

#### 7.4.4.5 Backwash Recycle

Plants that recycle the backwash water to the head of the plant should evaluate the impacts the backwash stream has on the coagulation, flocculation, and sedimentation processes. For example, the operator should know how the coagulation and flocculation processes need adjusting when there is a change in recycle flow. Ideally, the impacts of the recycle flow on these processes should be minimized.

For systems that recycle, the Filter Backwash Rule (FBR) requires spent filter backwash, thickener supernatant, or liquids from dewatering processes to be returned through all the processes of a system's existing conventional or direct filtration treatment train (40 CFR 141.76(c)). The rule allows for alternative recycle locations with state approval (40 CFR 141.76(c)).

#### 7.4.4.6 Filter Assessments

Filter assessments can provide valuable information for optimizing the performance of a filter. The IESTWR and LT1ESWTR require systems to conduct an individual filter self-assessment if a filter exceeds specified effluent turbidity criteria. However, systems seeking *Cryptosporidium* treatment credit for lower finished water turbidity should also consider conducting filter assessments to evaluate operating parameters and optimize filter performance. Chapter 5 of the IESWTR Turbidity Guidance Manual describes how to conduct an individual filter self- assessment.

### 7.4.5 Hydraulic Control

Proper hydraulic control throughout the treatment process is essential. In the coagulation and sedimentation processes it is important to minimize short circuiting so the majority of the water receives the designed coagulation and sedimentation treatment. Hydraulic surges can cause greater turbulence that may break up flocculating particles and resuspend settling particles. In the subsequent filtration process, hydraulic surges can cause particle breakthrough anytime during the filtration cycle. Systems should look at historical water demand data and other conditions that adversely affect the system's ability to control filter performance (e.g., backwashing, changes in flow). With these data, they should develop operating plans to address the condition and allow control of the filter effluent quality.

## 7.5 Process Management Techniques

### 7.5.1 Standard Operating Procedures (SOPs)

Developing SOPs for all aspects of the operation and maintenance of a water system is essential for running a high quality system. SOPs provide the basis for ensuring that activities are accomplished in a consistent manner. They should be kept as simple as possible in order to ensure that each operator is consistent in carrying out the task at hand. The title of the procedure should be clear, concise, and descriptive of the equipment, process, or activity. SOPs should be developed with input from staff, thus enabling them to understand and implement procedures in compliance with applicable requirements.

### 7.5.2 Prevention and Response Plan for Loss of Chemical Feed

Loss of chemical feed is a common cause of increased turbidity through the treatment processes. Plants should have equipment and SOPs for preventing such occurrences or reacting to them rapidly if they do occur. The following items are necessary to prevent an upset in water quality due to a chemical feed failure.

- SOPs to verify doses with feed response time (lag time) accounted for.
- Redundant feeds.
- Routine maintenance of all chemical feed parts (e.g., pump, feed line).
- Inventory of spare parts available so repairs can be made quickly.
- Pump or feed failure alarms.
- Process monitors detecting chemical feed failure (e.g., streaming current, zeta potential, and pH monitors).

### 7.5.3 Adequate Chemical Storage

Sufficient chemical storage is necessary to ensure continued operation of the plant at proper dosages, including enough to run at higher dosages if an unexpected turbidity spike should occur in the raw water. Care must also be taken, however, to follow manufacturer's suggestions on the useful life of the chemical. Many coagulants will degrade over time and will not perform properly and may even cause increased turbidity if allowed to age too long. Storage tanks should also be designed so that there are no dead spaces where chemicals may accumulate with much longer residence times than the hydraulic residence time of the tank.

### 7.5.4 Voluntary Programs

EPA, state regulatory agencies, AWWA, and other drinking water organizations have established voluntary programs for systems to ensure the delivery of safe water to their customers. These programs often focus on optimizing the treatment process and identifying the limiting factors of performance. Consequently, they are excellent aids for systems considering this toolbox option. This section discusses two programs, the Partnership for Safe Water and the CCP. (The CCP is also promoted as part of the Partnership for Safe Water).

#### 7.5.4.1 Partnership for Safe Water

The Partnership for Safe Water is a voluntary cooperative effort between EPA, AWWA, and surface water systems. The goal of the program is to "provide a new measure of safety to millions of Americans by implementing prevention programs where legislation or regulation does not exist. The preventive measures are based around optimizing treatment plant performance, and thus increasing protection against microbial contamination in America's drinking water supply." (http://www.awwa.org/partner/partner1.htm).

For further information about the Partnership for Safe Water and how to join, see AWWA's website: http://www.awwa.org/Resources/PartnershipMain.cfm?ItemNumber=51227&navItemNumber=51231.

Water systems that participate in the program go through four phases:

Phase I: Commitment – operators and management indicate their willingness to complete the program through phase III.

Phase II: Data Collection and Analysis – the water system must collect one year of raw, settled, and filter effluent turbidity data and submit to AWWA for analysis.

Phase III: Self Assessment – allows the system to examine the capabilities of the existing plant's operation and administration and identify factors that limit performance.

Phase IV: Procedures and Applications Package – systems demonstrate they addressed areas of limited performance and produce high quality water as measured by filter effluent turbidity.

Through the efforts of monitoring, data analysis, and evaluating the capabilities of unit processes, significant improvements in water quality can be achieved. In the Partnership's 2001 Annual report, AWWA reported an increase from 20 percent to 32 percent of plants completing Phase II with finished water turbidity levels less than 0.1 NTU (based on 95th percentiles). At the beginning of Phase III, approximately 51 percent of plants reported 95th percentile turbidity less than 0.1 NTU, and after completing Phase III approximately 70 percent of plants achieved less than 0.1 NTU.

#### 7.5.4.2 Composite Correction Program (CCP)

The CCP was developed in 1988 to optimize surface water treatment plant performance with respect to protection from microbial pathogens. The program consists of two parts, the comprehensive performance evaluation (CPE) and comprehensive technical assistance (CTA). The CPE is a thorough review and analysis of a facility's design capabilities and associated administrative, operational, and maintenance practices as they relate to achieving optimum

performance from the facility. It can be conducted by the system or by a third party over a period of roughly 3 to 4 days. The CTA builds on the results of the CPE by addressing the combination of factors that limit a facility's performance. If conducted by a third party, it should be implemented by a third party who is in a position to pursue corrective actions in all areas, including politically sensitive, administrative, or operational limitations.

EPA published a handbook, *Optimizing Water Treatment Plant Performance Using the* Composite *Correction Program* (1998), that fully describes the goals, methods, and procedures of the CCP. To obtain a copy, call the EPA Safe Drinking Water Hotline at 1-800-426-4791.

## 7.6 References

American Water Works Association. 2000a. *Operational Control of Coagulation and Filtration Processes*, 2nd Edition. American Water Works Association.

American Water Works Association. 2000. *Water Quality and Treatment 5th Edition*. McGraw Hill.

Kawamura, Susumu. 2000. Integrated Design and Operation of Water Treatment Facilities. John Wiley & Sons, Inc.

U.S. EPA. 1998. *Optimizing Water Treatment Plant Performance Using the Composite Correction Program*. Office of Water and Office of Research and Development. EPA 625/6-91/027.

# 8. Bag and Cartridge Filters

## 8.1 Introduction

Under the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR), bag and cartridge filters are defined as pressure driven separation devices that remove particles larger than 1 micrometer (μm) using an engineered porous filtration media. Bag filters are typically constructed of non-rigid, fabric filtration media housed in a pressure vessel in which the direction of flow is from the inside of the bag to the outside. Cartridge filters are typically constructed as rigid or semi-rigid self supporting filter elements housed in pressure vessels in which flow is from the outside of the cartridge to the inside (40 CFR 141.2).

A pressure vessel may contain either single or multiple filters in a series or in parallel. As the water flows through a bag or cartridge filter, particles collect on the filter and the difference in pressure from the inlet to the outlet, termed "pressure drop," increases. Once a "terminal pressure drop" is reached, the bag or cartridge filter is replaced.

Typically, bag and cartridge filters are used by small systems for protozoa or other particle removal. The pore sizes in the filter bags and cartridges designed for protozoa removal are small enough to remove protozoan cysts and oocysts but generally large enough that viruses, bacteria, and fine colloidal clays could pass through.

This chapter provides background information on the treatment performance, design, and operation of bag and cartridge filters, with emphasis on those issues that a system should consider for integrating bag or cartridge filters into its treatment process to comply with the LT2ESWTR. This chapter is organized as follows:

8.2 LT2ESWTR Compliance Requirements - describes criteria and reporting requirements that systems must meet to receive *Cryptosporidium* treatment credit.

8.3 Toolbox Selection Considerations - describes the advantages and disadvantages of integrating a bag and cartridge filtration process for compliance with the LT2ESWTR.

8.4 Challenge Testing - describes the challenge testing that a bag or cartridge filter must pass to be awarded *Cryptosporidium* treatment credit for the LT2ESWTR.

8.5 Design Considerations - discusses influent water quality, size of filter system and redundancy, layout features, filter cycling, pressure monitoring, valves and appurtenances, air entrapment, and National Science Foundation (NSF) certification.

8.6 Operational Issues - discusses pressure drop across the filter, and monitoring to assess performance and indicate possible process upsets with the bag or cartridge filter or other upstream processes.

## 8.2 LT2ESWTR Compliance Requirements

### 8.2.1 Credits

Bag and cartridge filtration processes that meet the EPA definition and demonstrate *Cryptosporidium* removal through challenge testing may receive the following *Cryptosporidium* removal credit for the LT2ESWTR (40 CFR 141.719(a)):

- Up to 2.0-log removal for individual bag or cartridge filters showing a minimum of 3.0-log removal in challenge testing.

- Up to 2.5-log removal for bag or cartridge filters in series showing a minimum of 3.0-log removal in challenge testing.

Challenge testing must be conducted according to the LT2ESWTR requirements outlined in section 8.4 of this chapter. A 1-log factor of safety for a single filter and 0.5-log factor of safety for multiple filters in series is applied to the allowable removal credit over that demonstrated by challenge testing because bag and cartridge filters cannot have their integrity directly tested; hence, there are no means of verifying their removal efficiency during routine use.

Recently, some cartridge filtration devices have been developed for drinking water treatment using membrane media, which can be direct integrity tested. These membrane cartridge filters (MCFs) could be considered a membrane filtration process for the purpose of compliance with the LT2ESWTR treatment requirements for *Cryptosporidium* (i.e., the MCF process would be eligible for the same credit, and subject to the same requirements, as a membrane filtration process). A direct integrity test is a physical test applied to a membrane unit to identify and isolate integrity breaches (i.e., one or more leaks that could result in contamination of the filtrate). Manufacturers can provide information on direct integrity testing and whether it is feasible with their products. Refer to the *EPA Membrane Filtration Guidance Manual* (U.S. EPA 2005) for direct integrity testing and other membrane filtration requirements.

### 8.2.2 Reporting Requirements

All reporting requirements for the Surface Water Treatment Rule (SWTR), Interim Enhanced Surface Water Treatment Rule (IESWTR), and Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) are still applicable; the LT2ESWTR does not modify or replace any previous rule requirements. The location of filter effluent turbidity monitoring for compliance with the IESWTR and LT1ESWTR does not change with the installation of a bag or cartridge filter as a secondary filtration process. That is, a system would still monitor filter effluent turbidity after the primary filters for compliance with the IESWTR and LT1ESWTR.

When bag and/or cartridge filters are used to comply with treatment requirements, the LT2ESWTR requires systems to submit an initial report that demonstrates the following (40 CFR 141.721(f)):

- Process meets the definition of a bag or cartridge filter.

- Removal efficiency from challenge testing (described in section 8.4). The removal demonstrated must be 1.0-log greater than the credit awarded for a single and 0.5-log greater than the credit awarded for multiple filters in series.

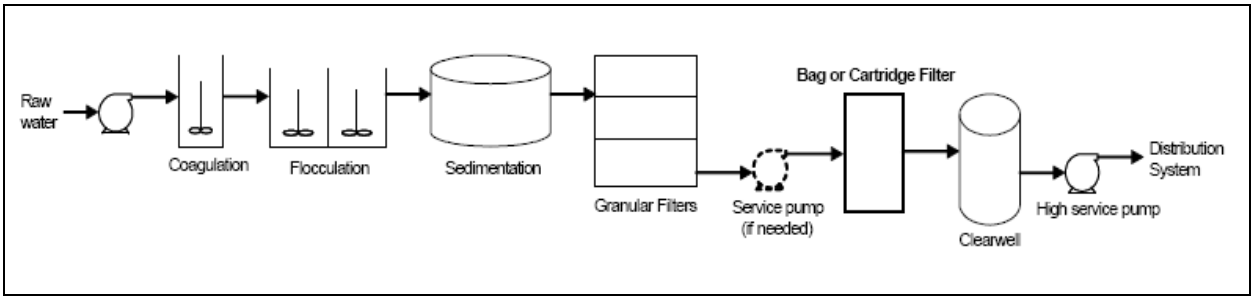
This initial report must be submitted by April 1, 2012 for systems serving more than 100,000, October 1, 2012 for systems serving between 50,000 and 99,999, October 1, 2013 for systems serving between 10,000 and 49,999 and October 1, 2014 for small systems serving fewer than 10,000 people.

For routine compliance reporting, systems must verify each month that 100 percent of plant flow was treated by the bag or cartridge filter (40 CFR 141.721(f)). One possible approach states may elect to use for flow verification is to have operators certify each month that all flow was treated by the filter. States may require additional reporting at their discretion. Section 8.6 provides recommendations for filter effluent and process monitoring.

### 8.2.3 Integration into a Treatment Process Train

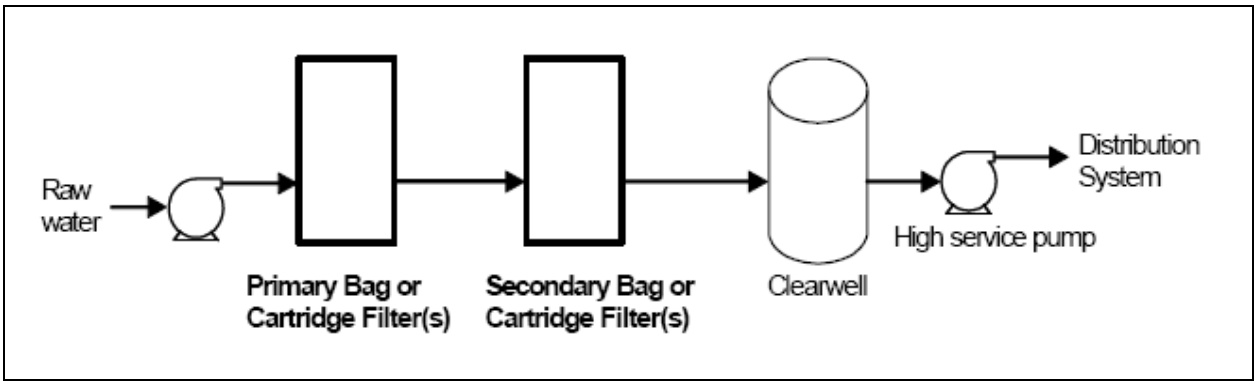
To achieve compliance with the IESWTR and LT1ESWTR, all plants (except those meeting the filter avoidance criteria in 40 CFR 141.71) must have a filtration process approved by the state. Approved processes receive 2-log *Cryptosporidium* removal credit under the IESWTR and LT1ESWTR. For compliance with additional treatment requirements for the LT2ESWTR, bag and cartridge filters should be added as an additional filtration process following the existing primary filtration (see Exhibits 8.1 and 8.2). The bag and cartridge filters provide additional removal of the smaller contaminants and any contaminants that break through the granular media filters during the end of a run cycle or process upsets.

**Exhibit 8.1 Schematic of Treatment Process with Bag/Cartridge Filters**



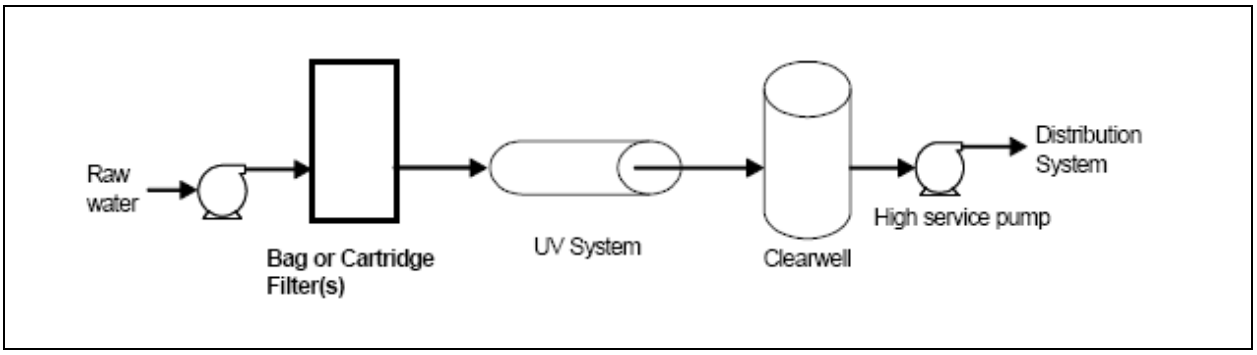

For those systems using a bag or cartridge filter process to meet LT1ESWTR requirements, thus serving as the primary filtration process, it may be possible to configure the bag or cartridge filters in a series (see Exhibit 8.2).

### Exhibit 8.2 Bag/Cartridge Filters in Series




Another possible configuration is a bag or cartridge filter followed by an ultraviolet light (UV) system (see Exhibit 8.3). This configuration would allow removal of particles and microbial pathogens as well as inactivation of *Cryptosporidium*, *Giardia*, and viruses. In this case, the bag or cartridge filter would serve as the primary filter and thus, be subject to SWTR, IESWTR, and LT1ESWTR requirements, while the UV system would be subject to the LT2ESWTR requirements. Refer to EPA's *UV Disinfection Guidance Manual* (U.S. EPA 2006) for information regarding UV systems and associated requirements with LT2ESWTR.

### Exhibit 8.3 Bag/Cartridge Filter with UV System




Factors that should be considered when developing a treatment process scheme include available space, hydraulic profile, and point of disinfection. Space requirements are small for bag and cartridge filter systems, but extra space for maintenance activities should be considered in the planning process. Because a significant headloss is associated with an additional filtration process, systems should consider their hydraulic profile when integrating new filters into an existing process sequence. Although the addition of a new bag filtration process does not necessarily require that the point of primary disinfection be changed, some bag filtration installations chlorinate prior to the bag filtration process to minimize biofilm growth on the bags. However, if systems are considering using a bag or cartridge filter as the primary filter as in Exhibit 8.3, chlorinating prior to filtration will likely cause higher disinfection byproduct formation compared to post-filter chlorination since the filtration process will remove some organic material.

## 8.3 Toolbox Selection Considerations

This section describes the advantages and disadvantages of integrating a bag and cartridge filtration process for compliance with the LT2ESWTR.

### 8.3.1 Advantages

The advantages of bag and cartridge filtration processes include low maintenance requirements, relatively low capital cost, minimal operator training, and low space requirements. The only routine maintenance required is filter replacement when a defined terminal pressure drop or other operating parameter, such as filter age or volume treated, is reached. The operation of these systems is straightforward and requires little technical skill. In addition, the filter materials are relatively inexpensive and the housing system is not complex, resulting in relatively low capital costs.

### 8.3.2 Disadvantages

A disadvantage of bag and cartridge filtration processes is most filters must be replaced instead of regenerated. For larger flows, or water with higher particle loads, frequent filter replacement increases operation and maintenance costs. Additional pumps may be required to provide needed pressure. Also, redundancy should be built into the process design, increasing costs. Bag and cartridge filters can also be subject to clogging by biofilm growth or excess coagulants. Maintaining a residual through the filter is one possible way to prevent biofilm growth. See the *Simultaneous Compliance Guidance Manual* (U.S. EPA 2007) for additional recommendations.

## 8.4 Challenge Testing

Manufacturers commonly rate fabric filters by pore size or pore distribution. However, there is no industry standard for measuring or reporting these characteristics. This lack of standardization causes problems for establishing design criteria to ensure that a given bag or cartridge filter will effectively remove a given percentage of *Cryptosporidium*. Furthermore, an oocyst has different structural characteristics than the markers used to determine pore size; thus, the rate of rejection may differ for an oocyst versus the test markers used to determine pore size or molecular weight cutoff. To compensate for these factors of uncertainty for *Cryptosporidium* removal, the LT2ESWTR requires bag or cartridge filters to be challenge tested to determine removal credit.

Challenge testing is a process in which a known quantity of *Cryptosporidium* oocysts (or an acceptable surrogate) is added to the filter influent and the effluent concentration is measured to determine the removal capabilities of the filter. This testing is product-specific, not site-specific, meaning it does not have to be tested at every water system seeking removal credit. Instead, a manufacturer (or independent third party) would challenge test each of its products in order to obtain a 2.0- or 2.5-log *Cryptosporidium* removal rating. Bag or cartridge filters must be

challenge tested, however, in the same configuration that the system will use, either as individual filters or as a series of filters.

For compliance with the LT2ESWTR, EPA defined a set of test conditions that must be met for an acceptable challenge test. These conditions provide only a framework for the challenge test; states may develop additional testing requirements. The EPA *Membrane Filtration Guidance Manual* (U.S. EPA 2005) contains detailed guidance on developing challenge test protocol and conducting the test for membrane processes that relate to these requirements. Additionally, NSF International, in cooperation with EPA, developed the *Protocol for Equipment Verification Testing for Physical Removal of Microbiological and Particulate Contaminants* (NSF International 2005) with a chapter for testing bag and cartridge filters (Chapter 4). Although the protocol was developed for compliance with the SWTR, some testing principles still apply[1].

Section 8.4.1 describes the test conditions required by the LT2ESWTR (40 CFR 141.719(a)(2)-(8)). Section 8.4.2 shows how to calculate the log removal value (LRV) for challenge testing results. Section 8.4.3 discusses modifications to the filter unit (e.g., change in filter media) occurring after challenge testing that may require additional challenge testing.

### 8.4.1 Testing Conditions

#### 8.4.1.1 Full Scale Filter Testing

Challenge testing must be conducted on full-scale bag or cartridge filters and the associated filter housing or pressure vessel that are identical in material and construction to the filters and housing the system will use for removal of *Cryptosporidium*.

#### 8.4.1.2 Challenge Particulate

Challenge testing must be conducted using *Cryptosporidium* or a surrogate which is removed no more efficiently than *Cryptosporidium*. The microorganism or surrogate used during challenge testing is referred to as the "challenge particulate." The concentration of the challenge particulate must be determined using a method capable of discreetly quantifying the specific organism or surrogate used in the test; gross measurements such as turbidity may not be used (40 CFR 141.719(a)(3)). Key physical characteristics to be considered for identifying an acceptable surrogate include size, shape, and surface charge. Other factors include ease of measurement and cost. Chapter 3 of EPA's *Membrane Filtration Guidance Manual* (U.S. EPA 2005) describes the characteristics of acceptable surrogates and lists potential and inert surrogates for *Cryptosporidium*. Examples of possible microbial surrogates are *P. dimunita* and *S. marcessans*.

[1] Specific sections of the EPA/NSF ETV Protocol that provide guidance for developing and conducting a challenge test for LT2ESWTR include: section 7.0, Characterization of Feed Water; section 11.0, Operating Conditions; section 12.3, Work Plan; section 13.0, Data Management; and section 14.0, QA/QC.

### 8.4.1.3 Test Solution Concentration

In order to demonstrate a removal efficiency of at least 3-log for bag or cartridge filters, it may be necessary to seed the challenge particulate into the test solution. A criticism of this approach is that the seeded levels are orders of magnitude higher than those encountered in natural waters, which could lead to artificially high estimates of removal efficiency. To address this issue, EPA set a limit on the maximum feed concentration applied to a filter during the challenge study. The limit is based on the detection limit of the challenge particulate:

*Equation 8-1*

$$\text{Maximum Feed Concentration} = 1.0 \times 10^4 \times \text{Filtrate Detection Limit}$$

These concentrations allow the demonstration of up to 4.0-log removal for bag filters and cartridge filters during challenge testing if the challenge particulate is removed to the detection limit.

Example 8-1 - Determining maximum allowable feed concentration

If the detection limit of the surrogate test is 2 units/L, then the maximum feed concentration is $1 \times 10^4 \times (2) = 2 \times 10^4$

### 8.4.1.4 Challenge Test Duration

Each filter must be tested for a duration sufficient to reach "terminal pressure drop" (40 CFR 141.719(a)(6)). Terminal pressure drop is a parameter specified by the manufacturer that establishes the end of the useful life of the filter. Continuous challenge particulate feed is not required (i.e., intermittent seeding is permitted). At a minimum, removal efficiency must be determined during three periods over the filtration cycle:

- Within 2 hours of start-up of a new filter.
- When the pressure drop is between 45 and 55 percent of the terminal pressure drop.
- At the end of the run after the pressure drop has reached 100 percent of the terminal pressure drop.

The rule does not specify the number of samples that must be collected during each of the three periods. Because the effluent concentration is often very low and near the detection limit, it may be beneficial to collect more effluent than influent samples to obtain a more accurate removal efficiency.

#### 8.4.1.5 Water Quality of Test Solution

Water quality can have a significant impact on the removal of particulate contaminants, such as *Cryptosporidium*. In general, bag and cartridge filters in water treatment do not experience influent turbidity concentrations much greater than 10 nephelometric turbidity units (NTU). For the application of the LT2ESWTR, they typically will receive filtered water and thus, very low turbidity.

A clean-water challenge test will generally provide the most conservative estimate of removal efficiency. However, since the challenge test must run until terminal head loss is reached, the challenge test solution should contain some solids to cause the head loss build-up across the filter, but not an excessive amount that will cause a rapid build-up. Particulate foulants that may be appropriate to add to the test solution include clay particles (such as bentonite or kaolin) or carbon powder, as long as they are not excessively fine-sized.

The following are recommended for the challenge test solution:

- High quality water with a low to moderate concentration of suspended solids should be used as the challenge solution. Suspended solids concentration should be high enough to achieve a reasonable rate of headloss buildup, but not so high that the headloss builds up too rapidly to conduct the challenges at the various headloss levels.

- No oxidants, disinfectants, or other pretreatment chemicals should be added to the test solution.

- Test water should be characterized with respect to basic water quality parameters, such as pH, turbidity, temperature, and total dissolved solids.

#### 8.4.1.6 Maximum Design Flow Rate

The challenge test must be conducted at the maximum design flow rate for the filter as specified by the manufacturer (40 CFR 141.719(a)(5)).

#### 8.4.1.7 Challenge Particulate Seeding Method

There are two basic approaches to seeding: batch seeding and in-line injection. In batch seeding, all of the challenge particulates are introduced into the entire volume of test solution and mixed to a uniform concentration. Batch seeding requires the entire test solution to be contained in a reservoir and for the reservoir to be well mixed to ensure a uniform concentration of the seeded particles. Generally, batch seeding is used for small scale systems that only require relatively small amounts of feed solution for testing.

In-line injection is the most common seeding approach used in challenge testing, allowing challenge particulates to be introduced into the feed on either a continuous or intermittent basis. While either could be used, intermittent seeding may be preferable to

continuous seeding for conducting the challenge test at the required intervals (i.e., a minimum of beginning, middle, and end-of-run). If intermittent injection is used, equilibrium should be achieved during each seeding event prior to the collection of feed and filtrate samples.

In-line injection delivers the challenge particles from a concentrated stock solution with a known feed concentration. Guidelines and examples for determining challenge test feed concentration and stock solution delivery rates are provided in Chapter 3 of the *Membrane Filtration Guidance Manual* (U.S. EPA 2005).

In-line injection requires additional equipment, such as chemical feed pumps, injection ports, and in-line mixers. These components should be designed to ensure a consistent challenge particulate concentration in the feed. A chemical metering pump that delivers a steady flow is recommended (pumps that create a pulsing action should be avoided). The injection port should introduce the challenge material directly into the bulk feed stream to aid in dispersion. An in-line static mixer should be placed downstream of the injection port, and a feed sample tap should be located approximately ten pipe diameters downstream of the mixer (U.S. EPA 2005).

#### 8.4.1.8 Challenge Test Solution Volume

The volume of the test solution depends on filtrate flow rate, test duration, and hold-up volume of the test system. For intermittent, in-line injection, the seeded test solution volume can be considerably less than that required for batch seeding. Formulas for calculating test solution volume and examples are provided in Chapter 3 of the *Membrane Filtration Guidance Manual* (U.S. EPA 2005).

#### 8.4.1.9 Sampling

An effective sampling program depends on a detailed sampling plan and the use of appropriate sampling methods, locations, and quality assurance/quality control (QA/QC) measures.

Samples can be collected using either grab or composite sampling methods. Grab samples consist of pre-determined amounts of water taken from the feed or filtrate streams, while composite samples are of the entire process stream. Grab sampling is commonly used to determine the concentration of challenge particulates in the feed solution, while grab or composite sampling is used to analyze the filtrate stream. Good sampling practices include flushing samples taps, using clean sample containers, and preventing cross contamination of samples. QA/QC measures include clearly identifying samples, collecting duplicates, and using blanks.

In many cases, it may be advantageous to collect more filtrate samples than feed samples, since the concentration of the challenge microorganism in the filtrate samples is expected to be very low and error of just a few particles could have significant impact on the demonstrated removal efficiency.

Sample port design is an important consideration and should ensure that a representative sample is obtained. Poorly designed ports contain large volumes where stagnation may occur (e.g., large valves and long sample tubes) and pull the sample from the edge of the pipe. A well designed port has a sample quill that extends into the center of the pipe to draw a more representative sample.

Chapter 3 of the *Membrane Filtration Guidance Manual* (U.S. EPA 2005) contains additional information on developing sampling plans and provides schematics of typical sampling apparatuses.

### 8.4.2 Calculating Log Removal (141.719(a)(7)-(9))

Removal efficiency of a filter must be determined from the results of challenge testing and calculated using Equation 8-2.

*Equation 8-2*

$$LRV = Log_{10}(C_f) - Log_{10}(C_P)$$

Where:
LRV = log removal value demonstrated during challenge testing
$C_f$ = feed concentration measured during the challenge test
$C_p$ = filtrate concentration measured during the challenge test

The feed and filtrate concentrations must be expressed in the same units (number of challenge particulate per unit volume). If the challenge particulate is not detected in the filtrate, then the filtrate concentration ($C_P$) must be set equal to the detection limit.

Example 8-2 - Calculating the LRV

Feed Concentration ($C_f$) 20,000 units/L
Filtrate Concentration ($C_P$) 3 units/L

LRV = Log(20,000) - Log(3)
LRV = 4.30 - 0.48 = 3.82

The LT2ESWTR does not specify how the feed and effluent concentration must be determined. A conservative approach would be to use the lowest feed concentration and highest filtrate concentration from each filter run.

A challenge test will likely evaluate multiple filters. An LRV must be calculated for each filter tested. The final log removal efficiency assigned to the filter process tested depends on the number of filters tested:

- If fewer than 20 filters were tested during a challenge study, the overall removal efficiency for the filter product line must be set equal to the lowest LRV observed among the filters.

- If 20 or more filters were tested during challenge testing, the overall removal efficiency for the product line must be set equal to the $10^{th}$ percentile of the LRVs observed during the challenge study. (The percentile is defined by [i/(n+1)] where *i* is the rank of *n* individual data points ordered lowest to highest. If necessary the system may calculate the $10^{th}$ percentile using linear interpolation).

### 8.4.3 Modifications to Filtration Unit after Challenge Testing (141.719(a)(10))

If a previously tested filter is modified in a manner that could change the removal efficiency of the filter product line, challenge testing to demonstrate the removal efficiency of the modified filter must be conducted and submitted to the state. Significant modifications may include, but not limited to:

- Changes to the filtration media (e.g., different fabric, change in the filter manufacturing process).

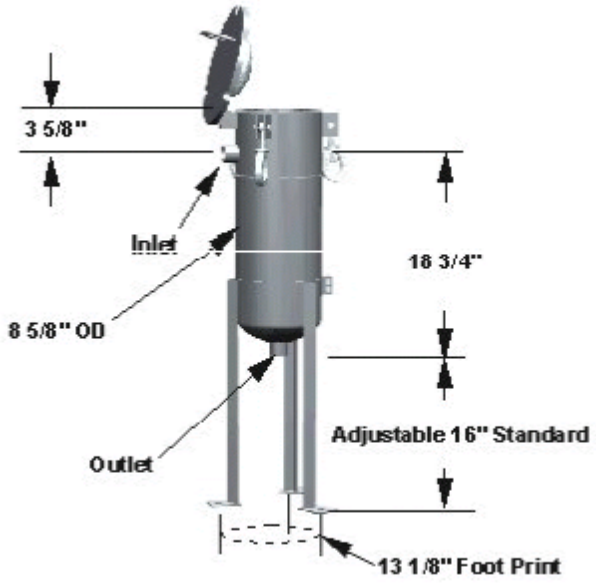
- Changes to the configuration of the filtration media.

- Modifications to the sealing system.

## 8.5 Design Considerations

Bag and cartridge filter systems may contain anywhere from one to over twenty filter units. There is no maximum number of filters a system can include; however, membrane or other filtration processes become more practical for larger flows since bag and cartridge filters are generally replaced instead of backwashed or regenerated. A single filter unit is comprised of the filter media (bag or cartridge), housing, and associated piping and valves. Exhibit 8.4 shows a typical single filter vessel (housing).

## Exhibit 8.4 Single Filter Vessel

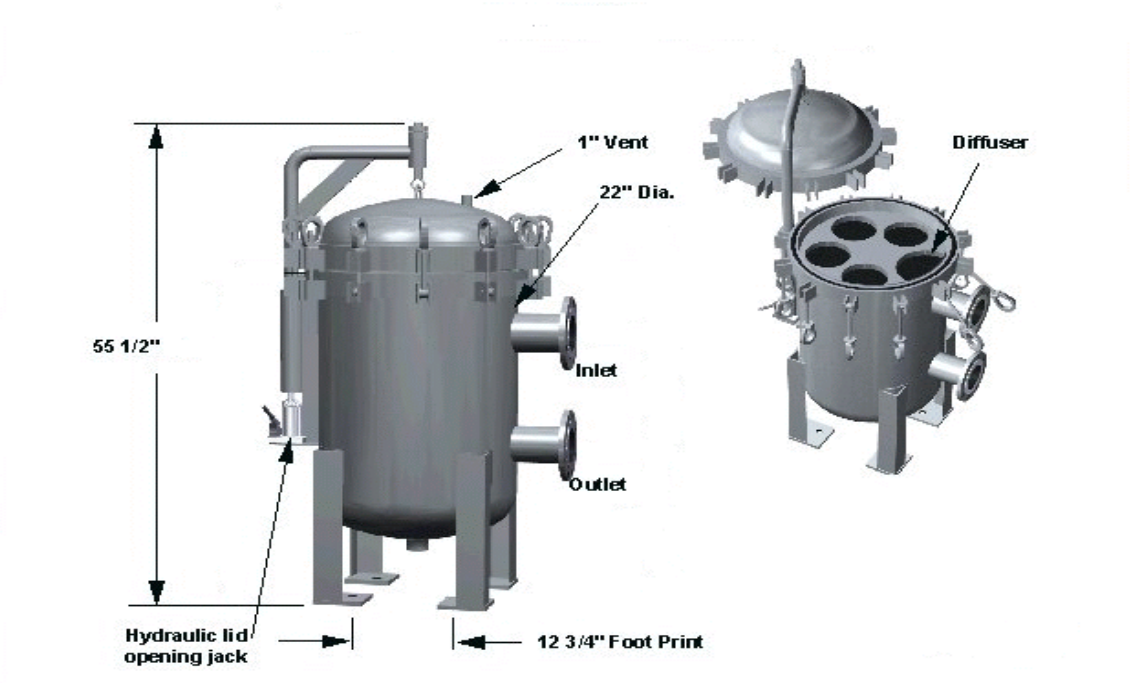



Source: U.F. Strainrite

Systems with multiple filters may be designed as a manifold with connective piping between the individual filters in separate housing or alternatively as multiple filters in a single housing. Exhibits 8.5 and 8.6 show the manifold design and multiple filter vessel design, respectively.

## Exhibit 8.5 Manifold Bag Filter Design

## Exhibit 8.6 Multiple Filter Vessel




Source: U.F. Strainrite

The designs of bag and cartridge filters are not complex; however, there are a couple of key issues that should be taken into consideration. First, the filter units should be designed integrally with their respective housing systems. Poor fittings can cause leaks and premature failure. Manufacturers can provide individual filter units that can be retrofitted into the existing process or complete filter houses that are skid mounted. *It is important to adhere to the manufacturer's instructions on filter installation*.

Second, the overall water treatment process design should minimize sudden changes in pressures applied to the bag or cartridge filters. Each time the flow to the filter is interrupted and then restarted, a sudden increase in pressure can occur across the filter unit unless steps are taken to allow for gradual pressure ramp-up. The particle load in the filter effluent often increases when the filter cycle begins. A study by McMeen (2001) reported that the increase in particle load could be occurring due to the seal at the top of the filter failing when the pressure suddenly increases. Bag filters are especially susceptible to cycling because these pressure fluctuations also increase wear on the fabric and seams, causing premature failure. Section 8.5.4 provides recommendations for reducing filter cycling.

### 8.5.1 Water Quality

As previously described, systems seeking compliance with the LT2ESWTR will most likely integrate a bag or cartridge filter process after the primary filtration process. As a result, influent water quality, with respect to high particulate levels, should not be an issue. However, for systems with existing processes that use coagulants, the presence of residual coagulant in the primary filter effluent may clog the pores of a bag or cartridge filter. Although this will not impair removal efficiency for *Cryptosporidium*, it will shorten the time until the terminal pressure drop is reached, thus reducing filter life.

Another water quality issue is the potential for biofilm growth on the bag or cartridge filter media. Systems can add a disinfectant prior to the bag or cartridge filters to prevent biofilm growth. (The filters must be compatible with the disinfectant.)

### 8.5.2 Size of Filter System and Redundancy

Systems should be adequately designed to handle maximum day or maximum instantaneous flow, depending on the existing treatment process design. Prolonged operation at maximum flow velocity clogs the filter media faster than operating at lower flow velocities. The total volume throughout is greater when operating at a flow velocity lower than maximum flow velocity rated for the filter.

A minimum of two bag or cartridge filter housings should be provided to ensure continuous water treatment in the event of failure in the filter operation and to allow for filter maintenance and replacement. For water systems that do not require continuous operation, a state may approve a single filter housing operation. Redundancy in pumps is also recommended to ensure continuous operation.

### 8.5.3 Design Layout

Design layout features that should be considered for most designs are as follows:

- Piping should be designed to allow isolation of the individual filter units or vessels for maintenance and filter replacement.
- Common inlet and outlet headers for the filter units.
- Sufficient available head to meet the terminal pressure drop and system demand.

### 8.5.4 Filter Cycling

Filter cycling refers to the starting and stopping of the pump or filter operation. This can be problematic with bag filter processes (cartridge filters are not known to have this problem) in which water is pumped directly from the source to the filter, and then out to the distribution system. In these situations, the filters operate on demand, similar to wells for small systems, and the sudden increase in pressure across the filter causes premature wear and filter failure. For LT2ESWTR compliance, systems with bag filters in a series or followed by UV disinfection should consider the following recommendations for controlling the flow into the filter process to minimize filter cycling.

- Lengthen the filter runs by reducing the flow as much as possible through the filter.
- Install or divert the flow to a storage facility (e.g., pressure tank, clearwell) after the bag filtration process. The stored water can supply the frequent surges in demand and thus reduce the bag or cartridge filter cycling.

During filter start-up and other hydraulic surges, bag and cartridge filters often experience an increase in filter effluent turbidity. Systems should consider the following options to improve filtered water quality.

- Design for filter to waste capability. EPA strongly recommends **filtering to waste for the first few minutes** of the filter cycle.
- Install a slow opening and closing valve ahead of the filter to reduce flow surges.

### 8.5.5 Pressure Monitoring, Valves, and Appurtenances

As previously mentioned, once the terminal pressure drop has been reached, the filter should be replaced. At a minimum, pressure gauges should be located before and after the bag or cartridge filter system and should be monitored at least daily. A valve or flow restricter should be installed on the inlet header pipe of the filters to maintain flows below the maximum operating flow for the filters.

### 8.5.6 Air Entrapment

An automatic air release valve should be installed on the top of the filter housing to release any air trapped in the filter. These valves should be checked routinely and properly maintained.

### 8.5.7 NSF Certification

All components used in the drinking water treatment process should be evaluated for contaminant leaching and certified under ANSI/NSF Standard 61.

## 8.6 Operational Issues

This section discusses two key issues associated with operating bag or cartridge filters, pressure changes and water quality monitoring.

### 8.6.1 Pressure Drop (Inlet/Outlet Pressures)

The pressure drop across the filter directly relates to the amount of particle build-up on the filter material and to the time when the filter should be replaced. Typical pressure drops across a clean filter are 1 to 2 psig (pounds per square inch-gauge) and can increase to a differential of 20 to 30 psig when the terminal pressure drop is achieved. The pressure differential does not increase linearly with run time; the differential pressure increases at a faster rate with the duration of the run or as more material accumulates on the filter. The time between filter replacement is primarily dependent on flow rate, but also on influent water quality and filter material (i.e., size of pores).

The differential pressure between the inlet and the outlet header should be monitored to determine when the filter needs replacement. Also the differential pressure should be monitored immediately after replacing a filter and placing the unit back in service to verify that the filter was properly installed to prevent bypassing around the seals. An alarm could also be linked to the pressure gauges to ensure the operator is alerted when a filter needs to be replaced due to fouling or failure of the filer or associated seals.

### 8.6.2 Water Quality Monitoring

In addition to monitoring the pressure drop across the filter, the influent and effluent turbidity or particle count can be monitored to assess performance and indicate possible process upsets with the bag or cartridge filter or other upstream processes. The recommended monitoring frequency depends on the influent water quality and its variability. At a minimum, the pressure differential and effluent turbidity can be checked daily. If the filter is used to meet the treatment requirements of IESWTR/LT1ESTWR, turbidity monitoring is required and the state will set a turbidity performance standard. During the initial start-up phase of a newly integrated bag or

cartridge filtration system, monitoring can be more frequent and then can be reduced once the operator becomes familiar with the system. If continuous monitoring of turbidity and/or pressure differential is employed, the output from the sensors should be sent to an alarm to warn operators of sudden changes in operation, or if the filter element needs replacing.

EPA recognizes turbidity has limitations as an indicator of filter failure or pathogen breakthrough. However, in the absence of a better indicator, monitoring both influent and effluent turbidity over a full run (i.e., from start to end of the filter life) can provide a performance baseline. The baseline can then be used to indicate process upsets. This method may not be applicable to systems with very low raw water turbidity or where the influent has been filtered; the difference between influent and effluent turbidity may be too low to provide meaningful data.

Particle counters can be another valuable monitoring tool. If available, periodic checks of influent and effluent particle counts are also recommended to ensure the filter is removing particles in the appropriate size range (i.e., 4-6 microns).

## 8.7 References

McMeen. 2001. *Alternate Filtration: Placing New Technology in an Old Regulatory Box*. American Water Works Association, Membrane Conference Proceedings.

NSF International. 2005. *Protocol for Equipment Verification Testing for Physical Removal of Microbiological and Particulate Contaminants*. 40 CFR 35.6450. http://www.epa.gov/etv/pubs/059205epadwctr.pdf.

U.S. EPA. 2005. *Membrane Filtration Guidance Manual*. Office of Water. EPA 815-R-06-009. November, 2005. http://www.epa.gov/ogwdw/disinfection/lt2/compliance.html.

U.S. EPA. 2006. *Ultraviolet Disinfection Guidance Manual for the Final Long Term 2 Enhanced Surface Water Treatment Rule*. Office of Water. EPA 815-R-06-007. November, 2006. http://www.epa.gov/safewater/disinfection/lt2/compliance.html.

U.S. EPA. 2007. *Simultaneous Compliance Guidance Manual for the Long Term 2 and Stage 2 DBP Rules*. EPA 815-R-07-017. March, 2007. http://www.epa.gov/safewater/disinfection/stage2/compliance.html.

# 9. Second Stage Filtration

## 9.1 Introduction

The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) 40 CFR 141.719(c) describes second stage filtration as the use of a rapid sand, dual media, granular activated carbon (GAC), or other fine grain media unit process applied in a separate stage following rapid sand or dual media filtration. Applying an additional layer of media, such as a GAC cap, on an existing single stage filtration unit does not qualify for this credit.

This chapter is organized as follows:

9.2 LT2ESWTR Compliance Requirements - discusses criteria and reporting requirements that systems must meet to receive *Cryptosporidium* removal.

9.3 Toolbox Selection Considerations - discusses issues specific to second stage filtration that water systems should consider when selecting toolbox options.

9.4 Design and Operational Considerations - discusses hydraulic issues, backwashing, and turbidity monitoring for systems that integrate a second stage filtration in their treatment train.

## 9.2 LT2ESWTR Compliance Requirements

### 9.2.1 Credits

Under the LT2ESWTR, a system that employs a second, separate filtration stage may receive 0.5 log credit for *Cryptosporidium* removal (40 CFR 141.719(c)) under the following conditions.

- The first stage of filtration is preceded by a coagulation step.
- The second stage of filtration is comprised of rapid sand, dual media, GAC, or other fine grain media.
- Both filtration stages treat 100 percent of plant flow.
- The state must approve the treatment credit based on an assessment of the design characteristics of the filtration process.

Under the LT2ESWTR, a system integrating a slow sand filtration process for the second stage of filtration can receive 2.5 log credit for *Cryptosporidium* removal (40 CFR 141719(d)) under the following conditions.

- No disinfectant residual is present in the influent to the slow sand filtration process.
- Both filtration stages treat 100 percent of plant flow.
- The state must approve the treatment credit based on an assessment of the design characteristics of the filtration process.

### 9.2.2 Reporting Requirements

To receive *Cryptosporidium* removal credit for compliance with the LT2ESWTR, systems must verify that 100 percent of the flow was filtered through both stages and that the first stage was preceded by a coagulation step (40 CFR 141.721(f)).

Reporting for LT2ESWTR does not take the place of the IESWTR and LT1ESWTR reporting requirements. Specifically, the turbidity of the combined and individual filter effluent from the *first* filtration stage must be reported as required by the IESWTR and LT1ESWTR (40 CFR 141.74, 40 CFR 141.174(a), 40 CFR 141.551, and 40 CFR 141.560).

## 9.3 Toolbox Selection Considerations

Plants already employing a second unit process that meets the requirements for this toolbox option (e.g., GAC columns to meet dissolved organic or taste and odor treatment goals) are in the ideal position to seek credit. Other plants that have enough excess filtration capacity or unused filter beds (e.g., built in anticipation of unrealized plant expansions), may be able to convert piping to enable these filters to operate in series for relatively low cost. However, many plants will find that integrating second stage filtration into an existing treatment train poses significant additional space, capital, and hydraulic requirements. These systems may want to consider this option if the additional treatment provides other benefits. For example, systems that use chloramination and/or ozone could run the second stage under biological filtration conditions to reduce assimilable organic carbon (AOC), which promotes biofilm growth and nitrification (for chloraminating systems) in the distribution system.

Additionally, plants experiencing taste and odor problems or dissolved organic contaminants in their raw water might consider installing GAC columns to alleviate these problems and also receive the *Cryptosporidium* removal credit.

Slow sand filtration plants who wish to consider this toolbox option should either have sufficient excess filtration capacity to allow filters to operate in series (with possible piping modifications) or have sufficient land area to build additional filters.

### 9.3.1 Advantages

The advantages of a second stage filtration process are the same for both rapid and slow sand plants and include operator familiarity with the process, ease of operation, and potential to reduce disinfection byproducts. For plants with existing processes and infrastructure meeting the two-stage requirements, implementation costs are likely to be relatively low.

### 9.3.2 Disadvantages

The disadvantages associated with second stage filtration apply primarily to those plants that do not have existing processes in place or cannot easily convert built-in infrastructure. In addition to the capital cost for new filters, these plants may need the following improvements to integrate a second stage of filtration:

- Space if there is currently no room for expansion in the existing plant grounds.
- Additional pumping to compensate for head loss associated with an additional filtration process.
- Increased backwash supply and treatment.

For those plants that have existing infrastructure available for a second stage of filtration, they still may have to account for an increased volume of backwash and loss of head due to the second stage.

Systems with rapid sand filtration plants that are considering integrating slow sand filtration into their treatment process should be aware of the following differences in operation and performance of slow sand plants compared to rapid sand plants:

- More space required for slow sand plants.
- Decreased filtering performance with cold temperatures.
- Maintenance of filters requires draining and scraping a thin layer off the top of the filter.

## 9.4 Design and Operational Considerations

The design of the second stage is site-specific and depends on existing infrastructure (e.g., some systems may have enough filtration capacity to operate filters in series) and space and hydraulic requirements. EPA does not specify or restrict certain configurations, beyond the requirement that all flow must be treated by both stages. Systems that have existing filters not in use or not used to capacity may reconfigure the piping to operate in series. Media sizing for the

second stage is also not specified; however, typical design standards for regular or deep bed filters should be followed.

### 9.4.1 Hydraulic Requirements

Additional pumps may be needed to provide the necessary head between the first and second stages of filtration. The number of pumps and total number of filters should allow for redundancy, to ensure that sufficient treatment capacity is in place to treat all the plant flow in the event of equipment breakdown or maintenance. However, the filter loading rate to the second stage does not necessarily need to be the same as for the first stage. The water influent to the second stage should be significantly cleaner, and may enable higher loadings. Final design loading rates should be determined in consultation with the state.

If the filter effluent from the first stage filters is not combined and sent to the second stage filters via a distribution box or other flow equalization device, plant operation may be more complex. For example, if the effluent from one first stage filter is sent to just one second stage filter, then as the flow from first filter decreases (or headloss through it increases), flow through the second filter will also decrease, unless automatic effluent control valves are installed on the second stage filter. Also, in this case, whenever the first stage filter is backwashed, the second stage filter will also be out of service.

### 9.4.2 Backwashing

Consistent with the Filter Backwash Recycling Rule, the filter backwash from the second stage (as well as the first stage) must be recycled to the head of the plant if it is recycled. The existing backwashing capacity may be limited and need to be increased. There may be insufficient finished water storage to supply backwash water or there may need to be additional pumping capacity, depending upon the design of the additional filtration stage (e.g., if the existing filters have a small area and the new filters are significantly larger, the existing backwash pumps may not be able to supply water at a high enough flow to properly expand the filter bed). It is likely that the second stage filters would need to be backwashed less frequently than the first stage ones, due to the lower solids loading.

Filter ripening and/or filter-to-waste times for the second filtration stage will most likely differ from the first stage.

### 9.4.3 Turbidity Monitoring

Depending on the removal performance of the first stage filtration process, it may be difficult to see differences in second stage removal performance if monitoring of the second stage process is limited to the combined filter effluent (CFE) of the second stage. Individual filter

effluent (IFE) monitoring of the second stage filters on a continuous or routine basis may identify performance issues that can be addressed proactively.

# 10. Chlorine Dioxide

## 10.1 Introduction

Chlorine dioxide ($ClO_2$) is used for disinfection, taste and odor control, and iron and manganese removal. Chlorine dioxide is effective for inactivation of bacteria, viruses, and protozoa, including *Cryptosporidium* while forming fewer halogenated byproducts than chlorine. It is stable only in dilute aqueous solutions and must be generated on-site. It can be generated using a variety of starting materials including chloride, chlorite, or chlorate.

The Surface Water Treatment Rule (SWTR), Stage 1 Disinfection Byproducts Rule (Stage 1 DBPR), and Interim Enhanced Surface Water Treatment Rule (IESWTR) all recognize the ability of chlorine dioxide to inactivate pathogens. As a result, there is much information and guidance available on the application of chlorine dioxide for disinfection, particularly in the following two guidance manuals:

- Guidance Manual for Compliance with the Filtration and Disinfection Requirements for Public Water Systems Using Surface Water Sources (U.S. EPA 1991) (commonly referred to as the Surface Water Treatment Rule Guidance Manual).
  - Describes how to calculate the contact time (CT) value for a given disinfectant, including methodologies for determining the residual concentration (C) and contact time (T).
  - Includes CT values for log-inactivation of *Giardia* and viruses.
- Alternative Disinfectants and Oxidants Guidance Manual (U.S. EPA 1999) provides full descriptions of:
  - Chlorine dioxide chemistry.
  - On-site generation.
  - Primary uses and points of applications.
  - Pathogen inactivation and disinfection efficiency.
  - Byproduct production.
  - Analytical methods.
  - Operational considerations.

The Alternative Disinfectants and Oxidants Guidance Manual is available on EPA's website, http://www.epa.gov/safewater/mdbp/implement.html.

The purpose of this chapter is to (1) describe what systems need to do to achieve *Cryptosporidium* inactivation treatment credit for disinfecting with chlorine dioxide, (2) discuss design and operational considerations that will assist water systems in deciding whether this toolbox option is a practical option for them, and (3) discuss key issues associated with using chlorine dioxide as a disinfectant. This chapter is organized as follows:

10.2 Log Inactivation Requirements - describes the concentration and time variables of the CT parameter, presents the chlorine dioxide CT table for *Cryptosporidium*, and provides a sample CT calculation.

10.3 Monitoring Requirements - describes monitoring requirements of both LT2ESWTR and Stage 1 DBPR.

10.4 Unfiltered Systems LT2ESWTR Requirements - describes the level of *Cryptosporidium* inactivation unfiltered systems must provide and monitoring requirements that must be met.

10.5 Disinfection with chlorine dioxide - describes chlorine dioxide chemistry and disinfection with chlorine dioxide.

10.6 Toolbox Selection Considerations - discusses the advantages and disadvantages of disinfection with chlorine dioxide.

10.7 Design Considerations - discusses effects of temperature and the point of chlorine dioxide addition on achieving the required CT value.

10.8 Operational Considerations - discusses water quality parameters that affect the disinfection ability of chlorine dioxide.

10.9 Safety Issues - describes considerations for chemical storage and discusses the acute health risks of chlorine dioxide.

## 10.2 Log Inactivation Requirements

Systems can achieve anywhere from 0.25- to 3.0-log *Cryptosporidium* inactivation with the addition of chlorine dioxide. The amount of *Cryptosporidium* inactivation credit a system may receive is determined by the CT provided in the treatment process (40 CFR 141.720(b)). This methodology provides a conservative characterization of the dose of chlorine dioxide necessary to achieve a specified inactivation level of *Cryptosporidium*. CT is the product of the disinfectant concentration and disinfectant CT and is defined in the LT2ESWTR (40 CFR 141.720(a)):

CT = Disinfectant (mg/L) × Contact Time (minutes)

"**T**" is the time (in minutes) it takes the water, during peak hourly flow, to move from the point of disinfectant application to a point where, C, residual concentration is measured at or prior to the first customer, or between points of residual measurement.

"**C**" is the concentration of chlorine dioxide present in the system, expressed in mg/L.

The concept of regulating surface water treatment disinfection processes through CT was first introduced in the SWTR. Tables of *Giardia* and virus log-inactivations correlated to CT values, commonly referred to as CT tables, were presented in the SWTR Guidance Manual. For the LT2ESWTR, EPA developed CT tables for the inactivation of *Cryptosporidium*. Alternatively, a system may conduct a site-specific study to determine the CT values necessary to meet a specified log-inactivation, using state approval (40 CFR 141.720(c)). Appendix A provides guidance for conducting a site-specific study.

### 10.2.1 CT Calculation

The methodology and calculations for determining CT have not changed from the SWTR to the LT2ESWTR requirements. This section briefly reviews how CT is used to determine log-inactivation for the SWTR and presents the chlorine dioxide CT table for *Cryptosporidium* inactivation. Refer to the SWTR Guidance Manual for descriptions of measuring C and determining T.

CT can be calculated for an entire treatment process or broken into segments and summed for a total CT value. C is measured at the end of a given segment. T is generally estimated by methods involving established criteria (flow, volume, and contactor geometry) or tracer studies. The following steps describe the CT calculation from measured C and T values for a segment of the entire treatment process:

1. Calculate $CT_{calc}$ by multiplying the measured C and T values.

2. From the CT table (see Exhibit 10.1 for the CT table for *Cryptosporidium*), find the CT value for the log-inactivation desired, this is $CT_{table}$.

3. Calculate the ratio of $CT_{calc}/CT_{table}$ for each segment.

4. If a system has multiple segments, sum the $CT_{calc}/CT_{table}$ ratios for a total inactivation ratio.

5. If the ratio of $CT_{calc}/CT_{table}$ is at least 1, then the treatment process provides the log-inactivation that the $CT_{table}$ represents (log-inactivation desired from step #2).

## Exhibit 10.1 CT Values (mg-min/L) for *Cryptosporidium* Inactivation by Chlorine Dioxide [1]

| Log credit | Water Temperature, °C | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | <=0.5 | 1 | 2 | 3 | 5 | 7 | 10 | 15 | 20 | 25 | 30 |
| 0.25 | 159 | 153 | 140 | 128 | 107 | 90 | 69 | 45 | 29 | 19 | 12 |
| 0.5 | 319 | 305 | 279 | 256 | 214 | 180 | 138 | 89 | 58 | 38 | 24 |
| 1.0 | 637 | 610 | 558 | 511 | 429 | 360 | 277 | 179 | 116 | 75 | 49 |
| 1.5 | 956 | 915 | 838 | 767 | 643 | 539 | 415 | 268 | 174 | 113 | 73 |
| 2.0 | 1275 | 1220 | 1117 | 1023 | 858 | 719 | 553 | 357 | 232 | 150 | 98 |
| 2.5 | 1594 | 1525 | 1396 | 1278 | 1072 | 899 | 691 | 447 | 289 | 188 | 122 |
| 3.0 | 1912 | 1830 | 1675 | 1534 | 1286 | 1079 | 830 | 536 | 347 | 226 | 147 |

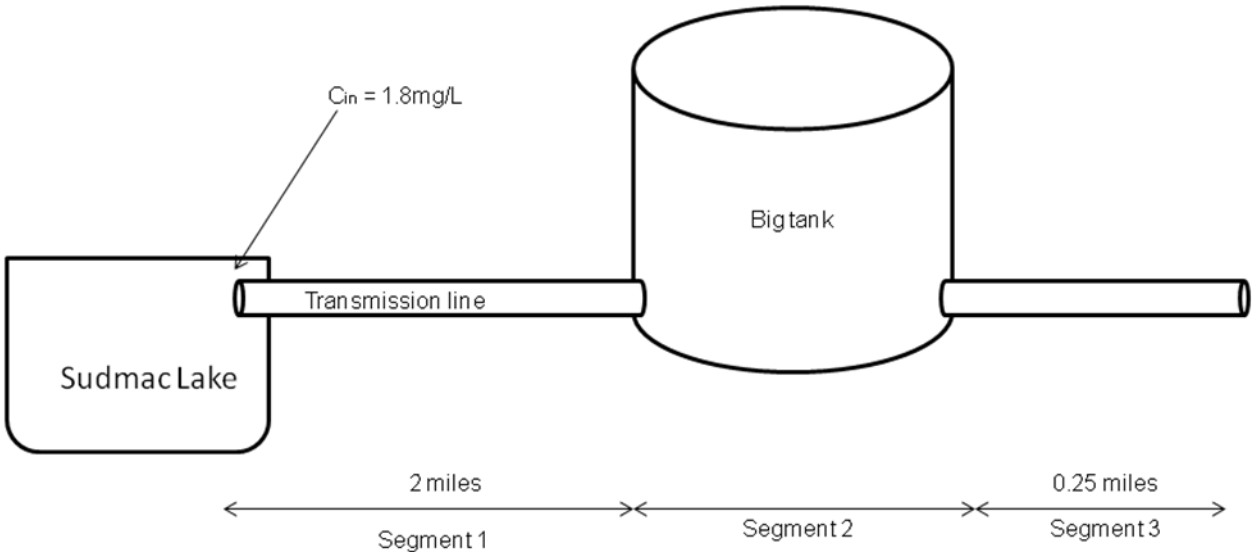
[1] Systems may use this equation to determine log credit between the indicated values:
Log credit=$(0.001506\times(1.09116)^{Temp})\times$ CT
Source: 141.720 (b)(1)

Example CT Calculation

A plant draws 1.5 million gallons per day (MGD) of 5 degrees Celsius water from a stream, adding 1.8 mg/L of chlorine dioxide at the intake. The water travels through 2 miles of 12 inch pipe to a settling tank. The detention time in the tank, as determined by a tracer study, is 150 minutes. After the tank, it travels through another 12-inch pipe to the plant. Exhibit 10.2 provides a schematic of an intake, piping, and tank. The concentration of chlorine dioxide at each point is measured as follows:

$C_{initial}$ = 1.8 mg/L
$C_{entering\ tank}$ = 1.6 mg/L
$C_{leaving\ tank}$ = 0.8 mg/L
$C_{leaving\ 2nd\ pipe}$ = 0.2 mg/L

## Exhibit 10.2 CT Calculation Example Schematic



The residence times of the two sections of pipe are determined assuming plug flow. Therefore, the time for each section is calculated as follows:

$T_1 = (A_1 * L_1/Q_1) = (\pi r^2 L_1/Q_1) * (7.48 \text{ gal}/1 \text{ ft}^3) * (\text{MG}/1{,}000{,}000 \text{ gal}) * (1{,}440 \text{ min/day})$

where:

- A is the cross-sectional area of the pipe in square feet;
- Q is the volumetric flow rate in MGD;
- L is the length of pipe in feet; and
- r is the radius of the pipe in feet.

Therefore the times for the two sections of the pipe are as follows:

$T_1 = 2 \text{ mi} * (5{,}280 \text{ ft/mi}) * \pi * (0.5 \text{ ft})^2 * (0.0108 \text{ MG*sec/ft}^3\text{*day})/(1.5 \text{ MGD}) = 59.7 \text{ min}$
$T_3 = 0.25 \text{ mi} * (5{,}280 \text{ ft/mi}) * \pi * (0.5 \text{ ft})^2 * (0.0108 \text{ MG*sec/ft}^3\text{*day})/(1.5 \text{ MGD}) = 7.4 \text{ min}$

The $T_{10}$, or time for 90 percent of a tracer to pass through the section for the tank is as follows:

$T_2 = 150$ minutes

CT Calculation:

Step 1. Calculate CT for each segment.

The concentrations and times for each segment are known. The T's are calculated above and the C is the concentration measured at the end of each segment. The CT for each segment is calculated as follows:

$CT_1 = (1.6 \text{ mg/L}) \times (59.5 \text{ min}) = 95.2 \text{ mg}\times\text{min/l}$
$CT_2 = (0.8 \text{ mg/L}) \times (150 \text{ min}) = 120 \text{ mg}\times\text{min/l}$
$CT_3 = (0.2 \text{ mg/L}) \times (7.4 \text{ min}) = 1.5 \text{ mg}\times\text{min/l}$

Step 2. Look up $CT_{table}$ in Exhibit 10.1. For 5°C and 0.5-log inactivation,

$CT_{table} = 214 \text{ mg}\times\text{min/L}$

Step 3. Calculate the ratio of $CT_{calc}/CT_{table}$ for each segment.

$(CT_{calc}/CT_{table})_1 = 95.2/214 = 0.44$
$(CT_{calc}/CT_{table})_2 = 120/214 = 0.56$
$(CT_{calc}/CT_{table})_3 = 1.5/214 = 0.01$

Step 4. Sum the $CT_{calc}/CT_{table}$ for each segment.

$(CT_{calc}/CT_{table})_{total} = 0.44 + 0.56 + 0.01 = 1.01$

Determine Log Inactivation:

If the result of Step 4 is greater than 1, the log-inactivation associated with the $CT_{table}$ values is achieved. If the result is less than 1, that level of log-inactivation is not achieved (if the log-inactivation was less than 1.0, the calculations should be repeated at a lower log-inactivation). In this example, the sum of the $CT_{calc}/CT_{table}$ for all the segments is greater than 1, so the system qualifies for a 0.5-log *Cryptosporidium* inactivation.

This example is intended to show a CT determination. Meeting log inactivation requirements using a disinfectant in a raw water supply may not be appropriate for raw water conditions (e.g., high turbidity) that may interfere with disinfectant efficacy.

## 10.3 Monitoring Requirements

### 10.3.1 LT2ESWTR

The LT2ESWTR requires **CT calculation at least once per day with both C and T measured during peak hourly flow (**40 CFR 141.720 (a)). Since systems may not know when the peak hourly flow is, EPA recommends monitoring flow on an hourly basis. Continuous flow monitoring and recording can also be used to determine peak flow. Systems should reevaluate CT whenever they modify a process and the hydraulics are affected (e.g., add a pump for increased flow, reconfigure piping).

The chlorine dioxide concentration should be measured using approved analytical methods, either DPD, (Standard Method 4500-ClO2 D) or Amperometric Method I or II, (Standard Method 4500-ClO2 C or E, respectively). Details on these methods can be found in Standard Methods for the Examination of Water and Wastewater, 20th edition, American Public Health Association, 1998.

Note, if a system changes its disinfection process, the LT2ESWTR requires the system to calculate a disinfection profile and benchmark (40 CFR 141.708) (see Chapter 1, section 1.4.5 for details).

### 10.3.2 Stage 1 DBPR

The Stage 1 DBPR requires all systems using chlorine dioxide for disinfection or oxidation to monitor daily for chlorine dioxide and chlorite at the distribution system entry point. In addition, systems must take monthly chlorite samples at three locations in the distribution system. Exhibit 10.3 lists the chlorine dioxide and chlorite distribution system monitoring requirements.

### Exhibit 10.3 Distribution System Monitoring Requirements at Each Plant

| Location | Frequency |
|---|---|
| **Chlorite** | |
| Distribution System Entry Point | Daily |
| Distribution System Sample Set of 3:<br>1 Near First Customer<br>1 At Average Residence Time<br>1 At Maximum Residence Time | Monthly |
| **Chlorine Dioxide** | |
| Distribution System Entry Point | Daily |

If the chlorine dioxide maximum residual disinfectant level (MRDL) of 0.8 mg/L or the chlorite maximum contaminant level (MCL) of 1.0 mg/L is exceeded in any of the samples, additional monitoring is required (see the Stage 1 DBPR, 40 CFR141.132(b) for further information). Depending on the results of the additional samples, the system could have an acute violation with more serious public notification requirements than for a chlorine MRDL violation. The monthly monitoring requirements for chlorite may be reduced if all chlorite samples are below the MCL for a 1-year period.

## 10.4 Unfiltered System LT2ESWTR Requirements

The LT2ESWTR requires unfiltered systems to provide at least 2.0-log *Cryptosporidium* inactivation (40 CFR 141.712(b)). If their source water *Cryptosporidium* concentration is greater than 0.01 oocyst/liter, then systems must provide 3.0-log *Cryptosporidium* inactivation (40 CFR 141.712(b)). The requirements of the previous SWTR regulations still apply— achieve 3-log inactivation of *Giardia* and 4-log inactivation of viruses and maintain a disinfectant residual in the distribution system (e.g., free chlorine or chloramines). LT2ESWTR also requires that a minimum of two disinfectants be used to meet overall disinfection requirements.

The monitoring requirements described in section 10.3 apply to unfiltered systems. Additionally, the LT2ESWTR requires unfiltered systems to meet the *Cryptosporidium* log-inactivation requirements determined by the daily CT value every day the system serves water to the public, except one day per calendar month (40 CFR 141.712(c)). Therefore, if an unfiltered system fails to meet *Cryptosporidium* log-inactivation two days in a month, it is in violation of the treatment technique requirement.

## 10.5 Disinfection with Chlorine Dioxide

Chlorine dioxide ($ClO_2$) is an uncharged compound of chlorine in the +IV oxidation state. It is a relatively small, volatile, and highly energetic molecule, and a free radical even in dilute aqueous solutions. At high concentrations, it reacts violently with reducing agents. However, it is stable in dilute solution in a closed container in the absence of light. When an aqueous solution is open to the atmosphere, chlorine dioxide readily comes out of solution. Aqueous solutions of chlorine dioxide are also susceptible to photolytic decomposition, depending on the time of exposure and intensity of ultraviolet light (UV) light.

Disinfection of protozoa is believed to occur by oxidation reactions disrupting the permeability of the cell wall (Aieta and Berg 1986). Chlorine dioxide functions as a highly selective oxidant due to its unique, one-electron transfer mechanism where it is reduced to chlorite ($ClO_2^-$) (Hoehn et al. 1996).

In drinking water, chlorite ($ClO_2^-$) is the predominant reaction end product, with approximately 50 to 70 percent of the chlorine dioxide converted to chlorite and 30 percent to chlorate ($ClO_3^-$) and chloride (Cl-) (Werdehoff and Singer 1987). This has a significant impact on disinfection capabilities for drinking water, since chlorite is a regulated drinking water contaminant with an MCL of 1.0 mg/L. Based on a 50 to 70 percent conversion of chlorine dioxide to chlorite, the maximum dose is limited to 1.4 to 2.0 mg/L unless the chlorite is removed through subsequent treatment processes.

## 10.6 Toolbox Selection Considerations

### 10.6.1 Advantages

There are several advantages to using chlorine dioxide as a primary disinfectant. Chlorine dioxide is approximately four times as effective as chlorine for the inactivation of *Giardia* and is a stronger disinfectant than chlorine for bacteria (White 1999). However, free chlorine is more effective for the inactivation of viruses. Other advantages of disinfection with chlorine dioxide include:

- A high oxidizing potential allows it to oxidize other compounds such as manganese and some taste and odor compounds.
- Chlorine dioxide does not form regulated halogenated organic byproducts.
- The effect of pH on the disinfection ability of chlorine dioxide is much smaller than for other disinfectants.
- Chlorine dioxide has shown a synergistic effect when combined with other disinfectants such as ozone, chlorine, and chloramines that leads to greater inactivation with the disinfectants added in series than by either disinfectant individually.
- Chlorine dioxide can be used in the control of zebra mussels.

### 10.6.2 Disadvantages

A major disadvantage of chlorine dioxide is the byproduct formation of chlorite and chlorate. Section 10.6 describes the dose limits of chlorine dioxide due to the formation of chlorite. Other disadvantages of disinfection with chlorine dioxide include:

- Difficulty in maintaining an effective residual. Additionally, residual will be lost in the filters.

- It decomposes upon exposure to sunlight, flourescent light bulbs, and UV disinfection systems.

- Ability to disinfect is reduced under colder temperatures.

- It can form brominated disinfection byproducts (DBPs) in the presence of bromide.

- If the ratio of reactants in the chlorine dioxide generator is incorrect, excess aqueous chlorine can remain, which can form halogenated disinfection byproducts.

- Chlorine dioxide must be generated on-site.

- There may be a need for three-phase power which may not be compatible with some water systems.

- Chlorine dioxide can be explosive at high temperatures or pressures.

- Storage of sodium chlorite solution can be problematic due to crystallization at low temperatures or high concentrations and stratification at temperatures below 40°F (or 4°C).

- Dialysis patients are sensitive to higher chlorite levels and should be notified if chlorine dioxide is going to be added where it has not routinely been used.

- Training, sampling, and analysis costs are high.

- If used together with granular activated carbon (GAC) it can react to form chlorate.

Systems considering using chlorine dioxide as a disinfectant should perform chlorine dioxide demand/decay tests on the water being considered for disinfection (raw water or filter effluent) under normal and poor water quality conditions. If chlorine dioxide is added where the demand is 1.4 mg/L or greater, the system may have difficulty complying with the chlorite MCL. If the raw water has a chlorine dioxide requirement greater than 1.4 mg/L, chlorine dioxide might still be able to be used for post disinfection since the oxidant demand will be less after the filters. The Simultaneous Compliance Guidance Manual can be consulted on the how to offset the disadvantages of chlorine dioxide use.

## 10.7 Design Considerations

### 10.7.1 Designing to Lowest Temperature

As the water temperature declines, chlorine dioxide becomes less effective as a disinfectant. LeChevallier et al. (1997) found that reducing the temperature from 20 degrees Celsius to 10 degrees Celsius reduced disinfection effectiveness by 40 percent. Since the

treatment achieved for chlorine dioxide addition is temperature dependent, systems need to consider the variability in water temperature to ensure they meet the CT level for the minimum treatment needed for compliance. For example, if a system is required to provide an additional 1-log *Cryptosporidium* treatment and plans to achieve that with chlorine dioxide alone, then it should determine the CT required for the lowest water temperature experienced and ensure it can meet those CT requirements.

### 10.7.2 Point of Addition

There are two main considerations for determining locations of chlorine dioxide addition for the purpose of *Cryptosporidium* inactivation—CT and chlorine dioxide demand. Additionally, systems using ozone should consider that ozone will degrade chlorine dioxide. The application point for chlorine dioxide should be well upstream of the ozone process or just after the ozone process.

*Contact Time*

The CT requirements for *Cryptosporidium* are much higher than for *Giardia* and viruses and when designing to the lowest water temperatures, the resulting CT requirements are relatively high for even the 0.5- and 1.0-log inactivation. Chlorine dioxide readily degrades when exposed to light from flourescent lamps or the sun; therefore, all the available concentration in open basins will most likely not be utilized for disinfection. For most systems, the point of application will be either at the raw water intake or after the filters, whichever can provide the necessary CT.

*Oxidant Demand*

The oxidant demand of the water affects chlorite and chlorate byproduct formation. If the chlorine dioxide requirement of the raw water is greater than 1.4 mg/L then chlorite concentration will likely exceed the MCL. However, chlorine dioxide could be added after the filters where the oxidant demand is frequently lower and, therefore, a lower dose of chlorine dioxide would result in a lower byproduct concentration of chlorite.

## 10.8 Operational Considerations

Of all the water quality parameters, water temperature has the strongest effect on the disinfection ability of chlorine dioxide. The concentration of suspended matter and pH also have an effect, but to a lesser extent than temperature. Although the disinfection potential of chlorine dioxide is not strongly affected by pH, studies have shown that chlorine dioxide disinfection is better under higher pH (LeChevallier et al. 1997).

Suspended matter and pathogen aggregation affect the disinfection efficiency of chlorine dioxide. Protection from chlorine dioxide inactivation due to bentonite was determined to be approximately 11 percent for water with turbidity values less than or equal to 5 NTU and 25 percent for turbidity between 5 and 17 NTUs (Chen et al. 1984).

Based on the research discussed above, the optimal conditions for *Cryptosporidium* disinfection with chlorine dioxide are low turbidity, high pH, and high temperature.

## 10.9 Safety Issues

Because chlorine dioxide can be explosive and pose acute health risks to those exposed to gaseous chlorine dioxide, a safety plan should be developed that includes precautions for generation, handling, storage, and emergency response.

> Airborne concentrations greater than 10 percent may cause explosions.

### 10.9.1 Chemical Storage

Most chlorine dioxide generators use sodium chlorite solutions as a raw material. If sodium chlorite solutions are accidently acidified or exposed to a reducing agent, uncontrolled production and release of gaseous chlorine dioxide can result. In addition to being toxic, if the gaseous chlorine dioxide reaches concentrations greater than 10 percent, it can spontaneously explode.

Sodium chlorite should be stored away from other chemicals, particularly any acid solutions or chemicals that could act as reducing agents. Construction materials in sodium chlorite storage areas, as well as chlorine dioxide generating areas, should be fire resistant such as concrete. Sodium chlorite fires burn especially hot and produce oxygen as a byproduct, so special fire fighting techniques are required to extinguish the fire. These firefighting techniques should be part of the safety plan and proper equipment and supplies should be stored nearby. Temperatures in storage and generation areas should be kept below 30 degrees Celsius.

### 10.9.2 Acute Health Risks of Chlorine Dioxide

Exposure to gaseous chlorine dioxide can cause shortness of breath, coughing, respiratory distress, and pulmonary edema. The Occupational Safety and Health Administration (OSHA) permissible exposure limit (PEL) is 0.1 ppm. Areas where chlorine dioxide is generated and stored should have appropriate monitoring to detect leaks of chlorine dioxide or other chlorine containing chemicals into the air. Proper ventilation and scrubbing systems should be installed. First aid kits and respirators should also be accessible outside the building. Operators should be trained to use the respirators.

## 10.10 References

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# 11. Ozone

## 11.1 Introduction

Ozone is commonly used in drinking water treatment for disinfection as well as taste and odor control. Ozone is a strong oxidant that can inactivate microorganisms, including *Cryptosporidium*, *Giardia*, and viruses, as well as oxidize and break down natural organic matter (NOM). It exists as a gas at room temperature and must be generated on-site. Ozone reacts rapidly with organic and inorganic compounds and does not maintain a residual over the time scales associated with secondary disinfection.

The Surface Water Treatment Rule (SWTR), subsequent Stage 1 Disinfectants and Disinfection Byproducts Rule (DBPR), and Interim Enhanced SWTR (IESWTR) all recognize the capability of ozone to inactivate pathogens. As a result, there is much information and guidance available on the application of ozone for disinfection, particularly in the following two guidance manuals:

- *Guidance Manual for Compliance with the Filtration and Disinfection Requirements for Public Water Systems Using Surface Water Sources (U.S. EPA 1991)* (commonly referred to as the SWTR Guidance Manual).
  - Describes how to calculate the contact time (CT) value for ozone (CT is described in the next section), including methodologies for determining the residual concentration (C) and contact time (T).
  - Includes ozone CT values for log-inactivation of *Giardia* and viruses.
- *Alternative Disinfectants and Oxidants Guidance Manual (U. S. EPA 1999)* provides full descriptions of:
  - Ozone chemistry
  - On-site generation
  - Primary uses and points of application
  - Byproduct production
  - Analytical methods
  - Operational considerations

The SWTR Guidance Manual and *Alternative Disinfectants and Oxidants Guidance Manual* are available on EPA's website:

http://www.epa.gov/safewater/mdbp/implement.html

The purpose of this chapter is to (1) describe what systems need to do to receive *Cryptosporidium* treatment credit for disinfecting with ozone, (2) discuss design and operational considerations that will assist water systems in deciding whether this toolbox option is a practical option for their system, and (3) discuss key issues associated with using ozone as a disinfectant. This chapter is organized as follows:

11.2 Credits - discusses *Cryptosporidium* inactivation credit systems can receive with the addition of ozone, and relates CT to *Cryptosporidium* inactivation credit.

11.3 CT Determination - summarizes how CT is used to determine log inactivation credit for the SWTR and highlights the changes in CT calculation methodologies from the SWTR to the LT2ESWTR.

11.4 Monitoring Requirements - discusses monitoring requirements of both Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) and Stage 1 DBPR.

11.5 Unfiltered Systems LT2ESWTR Requirements - discusses *Cryptosporidium* inactivation requirements that unfiltered systems must meet.

11.6 Toolbox Selection - discusses the potential advantages and disadvantages of ozone processes.

11.7 Disinfection with Ozone - describes reaction pathways of ozone in water as well as inorganic and organic byproduct formation.

11.8 Design - discusses similarities and differences of different types of ozone generators and contactors, general considerations in determining the locations of ozone addition, and filter media and operating conditions of biologically active filters.

11.9 Safety Considerations in Design - discusses various safety considerations that should be taken into account in the design of ozone generators.

11.10 Operational Issues - discusses how ozone disinfection and CT calculation are affected by ozone demand, pH, temperature, and residual disinfectant in the distribution system.

## 11.2 Credits

Systems can receive between a 0.25 to 3.0 log *Cryptosporidium* inactivation credit with the addition of ozone, depending on the ozone dose applied. The value of the *Cryptosporidium* inactivation credit that a system receives is determined by the CT or inactivation provided in the treatment process. CT values are established to provide a conservative characterization of the dose of ozone necessary to achieve a specified inactivation of *Cryptosporidium*. CT is defined as the product of the disinfectant concentration and disinfectant contact time:

CT = Disinfectant (mg/L) x Contact Time (minutes)

- "**T**" is the time it takes the water to move from the point where the initial disinfectant residual concentration is measured to the point where the final disinfectant residual concentration is measured in a specified disinfectant segment.

- "**C**" is the concentration of dissolved ozone in mg/L.

The concept of regulating surface water treatment disinfection through CT was first introduced in the SWTR. Tables relating *Giardia* and virus log inactivation with associated CT values, commonly referred to as CT tables, were presented in the SWTR Guidance Manual. For the LT2ESWTR, EPA developed CT values for *Cryptosporidium* inactivation by ozone (Exhibit 11.1).

### Exhibit 11.1 CT Values for *Cryptosporidium* Inactivation by Ozone (40 CFR 141.730)

| Log credit | Water Temperature, °C[1] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | <=0.5 | 1 | 2 | 3 | 5 | 7 | 10 | 15 | 20 | 25 | ≥30 |
| 0.25 | 6.0 | 5.8 | 5.2 | 4.8 | 4.0 | 3.3 | 2.5 | 1.6 | 1.0 | 0.6 | 0.39 |
| 0.5 | 12 | 12 | 10 | 9.5 | 7.9 | 6.5 | 4.9 | 3.1 | 2.0 | 1.2 | 0.78 |
| 1.0 | 24 | 23 | 21 | 19 | 16 | 13 | 9.9 | 6.2 | 3.9 | 2.5 | 1.6 |
| 1.5 | 36 | 35 | 31 | 29 | 24 | 20 | 15 | 9.3 | 5.9 | 3.7 | 2.4 |
| 2.0 | 48 | 46 | 42 | 38 | 32 | 26 | 20 | 12 | 7.8 | 4.9 | 3.1 |
| 2.5 | 60 | 58 | 52 | 48 | 40 | 33 | 25 | 16 | 9.8 | 6.2 | 3.9 |
| 3.0 | 72 | 69 | 63 | 57 | 47 | 39 | 30 | 19 | 12 | 7.4 | 4.7 |

[1]CT values between the indicated temperatures may be determined by linear interpolation.

The CT values reported in Exhibit 11.1 were developed from the following equation:

$$Cryptosporidium\ Log\ Credit = 0.0397 \times (1.09757)^{Temp} \times CT \qquad \text{Equation 11-1}$$

where "*Temp*" is the water temperature expressed in degrees Celsius between 0.5 and 25 °C. A water system may use the above equation in lieu of the table. Equations for *Giardia* and virus shown below for ozone were derived from the $k_{10}$ values presented in Appendix O to the SWTR Guidance Manual for *Giardia* and virus.

$$Giardia\ Log\ Credit = 1.0380 \times (1.0741)^{Temp} \times CT \qquad \text{Equation 11-2}$$

$$virus\ Log\ Credit = 2.1744 \times (1.0726)^{Temp} \times CT \qquad \text{Equation 11-3}$$

If a utility believes that the CT values for *Cryptosporidium* presented in Exhibit 11.1 or calculated by Equation 11-1 do not accurately represent the conditions needed to achieve the desired level of inactivation in their system, they have the option of conducting a site specific study to generate a set of CT tables for their facility. The study would involve measuring actual *Cryptosporidium* inactivation performance under site conditions. If accepted by the state, the CT tables generated by the site study would replace the tables given in this guidance for the site at which the study was performed. Guidance on site specific studies of *Cryptosporidium* inactivation is presented in Appendix A.

## 11.3 CT Determination

The recommended methodologies and calculations for determining CT have two modifications from the SWTR to the LT2ESWTR.

For *Cryptosporidium*, EPA recommends that **no inactivation credit** be granted for the first dissolution chamber due to the higher CT requirements of *Cryptosporidium* compared to *Giardia* and virus. This differs from the SWTR guidance manual, where EPA recommends granting *Giardia* and virus inactivation credit for a "first chamber" (i.e. dissolution chamber; see section O.3.3 of Appendix O of the SWTR guidance manual ) of an ozone contactor, provided that the residual ozone concentration measured at the outlet from the first chamber met minimum concentration levels. For *Cryptosporidium*, the relatively small CT values normally achieved due to oxidant demand in the first dissolution chamber and the resources required for routine ozone monitoring would likely offset the benefit from the small *Cryptosporidium* credit achieved**.**

If no tracer study data are available for determining $T_{10}$, EPA recommends using the Continuous Stirred Tank Reactor (CSTR) approach (described below) or the Extended CSTR approach (described in Appendix B). In the SWTR Guidance Manual methods were presented for determining the ratio of $T_{10}$ to the theoretical hydraulic detention time (HDT) based on baffling characteristics (see Table C-5 of the SWTR Guidance Manual). However, these $T_{10}$/(HDT) ratios based on baffling characteristics were based on baffling characteristics of clearwells and basins. At this time, EPA is not aware of similar studies for ozone contactors that could be used to develop comparable recommendations and hence tracer studies are recommended for determining $T_{10}$/HDT ratios.

This guidance manual presents four methods for calculating CT in an ozone contactor:

- $T_{10}$ Method.
- CSTR Method.
- Extended $T_{10}$ Method.
- Extended CSTR Method.

These methods differ in the level of effort associated with them. Selecting the appropriate method(s) to use depends on the configuration of the ozone contactor, the availability of state-approved tracer testing results, and the amount of process evaluation and monitoring that a

system wishes to undertake. The $T_{10}$ and CSTR methods are the simplest methods and are described in this chapter. Appendix B provides more information for choosing the appropriate method and detailed guidance for the Extended $T_{10}$ and Extended CSTR methods. Exhibit 11.2 summarizes the current methods, including a description of the situations when their use is appropriate. It should be noted that, while this Manual is focused on *Cryptosporidium* inactivation, any of the four CT calculation methods discussed herein can also be applied to calculate the CT for obtaining credits for *Giardia* or virus inactivation in an ozone contactor under the requirements of the SWTR, recognizing that the first chamber credit for *Giardia* and virus inactivation provided under the SWTR remain valid, while no such credit is recommended for *Cryptosporidium*.

## Exhibit 11.2 Recommended Methods and Terminology for Calculating the Log-Inactivation Credit in an Ozone Contactor

| | Section Description | Terminology | Method for Calculating Log-Inactivation | Recommended Restrictions |
|---|---|---|---|---|
| Without Tracer Data | Chambers where ozone is added | | | |
| | First chamber | First Dissolution Chamber | No *Cryptosporidium* log-inactivation credit is recommended | The SWTR criteria for 1st chamber credit should still be used if calculating inactivation of *Giardia* and virus |
| | Other chambers | Co-Current or Counter-Current Dissolution Chambers | CSTR Method in each chamber with a measured effluent ozone residual concentration | No credit should be given to a dissolution chamber unless a detectable ozone residual has been measured upstream of this chamber |
| | Reactive Chambers | | | |
| | ≥ 3 consecutive chambers | Extended Reactive Zone | Extended CSTR Method in each chamber | Detectable ozone residual should be present in at least 3 chambers in this zone, measured via in-situ sample ports. Otherwise, the CSTR method should be applied individually to each chamber having a measured ozone residual |
| | < 3 consecutive chambers | Reactive Chamber(s) | CSTR Method in each chamber | The SWTR criteria for 1st chamber credit should still be used if calculating inactivation of *Giardia* and virus |
| With Approved Tracer Data | Chambers where ozone is added | | | |
| | First chamber | First Dissolution Chamber | No *Cryptosporidium* log-inactivation is credited to this section | The SWTR criteria for 1st chamber credit should still be used if calculating inactivation of *Giardia* and virus |
| | Other chambers | Co-Current or Counter-Current Dissolution Chambers | $T_{10}$,or CSTR Method in each chamber with a measured effluent ozone residual concentration | No credit should be given to a dissolution chamber unless a detectable ozone residual has been measured upstream of this chamber |
| | Reactive Chambers | | | |
| | ≥ 3 consecutive chambers | Extended Reactive Zone | Extended $T_{10}$ or Extended CSTR Method in each chamber. The Extended CSTR method is not appropriate for non-conventional contactors. | Detectable ozone residual should be present in at least 3 chambers in this zone, measured via in-situ sample ports. Otherwise, the $T_{10}$ or CSTR method should be applied to each chamber having a measured ozone residual |
| | < 3 consecutive chambers | Reactive Chamber(s) | $T_{10}$ or CSTR Method in each chamber | None |

The remainder of this section describes how to calculate C for the $T_{10}$ and CSTR methods and then describes the $T_{10}$ and CSTR methodologies.

### 11.3.1 Measuring C for $T_{10}$ and CSTR Methods

The recommended methods for determining C have not been modified from those presented in the SWTR Guidance Manual. The two methods for determining C are:

1) Direct measurement of the concentration profile of dissolved ozone in each contact chamber (described in section O.3.2 of the SWTR Guidance Manual).

2) Indirect prediction of the characteristic C based on dissolved ozone measurements at the contact chamber outlet (described in section O.3.3 of the SWTR Guidance Manual).

For the second method that involves predicting the characteristic C based on outlet measurements, the correlations presented in Exhibit 11.3 should be used based on $C_{in}$ and $C_{out}$ measurements. To be granted inactivation credit for a chamber, its final ozone concentration should be above the detection limit (i.e., have a positive $C_{out}$ value).

### Exhibit 11.3 Correlations to Predict C* Based on Ozone Residual Concentrations in the Outlet of a Chamber

| Method | Turbine | Dissolution Chamber Co-Current Flow | Dissolution Chamber Counter-Current Flow | Reactive Chamber |
|---|---|---|---|---|
| $T_{10}$ | $C_{out}$ | $C_{out}$ or $(C_{in}+C_{out})/2$ | $C_{out}/2$ | $C_{out}$ |
| CSTR | $C_{out}$ | $C_{out}$ or $(C_{in}+C_{out})/2$ | $C_{out}/2$ | $C_{out}$ |

C* - Characteristic concentration, used for CT calculation.
$C_{out}$ - Ozone residual concentration at the outlet from the chamber.
$C_{in}$ - Ozone residual concentration at the inlet to the chamber, which can be $C_{out}$ of the immediate upstream chamber.

### 11.3.2 $T_{10}$ Method

Using the $T_{10}$ approach, the $T_{10}$ is the time at which 10 percent of the water in the contactor or segment has passed through the contactor or segment. EPA recommends that tracer studies be used to determine the $T_{10}$/HDT ratio for ozone contactors. The *SWTR Guidance Manual* and *Tracer Studies in Water Treatment Facilities: A Protocol and Case Studies* describe how to conduct a tracer test.

Appendix C of the SWTR Guidance Manual provides a description of tracer studies and tracer study methods. Appendix E of this guidance provides a description of tracer test development and analysis. In general, tracer studies should represent the range of flow and

operational conditions expected for the ozone process and should include data quality criteria (i.e. percent tracer recovered). Tracer chemicals should be conservative (high percent recovery) and should be acceptable to the primacy agency for use in public water systems.

CT can be calculated for an entire treatment process (e.g., an entire ozone contactor) or broken into segments (e.g., individual contact chambers) and summed for a total CT value for all segments. *C* is measured either at the end of a given segment or both the beginning and end of the segment.

The following steps describe the CT calculation from measured C and T values for a segment or the entire treatment process:

1) Calculate $CT_{calc}$ by multiplying the measured C and T values. Sum $CT_{calc}$ for individual segments to obtain $CT_{calc}$ for the entire ozone contactor.

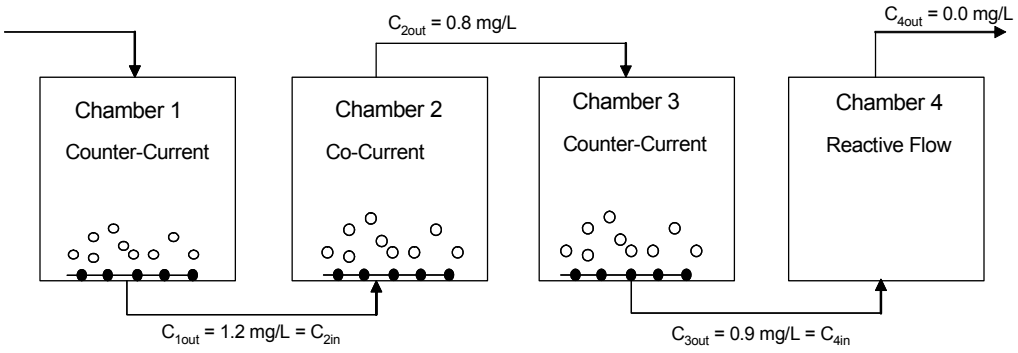
2) Calculate log inactivation credit using CT tables, or Equations 11.1, 11.2 or 11.3.

3) Calculate the Inactivation Ratio as Log-Credit Achieved / Log-Credit Required.

4) If the Inactivation Ratio is at least 1.0, then the treatment process provides the level of log inactivation desired.

*Example CT Calculation and Log Credit Determination using the $T_{10}$ Method*

A water system employs a four chamber ozone contactor to achieve a 0.5-log *Cryptosporidium* inactivation credit. The contactor is designed and operated as shown in the following diagram.




The water temperature is 5 degrees Celsius. Each chamber has a volume of 1,000 gallons. The flowrate through the contactor is 100 gpm. Results from a tracer test showed the $T_{10}$/HDT ratio for the contactor is 0.60.

The first step is to determine the ozone concentration for each chamber (segment). EPA recommends that inactivation credit not be granted for the first chamber, therefore concentrations are only calculated for Chambers 2, 3, and 4. Using Exhibit 11.3, C can be determined with the following equations:

Chamber 2: $C = (C_{in} + C_{out}) / 2$ or $C = C_{out}$
Chamber 3: $C = C_{out}/2$
Chamber 4: $C = C_{out}$

Therefore for:

Chamber 2: C = (1.2 + 0.8) / 2 = 1.0 mg/L (this equation gives the higher C value)
Chamber 3: C = 0.9 / 2 = 0.45 mg/L
Chamber 4: C = 0.0 mg/L

2) Calculate the T for each chamber.

The $T_{10}$ measured across all four chambers is divided proportionally by volume among the four chambers. This method should not be used if the sum of the volumes of the chambers with effluent ozone concentrations of zero (non-detectable) is 50 percent or greater than the entire volume of the chambers. In this example, only the last chamber had a non-detectable final concentration and that chamber is 25 percent the volume of all the chambers. Therefore the overall $T_{10}$ can be extrapolated among the chambers to estimate individual chamber $T_{10}$ values.

$$T_{10} \text{ of each chamber} = \left[\frac{T_{10}}{HDT}\right] \times HDT_{chamber} = \left[\frac{T_{10}}{HDT}\right] \times \left[\frac{Vol_{chamber}}{Flowrate}\right] = [0.6] \times \left[\frac{1000}{100}\right] = 6 \text{ min.}$$

(In this example, the volume of each chamber is the same. Therefore, the $T_{10}$ of each chamber is also 6 minutes)

3) Calculate the CT for each chamber. CT for the total contactor is the sum of the CT for individual chambers.

Chamber 1: not calculated
Chamber 2: CT = 1.0 mg/L × 6 min = 6.0 mg-min/L
Chamber 3: CT = 0.45 mg/L × 6 min = 2.7 mg-min/L
Chamber 4: CT = 0 mg/L × 6 min = 0 mg-min/L
Total CT = 8.7 mg-min/L

4) Calculate the *Cryptosporidium* log credit using Equation 11-1:

$$Cryptosporidium\ Log\ Credit = 0.0397 \times (1.09757)^{5} \times 8.7 = 0.55 \text{ logs}$$

5) Calculate *Cryptosporidium* inactivation Ratio as 0.55/0.5 = 1.1.

The inactivation ratio is greater than 1.0, and therefore this system achieves 0.5 log *Cryptosporidium* inactivation credit.

### 11.3.3 CSTR Method

The CSTR method is recommended for contactors that experience significant back mixing ($T_{10}$/HDT ≤ 0.5) or when no tracer data is available. This method uses the HDT of the ozone contactor, as described below, for estimating the contact time. The CSTR method should be applied to the individual chambers in the contactor.

For the CSTR approach, log inactivation is calculated with the following equation:

$$-\text{Log}\,(I/I_0) = \text{Log}\,(1 + 2.303 \times k_{10} \times C \times HDT) \qquad \text{Equation 11-4}$$

where:

-Log ($I/I_0$) = the log inactivation
$k_{10}$ = log base 10 inactivation coefficient (L/mg-min)[1]
C = Concentration from Exhibit 11-3 (mg/L)
HDT = Hydraulic detention time (minutes)

Exhibit 11.4 presents the $k_{10}$ values for *Cryptosporidium* ($k_{10}$ values are calculated from the CT table).

**Exhibit 11.4 Inactivation Coefficients for *Cryptosporidium*, Log base 10 (L/mg-min)**

| | Water Temperature, °C | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | ≤0.5 | 1 | 2 | 3 | 5 | 7 | 10 | 15 | 20 | 25 | ≥30 |
| $k_{10}$ | 0.0417 | 0.0430 | 0.0482 | 0.0524 | 0.0629 | 0.0764 | 0.101 | 0.161 | 0.254 | 0.407 | 0.648 |

To interpolate between the temperatures in the table, Equation 11-5 can be used.

$$k_{10} = 0.0397 \times (1.09757)^{Temp} \qquad \text{Equation 11-5}$$

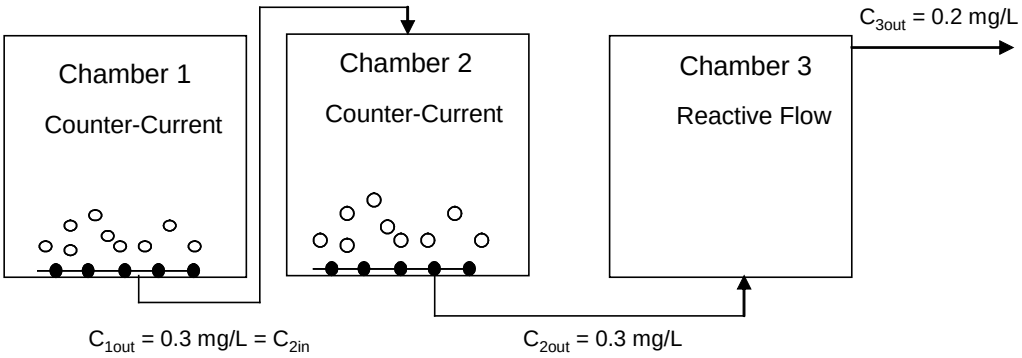
[1] $k_{10}$ is calculated from the CT table with the following equation: Log inactivation = $k_{10}$ x CT

In order to apply Equation 11-4, both C and HDT must be known. These two parameters can be determined for individual chambers or for zones consisting of multiple, adjacent chambers. In general, if the concentration is measured at 3 or more points in the contactor, the Extended CSTR method will provide a larger CT credit than CSTR method alone, so the basic CSRT method will likely not be used.

EPA recognizes that, for many situations, either the CSTR or the $T_{10}$ method can be used to calculate inactivation credit, and that they may generate two different estimates of log inactivation. EPA recommends that systems use, and states accept, the higher estimate of the log inactivation credit. However, systems should select one method to be used and use that method consistently.

***Example - CT Calculation and Log Credit Determination using the CSTR Method with the concentration measured for each chamber***

A system employs a three-chamber ozone contactor, with ozone addition in the first two chambers. The second chamber is a counter-current flow dissolution chamber with influent and effluent ozone concentrations of $C_{in}$ = 0.3 mg/L and $C_{out}$ = 0.3 mg/L. The effluent ozone concentration in the third, reactive chamber is $C_{out}$ = 0.2 mg/L. At 10°C, Exhibit 11-4 shows the $k_{10}$ value at 0.101 L/mg-min. The HDT for each chamber is 20 minutes.




1) Determine the C values for each chamber for the CSTR calculation

Chamber 1: No *Cryptosporidium* inactivation credit recommended
Chamber 2: $C = C_{2out}/2 = 0.3 / 2 = 0.15$ mg/L
Chamber 3: $C = C_{3out} = 0.2$ mg/L

2) Calculate the log inactivation for each chamber using Equation 11-4

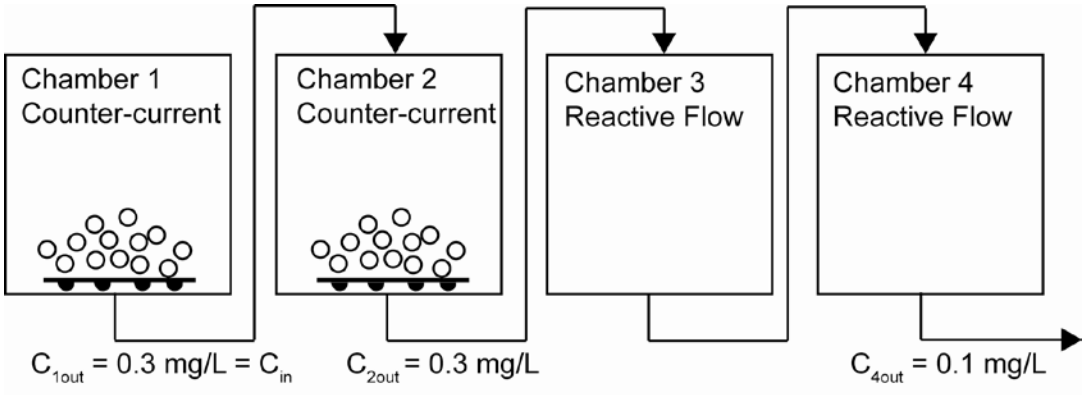
Chamber 2: Log inactivation = Log(1 + 2.303×0.101×0.15×20) = 0.23
Chamber 3: Log inactivation = Log(1 + 2.303×0.101×0.20×20) = 0.28

3) Sum the log inactivation values to determine the total log credit achieved.

The total log-inactivation across the contactor is 0.23 + 0.28 = 0.51 log inactivation, which is greater than the target of 0.5 logs. Therefore, 0.5 log credit is achieved.

***Example - CT Calculation and Log Credit Determination using the CSTR Method with the concentration <u>not</u> measured for each chamber***

A system employs a four chamber ozone contactor, with ozone addition in the first two chambers. The second chamber is a counter-current flow dissolution chamber with influent and effluent ozone concentrations of $C_{in}$ = 0.3 mg/L and $C_{out}$ = 0.3 mg/L. The effluent ozone concentration in the third, reactive chamber is unknown, and in the fourth, reactive chamber is 0.1 mg/L. At 10 °C, $k_{10}$ = 0.101 L/mg-min. The HDT for each chamber = 20 minutes. Chambers 3 and 4 are considered a single zone, and the effluent concentration of Chamber 3 is assumed to be equal to that of Chamber 4.




1) Determine the C values for each chamber

Chamber 1: No *Cryptosporidium* inactivation credit recommended
Chamber 2: $C = C_{2out}/2 = 0.3 / 2 = 0.15$ mg/L
Chamber 3: $C = C_{4out} = 0.1$ mg/L
Chamber 4: $C = C_{4out} = 0.1$ mg/L

2) Calculate the log inactivation for each chamber using Equation 11-4

Chamber 2: Log inactivation = Log(1 + 2.303×0.101×0.15×20) = 0.23
Chamber 3: Log inactivation = Log(1 + 2.303×0.101×0.10×20) = 0.17
Chamber 4: Log inactivation = Log(1 + 2.303×0.101×0.1×20) = 0.17

3) Sum the log inactivation values to determine the log credit achieved.

The total log-inactivation across the contactor is 0.23 + 0.17 + 0.17 = 0.57 log inactivation. Therefore, the minimum 0.5 log credit is achieved.

### 11.3.4 Extended $T_{10}$ and Extended CSTR Methods

The Extended $T_{10}$ and Extended CSTR methods require the measurement of the ozone concentration at a minimum of three points within the contactor. These data are used to predict an ozone concentration profile through the contactor. The Extended methods generally result in greater CT credit and hence lower doses of ozone needed to achieve the same calculated level of inactivation, when compared to the direct $T_{10}$ or CSTR method. Appendix B provides a complete description of the extended methods and how they are applied to an ozone contactor.

## 11.4 Monitoring Requirements

### 11.4.1 LT2ESWTR

The LT2ESWTR (40 CFR 141.720(a)) requires **daily CT monitoring** conducted **during peak hourly flow**. Since systems may not know when the peak hour flow will occur, EPA recommends monitoring on an hourly basis. Systems should re-evaluate contact time whenever they modify a process and the hydraulics are affected (e.g., add a pump for increased flow, reconfigure piping).

The concentration of ozone is measured with the indigo methods, Standard Method 4500-O3 B and Standard Method 4500 O3 B-97. Details on these methods can be found in *Standard Methods for the Examination of Water and Wastewater*, 20th edition, American Public Health Association, 1998. Appendix C provides information on sample collection, preparation and stability of reagent, and calibration and maintenance of online monitors.

### 11.4.2 Stage 1 DBPR and Stage 2 DBPR

The Stage 1 DBPR requires all systems using ozone for disinfection or oxidation to take at least one bromate sample per month for each treatment plant using ozone (See the Stage 1 DBPR, 40 CFR 141.132(b) for further information). Samples must be taken at the distribution system entry point when the ozone system is operating under normal conditions. Under the Stage 2 DBPR (40 CFR 141.132 (b) (3) (ii) (B), beginning April 1, 2009, systems may reduce monitoring from monthly to quarterly if the system demonstrates that the running annual average

raw water bromide concentration is less than 0.0025 mg/L, based on monthly measurements for the most recent four quarters. Systems that were allowed to reduce bromate monitoring to quarterly prior to April 1, 2009, may remain on quarterly monitoring if the running annual average raw water bromide concentration is less than 0.0025 mg/L. The MCL for bromate is 10 µg/L based on a running annual average.

## 11.5 Unfiltered System LT2ESWTR Requirements

The LT2ESWTR requires unfiltered systems to meet the following requirements (40 CFR 141.712(b), (c) and (d)):

- Provide at least 2.0 log *Cryptosporidium* inactivation.
- If their source water *Cryptosporidium* concentration is greater than 0.01 oocyst/liter, then the system must provide 3.0 log *Cryptosporidium* inactivation.
- Use a minimum of two disinfectants to meet overall disinfection requirements.

The requirements of the previous SWTR regulations still apply—achieve 3 log inactivation of *Giardia* and 4 log inactivation of viruses, and maintain a disinfectant residual in the distribution system (e.g., free chlorine or chloramines).

The monitoring requirements described in section 11.4 apply to unfiltered systems. Additionally, unfiltered systems must meet the *Cryptosporidium* log-inactivation requirements every day the system serves water to the public, except one day per calendar month (40 CFR 141.712(c)). Therefore, if an unfiltered system fails to meet *Cryptosporidium* log-inactivation two days in a month, it is in violation of the treatment technique requirement.

## 11.6 Toolbox Selection

Selecting ozone disinfection to receive *Cryptosporidium* inactivation credit for compliance with the LT2ESWTR has cost, operational, and upstream and downstream process implications. The ozone CT requirements for *Cryptosporidium* inactivation are significantly higher than those for *Giardia* and virus, and capital requirements could be substantial for a system seeking higher than 0.5 log credit. As a result, ozone is likely a better option for systems that will benefit from its other treatment effects. This section discusses the potential advantages and disadvantages of ozone processes.

### 11.6.1 Advantages

Ozonation reduces many other contaminants and improves process performance, both directly and indirectly. The indirect benefits are those where other aspects of the treatment process can be improved or changed, resulting in a higher finished water quality. The advantages of ozone use include:

- Total organic carbon (TOC) reduction.
- Iron, manganese, and sulfide oxidation.
- Taste, odor, and color control.
- Lower formation of trihalomethanes (THMs) and haloacetic acids (HAAs) upon post chlorination due to precursor removal and generally lower chlorine doses.
- Biological stability when ozonation is followed with biological filtration.

### 11.6.2 Disadvantages

Considering only benefits from *Cryptosporidium* inactivation credit, the capital, operational, and maintenance costs are relatively high compared to other toolbox options for similar credit, especially for systems treating colder water. Other disadvantages include:

- Higher level of maintenance and operator skill required.
- Additional safety and containment issues with ozone contactors.
- Possible need for three-phase power which may not be compatible with some water systems.
- Bromate formation (bromate is a regulated disinfection byproduct (DBP)).
- Potential enhanced formation of other unregulated DBPs either from ozonation alone (e.g. formaldehyde) or upon secondary chlorination/chloramination (e.g. chloropicrin).
- Upstream processes can cause fluctuations in ozone demand, thus affecting ozone residual control.
- Assimilable organic carbon (AOC) production, which can contribute to biofilm growth in the distribution system if not removed.
- High capital requirements to achieve CT requirements with low water temperatures (below 10 $^{o}$C).

## 11.7 Disinfection With Ozone

### 11.7.1 Chemistry

The stability of ozone upon dissolution into natural waters is governed by both the direct reaction of ozone with various constituents in the water, as well as the so-called "auto catalytic chain decomposition" reaction. The general behavior in the dissolved ozone concentration over time has been described by numerous investigators (e.g. Park et al. 2001; Elovitz et al. 1999, 2000) as a two-stage process: an initial rapid consumption step (ozone consumed after a few seconds to 30 seconds) followed by a slower ozone decay step which can often be described by first-order kinetics. Various water quality parameters, including temperature, pH, DOC concentration, DOC character, and alkalinity, as well as the ozone dose can affect the amount and rate of ozone consumed in these two stages (Elovitz et al. 2000; Park et al. 2001; Buffle et al. 2006). For example, increasing ozone dose can increase the amount of ozone consumed in the initial reaction phase; however, the rate of ozone decay in the second phase is slower. This is of importance for considerations to ozone dose requirements in order to maintain dissolved ozone for sufficient disinfection (i.e. CT requirements).

One of the consequences of the spontaneous decomposition of ozone during water treatment is the generation of hydroxyl free radicals (Hoigné and Bader 1983a, 1983b; Glaze et al. 1987). The hydroxyl free radicals are among the most reactive oxidizing agents in water, with most reaction rates on the order of $10^{8}$ - $10^{10}$ $M^{-1}$ $s^{-1}$. Because of their high reactivity, the half-life of hydroxyl free radicals is on the order of microseconds. The formation of hydroxyl radicals is affected by the same water quality parameters that affect ozone decomposition (see above), and is also vastly different in the initial reaction phase versus the secondary reaction phase (Elovitz et al. 1999, 2000; Buffle et al. 2006). Under typical conditions for ozonation of drinking water source waters, the transient concentration of the hydroxyl free radicals can reach as high as $10^{-11}$ M during the first 20 milliseconds of the initial reaction phase (Buffle et al. 2006). During the secondary phase, hydroxyl radical concentrations are typically on the order of $10^{-13}$ to $10^{-14}$ M (Elovitz et al. 1999, 2000) and reach levels above $10^{-12}$ M under typical ozonation conditions (Glaze and Kang 1988). Despite these low transient concentrations, the hydroxyl free radical can be a very important reactant for the oxidation of constituents that are slow reacting with respect to molecular ozone. Under certain conditions, an ozonation process can be operated with the intent of creating an "Advanced Oxidation Process" (AOP) which purposefully enhances the formation of hydroxyl free radicals from the decomposition of ozone. While the enhanced formation of hydroxyl radicals can lead to enhanced oxidation of certain micropollutants, it will also lower the overall ozone CT. The application of AOPs is discussed further in Chapter 7 of the *Alternative Disinfectants Guidance Manual* for information on advanced oxidation processes).

Exhibit 11.5 illustrates the major pathways that develop during ozonation:

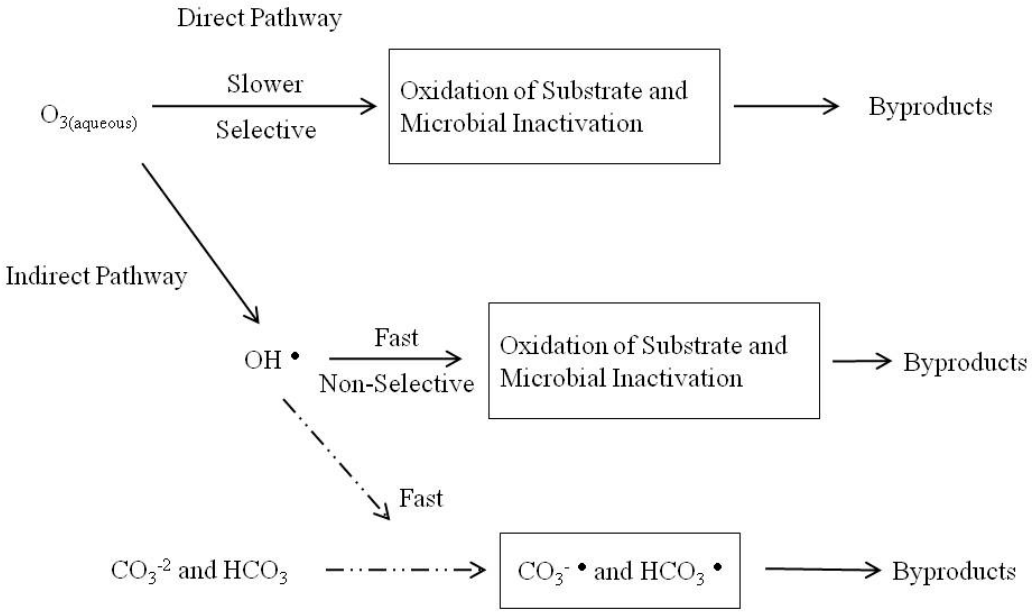
- Direct oxidation of compounds by molecular ozone in the aqueous phase.

- Oxidation of compounds by hydroxyl free radicals produced during the decomposition of ozone.

As indicated in Exhibit 11.5, the direct reaction with molecular ozone is relatively slow compared to the hydroxyl reaction. However, the reaction with many aqueous species is still very rapid compared to other disinfectants. In addition, the concentration of ozone is orders of magnitude greater then the concentration of hydroxyl radicals. Hence, when assessing the role of the two pathways, both the intrinsic reaction rate as well as the concentration of the two oxidants must be considered.

The reaction of hydroxyl free radicals with carbonate and bicarbonate produces carbonate and bicarbonate radicals. These free radicals may also participate in the oxidation of chemical and microbial species. However, these reactions tend to be selective and, with some exception, their contribution to micropollutant oxidation is negligible compared to the two main pathways.

**Exhibit 11.5 Reaction Pathways of Ozone in Water**



### 11.7.2 Byproduct Formation

Reactions between ozone and NOM can form a variety of organic byproducts including aldehydes, ketones, and acids. Inorganic byproducts are also formed. Bromide reacts with ozone and hydroxyl radicals to form bromate, a regulated drinking water contaminant with an MCL of 10 µg/l. Oxidation of bromide to hypobromous acid during ozonation, and subsequent reaction of he hypobromous acid with natural organic matter can lead to the formation of brominated organic compounds, such as bromoform and dibromoacetic acid, which are also regulated through the total trihalomethanes (TTHMs) and haloacetic acids (HAA5) MCLs under the Stage 2 DBPR.

#### 11.7.2.1 Bromate and Brominated Organic Compounds

Bromate and brominated organic compound formation is dependent on water quality and treatment conditions, and only occurs in waters with bromide ion present. Bromate formation occurs through a very complex, nonlinear mechanism involving both ozone and hydroxyl free radical pathways. Bromate formation generally increases with increasing pH, carbonate alkalinity, ozone dose, and temperature, and perhaps most importantly, bromide concentration. However, attempts at reducing bromate formation by lowering pH may increase the formation of brominated organic byproducts. Other methods for minimizing bromate formation during ozonation include ammonia addition (Hoffman et al. 2001), and more recently the so-called Chlorine-Ammonia Process (Buffle et al. 2004). Overall, the source water bromide concentration is a very important factor when considering adding ozone to a treatment process. Source waters with bromide concentrations greater than 50 ppb likely need to consider the possibility of significant bromate formation (von Gunten, 2003)

#### 11.7.2.2 Non-Brominated Organic Compounds

Ozone reacts with NOM and breaks larger organic molecules down into simpler, more biodegradable compounds such as aldehydes, ketones, and acids. These biodegradable organic molecules are a food source for microorganisms and can affect biological growth in the distribution system. Escobar and Randall (2001) conducted a case study at a ground water treatment plant that was adding ozone to improve the aesthetic quality of the water. They found that the AOC (the fraction of TOC that is most readily utilized by bacteria) concentrations significantly increased in the distribution system however, with diligent maintenance of chlorine residual biological growth was suppressed. Biofilters can be used to reduce the AOC entering the distribution system. (Section 11.8.3 describes biofilters and their operation.)

## 11.8 Design

### 11.8.1 Generators and Contactors

There are several types of ozone generators and contactors. All generators use oxygen as a raw material and convert it to ozone using electrochemical reactions. They differ from each other in the source of oxygen used and the configuration of generator elements. Generators can use either air or pure oxygen as an oxygen source. The *Alternative Disinfectants and Oxidants Guidance Manual* describes the type of generators and contactors in detail.

### 11.8.2 Point of Addition

Raw water quality, turbidity, and ozone demand are commonly used to assess the possible locations for adding ozone. The *Alternative Disinfectants and Oxidants Guidance Manual* describes the water quality characteristics, advantages, and disadvantages of feed points at a raw water location, after sedimentation, and after first-stage filtration of a two-stage process. The general considerations are:

- Integrating ozone into overall disinfection strategy for the treatment facility.
- Placing the ozone addition point further downstream, particularly after physical removal processes, generally reduces both the ozone demand and byproduct formation.
- Adding ozone ahead of filtration allows any biodegradable organics, formed from the ozonation of more recalcitrant TOC, to be removed by subsequent biological activity in the filters. Also, solid-phase manganese and iron formed through oxidation by ozone can be removed by the filters.

In general, applying ozone prior to coagulation can enhance clarification. Applying prior to filtration can also improve filtration performance; however these effects are site-specific and are likely to depend on ozone dose.

Detrimental impacts on filtration operation have also been reported. Bishop et al. (2001) investigated the effects of ozone on filtration with a raw water of moderate turbidity, TOC, iron, and manganese concentrations. With ozone doses of 0.5 to 1.0 mg/L, turbidity increased in the contactors with visible floc formation. At lower ozone doses, 0.16 to 0.35 mg/L, the turbidity still increased, but not as much as the higher ozone dose. Because of the higher filter loadings, the duration of filter cycles decreased. The authors believed the increased turbidity was partially due to solid-phase manganese formation, and also likely due to the organic matter and residual metals.

### 11.8.3 Biologically Active Filters

When ozone oxidizes organic matter, the AOC in the water typically increases. Some systems use biologically active filters to remove the AOC prior to chlorination and entry to the distribution system. Microbes present in the upper portion of the filters consume the AOC, mineralizing them to carbon dioxide and water, and reducing the amount available to microorganisms in the distribution system (e.g., microorganisms in pipeline biofilm) and for DBP formation.

#### 11.8.3.1 Media for Biologically Active Filters

Any filter media that has sufficient surface area for microbes to attach to can be used for biological filtration. Slow sand, rapid sand, and granular activated carbon (GAC) filters have all been successfully used for biologically active filtration. Research indicates that both sand/anthracite and sand/GAC filters can support the total amount of biomass to sufficiently remove organic components (LeChevallier et al. 1992, Krasner et al. 1993, Coffey et al. 1995). Wang and Summers (1996) and Zhang and Huck (1996) have shown that the contact time with the biofilm is more important than the mass of biofilm above a minimum level of biomass. Generally, the longer the contact time the greater the removal of AOC. However, the increase in removal is not a linear relationship—the removal rate decreases at extended contact times (Zhang & Huck 1996). DBP precursors most often take longer to biodegrade, making extended contact times necessary if this is the process goal. This can be achieved with deep anthracite filter beds or GAC filters (Prevost et al. 1990). The adsorption capacity of GAC provides a longer time for the organic compounds to be consumed by the biomass as the particles are adsorbed by the GAC (LeChevallier et al. 1992).

#### 11.8.3.2 Operating Biologically Active Filters

It is not necessary to seed a biological filter in order to obtain the necessary biological growth. The organisms naturally present in the system are sufficient to obtain the needed growth. The only additional requirement is to provide the conditions for biological growth. These conditions include necessary food sources, sufficient dissolved oxygen, nutrients, proper pH and temperature. The products from ozone and NOM reactions will provide the needed food for the microorganisms to grow. The reaction of ozone also produces oxygen as one of its products, so the dissolved oxygen concentration should be sufficiently high. Generally, the pH and nutrient levels in most waters will also be sufficient to allow the necessary growth. Organic removal will generally be higher at higher temperatures. Several studies have found significantly decreased removal at temperatures below 15 degrees Celsius (Krasner et al. 1993, Coffey et al. 1995, Daniel and Teefy 1995).

In order to maintain biological growth, a disinfectant other than ozone cannot be added prior to the filters. GAC filters can reduce small disinfectant residuals through reaction with the carbon; however, this can lead to physical breakdown of the GAC and more frequent media replacement. Using chlorinated or chloraminated backwash water can also be a concern. Studies

have shown mixed results with chlorinated backwash water, with some showing no effect and others showing significantly reduced removal (Miltner et al. 1996, Miltner et al. 1995, Hacker et al. 1994, Reckhow et al. 1992, McGuire et al. 1991). Short vigorous backwashes with a relatively low chlorine dose may be more effective in maintaining biological filtration than less vigorous backwashes at longer times with higher chlorine doses (Urfer et al. 1997).

## 11.9 Safety Considerations in Design

Ozone is a corrosive gas and according to Occupational Safety and Health Administration (OSHA) Standards, exposure to airborne concentrations should not exceed 0.1 ppm (by volume) averaged over an eight-hour work shift.

Ozone generators should be housed indoors for protection from the environment, and to protect personnel from leaking ozone in the case of a malfunction. Ventilation should be provided to prevent excess temperature rise in the generator room, and to exhaust the room in the case of a leak. Adequate space should be provided to remove the tubes from the generator shell and to service the generator power supplies. Off-gas destruct units can be located outside if the climate is not too extreme. If placed inside, an ambient ozone detector should be provided in the enclosure. All rooms should be properly ventilated, heated, and cooled to match the equipment-operating environment.

## 11.10 Operational Considerations

When using ozone for disinfection, it is important to evaluate all the factors that could affect the CT achieved. For example, if raw water quality fluctuates and ozone demand increases, the residual concentrations will decrease unless ozone dose is adjusted. The system is now at risk of not achieving the required level of CT. The ozone demand, pH, and temperature of the raw water, under worst-case to best-case conditions, should be evaluated to determine their effect on ozone disinfection. Systems should develop standard operating procedures (SOPs) for addressing changes in raw water quality. The remainder of this section discusses how these factors affect ozone disinfection and the CT calculation.

### 11.10.1 Ozone Demand

The following water quality constituents contribute to ozone demand:

- Natural organic matter (NOM)—Ozone will oxidize organic matter, which includes compounds causing taste and odor. As discussed in section 11.8.2 organic byproducts are also produced.

- Synthetic organic compounds (SOCs)—Some SOCs can be oxidized and mineralized under favorable conditions.

- Bromide—Ozone will oxidize bromide forming, hypobromous acid, hypobromite ion, bromate ion, brominated organics, and bromamines.

- Bicarbonate or carbonate ions—The hydroxyl radical reacts with bicarbonate and carbonate ions and form carbonate radicals.

Ozone demand is particularly important to the CT calculation since it directly affects the residual ozone used in the CT calculation. Ozone concentrations in water are generally monitored continuously using an on-line residual monitor, and confirmed periodically with a bench top instrument. As the ozone demand changes, the amount of ozone applied can be adjusted to maintain the desired CT.

### 11.10.2 pH

The pH of water does not have a significant effect on ozone disinfection capabilities. However, there is strong impact of pH on ozone demand and decay rate. As pH increases, ozone decomposition increases, and there is a concomitant increase in the formation of the hydroxyl radical. Under the pH range typically used for drinking water treatment (e.g. pH 6 – 9), the initial demand may be only increased by increasing pH due to changing pH speciation of compounds that react directly with ozone (Buffle et al, 2006b). However, increasing pH has a significant effect on the secondary reaction phase due to increased contribution of the autocatalytic chain decomposition (Elovitz et al., 2000; Buffle et al, 2006b).

### 11.10.3 Temperature

Like all chemical reactions, the reaction rate between ozone and a pathogen increases with increasing temperature. The CT requirements are based on temperature; as temperature decreases, the CT required to achieve a given level of inactivation increases. Conversely, the rate of ozone decay decreases as temperature decreases, generally resulting in a higher CT for a given ozone dose and hydraulic resident time. The ozone process should be designed to provide the necessary log inactivation under all conditions. Standard operating procedures (SOPs) should also describe process adjustments required to operate at the lowest water temperatures experienced by the system in the past 10 years.

### 11.10.4 Maintaining Residual Disinfectant in the Distribution System

It is necessary to maintain a residual in the distribution system to prevent microbial regrowth. Because of the reactive nature of ozone, its residual tends to dissipate within minutes and cannot be relied upon to maintain a disinfectant throughout the distribution system. Therefore, a secondary disinfectant should be used, usually either chlorine or chloramine.

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# 12. Demonstration of Performance (DOP)

## 12.1 Introduction

The purpose of the "demonstration of performance" (DOP) toolbox component is to allow a system to demonstrate that a plant, or a unit process[1] within a plant, should receive a higher *Cryptosporidium* treatment credit than is presumptively awarded under the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). Presumptive treatment credits are applicable to any physical removal process that complies with the provisions of the Interim Enhanced Surface Water Treatment Rule (IESWTR) and Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR) (40 CFR 141.552). Exhibit 12.1 gives some examples of typical plants that could be eligible for a DOP credit. DOP credits can be granted for any process, including inactivation processes; however, this chapter is limited to a discussion of physical removal processes. Membrane processes receiving DOP credit must still meet challenge testing and direct integrity testing requirements as specified in 40 CFR 141.719.

### Exhibit 12.1 Example Filtration Plant Types Eligible for DOP

| Plant Type | Minimum Elements of Process Train |
|---|---|
| Conventional | Coagulation/Flocculation, Sedimentation, High Rate Granular Media Filtration |
| Slow Sand Filtration | Slow Sand Filtration |
| Diatomaceous Earth | Diatomaceous Earth Filtration |
| Softening/Granular Media Filtration | Single Stage Lime Softening, High Rate Granular Media Filtration |
| Direct Filtration | Coagulation/Flocculation, High Rate Granular Media Filtration |

Where a system can demonstrate that a plant, or a unit process within a plant, consistently achieves a *Cryptosporidium* treatment efficiency greater than the presumptive credit specified in the LT2ESWTR, the state may allow the system to receive a higher *Cryptosporidium* treatment credit for compliance with the LT2ESWTR (40 CFR 141.718(c)). To demonstrate the higher level of *Cryptosporidium* treatment, systems must conduct a site-specific study using a protocol approved by the state. This study must account for all expected operating conditions and, at the discretion of the state, determine ongoing monitoring and/or performance requirements to ensure conditions under which the DOP was awarded are maintained during routine operations.

In general, the term "treatment" in the LT2ESWTR refers to both physical removal and inactivation of *Cryptosporidium*. Treatment credits discussed in this chapter pertain to physical removal.

[1] A system would conduct a DOP of a unit process while ensuring the other parts of the treatment process were achieving their assumed *Cryptosporidium* treatment. For example, maximizing removal in a pre-sedimentation basin can cause reduced removal in the subsequent sedimentation basin and filters.

This chapter provides guidance for implementing the DOP toolbox option and is organized as follows:

12.2 LT2ESWTR Compliance Requirements - discusses DOP treatment credit with respect to other toolbox options and reporting requirements.

12.3 Toolbox Selection Considerations - describes selection considerations for plants to consider before conducting a DOP study, the duration of a DOP study, and an approach for conducting a DOP study.

12.4 DOP Criteria Development - discusses key issues of DOP design including process evaluation criteria, selection of performance indicators, and full-scale versus pilot-scale testing.

12.5 Demonstration Protocol - discusses the minimum elements that should be included in the DOP protocol - DOP test matrix, DOP monitoring plan, DOP implementation, and data analysis and reporting.

## 12.2 LT2ESWTR Compliance Requirements

### 12.2.1 Credits

The LT2ESWTR does not specify how treatment performance must be demonstrated; however the protocol used must be approved by the state (40 CFR 141.718(c)). Determination of an increased *Cryptosporidium* treatment credit will be made by the state.

The LT2ESWTR does not allow systems to claim presumptive credit for the toolbox options listed below, if that component is included in the DOP credit (40 CFR 141.718(c)(1)).

- Presedimentation
- Two-stage lime softening
- Bank filtration
- Combined or individual filter performance
- Membrane filters
- Bag and cartridge filters
- Second stage filtration

For example, if a plant receives a DOP credit for a treatment train, the system may not also receive credit for a presedimentation basin or achieving the lower finished water turbidity of the combined filter performance option.

States may award a lower level of *Cryptosporidium* treatment credit towards compliance for the LT2ESWTR to a system where, based on site-specific information, a plant or a unit process achieves a *Cryptosporidium* treatment efficiency less than a presumptive credit specified in the LT2ESWTR (40 CFR 141.718(c)).

### 12.2.2 Reporting Requirements

The LT2ESWTR requires results from the testing be submitted no later than the *Cryptosporidium* compliance date (40 CFR 141.721):

- Schedule 1 – April 1, 2012
- Schedule 2 – October 1, 2012
- Schedule 3 – October 1, 2013
- Schedule 4 – October 1, 2014

The state may require systems to report operational data on a monthly basis to verify that conditions under which DOP credit was awarded are maintained during routine operation (40 CFR 141.721).

## 12.3 Toolbox Selection Considerations

The DOP toolbox option is intended for plants that operate at a high level of performance. A system should review existing performance data to verify that it can meet high performance levels under a range of operating conditions (including filters out of service, returning to service, and flow rate changes) before conducting a DOP study. EPA recommends systems achieve less than 0.1 nephelometric turbidity units (NTU) in each individual filter effluent as an indicator for considering whether the DOP option is practical.

Before applying the DOP approach to an individual unit process, facilities should carefully consider the potential advantages and disadvantages of such an approach. The microbial toolbox allows for treatment credits for unit processes based on specified design and/or operational criteria described in other chapters of this manual. It is possible that a detailed DOP program may result in a lower credit than already granted by the LT2ESWTR.

A DOP study should address the range of operating conditions (e.g., flow rates, chemical and disinfection practices and dosages) and seasonal raw water quality variations based on a

review of plant operating records and historical water quality records. If source or operating conditions are expected to change (e.g., turbidity events, high watershed runoff, increased system demands) these should be addressed in the DOP study. Systems should have a contingency plan for achieving compliance with the LT2ESWTR if the DOP does not provide the anticipated credit.

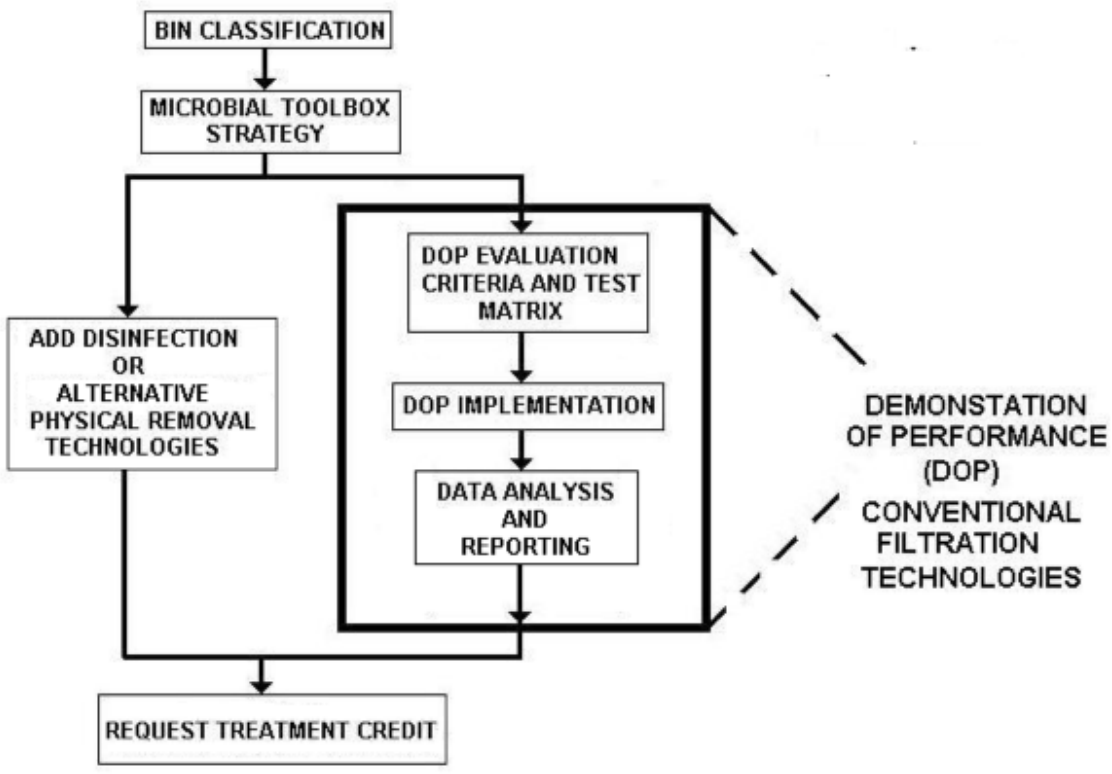
### 12.3.1 Overview of the Demonstration Protocol

This chapter presents one approach for conducting a DOP study. Other approaches or modifications to this approach may be approved by the state. Major elements of the DOP protocol include the following:

- Development of DOP evaluation criteria and test matrix.
- DOP implementation.
- Data analysis and reporting.

Exhibit 12.2 presents a flowchart relating these elements to the overall microbial toolbox framework. Each of these topics is discussed in detail in this chapter.

**Exhibit 12.2 Flowchart for DOP Protocol**



## 12.4 DOP Criteria Development

Source water *Cryptosporidium* levels and water quality characteristics vary from system to system. Accordingly, DOP programs should be tailored to address site-specific process issues associated with each water treatment plant (WTP). Major questions that should be resolved during the design of the DOP include (but are not limited to) the following:

- What are the governing process evaluation criteria and treatment objectives?
- What microorganism or surrogate parameter(s) should be used to demonstrate removal efficiency of *Cryptosporidium*?
- Should the DOP be conducted at full-scale or pilot-scale?

Each of these questions is addressed in the following sections.

### 12.4.1 Process Evaluation Criteria

Process evaluation encompasses the treatment objectives of the plant, influent water quality, system demand, and operating conditions or treatment techniques. The DOP plan should address all critical operating conditions, whether conducted in full-scale or pilot-scale. Influent water quality, flow rates, process configurations, and operating conditions need to be clearly defined during the development of the DOP plan. Common process evaluation criteria are discussed in this section.

#### 12.4.1.1 Treatment Objectives

The DOP toolbox option primarily relates to *Cryptosporidium* removal by physical methods such as clarification and filtration. However, WTPs are tasked to remove or control multiple contaminants in the source water besides *Cryptosporidium*. The impact of operational strategies and treatment methods for other contaminants on the efficiency of *Cryptosporidium* removal should be considered during the DOP criteria development stage. The system should not change its operational strategy between the DOP study conditions and routine operation after the study has ended—the DOP credit is based on the operational strategy used in the study. For example, a system that uses enhanced coagulation throughout the study period should also use it during routine operation for compliance with the LT2ESWTR.

Other examples of treatment techniques that can affect *Cryptosporidium* removal and thus should be considered in the development stage include the following:

- Prechlorination may be used to enhance floc formation (and *Cryptosporidium* removal) in filtration trains. However, prechlorination may also promote trihalomethane (THM) and haloacetic acid (HAA) formation. Therefore, prechlorination doses used during the DOP study should be set to balance floc and disinfection byproduct formation. Operational guidelines should be documented in the DOP plan.

- Granular media filter run times may be extended to increase unit filter run volumes (UFRVs) and filter efficiency. However, increased UFRVs also increase the potential for *Cryptosporidium* breakthrough. Maximum UFRVs should be established to minimize *Cryptosporidium* breakthrough.

- Alternative coagulation strategies may be used to enhance *Cryptosporidium* removal in granular media filters but may also result in post-filtration flocculation that can cause deposition or scaling in water distribution systems. Coagulant dosing rates should be set during the DOP study to minimize downstream floc formation.

Additionally, if a treatment process or plant technique is used intermittently for a seasonal or sporadically occurring contaminant, this treatment should also be used as needed during the DOP study, consistent with routine operation.

#### 12.4.1.2 Influent Water Quality Characteristics

Source water quality characteristics that may affect *Cryptosporidium* removal efficiencies should be identified. These will depend on the treatment processes employed and may include (but are not limited to) turbidity, pH, alkalinity and temperature. Critical (or worst-case) ranges for these parameters that are anticipated over the plant design life or permit period should be clearly defined. The demonstration study should include tests run under the worst-case source water conditions. In pilot-scale DOP studies, raw source water can be modified to simulate worst-case water qualities.

#### 12.4.1.3 System Flow Rate

The system flow rate or range of flow rates to be evaluated during the DOP should be clearly defined. Where possible, plant performance should be demonstrated for the critical flow condition that defines permitted plant capacity (e.g., peak instantaneous flow or peak daily flow). For full-scale studies, this may not be feasible for facilities that operate significantly below permitted or maximum capacity. For pilot-scale studies, the range of system unit process flow rates should replicate the full-scale low, intermediate, and maximum flow and recycles rates.

#### 12.4.1.4 Plant Operating Conditions

WTP operations can vary significantly over the course of the demonstration period due to various factors including, but not limited to, raw water quality, system flow rate, and maintenance activities. The critical operating conditions that may impact *Cryptosporidium* removal at the WTP should be defined. Issues to consider include the following:

- What are the normal and worst-case operating conditions for each unit process with respect to *Cryptosporidium* removal?
- How many process trains or elements are normally in service? How will the plant perform when units are out of service for maintenance and repair, thereby increasing unit process flow rates (particularly in filters)?
- What is the process control strategy for chemical addition? How does this relate to *Cryptosporidium* removal?
- What is the process control strategy for filter operations? How does this relate to *Cryptosporidium* removal?
- How will the plant's recycle, backwash, and filter-to-waste schemes affect *Cryptosporidium* removal?

In the case of pilot-scale studies, performance demonstrations should replicate full-scale operating conditions in any respect that may influence *Cryptosporidium* removal.

### 12.4.2 Selection of Performance Indicators

Although the LT2ESWTR mandates treatment controls for *Cryptosporidium*, it is not currently feasible to demonstrate actual *Cryptosporidium* removal at full-scale facilities. In most cases, influent *Cryptosporidium* levels are not consistently high enough to demonstrate significant (such as 4 log) removal across the process train. Raw water spiking of *Cryptosporidium* is not a feasible option at full-scale facilities due to the potential health risk to system users and the number of oocysts required. Consequently, alternative indicators of *Cryptosporidium* removal will be needed for facilities that plan to conduct DOP studies at full-scale.

#### 12.4.2.1 Surrogate Parameters for *Cryptosporidium*

EPA has reviewed a number of studies that suggest aerobic bacteria spores are a suitable indicator of *Cryptosporidium* removal in conventional treatment trains (coagulation, flocculation, sedimentation and filtration). Some characteristics of aerobic spores (as summarized by Cornwell et al. 2001) are:

- Naturally occurring (Nieminski and Bellamy 2000, Jakubowski et al. 1996).
- Do not pose health risks (Jakubowski et al. 1996, Rice et al. 1996).
- Can be detected at low concentrations (< 1 cfu/100 mL).
- Are slightly smaller than *Cryptosporidium* oocysts (Rice et al. 1996).
- Spore removal by water treatment is a conservative indicator of *Cryptosporidium* removal (Rice et al. 1996, Dugan et al. 1999, Nieminski and Bellamy 2000, Emelko 2001).
- Reduction of indigenous spores by inactivation is expected to be negligible in comparison with removal of spores by physical processes (Jakubowski et al. 1996, Rice et al. 1996).
- Aerobic spores do not undergo re-growth during treatment.

Although aerobic spores appear to be a suitable indicator for *Cryptosporidium* removal in filtration plants, raw source water spore concentrations will likely not be high enough throughout the study period to demonstrate high log removal across a full-scale treatment train.

Microspheres may also be used as an alternative indicator for *Cryptosporidium* removal. Microspheres are chemically inert, easy to handle, and relatively inexpensive. They can be manufactured with a uniform particle size and smooth surface, making them appropriate as a conservative indicator. They can also be manufactured without a significant surface charge to minimize particle interaction and loss to processes such as adsorption. Microspheres are easily obtainable in concentrations of $10^7$ to $10^9$ particles per mL, which should be adequate to prove desired removal rates.

The biggest disadvantage of the use of microspheres is in detection methods. Particle counters are effective in counting microspheres. In fact, microspheres are often used to calibrate particle counters. However, particle counters may not be able to distinguish between microspheres and other particles and may not be able to distinguish conglomerated particles. Other problems with particle counters such as coincidence error and the limited dynamic range can also skew results. A more effective method of measurement involves capturing the microspheres by filtering the sample and then counting microspheres. Use of fluorescent microspheres can aid in the counting process (Abbaszadegan et al. 1997, Li et al. 1997). Microspheres also tend to have lower zeta potentials than live Cryptosporidium oocysts (Dai and Hozalski 2003). Recent work, however, has found ways to adjust the surface charge on microspheres to more closely mimic natural pathogens (Pang et al. 2009).

If appropriate detection methods are used and the microspheres are conservative representatives of *Cryptosporidium* oocysts, microspheres can be a good surrogate for

*Cryptosporidium*. Emelko and Huck (2004) found that 4.6 micron carboxylated fluorescent dyed polystyrene microspheres acted as a good indicator for *Cryptosporidium* over a wide range of log removals. For example; neutrally charged, 1 micron, spherical latex microspheres could provide an acceptable conservative indicator for *Cryptosporidium* removal.

The state may accept alternative indicators for *Cryptosporidium*; however, they should not be more easily removed than *Cryptosporidium*. The surrogate parameter should give a direct view of removal and should be an element that is not created in the plant (e.g., particle counts caused by chemical precipitation). Furthermore, the method of measurement should be sensitive enough to detect temporal variations in the parameter. Parameters such as turbidity or particle counts may be used in the DOP study, but are not suitable as stand-alone surrogates.

#### 12.4.2.2 Long-Term Performance Indicators

As discussed previously, plants that implement a DOP plan should document long-term performance of filtration facilities for turbidity and/or particle count reduction. While turbidity and particle counts are not suitable as stand-alone indicators for full-scale *Cryptosporidium* removal, such data can be used to identify changes in the filtration performance.

It is recommended that individual filter efficiency be monitored frequently to identify differences in individual filter performance. This will allow the plant to assess temporal variations in filter effluent quality and will provide improved process control.

### 12.4.3 Full-Scale Versus Pilot-Scale Testing

In general, full-scale testing is preferred over pilot-scale testing since the performance of existing process trains is demonstrated directly. However, full-scale studies may not be feasible for many facilities for the following reasons:

- Influent *Cryptosporidium* levels will not be high enough to demonstrate high log removal. Likewise, influent aerobic spore concentrations may not be high enough to demonstrate significant log removal.
- Full-scale spiking with aerobic spores may not be feasible due to larger flows.
- Facilities may operate well below design or permitted flow capacity for the entire study period.
- Demonstration of worst-case operating conditions at full-scale may be difficult to plan, especially with regard to raw water quality and flow rates.

The major concern with the use of pilot-scale testing is the uncertainty associated with scale-up of pilot results to predict the performance of full-scale systems. Other potential limitations of pilot-scale studies are:

- Pilot-scale data generally represent steady-state conditions; however, sudden changes in flow or water quality may have a significant effect on *Cryptosporidium* removal; such changes are difficult to capture in a pilot-scale plant.

- Pilot-scale plants generally have much tighter process controls and higher levels of attention than full-scale plants; and thus, may not be indicative of actual full-scale performance.

- A pilot-scale plant cannot represent expected individual differences between multiple filters in a full-scale plant.

- Particle loadings to the treatment process in a pilot-scale study may be much higher than actual full-scale loadings, and thus, may not represent actual operating conditions.

- It may be too difficult to construct a pilot plant that represents the entire full-scale process train.

Pilot system dimensions and flow rates should be sufficiently large to minimize scale-up issues. Some recommended guidelines for pilot filter sizing include the following (U.S. EPA 1991):

- Unit filtration rate in the pilot system should be identical to that of the full-scale plant.

- Pilot filter diameter should be greater than or equal to 100 times the media diameter.

- Media diameter and depth should be identical to that of the full-scale system.

Pilot systems should also incorporate all major process elements of the full-scale process train, including chemical addition systems and recycle streams. Such systems must be able to simulate flow rate and water quality perturbations (i.e., temporal disturbances to steady state conditions).

## 12.5 Demonstration Protocol

Once the DOP criteria have been developed, the DOP protocol can be formulated. This section outlines the minimum elements that should be included in the DOP protocol. Participation from the governing regulatory agency should be solicited during the DOP protocol development phase.

### 12.5.1 DOP Test Matrix

The first step in the formulation of the specific DOP protocol is the development of a matrix of test conditions to be evaluated during the DOP period. These test conditions should be formulated to assess *Cryptosporidium* removal (or other suitable parameters) under a range of normal and worst-case scenarios. The DOP matrix should clearly define specific test scenarios to be evaluated, incorporating the following criteria:

- Source water quality ranges– including minimum/maximum limits for critical water quality parameters that influence *Cryptosporidium* removal in the plant.

- Influent flow rates– including the maximum flow rate that defines plant capacity.

- Operating scenarios– including all operations that may cause process upset in the treatment train (e.g., events that cause temporal changes in water quality, and flow loadings to process units). These operations include, but are not limited to: filter backwashing, filter-to-waste practices, intermittent recycles, returning filters to service, and routine maintenance practices.

Critical influent flow ranges and operating conditions should be identified during the DOP criteria development phase, as described in section 12.2. The demonstration period should be at least one year, and should encompass all critical operating conditions. An example test matrix format is presented in Exhibit 12.3.

**Exhibit 12.3 Example DOP Test Matrix**

| Scenario Condition | (Normal or Worst-Case) Influent Concentration Range Flow | Influent Concentration Range | | Flow Rate Range | Units in Service | Backwash Conditions | Date of Scenario Test |
|---|---|---|---|---|---|---|---|
| | | Surrogate | Turbidity | | | | |
| S1 | Normal | Average | Average | Average | 4 (All) | | |
| S2 | Normal | Average | Average | Average | 3 | | |
| S3 | Worst Case A | Average | Average | High | 3 | | |
| S4 | Worst Case B | High | High | Average | 3 | | |
| S5 | Worst Case C | Low | Low | Average | 3 | | |

### 12.5.2 DOP Monitoring Plan

The DOP involves sampling and analysis of *Cryptosporidium* indicators in the raw source water and filtration train effluent over the course of a demonstration period defined by the DOP test matrix. Once the test matrix is established, the DOP monitoring plan should be formulated to define the following protocol details:

- Monitoring locations.
- Test parameters (field and laboratory).
- Monitoring frequency.
- Quality assurance/quality control (QA/QC) procedure for/during sampling.

A sample DOP monitoring plan is presented in Exhibit 12.4

## Exhibit 12.4 Example DOP Monitoring Plan

| Monitor Event Number | Date | Test Scenario ID (See Exhibit 12.2) | Effluent Sample Locations[A] | | | | Number of Samples per Location | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Filter 1 | Filter 2 | Filter 3 | Filter 4 | Crypto/ Aerobic Spores | Particle Count | pH | Temp. |
| 1 | Week 1 | S1 | X | X | X | X | 2[B] | 1 | 1 | 1 |
| 2 | Week 2 | S3 | X | X | X | X | 1 | 1 | 1 | 1 |
| 3 | Week 3 | S2 | X | X | X | X | 1 | 1 | 1 | 1 |
| … | ... | … | ... | … | ... | … | … | … | … | … |
| 52 | Week 52 | S4 | X | X | X | X | 2[B] | 1 | 1 | 1 |

A - Influent sample location identical for all test scenarios.
B - Duplicate samples.

#### 12.5.2.1 Sampling Location

Paired samples should be collected from the plant influent (raw source sample) and the combined filter effluent for a DOP study of an entire plant. The plant influent location should be before the pre-sedimentation basins and off-stream storage facilities and follow any process recycles added prior to the first major unit process element of the treatment train. For pilot studies involving microbial dosing, the influent monitoring point should follow complete mixing of the source water and injection stream. The plant effluent sample should be comprised of composite samples from the effluent of all operating filters. It is recommended that at least five sample pairs (influent/effluent) be collected during each test run to capture temporal changes in filter and effluent quality.

#### 12.5.2.2 Monitoring Parameters

Samples should be analyzed for all parameters required to assess *Cryptosporidium* removal in the treatment trains, as discussed in section 12.2. Parameters such as pH, alkalinity, temperature, and turbidity should be measured and recorded in the field.

#### 12.5.2.3 Monitoring Frequency

A monitoring event is defined as a paired (concurrent) sampling of plant influent and filter effluent samples. At a minimum, monitoring should be performed once per week for 52 consecutive weeks. More frequent monitoring may be required to capture all critical operating scenarios defined by the DOP Test Matrix. The DOP database should be sufficiently large to allow for statistical analysis.

If a DOP credit is issued by the state, the credit will be conditional on continuing demonstration of a higher level of performance. The DOP Monitoring Plan can be modified to document continuing performance at a reduced sampling frequency. However, sampling events should still capture critical operating scenarios.

#### 12.5.2.4 Quality Assurance/Quality Control (QA/QC)

Quality assurance/quality control (QA/QC) sampling should be performed to allow assessment of data variability and quantification errors due to sample collection procedures and analytical methods. At a minimum, duplicate samples should be collected during one monitoring event per month.

### 12.5.3 DOP Implementation

The DOP should commence only after the state approves the DOP test matrix and monitoring protocol. The DOP plan should be administered by a qualified WTP operator or water process engineer. Data review and QA/QC practices should be conducted routinely to ensure that the objectives of the DOP program are met. Particular attention should be given to verification of the plant operating conditions (influent loadings, unit process loadings, etc.) to confirm that all critical operating scenarios identified in the DOP test matrix are evaluated during the demonstration period.

Personnel responsible for implementing the DOP monitoring plan should be properly trained in sample collection techniques, QA/QC procedures and operational data acquisition. Specific procedures should be used to collect and analyze samples as described in the following sections:

- Sample collection and preservation methods.
- Analytical methods.
- Microbial dosing methods (for pilot tests).
- Documentation procedures.

#### 12.5.3.1 Sample Collection Methods

Influent and effluent samples should be collected in a manner that is representative of the entire cross sectional flow at each monitoring point. If possible, monitoring points should be located in straight sections of pipe or channel well downstream of bends. For open channel flows, samples should be collected from mid-depth and mid-width of the channel. For pipe flow, samples should be collected from the tap directly into the sample containers. In each case, the sampling method should not reduce or prevent transfer of suspended solids from the process stream to the sample container. Parameters such as pH, turbidity, alkalinity and temperature should be directly measured in the field.

#### 12.5.3.2 Analytical Methods

The analytical methods for monitoring *Cryptosporidium* under the LT2ESWTR are prescribed at 40 CFR 141.704 and described in the Public Water System Guidance Manual for Source Water Monitoring under the LT2ESWTR. Analytical methods for all other water quality parameters should be performed in accordance with Standard Methods for the Examination of Water and Wastewater, 20th edition, or the most recent edition.

### 12.5.3.3 Microbial Dosing

For pilot testing that involves spiking of *Cryptosporidium*, aerobic spores or other indicators, microbial dosing procedures should be clearly established. Guidelines for microbial stock preparation and dosing are presented in this section.

A concentrated mixture of microorganisms should be prepared and fed to the raw source stream at a known feed rate, based on the microbial density in the concentrated stock, the flow rate of the pilot system, and the desired microorganism concentration in the pilot system. An equation that describes this relationship is:

Equation 12-1

$$Q_{feed} = \left( \frac{C_{pilot}}{C_{feed} - C_{pilot}} \right) Q_{pilot}$$

where:

$C_{pilot}$ = the microbial concentration in the pilot system
$C_{feed}$ = the microbial concentration in the concentrated stock solution
$Q_{pilot}$ = the flow rate of the pilot system (includes all process recycles present at the influent feed point, if applicable)
$Q_{feed}$ = the flow rate of the concentrated stock solution

For each trial, the test microorganisms should be completely mixed in a volume of raw water sufficient to supply the pilot plant for the duration of the experiment. The tank containing the suspension of test microorganisms should be continuously mixed for the duration of each experiment to promote homogeneity of the mixture. The concentrated stock should be delivered by a positive displacement pump (e.g., peristaltic) to the main process flow at a flow rate dictated by Equation 12-1. $C_{pilot}$ and $C_{feed}$ should be selected to provide a high enough influent microbial concentration to demonstrate at least 4 log removal in the pilot system. Based on this approach, $C_{pilot}$ should be set at least $10^4$ higher than the method detection limit for the test microorganism. The microbial density in the stock solution should be sampled at least twice, and preferably three times, during a feeding interval to verify consistent densities.

### 12.5.3.4 Documentation of WTP Operating Conditions

It is important to document WTP operating conditions during monitoring events to evaluate the effect of varying operating scenarios on *Cryptosporidium* removal. Standardized reporting forms should be developed to provide, at a minimum, the following information:

- System flow rate (instantaneous/flow chart, hourly and daily average).
- Operating mode (process scheme, number of trains, number of units in service).

- Water pH, alkalinity, turbidity and temperature.
- Performance data.
- Chemical addition rates/doses.
- Mechanical equipment in operation, with flow rates (major pumps, blowers, etc.).
- Recycle and backwash flows/rates.
- Related maintenance activities occurring prior to or during sampling event.

### 12.5.4 Data Analysis and Reporting

#### 12.5.4.1 Evaluation of Performance

To receive DOP treatment credits above presumptive credits in the LT2ESWTR, a plant should demonstrate consistent attainment of a specific log reduction of *Cryptosporidium* (or suitable indicators). To meet this objective, log reduction should first be computed for each monitoring event according to:

$$\text{Log Removal} = -\log (C_{inf}/C_{eff}) \qquad \text{Equation 12-2}$$

where: $C_{inf}$ = influent *Cryptosporidium* or indicator concentration
$C_{eff}$ = effluent *Cryptosporidium* or indicator concentration

For effluent samples in which no *Cryptosporidium*, spores, or other indicators are detected, the concentration should be set to the method detection limit.

The state will determine the level of DOP credit a facility receives based on review of the log removal data.

For the case of pilot testing and the use of multiple indicators for *Cryptosporidium* removal calculations will be site specific.

#### 12.5.4.2 Reporting for the DOP

At the conclusion of the DOP test period, a detailed report summarizing the major findings of the DOP program must be submitted to the governing regulatory agency. At a minimum, the DOP report should include the following information:

- Detailed description of full-scale WTP, including process flow schematics.
- Summary of treatment objectives and WTP design criteria.
- DOP test matrix and monitoring plan.
- DOP data summary.
- Detailed pilot plant design data (if applicable).
- Data analysis for estimate of *Cryptosporidium* log reduction.
- Appendices for raw full-scale/pilot-scale analytical and operational data.
- Monitoring plan to verify that on-going performance is equivalent to treatment credit. Source water indicators used in the study should be monitored to ensure performance is met.
- Plan for addressing operating conditions (e.g., influent water turbidity) out of the range tested in the study. The DOP test matrix generally sets the range of operating conditions under which the LT2ESWTR treatment credit is applicable. Therefore, it is advisable to develop a plan for addressing potential out-of compliance conditions. For example, if the influent source water quality conditions ranged from 5 NTU to 25 NTU during the study, the system may plan to make operational adjustments for influent water with turbidity greater than 25 NTU and increase filter effluent monitoring. Any such deviations would be reported to the state.

#### 12.5.4.3 Ongoing Reporting

As discussed previously, if a DOP credit is issued by the state, the credit will be conditional on continuing demonstration of a high level of performance. The DOP Monitoring Plan should be modified to document continuing performance at a reduced sampling frequency, while still capturing critical operating conditions. States may require systems receiving a DOP credit to report operational and progress monitoring data on a routine basis. Operational data should verify that continuous process control and optimization procedures are in place.

The DOP credit is applicable to minimum and maximum raw source water and finished water quality limits defined in the DOP Test Matrix. Routine reporting should be performed to verify that plants operate within these limits. If an exception occurs, it should be reported to the state in a timely manner. Frequent exceptions may prompt the state to require the plant to conduct a comprehensive performance evaluation (CPE) or similar operational evaluation to identify causes and solutions for exceptions.

## 12.6 References

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# 13. Ultraviolet Light

## 13.1 Introduction

The use of ultraviolet (UV) light for disinfection of drinking water is a relatively new application in the United States, although used for decades in the wastewater industry. UV disinfection is the process of irradiating water with UV light. The UV light is absorbed by the genetic material of microorganisms, damaging it, and preventing the microorganisms from reproducing. UV disinfection has been found to be particularly effective against protozoa and bacteria.

This chapter summarizes the requirements for water systems using UV disinfection to achieve compliance with the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) and provides considerations for toolbox selection. Water systems and states should refer to the *UV Disinfection Guidance Manual* (U.S. EPA 2006) for detailed guidance on design and operation of UV systems and the validation testing that must be conducted for compliance with the LT2ESWTR.

## 13.2 UV Disinfection Requirements for Filtered and Unfiltered PWSs

The LT2ESWTR has several requirements related to the use of UV disinfection, they address:

- UV doses for different levels of inactivation credit.
- Performance validation testing of UV reactors.
- Monitoring.
- Reporting.
- Off-specification operation.

### 13.2.1 UV Dose and Validation Testing Requirements

EPA developed UV dose requirements for PWSs to receive credit for inactivation of *Cryptosporidium*, *Giardia*, and viruses (Exhibit 13.1). The UV dose values in Exhibit 13.1 are applicable only to post-filter applications of UV disinfection in filtered systems and to unfiltered systems meeting turbidity requirements.

Unlike chemical disinfectants, UV light does not leave a chemical residual that can be monitored to determine UV dose and inactivation credit. The UV dose depends on the UV intensity (measured by UV sensors), the flow rate, and the UV absorbance. To determine the operating conditions under which the reactor delivers the required dose for treatment credit, the

LT2ESWTR requires PWSs to use UV reactors that have undergone validation testing [40 CFR 141.720(d)(2)]. These operating conditions must include flow rate, UV intensity as measured by a UV sensor, and UV lamp status.

### Exhibit 13.1 UV Dose Requirements – millijoules per centimeter squared ($mJ/cm^2$)[1]

| Target Pathogens | Log Inactivation | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | 0.5 | 1.0 | 1.5 | 2.0 | 2.5 | 3.0 | 3.5 | 4.0 |
| ***Cryptosporidium*** | 1.6 | 2.5 | 3.9 | 5.8 | 8.5 | 12 | 15 | 22 |
| ***Giardia*** | 1.5 | 2.1 | 3.0 | 5.2 | 7.7 | 11 | 15 | 22 |
| **Virus** | 39 | 58 | 79 | 100 | 121 | 143 | 163 | 186 |

[1] 40 CFR 141.720(d)(1).

Validation testing must meet the following requirements:

- Validated operating conditions must account for UV absorbance of the water, lamp fouling and aging, measurement uncertainty of online sensors, UV dose distributions arising from the velocity profiles through the reactor, failure of UV lamps or other critical system components, and inlet and outlet piping or channel configurations of the UV reactor [40 CFR 141.720(d)(2)(i)].

- Validation testing must involve full-scale testing of a reactor that conforms uniformly to the UV reactors used by the PWS, and it also must demonstrate inactivation of a test microorganism whose dose-response characteristics have been quantified with a low-pressure mercury vapor lamp [40 CFR 141.720(d)(2)(ii)].

- Using the above requirements as a basis, EPA developed a recommended validation protocol, presented in Chapter 5 of the *UV Disinfection Guidance Manual* (U.S. EPA 2006). Water systems are not required to follow this protocol but may follow alternatives that achieve compliance with the regulatory requirements as long as they are acceptable to the state. Also, states may have additional requirements than are provided in the LT2ESWTR.

### 13.2.2 UV Disinfection Monitoring Requirements

The LT2ESWTR requires PWSs to monitor their UV reactors to demonstrate that they are operating within the range of conditions that were validated for the required UV dose. At a minimum, PWSs must monitor each reactor for flow rate, lamp status, UV intensity as measured by a UV sensor, and any other parameters required by the state. UV absorbance should also be measured when it is used in a dose-monitoring strategy. PWSs must verify the calibration of UV sensors and recalibrate sensors in accordance with a protocol the state approves [40 CFR 141.720(d)(3)(i)].

### 13.2.3 UV Disinfection Reporting Requirements

The LT2ESWTR requires PWSs to report the following items [40 CFR 141.721(f)(15]:

- Initial reporting – Validation test results demonstrating operating conditions that achieve the UV dose required for compliance with the LT2ESWTR.

- Routine reporting – Percentage of water entering the distribution system that was not treated by the UV reactors operating within validated conditions on a monthly basis.

### 13.3.4 Off-specification Operational Requirement for Filtered and Unfiltered Systems

To receive disinfection credit for UV disinfection, both filtered and unfiltered PWSs must treat at least 95 percent of the water delivered to the public during each month by UV reactors operating within validated conditions for the required UV dose [40 CFR 141.720(d)(3)(ii)].

## 13.3 Toolbox Selection Considerations

UV disinfection is a relatively simple to use and highly effective technology for inactivating *Cryptosporidium*. Its main advantages include:

- It can inactivate chlorine-resistant pathogens such as *Cryptosporidium* oocysts and *Giardia* cysts at relatively low doses. It is often the lowest cost treatment option for inactivating *Cryptosporidium*.

- It does not produce regulated disinfection byproducts (DBPs).

- Its effectiveness is not pH or temperature dependent.

The disadvantages of UV disinfection include:

- UV disinfection effectiveness cannot be measured in "real-time" like chemical disinfectants.

- UV disinfection provides no distribution system residual.

- Much higher UV doses are required for virus inactivation.

- Power quality problems can disrupt disinfection in some cases.

## 13.4 Design and Operational Considerations

UV reactors for drinking water treatment typically consist of a closed-vessel containing UV lamps, UV sensors, and temperature sensors. UV lamps are usually housed within lamp sleeves to protect and insulate them. Some reactors include automatic cleaning mechanisms to keep the lamp sleeves free of deposits. UV sensors, flow meters, and in some cases, analyzers for UV absorbance (or a related parameter, UV transmittance) are used to determine the dose delivered by the reactor. UV lamps can be low pressure, low pressure-high output, or medium pressure mercury vapor lamps. Low pressure lamps emit light at one wavelength (i.e., monochromatic) and operate with the mercury under low vapor pressures. Low pressure high-output lamps are similar to low pressure lamps but operate at higher temperatures and have a higher UV light output. Medium-pressure lamps are polychromatic and operate at higher temperatures and mercury vapor pressures.

Below is an example of some key design and operational issues that should be considered when evaluating UV treatment options. Refer to the *UV Disinfection Guidance Manual* (U.S. EPA 2006) for more detailed guidance on UV facility design and operation.

- Water quality – The UV absorbance of the water to be treated is very important in the design of UV facilities. This is because UV absorbance influences UV dose delivery and therefore affects the UV reactor selection, validation requirements, and the UV facility size and cost. Compounds in the water can also foul lamp sleeves and other UV reactor components. Fouling is dependent on calcium, hardness, alkalinity, lamp temperature, pH, oxidation-reduction potential (ORP) and certain inorganic constituents (e.g., iron and manganese). UV facilities are typically equipped with cleaning systems to prevent fouling.

- Power quality – UV lamps can turn off if a voltage fluctuation, power quality anomaly, or power interruption occurs. Power quality tolerances depend on the UV equipment design and vary significantly among UV manufacturers. If power quality may be a problem at the intended installation location, a power quality assessment may be needed to quantify and understand the potential for off-specification operations.

- Hydraulic needs and limitations - Headloss through a UV reactor depends on the specific reactor, piping configuration, and flow rate. Typical headloss ranges from 0.5 to 3.0 feet for a reactor.

- Maintenance - UV reactors will need to be periodically shut down for regular maintenance. Typical maintenance tasks include checking UV sensor calibration, checking lamp cleaning efficiency, and replacing lamps and sleeves.

- LT2 Reporting Requirements - Systems are required to submit validation test results demonstrating operating conditions that achieve required UV dose as well as with a monthly report summarizing the percentage of water entering the distribution system that was not treated by UV reactors operating within validated conditions for the required dose.

## 13.5 References

U.S. EPA. 2006. *Ultraviolet Disinfection Guidance Manual for the Final Long Term 2 Enhanced Surface Water Treatment Rule*. Office of Water. EPA 815-R-06-007. November, 2006. http://www.epa.gov/safewater/disinfection/lt2/compliance.html.

# 14. Membrane Filtration

## 14.1 Introduction

The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) defines membrane filtration as follows:

> *Membrane filtration is a pressure or vacuum driven separation process in which particulate matter larger than 1 micrometer is rejected by an engineered barrier, primarily through a size-exclusion mechanism, and which has a measurable removal efficiency of a target organism that can be verified through the application of a direct integrity test. This definition includes the common membrane technologies of microfiltration, ultrafiltration, nanofiltration, and reverse osmosis. [40 CFR 141.2]*

Membrane processes that meet the requirements of LT2ESWTR will receive *Cryptosporidium* removal credit.

The U.S. Environmental Protection Agency (EPA) recently published the *Membrane Filtration Guidance Manual* for systems considering using membranes to comply with the requirements of the LT2ESWTR (U.S. EPA 2005). Readers interested in detailed information on membrane filtration should consult the *Membrane Filtration Guidance Manual*. This chapter summarizes rule requirements and lists advantages and disadvantages of membrane filtration compared with other toolbox technologies.

## 14.2 Membrane Filtration Requirements under the LT2ESWTR

In order to receive removal credit for *Cryptosporidium* under the LT2ESWTR, a membrane filtration system must meet the following three criteria:

1. The process must comply with the definition of membrane filtration as stipulated by the rule.

2. The removal efficiency of a membrane filtration process must be established through a product-specific challenge test and ongoing, site-specific direct integrity testing during system operation.

3. The membrane filtration system must undergo periodic direct integrity testing and continuous indirect integrity monitoring during operation.

The rule does not prescribe a specific removal credit for membrane filtration processes. Instead, removal credit is based on system performance as determined by challenge testing and verified by direct integrity testing. Thus, the maximum removal credit that a membrane filtration process may receive is the lower value of either [40 CFR 141.719(b)(1)]:

- The removal efficiency demonstrated during challenge testing.

**OR**

- The maximum log removal value that can be verified by the direct integrity test used to monitor the membrane filtration process.

Based on this framework, a membrane filtration process could potentially meet the Bin 4 *Cryptosporidium* treatment requirements, as shown in Exhibit 1.1 of this guidance manual.

These primary elements of the regulatory requirements for membrane filtration under the LT2ESWTR, including challenge testing, direct integrity testing, and continuous indirect integrity monitoring, are summarized in the following sections.

### 14.2.1 Challenge Testing

Since there are no uniform design criteria that can be used to ensure the removal efficiency of a membrane process, challenge testing is required to demonstrate the ability of a membrane process to remove a specific target organism. The removal efficiency demonstrated during challenge testing establishes the *maximum* removal credit that a membrane process would be eligible to receive, provided that this value is less than or equal to the maximum log removal value that can be verified by the direct integrity test [40 CFR 141.719(b)(1)], as described in the next section. The LT2ESWTR only requires product-specific challenge testing; once the removal efficiency has been demonstrated, additional testing is not required unless the product is significantly modified.

### 14.2.2 Direct Integrity Testing

While challenge testing can demonstrate the ability of an integral membrane process to remove the target organism, integrity breaches can develop in the membrane during routine operation that could allow the passage of microorganisms. In order to verify the removal efficiency of a membrane process during operation, direct integrity testing is required for all membrane filtration processes used to comply with the LT2ESWTR [40 CFR 141.719(b)(3)]. A direct integrity test is defined as a physical test applied to a membrane unit in order to identify and isolate integrity breaches. The rule does not mandate the use of a specific type of direct integrity test, but rather performance criteria that any direct integrity test must meet. These criteria include requirements for resolution, sensitivity, and frequency [40 CFR 141.719(b)(3)]:

- **Resolution**: The direct integrity test must be applied in a manner such that a 3 micrometer breach contributes to the response from the test.

- **Sensitivity**: The direct integrity test must be capable of verifying the ability of a membrane filtration system to achieve the log removal value awarded to the process by the state.

- **Frequency**: The direct integrity test must be applied at a frequency of at least once per day, although less frequent testing may be permitted by the state at its discretion if appropriate safety factors are incorporated.

A control limit must also be established for a direct integrity test, representing a threshold response which, if exceeded, indicates a potential integrity problem and triggers subsequent corrective action. For the purposes of LT2ESWTR compliance, this threshold response must be indicative of an integral membrane unit capable of achieving the *Cryptosporidium* removal credit awarded by the state.

### 14.2.3 Continuous Indirect Integrity Monitoring

Systems must conduct continuous indirect integrity monitoring on each membrane unit [40 CFR 141.719(b)(4)]. For the purposes of the LT2ESWTR, indirect integrity monitoring is defined as monitoring some filtrate water parameter that is indicative of the removal of particulate matter, and "continuous" is defined as monitoring at a frequency of no less than once every 15 minutes [40 CFR 141.719(b)(4)(ii)]. Although turbidity monitoring is specified as the default method of continuous indirect integrity monitoring under the rule, other methods, such as particle counting or particle monitoring, may be used in lieu of turbidity monitoring at the discretion of the state [40 CFR 141.719(b)(4)(i)]. For any indirect method used, a control limit must be established that is indicative of acceptable performance. Monitoring results exceeding the control limit for a period of more than 15 minutes must trigger immediate direct integrity testing [40 CFR 141.719(b)(4)(iv)].

## 14.3 Toolbox Selection Considerations – Advantages and Disadvantages

Membrane filtration is a highly efficient technology for removing pathogens and other particulates from drinking water. Its main advantages are:

- Removes bacteria and protozoa.

- Can lower DBPs by allowing lower disinfectant doses and removing DBP precursors.

- Can remove arsenic. Microfiltration (MF) and ultrafiltration (UF) can remove particulate arsenic (dissolved arsenic can be converted to particulate arsenic by coagulation prior to the MF/UF system). NF and reverse osmosis (RO) can remove dissolved arsenic.

- UF, NF, and RO can remove viruses, however it is very difficult to perform a direct integrity test that can detect a defect as small as a virus.

Membrane filtration is an advanced technology and can be more expensive than conventional technologies. Its major disadvantages are:

- Total cost may exceed that of conventional technologies.
- Can be fouled by organics and minerals.
- Increased loss of process water.

## 14.4 Design and Operational Considerations

There are a number of different types of membrane materials and module system designs for different classes of membranes. In general, MF and UF use hollow-fiber membranes, while NF and RO use spiral-wound membranes. Hollow-fiber membrane systems may be either pressure-driven (i.e., positive pressure as a driving force for filtration) or vacuum-driven (i.e., utilizes negative pressure as a driving force for filtration). In pressure-driven systems, upstream pumps are employed to push water across the membrane barrier. Vacuum-driven systems employ downstream pumps to induce suction on the inside the membrane fibers, pulling water across the barrier. Membrane systems using spiral-wound modules are pressure-driven, with six to eight modules usually arranged in series inside a containment vessel.

Membrane systems are typically designed and constructed in one or more discrete water production units, also called racks, trains, or skids. Production unit design varies widely by manufacturer and type of system (i.e., hollow-fiber vs. spiral-wound) but typically contains the membrane treatment system, associated piping, appurtenances, and other features. A typical membrane treatment system is composed of a number of identical units that combine to produce the total filtrate flow.

A major design variable for membrane systems is the flux, or the flow per unit of membrane area. Membranes are most often designed to operate at constant flux (or within a specific range fluxes, with the applied pressure (i.e., positive or negative) varying with the degree of resistance to flow. This resistance may be caused by fouling or changes in temperature, which affect water viscosity. Because the flux can vary significantly with temperature, the average, minimum, and maximum temperature of the water to be treated should be considered when designing the system. Pilot studies are often performed to optimize the flux, pretreatment, and cleaning regime (chemicals, doses, and intervals) for a particular application.

Core membrane process operations include backwashing, chemical cleaning, and integrity testing. The frequency of these processes is usually determined during pilot testing, but in the case of integrity testing may also be dictated by regulatory requirements. Backwashing is similar in principle to that for conventional media filters and is intended to remove contaminants accumulated on the membrane surface. Note that backwashing is only applicable to the microporous MF/UF membranes, but does not apply to the semi-permeable NF/RO membranes, which cannot be backwashed. Chemical cleaning is periodically conducted to remove any accumulated foulants; for MF/UF systems, this constitutes any fouling that is not removed on a routine basis via backwashing. Integrity testing is conducted to ensure that the membrane is free of any breaches, leaks, or defects that might allow unfiltered water to bypass the membrane

barrier. This testing is required by many states and at the federal level for applications in which membrane filtration is used to comply with the *Cryptosporidium* removal requirements of the LT2ESWTR.

Feed water quality is also a primary design consideration for membrane systems, as this can affect both the flux and rate of membrane fouling. For MF/UF systems, high levels of turbidity and TOC can increase backwashing requirements and chemical cleaning frequencies, causing poor performance and shortening membrane life. Additionally for NF/RO systems, high levels of scaling ions can increase energy consumption and chemical cleaning frequency and can result in poor performance and shortened membrane life. In many cases, pretreatment may improve feed water quality at lower cost than incorporating additional membrane area. Other important issues to consider in the design of membrane filtration systems include cross connection control, system reliability, chemical cleaning and residuals management.

## 14.5 References

U.S. EPA. 2005. *Membrane Filtration Guidance Manual*. Office of Water. EPA 815-R-06-009. November, 2005. http://www.epa.gov/ogwdw/disinfection/lt2/compliance.htm.

# Appendix A
# Site Specific Determination of Contact Time for Chlorine Dioxide and Ozone

A water system may perform a site specific study to generate a set of chlorine dioxide or ozone contact time (CT) values for that site if it believes those developed by the U.S. Environmental Protection Agency (EPA) do not reflect the true inactivation achieved. Such a study would involve measuring actual *Cryptosporidium* inactivation under site conditions, with a full range of temperature and contact times. If accepted by the state, the CT values may be used instead of those developed by EPA.

The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) does not specify any requirements for the chlorine dioxide or ozone site-specific study, only that it be approved by the state (40 CFR 141.720(c)). This appendix describes the different recommended elements of a study and discusses some of the issues involved in the statistical analysis of the results.

## A.1 Experimental Design

Experiments should be conducted with water that is representative of the water to be treated with respect to all conditions that can affect *Cryptosporidium* inactivation. Inactivation experiments should be performed with water exerting the highest oxidant demand (i.e., spring run-off or summer conditions) at high temperature to obtain the worst-case scenario in terms of chlorine dioxide or ozone demand/decay rate. In addition, experiments should also be conducted with water obtained during the winter months at the lowest temperatures observed at the treatment plant. These experiments would allow for the determination of the highest CTs that would be necessary to achieve the required level of inactivation. Additional experiments may be necessary to characterize the effects of other water quality parameters.

> *In order to obtain the most challenging water to assess the chlorine dioxide or ozone process, a predetermined testing schedule should be established based on source water TOC and $UV_{254}$ levels. Testing can occur when source water values for these parameters fall within defined worst-case ranges. Experiments should then be performed in the laboratory at worst-case temperatures for a given month.*

In order to obtain a complete data set, testing should occur at least every other month over the course of an entire year. Each sample date should be determined by the first time the total organic carbon (TOC) or $UV_{254}$ levels are within 75 percent of the maximum historical value for that month. At the time of sampling, sufficient water should be acquired to allow for three sets of experiments to be conducted, with each experiment having six data points (CT values) and a control. Two independent sets of experiments should be conducted with the water. Should significant discrepancies develop between the data sets, a third set of experiments would need to be conducted. An example experimental matrix is provided in Exhibit A.1.

**Exhibit A.1 Example Experimental Test Matrix**

| Date | Temperature to be Tested (Historical Record) | Water Quality Criteria | Schedule of Experiments | | |
|---|---|---|---|---|---|
| | | | Test 1 | Test 2 | Test 3 |
| February | Lowest Annual | TOC or $UV_{254}$ > 75% of max historical value | X | X | If Required |
| April | Highest in April | Same | X | X | If Required |
| June | Highest in June | Same | X | X | If Required |
| August | Highest Annual | Same | X | X | If Required |
| October | Highest in October | Same | X | X | If Required |
| December | Lowest in December | Same | X | X | If Required |

## A.2 Experimental Procedure

### A.2.1 Preparation of Oocysts

High oocyst quality is imperative to the success of the study because sub-standard oocysts could dramatically affect the data in a way that would underestimate the CT required to achieve a desired level of inactivation. Traditionally, *Cryptosporidium parvum* oocysts are derived from two host sources, bovine and rodent. The most common strain of *Cryptosporidium parvum* used to date is the Iowa strain, developed by Dr. Harley Moon. It is recommended that the utility perform all experiments using fresh (< 1 month old) Iowa-strain oocysts obtained from a reputable supplier. The utility should ensure that after purification the supplier stores the oocysts at 4$^{o}$C in a solution of dichromate or 0.01 M phosphate buffer saline solution (pH 7.4) containing two antibiotics (1,000 U/mL penicillin, and 1,000 mg/mL streptomycin), and an antimycotic (2.5 mg/mL amphotericin B). The oocysts should be shipped in a cooler on ice to the utility via next-day service. Upon arrival, the oocysts should be placed in a refrigerator and stored at 4 degrees C until needed.

When ready for use, the oocysts should be suspended in 0.01 M pH 7 buffer and centrifuged at a relative centrifugal force of approximately 1,100 for at least 10 minutes. Following centrifugation, the oocysts should be aspirated and re-suspended in the buffer, then centrifuged again at the same conditions. This step should be repeated once more to remove as much of the antibiotic or dichromate solution as possible. Following the last aspiration, the oocysts should be re-suspended in approximately 10 mL of the pH 7 buffer. The oocysts should then be stored at 4$^{o}$C until the experiment is initiated. The oocysts should be vortexed thoroughly prior to initiation of the experiment. Additional details regarding this procedure can be found in Rennecker et al. 1999.

### A.2.2 Source Water Preservation

Testing should be conducted as close as possible to the date that the experimental water is collected. If testing is to be performed at a location other than the utility where the water was collected, the water should be sent to the laboratory via an overnight delivery service and stored at 4 degrees Celsius until the start of testing.

### A.2.3 Experimental Apparatus

#### A.2.3.1 Chlorine Dioxide

It is recommended that chlorine dioxide be generated using the equipment and procedures outlined in Standard Methods for the Examination of Water and Wastewater (APHA 1998). With this as a basis, all inactivation experiments using chlorine dioxide should be performed using a batch-reactor configuration. An example of such a system is provided by Ruffell et al. 2000. This system uses an enclosed recirculating water bath to maintain the desired temperature inside the reactor vessels, which consist of 2-liter amber glass bottles. During the experiment, care should be taken to minimize the exposure of the reactors to light. Mixing of the reactor contents is provided with a magnetic stir bar and stir plate.

#### A.2.3.2 Ozone

Inactivation experiments can be performed with either a semi-batch or batch reactor configuration. When performing experiments with a semi-batch system, it is recommended that analytical components similar to those described by Hunt and Mariñas (1997) be used. Using this system, the reactor vessel containing the experimental water is maintained at the experimental temperature by immersion in a water bath. Ozone can be generated from either compressed air or oxygen and passed through a continuously-stirred glass bottle, which serves to dampen the effect of fluctuating ozone concentration. The ozonated gas leaving the dampening bottle is then introduced to the experimental water via a fine-bubble diffuser. The ozonated water is stirred continuously using a magnetic stirring plate and a stir bar.

It is recommended that inactivation experiments performed using a batch reactor configuration use analytical components similar to those described by Kim (2002). This reactor used a 100-mL gas-tight syringe to prevent ozone in solution from volatilizing into the atmosphere. The temperature inside the reactor is held constant by immersion in a recirculating water bath, and mixing is provided by a stir bar in the syringe controlled by a magnetic stir plate. Ozone can be produced from either compressed air or oxygen. A concentrated ozone stock solution should be prepared using distilled de-ionized or reverse osmosis-filtered water.

Other, less complex, batch reactor systems are also available which simply use an open vessel such as an Erlenmeyer flask or beaker (Finch et al. 1993a). With these systems, the reactor containing the experimental water is typically maintained at the desired temperature using a water bath. An ozonated solution, prepared with distilled de-ionized or reverse osmosis water, is added to the experimental water, and the ozone dose is measured from the diluted experimental water. When using this type of batch-reactor configuration that is open to the atmosphere, the user should take into account that ozone is lost to volatilization. This loss of ozone should be considered and minimized when performing any inactivation or demand/decay experiment.

### A.2.4 Inactivation experiments

The CT values obtained from each of the site-specific inactivation experiments are expected to be similar to those provided in the standard LT2ESWTR tables. Therefore, utilities wishing to determine site-specific inactivation data are advised to use the standard tables as a baseline. Each experiment should be designed such that six data points span the range of the standard inactivation curve for a given temperature. One control point with no disinfectant should also be taken.

#### A.2.4.1 Chlorine Dioxide

An experimental protocol developed from Ruffell et al. 2000 is provided here as an example. The reactor bottle should be filled with experimental water to a total volume corresponding to the desired sample volume times the number of samples expected per bottle (6 is recommended). The bottle is then placed in the water bath and allowed to equilibrate to the target experimental temperature. At this point, chlorine dioxide stock solution is added to the reactor bottle at the target dose. The reactor bottle is then capped to minimize chlorine dioxide volatilization. The chlorine dioxide concentration is measured approximately 10 min after dosing. An experiment was conducted by adding approximately a pre-determined number of oocysts to the reactor that will be sufficient for at least six data points. Note the volume of the oocyst aliquot should be less than 1 mL. Samples are then taken periodically at the contact times that correspond to the desired CT. The samples are immediately filtered through a 1μm filter. The filter is then placed in a clean 50 mL beaker and rinsed with approximately 15 mL of the dilute surfactant. The resulting oocyst suspension is transferred into a sterile 15 mL centrifuge tube.

These steps are repeated at various contact times corresponding to target CT parameters. After the last sample is taken, the chlorine dioxide dose is measured again. “Control” samples are also taken for each experiment by placing a sample of oocysts inside a similar reactor containing the experimental water minus the disinfectant at the target temperature. The oocysts are typically exposed to this condition for the duration of the experiment and subsequently processed for viability assessment with methods similar to those for the disinfected samples.

### A.2.4.2 Ozone

If a semi-batch reactor configuration is used, the protocol described by Rennecker et al. (1999) is recommended. The protocol is described briefly as follows. Ozonated gas is applied to the temperature-acclimated experimental water via a fine bubble diffuser. The ozone gas concentration is adjusted to achieve steady-state at dissolved ozone concentrations representative of what would be observed at the facility. The actual dissolved ozone concentration achieved for each experiment is measured. Mixing of the ozonated water is performed with a magnetic stir bar and stirring plate. An inactivation experiment is initiated by injecting a suspension containing a sufficient number of oocysts into the reactor, and ends by simultaneously removing the bubble diffuser and injecting a quenching agent. It should be noted that the number of oocysts necessary for each data point is dependent on the viability assessment method selected. Oocysts are then removed from the quenched solution by filtration through a 1 μm filter. The reactor is then rinsed with approximately 50 mL of a dilute surfactant, and then again with approximately 100 mL of the experimental water to remove any residual surfactant. Both eluents are passed through the filter that is then placed in a clean 50 mL beaker and rinsed with approximately 15 mL of the dilute surfactant. The resulting oocyst suspension is transferred into a sterile 15 mL centrifuge tube. These steps are repeated at various contact times corresponding to target CT parameters (i.e., the product of dissolved ozone concentration and contact time).

Control samples are prepared with each daily experimental set by shutting off the ozone generator, but allowing the oxygen gas to flow through the system. Oxygen gas is allowed to by-pass the semi-batch reactor after shutting off the generator to purge residual ozone gas from the system. All other conditions used for the control are consistent with the experimental conditions previously described. The contact time for control samples is 1 minute. After completion of the experiment, the samples are generally centrifuged at 1,100g for 10 minutes and stored in a phosphate buffer solution for a period of time not to exceed 48 hours prior to viability assessment procedures.

Experiments performed with a head-space free reactor can follow the following protocol (described previously in Kim 2002). The experimental temperature is maintained by immersing the 100-mL syringe, which serves as the reactor in a water bath. Mixing inside the reactor is provided using a stir bar and magnetic stir plate. The syringe is filled with the experimental water containing enough oocysts for all six data points. At this point, an aliquot of temperature-adjusted ozone stock solution of known concentration is added. Samples are then taken at time intervals corresponding to the pre-determined estimated CT using a syringe containing a quenching reagent. The samples are then processed using filtration and centrifugation, similar to those described above. A control should be performed for each experiment by placing the sample number of oocysts in the experimental water at the desired temperature. The oocysts should remain there for a period of time equal to the duration of the inactivation experiment. After this time, the oocysts should be processed in a manner consistent with the disinfected samples.

Experiments performed with batch reactor components that are not head-space free typically follow a similar, although less complex protocol. An example of such a system and the associated experimental protocol can be obtained from Finch et al. 1993a.

It should be noted that for all batch-reactor systems, a careful characterization of the ozone demand and decay kinetics of the experimental water should be performed prior to any disinfection testing. In addition, it is also recommended that ozone concentration samples be procured alternately between inactivation samples to verify ozone concentrations observed during the disinfection study.

### A.2.5 Sample Processing

After procuring each sample point, the samples should be stored at 4$^{o}$C until the end of the experiment. At the end of each experiment, the samples should be centrifuged at a relative centrifugal force of 1,100 for at least 10 minutes to remove quenching agents or surfactants. Following centrifugation, the samples should be carefully aspirated and re-suspended in 0.01 M pH 7 buffer solution. The samples should be stored at 4 degrees until the time of viability assessment.

### A.2.6 Viability Assessment

Determining the viability of oocysts for varying levels of disinfection is one of the most critical components of the inactivation experiments. At present, there are three methods available to assess *Cryptosporidium parvum* viability, each presenting unique advantages and disadvantages. These methods include the following techniques:

1) Animal infectivity.

2) Cell culture (*in vitro* infectivity).

3) *In vitro* excystation.

The most established of these methods is animal infectivity. This viability assessment method typically involves inoculating immuno-suppressed neonatal mice with varying numbers of oocysts exposed to a particular CT. After a certain incubation period, the mice are then sacrificed and their intestinal tracts are examined for signs of *Cryptosporidium*-induced infection (cryptosporidiosis). The primary benefit of this method is that it demonstrates that the treated oocysts are capable of reproduction inside a mammalian host and therefore are able to induce an infection. One criticism of this method is that although an infection is capable of being observed, mouse infectivity has not been correlated to human infectivity. In addition, the protocol associated with this method is difficult and expensive. It requires specialized laboratory training, facilities, and equipment. An example of this protocol can be found in Finch et al. 1993b.

A second method used to assess the viability of *Cryptosporidium parvum* is known as *in vitro* infectivity or cell culture. At present, cell culture methodologies used for this purpose are based on either microscopic evaluation (Slifko et al. 1997) or polymerase chain reaction (PCR) (Rochelle et al. 1997). The first step in using cell culture to assess oocyst viability involves applying the treated oocysts to a lawn of cells (typically derived from human or canine cell lines). After an incubation period, using microscopic evaluation-based culture methods, the cells are stained with fluorescent chemicals and then examined microscopically for various *Cryptosporidium* life stages. The presence of these life stages suggests that the oocysts were capable of reproduction and thus were viable and likely able to cause an infection in humans.

When using a PCR-based technique, after incubation the cells are processed and the *Cryptosporidium parvum* Ribonucleic acid (RNA) is extracted. Infectivity is then determined by targeting specific genetic sequences in the RNA. The primary advantage of using cell culture to assess *Cryptosporidium parvum* infectivity is that it can measure very low concentrations of oocysts. Therefore, cell culture is capable of demonstrating high levels of inactivation. In contrast, the disadvantages associated with using cell culture include a lack of agreement over the preferred cell lines and viability assessment technique. In addition, there has been no extrapolation between cell culture techniques and human infectivity. Lastly, cell culture techniques are complex and typically require specialized equipment and rigorous training, which makes this procedure somewhat expensive.

A third method known as *in vitro* excystation has also been developed to assess the viability of *Cryptosporidium parvum* (Rennecker et al. 1999). This method involves exposing oocysts to a simulation of a mammalian digestive tract. Following the simulation, the oocysts are then examined microscopically for oocyst life stages that are indicative of viability. The advantages of this method are that it is cost-effective, offers the ability to rapidly develop data, and requires minimal training. The main disadvantage of the method is that of the three methods described, *in vitro* excystation has the least similarity to an actual infection. However, it should be noted that in spite of this fact, two published studies have shown that inactivation data obtained with *in vitro* excystation closely matches animal infectivity and/or cell culture data (Rennecker et al. 2000, Owens et al. 1999).

## A.3 Statistical Analysis

A general approach for calculating a set of CT values involves the following steps:

1) Fitting an inactivation model(s) to the experimental inactivation data (for the entire year).

2) Calculating the predicted average CT requirements from the best fit model.

1) Calculating and applying a factor of safety for the average predicted CT requirement.

One approach by Clark et al. (2002) used a one-parameter Chick-Watson model to fit experimental data sets and develop standard CT curves, relative to inactivation level and temperature. As described in the LT2ESWTR Preamble, EPA used the Clark et al. approach for developing CT values but adjusted the analysis to account for different types of uncertainties and variability inherent in the data. EPA wanted to account for variability among different water matrices and oocyst strains, but not variability within the same group (i.e., same oocyst lot and water), and uncertainty in the regression. While such a complex approach may not be necessary for a site-specific study, the water system should be aware of the uncertainties and variability of the experimental data and use a statistical method that builds in a reasonable safety factor to ensure public health is protected.

Two types of confidence bounds that are commonly used when assessing relationships between variables, such as disinfectant dose (CT) and log inactivation, are confidence in the regression and confidence in the prediction. Confidence in the regression accounts for uncertainty in the regression line (e.g., a linear relationship between temperature and the log of the ratio of CT to log inactivation). Confidence in the prediction accounts for both uncertainty in the regression line and variability in experimental observations it describes the likelihood of a single future data point falling within a range. Bounds for confidence in prediction are wider (i.e., more conservative) than those for confidence in the regression. Depending on the degree of confidence applied, most points in a data set typically will fall within the bounds for confidence in the prediction, while a significant fraction will fall outside the bounds for confidence in the regression.

## A.4 References

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# Appendix B
# Ozone CT Methods

## Abbreviations and Glossary

| | |
|---|---|
| $BrO_3^-$ | Bromate ion |
| $C_{eff}T_{10}$ | Chamber effluent ozone residual in mg/L times chamber $T_{10}$ time in minutes |
| Co-current chamber | A chamber in an ozone contactor where the water is flowing upward and the ozone gas bubbles are rising. The direction of flow of the water and the gas is the same. |
| Counter-current chamber | A chamber in an ozone contactor where the water is flowing downward and the ozone gas bubbles are rising. The direction of flow of the water is in the opposite direction of the gas flow. |
| CSTR | Completely Stirred Tank Reactor – fully mixed volume |
| CT | The product of Concentration and Time in mg/L-min |
| DBP | Disinfection byproduct |
| gpm | Gallons per minute |
| HDT | Hydraulic detention time calculated as the volume divided by the flow. When volume is expressed in gallons, and flow expressed in gallons/minute, then the calculated HDT is in minutes. |
| In-situ sample ports | Sample ports that take a sample from the flow of the chamber, typically through tubing that projects into the flow. |
| $k^*$ | The first-order ozone decay coefficient, $min^{-1}$ |
| $k_{10}$ | Log-base-10 value of the lethality coefficient for the inactivation of *Cryptosporidium*, *Giardia* or virus with ozone. The units of $k_{10}$ in this document are L/mg-min. |
| -Log ($I/I_o$) | Log inactivation. Negative log-base-10 of the survival rate ($N/N_o$) of the microorganisms, where $I_o$ is the number of viable organisms entering the contactor, and I is the number of viable organisms leaving the contactor. |
| PQL | Practical Quantitation Limit: The minimum concentration of an analyte (substance) that can be measured with a high degree of confidence that the analyte is present at or above that concentration. |
| Q | Water flow – usually expressed in gallons per minute (gpm) or million gallons per day (MGD). |
| RTD | Residence Time Distribution probability distribution function describing the residence time of a fluid element within a contactor. |
| segment | A theoretical or physically real chamber within a contactor. Used predominantly to denote a theoretical chamber for non-conventional contactors. |

| | |
|---|---|
| $T_{10}$ | The time at which 10 percent of the water in the contactor or segment has passed through the contactor or segment. EPA recommends that tracer studies be used to determine the $T_{10}$/HDT ratio for ozone contactors. The *SWTR Guidance Manual* and *Tracer Studies in Water Treatment Facilities: A Protocol and Case Studies* describe how to conduct a tracer test. |
| Up flow chamber | A chamber within an over-under baffled bubble-diffuser ozone contactor in which the direction of water flow is upward. |
| V | Volume of the contacting zone in question – usually expressed in gallons or million gallons. |

# B.1 Introduction

## B.1.1 Background

Appendix O of the Surface Water Treatment Rule (SWTR) Guidance Manual (U.S. EPA 1991) includes a description of different methods for determining inactivation credit using an ozone contactor. These recommended methods differ in the level of effort associated with them and, in general, the ozone dose needed to achieve a given level of inactivation. This appendix provides guidance to help water systems select the more appropriate methods for their ozone process. More importantly, it builds on the information presented in the SWTR Guidance Manual with detailed descriptions of the extended $T_{10}$ method and extended continuous stirred tank reactor (CSTR) method. Appendices C, D and E compliment this appendix with descriptions of ozone residual sampling and laboratory analysis (Appendix D) and derivations of equations used in the extended CSTR approach (Appendix E).

The four methods for calculating LT2ESWTR ozone inactivation credit, presented in Chapter 11 and this appendix, are described below.

1. **$T_{10}$** --calculates CT through a contactor assuming hydraulic conditions similar to plug flow and can only be used with tracer study data. Using the $T_{10}$ approach, the contact time (T) is the time at which 10 percent of the water in the contactor or segment has passed through the contactor or segment. Even in well-baffled contactors, the $T_{10}$ is most often less than 65 percent of the average hydraulic detention time (HDT) through the contactor, and generally underestimates the true CT achieved. (The $T_{10}$ approach is described in Chapter 11, section 11.3.)

2. **CSTR**--calculates log inactivation credit in a chamber by assuming it to be completely mixed. It is applicable to contactors that experience significant back mixing or when no tracer study data are available. EPA recommends using this method (or the Extended CSTR) when no tracer study data are available. (The CSTR approach is described in Chapter 11, section 11.3.)

3. **Extended $T_{10}$**--a method that utilizes three measured ozone residuals at three chamber effluents in a reactive zone to predict the ozone residual concentrations at the effluents of the non-monitored chambers in the zone. It then uses the standard $T_{10}$ method to calculate the CT from all chambers using both measured and predicted ozone residuals. This method is only applied to reactive chambers and not to dissolution chambers, and requires tracer study data.

4. **Extended CSTR**--a method that utilizes three measured ozone residuals at three chamber effluents in a reactive zone to predict the ozone residual concentrations at the effluents of the non-monitored chambers in the zone. It then uses the standard CSTR method to calculate the CT from all chambers using both measured and predicted ozone residuals. This method is only applied to reactive chambers and not to dissolution chambers, and does not require tracer study data.

While this guidance manual describes four methods, other methods or modifications to these methods may be used at the discretion of the state.

## B.2 Selection of Methods for Calculating Inactivation Credit

Selecting the appropriate methods to use depends on the configuration of the ozone contactor and amount of process evaluation and monitoring that a water system is willing to undertake. It is also possible that combinations methods can be used. For contactors with multiple segments it is likely that the CT of one or two segments would be calculated using either the $T_{10}$ or CSTR methods, while the CT for the remaining segments would be calculated with the Extended $T_{10}$ or the Extended CSTR method.

Of the four methods described in the previous section, the two Extended methods are more complex. The Extended methods require measurements of the ozone concentration at a minimum of three points within this portion of the contactor. The residual measurements are then used to develop a predicted ozone concentration profile through this portion of the contactor. While many mathematical principles are discussed in these methods, their implementation is fairly straightforward. In fact, the methods presented in this appendix can be programmed into a conventional spreadsheet or a plant computer control system.

The following exhibits define the types of chambers potentially present in an ozone contactor and show the recommended methods for calculating the inactivation credit achieved. Only the $T_{10}$ or CSTR methods can be applied to dissolution chambers. However, they can be applied to the reactive chambers as well. If no tracer test data are available, it is recommended that the CSTR method be used. The Extended methods are applied over a minimum of three consecutive reactive chambers. Exhibit B.1 shows the recommended methods.

## Exhibit B.1 Recommended Methods and Terminology for Calculating the Log Inactivation Credit

| | Section Description | Terminology | Method for Calculating Log Inactivation | Recommended Restrictions |
|---|---|---|---|---|
| **Without Tracer Data** | ***Chambers where ozone is added*** | | | |
| | First chamber | First Dissolution Chamber | No log *Cryptosporidium* inactivation credit is recommended | The SWTR criteria for 1st chamber credit should still be used if calculating inactivation of *Giardia* and virus |
| | Other chambers | Co-Current or Counter-Current Dissolution Chambers | **CSTR Method** in each chamber with a measured effluent ozone residual concentration | No credit should be given to a dissolution chamber unless a detectable ozone residual has been measured upstream of this chamber |
| | ***Reactive Chambers*** | | | |
| | ≥ 3 consecutive reactive chambers | Extended-CSTR Zone | **Extended CSTR Method** in each chamber | Detectable ozone residual should be present in at least 3 chambers in this zone, measured via in-situ sample ports. Otherwise, the **CSTR method** should be applied individually to each chamber having a measured ozone residual |
| | < 3 consecutive reactive chambers | CSTR Reactive Chamber(s) | **CSTR Method** in each chamber with a measured effluent ozone residual concentration | None |
| **With Tracer Data** | ***Chambers where ozone is added*** | | | |
| | First chamber | First Dissolution Chamber | No log *Cryptosporidium* inactivation is credited to this section | The SWTR criteria for 1st chamber credit should still be used if calculating inactivation of *Giardia* and virus |
| | Other chambers | Co-Current or Counter-Current Dissolution Chambers | **$T_{10}$** or **CSTR Method** in each chamber | No credit should be given to a dissolution chamber unless a detectable ozone residual has been measured upstream of this chamber |
| | ***Reactive Chambers*** | | | |
| | ≥ 3 consecutive chambers with in-situ sample ports | Extended-CSTR Zone | **Extended $T_{10}$ or Extended CSTR Method** in each chamber. The Extended CSTR method is not appropriate for non-conventional contactors. | Detectable ozone residual should be present in at least 3 chambers in this zone, measured via in-situ sample ports. Otherwise, the **$T_{10}$** or **CSTR method** should be applied to each chamber having a measured ozone residual |
| | < 3 consecutive chambers | $T_{10}$ or CSTR Reactive Chamber(s) | **$T_{10}$** or **CSTR Method** in each chamber | None |

## B.3 Dissecting an Ozone Contactor

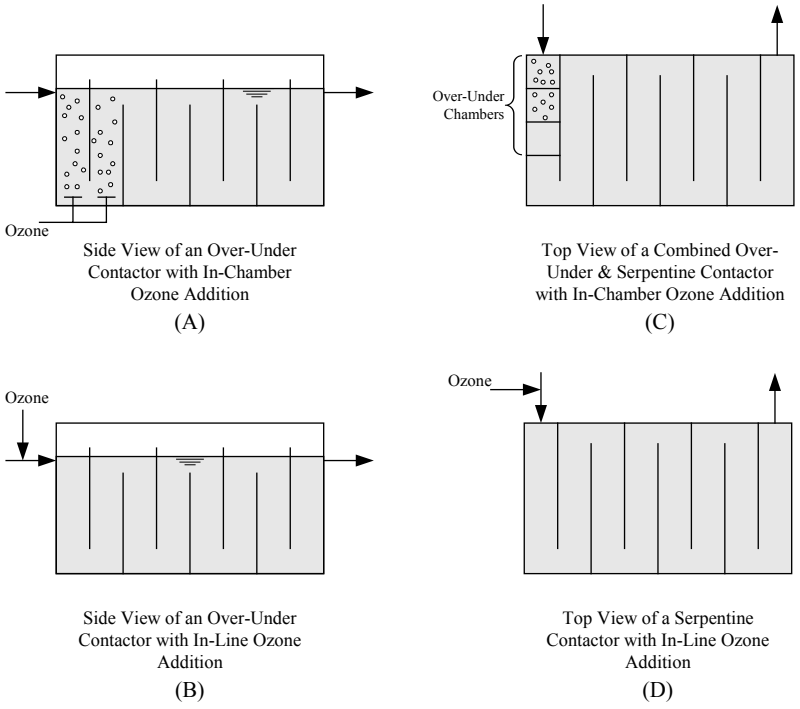
### B.3.1 Ozone Contactor Configurations

Ozone contactors are designed in a wide variety of configurations. Different configurations are adaptable to the Extended $T_{10}$ or Extended CSTR methods, but implementation details vary with contactor configuration. It is important for a water system to identify the type of configuration and become familiar with the terminology used in this guidance manual.

Exhibit B.2 shows configurations with multiple, consecutive well-defined reactive chambers. The water flow pattern in such contactors can be an "over-under" pattern, a "serpentine" pattern, or a combination of both. Gaseous ozone is added to the water by one of two procedures. Gaseous ozone can be injected into the influent water before the water enters the contactor, a process often called "in-line" ozone addition (see schematic B & D in Exhibit B.2). Alternatively, ozone enriched gas can be bubbled into one or more chambers, a process called "in-chamber" ozone addition (see schematic A & C in Exhibit B.2). In-chamber ozone addition takes place in chambers that have an over-under flow pattern and not in chambers that have a serpentine flow pattern (Exhibit B.2-C) in order to ensure full and complete ozone dissolution into all the water flow. These so-called bubble columns can be counter-current or co-current, describing the directional flow of the water with respect to the upward flowing bubbles. Note, Exhibit B.2 only shows example configurations; size and geometry of the chambers will vary.

In contrast to the multi-chamber configuration, ozone contactors may also be comprised of only one or two reactive zones. Examples of such contactors are shown in Exhibit B.3, which include a closed-pipe contactor (see schematic A) and two open-channel contactors (see schematics B & C). All three contactors depict a long and narrow water flow path that may promote more plug-flow hydrodynamics within the majority of the chamber. As with multi-chamber contactors, ozone can be added in-line, or in-chamber. Contactors A and B illustrate in-line ozone addition. Contactor C illustrates in-chamber ozone addition.

## Exhibit B.2 Schematics of Typical Conventional Configurations of Ozone Contactors with Multiple Chambers




Side View of an Over-Under Contactor with In-Chamber Ozone Addition

(A)

Top View of a Combined Over-Under & Serpentine Contactor with In-Chamber Ozone Addition

(C)

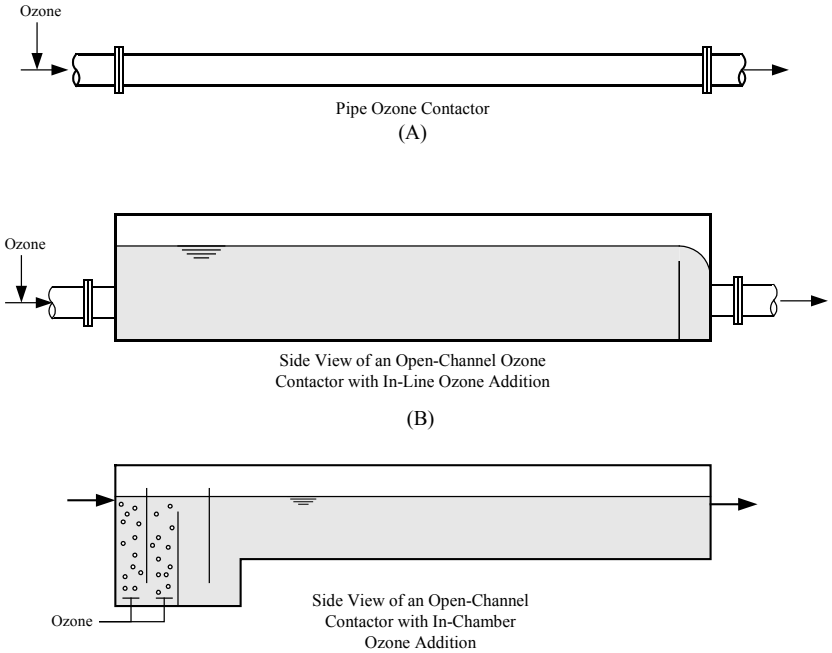
Side View of an Over-Under Contactor with In-Line Ozone Addition

(B)

Top View of a Serpentine Contactor with In-Line Ozone Addition

(D)

## Exhibit B.3 Example Schematics of Non-conventional Configurations of Ozone Contactors



Whether the contactor is configured with multiple chambers or predominantly a single long and narrow chamber, a major consideration of determining inactivation credit is characterizing the hydrodynamics within the contactor. The EPA recommends performing a tracer study to ascertain a description of the hydrodynamics. A tracer study will afford the necessary information for utilizing the $T_{10}$ or Extended $T_{10}$ methods. However, as with SWTR guidance manual, this guidance offers possibilities to apply an assumed, theoretical hydrodynamic condition to the contactor if a tracer study is not performed. However, since this option involves a major assumption, the guidance recommends assuming a hydrodynamic condition that is somewhat conservative with respect to the efficiency of disinfection kinetics. In particular, for utilities that opt to not perform a tracer study, this guidance offers the CSTR and Extended CSTR methods, which assumes that each chamber within a contactor has a high degree of mixing equivalent to an ideal CSTR.

### B.3.2 Classification of the Chambers and Contactor Zones

To properly apply the methods discussed in this manual, the contactor should be divided into specific sections or zones. To ensure clarity, certain terminology is adopted for unique sections of an ozone contactor, as presented in Exhibit B.1.

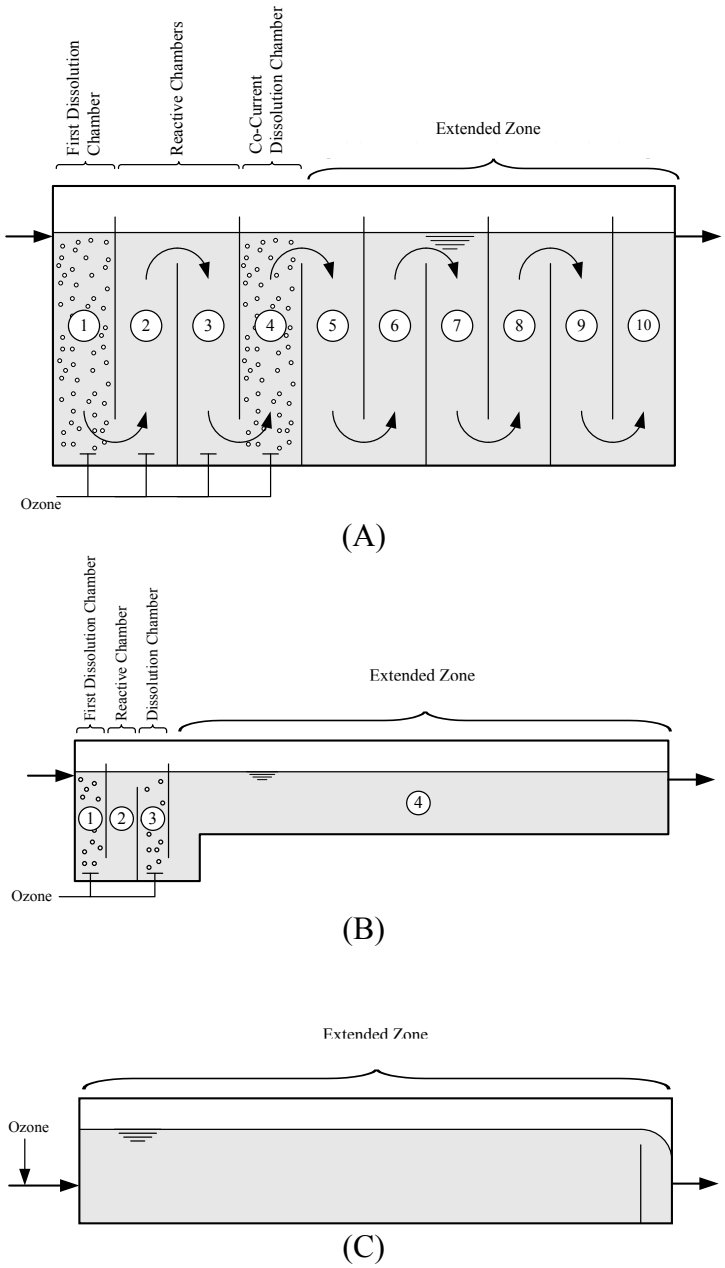
Exhibit B.4 shows example schematics of conventional and non-conventional configurations of ozone contactors. Schematic A is that of a conventional configuration of a 10-chamber over-under baffled ozone contactor with in-chamber ozone addition. Ozone is being added only in Chambers 1 and 4 in this example. Chamber 1 is classified as a "First Dissolution Chamber" and it is recommended that no disinfection credit be granted for this chamber. Rapid, initial ozone reactions and the transitional development of the ozone residual occur in the first dissolution chamber. As such, a representative dissolved ozone profile is difficult to estimate without multiple sample ports along the bubble column. The second and third chambers in the contactor shown in schematic A of Exhibit B.4 are reactive chambers through which ozone is decaying. These chambers are called "Reactive Chambers." The $T_{10}$ or CSTR method could be used to calculate the log inactivation across such Reactive Chambers when ozone residual values are available from the effluent of the chamber. The $T_{10}$ and CSTR methods are described in Chapter 11.

The fourth chamber in the contactor shown in schematic A of Exhibit B.4 includes ozone addition. This chamber is called a Co-Current "Dissolution Chamber." It should be emphasized that there is a distinction between a "Dissolution Chamber" and "First Dissolution Chamber." A chamber is given the "Dissolution Chamber" notation only when ozone residual has been detected at any point upstream of the influent to that chamber, thus signifying that the initial (i.e. instantaneous) ozone demand has been met. In other words, chamber 4 in schematic A of Exhibit B.4 can be classified as a Dissolution Chamber only if ozone residual has been detected at the effluent of either chamber 1, 2, or 3. The $T_{10}$ or CSTR method could be used to calculate the log inactivation credit across a Dissolution Chamber. If no ozone residual was detected upstream of this chamber location, then chamber 4 takes on the classification of, and is treated as, a "First Dissolution Chamber" and as with chamber 1, no log inactivation credit is granted.

Chambers 5 through 10 in schematic A in Exhibit B.4 represent the "Extended zone" since they meet the criterion of containing a minimum of three consecutive reactive chambers. If tracer data are unavailable, the Extended-CSTR approach is used to calculate the log inactivation across each chamber in this zone. If tracer data are available and can be used to calculate the $T_{10}$/HDT ratio of the contactor, then the Extended $T_{10}$ approach or the Extended CSTR approach could be used to calculate the log inactivation across each chamber in this zone. Modeling is used to calculate the ozone residual concentration at the effluent of each chamber within the Extended zone. Either Extended method requires an accurate estimation of the ozone decay coefficient, $k^*$, and the initial ozone residual at the entrance to the zone, $C_{in}$. Estimation of these two parameters, which is discussed in sections B.4.3.1 and B.4.3.2, requires the measurement of three ozone residual values across the minimum span of three chambers.

In the case of a contactor with in-line ozone addition, the entire contactor potentially becomes an Extended zone. If the contactor has at least three chambers equipped with in-situ sample ports and a measurable ozone residual, then the requirements for calculating $k^*$ and $C_{in}$ have been met and the entire contactor can be treated as an Extended zone. Care should be taken in locating the first ozone sample port such that enough reaction time is allowed for the immediate ozone demand to be fully met before the sample port.

## Exhibit B.4 Names of Various Sections of Different Types of Ozone Contactors




Schematics B and C in Exhibit B.4 represent non-conventional configurations of ozone contactors used in water treatment. While these contactors do not necessarily have clearly defined chambers divided by baffle walls, this guidance provides for methods such that they can be evaluated in a similar manner as conventional contactors with chambers are evaluated under the Extended $T_{10}$ method. For such contactors, the Extended zone could be divided into segments that represent theoretical chambers; the number of theoretical chambers determined by a tracer test. Subsequent sections of this Appendix and Appendix E include examples and calculations that illustrate this approach.

## B.4 Extended $T_{10}$ or Extended-CSTR Approaches for Ozone Contactors

### B.4.1 Introduction

The methods described in this chapter represent a more sophisticated approach to calculating inactivation credit in an ozone contactor as compared to the $T_{10}$ and CSTR approaches. This approach could potentially provide a higher and more accurate estimate of the level of microbial inactivation than that obtained using the $T_{10}$ approach. The potential benefits of using these more sophisticated measures are lower ozone doses and consequently lower formation of some ozonation disinfection byproducts, (e.g., bromate). However, as a consequence of this added sophistication, a higher degree of system evaluation and monitoring is needed for a given inactivation credit. Whether use of these more sophisticated approaches actually benefit the utility depends on many factors including the sought-after level of inactivation, the reactor configuration, and the water quality.

The approach described in this chapter is called the ***Extended Approach***. Certain aspects of this methodology were introduced in Appendix O of the SWTR Guidance Manual. However, the material presented here greatly expands upon the SWTR Guidance Manual, and may provide beneficial new tools for the utility. As indicated in Chapter 11, the approached described herein can be used for calculating CT for *Cryptosporidium*, *Giardia*, or virus inactivation credit.

The Extended method relies on measured ozone residuals across the Extended zone to model the ozone decay through the zone. The outcome is then used to project the ozone residual at any location in the Extended zone. This approach, which is applied only to an Extended zone as defined in the earlier section, includes the following four steps:

Step 1 – Measure the ozone residual at the effluent of at least three chambers in the Extended zone.

Step 2 – Utilizing the measured residuals and flowrate through the contactor, calculate the empirical ozone decay coefficient, $k^*$, and the ozone residual in the influent of the Extended zone, $C_{in}$.

Step 3 – Utilize the calculated $k^*$ and $C_{in}$ to predict the ozone residual concentration at the influent or effluent of any chamber in the Extended zone.

Step 4 – Utilize the calculated ozone residual values to calculate the CT and subsequent log inactivation across each chamber in the Extended zone.

Ozone residual measurement at the three locations might be conducted manually using the Indigo Trisulfonate method, or continuously using on-line ozone analyzers. The Quality Assurance protocols discussed in Appendix C should be implemented to ensure that the ozone residual measurements are accurate. For on-line control systems that utilize continuous residual monitoring, instantaneous disinfection calculation will not be possible because of general fluctuations in the residual monitor's responses to small changes in system operation. Most control systems include a function to conduct a rolling average of monitor readings at a preset interval. If this approach is used, EPA recommends that the averaging interval not exceed the

HDT of the contactor at design flowrate. For example, if the contactor is designed with an HDT of 10 minutes at its design flowrate, then the control system's averaging interval should not exceed 10 minutes. Therefore, if the monitor collects an ozone residual reading every two (2) minutes, then the control system would report an average of the previous five (5) consecutive readings every two minutes. The $k^*$ and $C_{in}$ values for the Extended zone are calculated using these rolling average residual values.

### B.4.2 Extended $T_{10}$ Method

To utilize this method for calculating the CT across an Extended zone of an ozone contactor, the contactor must have a representative set of tracer test results that have been used to set the $T_{10}$/HDT ratio for the contactor. Guidance on determining the $T_{10}$/HDT ratio for the contactor is found in *SWTR Guidance Manual*, *Tracer Studies in Water Treatment Facilities: A Protocol and Case Studies*, and Appendix E. The Extended zone comprises three or more individual chambers. Inactivation within each individual chamber is calculated in accordance with the $T_{10}$ method described in Section 11.3.2. The sum of CT and log inactivation values for individual chambers gives the CT and log inactivation across the entire zone. The distinction between a standard Reactive Chamber and a chamber that is a component of an Extended zone is the manner in which each chamber's $C_{out}$ value is obtained. In the case of a standard Reactive Chamber, $C_{out}$ is obtained from an actual measurement of the dissolved ozone residual at the exit of the chamber. In contrast, $C_{out}$ for a chamber in an Extended $T_{10}$ zone is a calculated value. It is important to note that the calculation of $C_{out}$ for a chamber is performed only on as many physical chambers (as in the case of multi-chamber contactors) or theoretical segments (as in the case on non-conventional contactors) within the Extended zone.

In addition to enabling the calculation of $C_{out}$ for all individual chambers, the Extended the $T_{10}$ method follows the guidance established in Appendix O of the SWTR Guidance Manual in allowing the overall the $T_{10}$/HDT ratio calculated for the entire contactor to be applied to each individual chamber – the so-called linear extrapolation of the $T_{10}$/HDT ratio. As noted in Appendix O of the SWTR Guidance Manual and in Lev and Regli (1992), this allowance is not theoretically correct, and except for atypical contactor designs where a more plug-flow region is followed by a highly mixed zone of similar volume, this allowance will lead to a higher calculation of CT credit than if the actual $T_{10}$/HDT ratio of each chamber was used. As described in Appendix O (and also alluded to in Section 11.3.2. of this guidance), this allowance comes with the proviso that the linear extrapolation of the overall $T_{10}$/HDT ratio is not appropriate if more than 50% of the volume of the contactor has an ozone concentration of "zero." In the context of the SWTR, a *measurement* of "zero" ozone residual is equivalent to "below the ozone method practical quantitation limit (PQL)." Because the Extended $T_{10}$ method allows ozone concentrations to be *calculated* to values that may be below the PQL of the allowable ozone methods (see Appendix C), essentially a value of "zero", the proviso in Appendix O needs to be revised. As such, the EPA recommends that the linear extrapolation of the overall $T_{10}$/HDT ratio is not applicable if more than 50% of the contactor volume has a measured or calculated (using the more conservative k* value) ozone residual of less than 0.05 mg/L. If the proviso is not met, the system may estimate its CT credit using other methods, or it may conduct a tracer test to evaluate the $T_{10}$/HDT ratio for a shorter section of the contactor, and use that ratio in the same manner. Because of the possibility of the 50% criteria not being met, the EPA recommends that

if a system is conducting its first or new tracer tests to characterize the contactor, that they collect tracer data at intermediate points in the contactor in addition to the exit of the entire contactor.

The procedure for calculating $C_{out}$ for a chamber in an Extended $T_{10}$ zone is described in this section. The value of $C_{out}$ for a chamber in an Extended $T_{10}$ zone is calculated using an empirical ozone decay coefficient, $k^*$, and the ozone residual concentration at the entrance to the zone, $C_{in}$. Equation B-1 shows how to calculate the ozone residual at any location X along the Extended zone:

$$C_X = C_{in} \times Exp\left(-k^* \times \left(\frac{[Volume]_{0-X}}{Q}\right)\right) \quad \text{(B-1)}$$

where:
$k^*$ = Empirical ozone decay coefficient, $min^{-1}$, calculated as described in section B.4.2.1.
$C_{in}$ = Calculated ozone residual concentration at the entrance to the Extended zone, mg/L, calculated as described in section B.4.2.2.
$[Volume]_{0-X}$ = Volume, in gallons, from the beginning of the Extended zone to a location X along the water path in the Extended $T_{10}$ zone.
$Q$ = Water flow through the contactor, gpm

Equation B-1 describes the Extended zone as a plug flow reactor for the purpose of calculating the profile of the ozone residual, *C*, along the zone.

Once the values of the ozone residual concentrations at the effluent of each chamber in the Extended $T_{10}$ zone are calculated, Equation 11-4 can then be used to calculate the log inactivation achieved across that chamber. The total log inactivation achieved across the entire contactor is equal to the sum of the log inactivation values calculated for each chamber.

$$-Log\left(\frac{I}{I_0}\right) = k_{10} \times C \times HDT_{chamber} \times \left(\frac{T_{10}}{HDT}\right) \quad \text{Equation 11-4}$$

where:

-Log ($I/I_0$) = the log inactivation
$k_{10}$ = log base-ten inactivation coefficient for the target organism (L/mg-min)
$C$ = Ozone residual concentration from Exhibit 11.3 (mg/L)
$HDT_{chamber}$ = Hydraulic detention time through the chamber (minutes)
($T_{10}$/HDT) = approved $T_{10}$/HDT ratio for the contactor

The values of $k_{10}$ for the inactivation of *Cryptosporidium*, *Giardia*, and virus with ozone can be expressed by the following equations (Temp = water temperature in $^oC$):

Inactivation of *Cryptosporidium* with Ozone: $k_{10} = 0.0397 \times (1.09757)^{Temp}$

Inactivation of *Giardia* with Ozone: $k_{10} = 1.0380 \times (1.0741)^{Temp}$

Inactivation of virus with Ozone: $k_{10} = 2.1744 \times (1.0726)^{Temp}$

The values of $k_{10}$ for the inactivation of *Giardia* and virus were derived from the $k_{10}$ values for *Giardia* and virus inactivation listed in Appendix O of the SWTR Guidance Manual.

The values of $k^*$ and $C_{in}$ should be determined every time log inactivation credit is calculated (i.e., at least daily). The following sub sections describe the procedures for estimating the values of $k^*$ and $C_{in}$ across the Extended zone.

### B. 4.2.1 Determining the Value of *k**

The empirical ozone decay coefficient, $k^*$ is calculated using ozone sample measurements, taken from in-situ sample ports, and an assumed theoretical model of the chamber's hydrodynamics. In the application of the Extended $T_{10}$ method, the Extended zone is modeled as a plug-flow reactor. It is important to note that the empirical ozone decay coefficient, $k^*$, will unlikely be the same value as the first-order decay rate constant that would be measured in a batch or plug-flow reactor. Only when the Extended zone hydrodynamics are that of a plug-flow system will the empirical $k^*$ be equivalent to the true first-order rate constant. Nonetheless, the simplifying assumption of plug-flow hydrodynamics and use of the two-parameter exponential (i.e. first-order kinetic) equation to fit two or three ozone residual measurements and subsequently predict the ozone concentration at other points in an Extended zone has been shown to be useful (Rakness, Najm et al. 2005).

The steps outlined below pertain to an Extended zone with a minimum of three consecutive chambers with measurable ozone residuals. That is, there should be at least three in-situ sample ports from the Extended zone with measurable ozone residual. The three ozone residual measurements, $C_1$, $C_2$, and $C_3$, are needed to estimate the value of the ozone decay coefficient, $k^*$. For example, the Extended zone in the contactor shown in Exhibit B.4(A) includes chambers 5 through 10. The ozone residual values at any three chambers in that span can be used to represent $C_1$, $C_2$, and $C_3$ in this analysis. The following steps should be followed to calculate the $k^*$ value:

Step 1 – Use Equation B-2 and residual measurements $C_1$ and $C_2$ to calculate the $k^*$ value representing the ozone decay between locations 1 and 2, $k^*_{1-2}$. (A derivation and explanation of Equation B-2 is presented in Appendix E):

$$k^*_{1-2} = \frac{Q}{[Volume]_{1-2}} \times Ln\left[\frac{C_1}{C_2}\right] \qquad \text{(B-2)}$$

where: $k^*_{1-2}$ = Empirical ozone decay coefficient between sampling locations 1 & 2, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_2$ = Measured ozone residual at location 2, mg/L
$[Volume]_{1-2}$ = Volume between sampling locations 1 and 2, gallons
$Q$ = Water flow through the contactor, gpm

Step 2 – Use residual measurements $C_1$ and $C_3$ along with Equation B-3 to calculate the $k^*$ value representing ozone decay between sampling locations 1 and 3, $k^*_{1-3}$:

$$k^*_{1-3} = \frac{Q}{[Volume]_{1-3}} \times Ln\left[\frac{C_1}{C_3}\right] \qquad \text{(B-3)}$$

where: $k^*_{1-3}$ = Empricial ozone decay coefficient between sampling locations 1 & 3, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_3$ = Measured ozone residual at location 3, mg/L
$[Volume]_{1-3}$ = Volume between sampling locations 1 and 3, gallons
$Q$ = Water flow through the contactor, gpm

It should be emphasized that sampling location 1 should not be at the entrance to the Extended zone, but should be at least one chamber into the zone. For example, in Exhibit B.4(A), $C_1$ should not be measured at the entrance to chamber 5, since that is the entrance to the Extended zone (notice in Figure B.4(A) that ozone is added to chamber 4). Instead, the first Extended zone sampling location should be located at the effluent of chamber 5, or downstream of that location. Section O.3.2 of Appendix O of the SWTR Guidance Manual provides guidance on the use of in-situ sample ports for direct ozone measurements.

Step 3 – The value of $k^*$ that is to be used in Equation B-1 will be calculated as the average of $k^*_{1-2}$ and $k^*_{1-3}$ as shown in Equation B-4.

$$k^* = \left[\frac{k^*_{1-2} + k^*_{1-3}}{2}\right] \qquad \text{(B-4)}$$

It is normal for the individual values of $k^*_{1-2}$ and $k^*_{1-3}$ to be somewhat different. However, it is recommended that they be within the range of 80 percent to 120 percent of the average *k** value calculated in Step 3. That is,

$$\frac{abs\left\langle k^* - k_{1-i}\right\rangle}{k^*} \leq 20\%$$

If the two *k** values do not meet this criterion, the utility may 1) reject the measured residual values and collect new samples until this quality assurance (QA) criterion is met, or 2) select the higher of the two *k** values as the more conservative estimate.

### B.4.2.2 Determining the Value of $C_{in}$

While it is possible to measure the ozone residual at the entrance to the Extended zone (e.g., an in-situ sample port), it is not recommended that the measured value be used because it is usually higher than the residual predicted by the first-order decay profile (Amy et al., 1997; Carlson et al., 1997; Hoigné and Bader, 1994; Rakness and Hunter, 2000; Rouston et al., 1998). This phenomenon is commonly attributed to the more rapid initial ozone decay, which is followed by a somewhat slower first-order decay profile. For this reason, the $C_{in}$ representing the ozone decay in the Extended zone should be back-extrapolated using the three downstream rolling average ozone residual values.

The value of $C_{in}$ can be calculated once the value of *k** is estimated from the three rolling average residual ozone values. Maintaining the assumption of a first-order decay rate, Equations B-5 through B-7 can be used to estimate the value of $C_{in}$ from the three rolling average ozone residual concentrations:

$$C_{in} = C_1 \times \exp\left(k^* \times \frac{[Volume]_{0-1}}{Q}\right) \quad \text{(B-5)}$$

$$C_{in,2} = C_2 \times \exp\left(k^* \times \frac{[Volume]_{0-2}}{Q}\right) \quad \text{(B-6)}$$

$$C_{in,3} = C_3 \times \exp\left(k^* \times \frac{[Volume]_{0-3}}{Q}\right) \quad \text{(B-7)}$$

where:
$k^*$ = Average decay coefficient from Equation B-4, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_2$ = Measured ozone residual at location 2, mg/L
$C_3$ = Measured ozone residual at location 3, mg/L
$[Volume]_{0-1}$ = Volume, in gallons, between the entrance of the Extended zone and sampling location 1

$[Volume]_{0-2}$ = Volume, in gallons, between the entrance of the Extended zone and sampling location 2

$[Volume]_{0-3}$ = Volume, in gallons, between the entrance of the Extended zone and sampling location 3

$Q$ = Water flow through the contactor, gpm

The $C_{in}$ value is then calculated as the average of the three values determined by Equations B-5 through B-7:

$$C_{in} = \left[ \frac{C_{in,1} + C_{in,2} + C_{in,3}}{3} \right] \quad \text{(B-8)}$$

A systematic example of the Extended $T_{10}$ approach is presented in section B.4.5

### B.4.3 Extended CSTR Approach

The direct CSTR approach for calculating log inactivation across a chamber in an ozone contactor is discussed in Section 11.3.3. This section includes a discussion of the application of the Extended CSTR method to an Extended zone as defined in Sections B.3.2 and B.4.1. The Extended CSTR approach can be used whether or not tracer test results are available for the ozone contactor. However, it was developed primarily to afford a method for systems that choose not to perform a tracer study. In particular, in cases where a system chooses not to perform a tracer study, the Extended CSTR method makes an assumption that the hydrodynamics of each individual chamber is equivalent to an ideal CSTR. The assumption of CSTR hydrodynamics is considered somewhat conservative in terms of the efficiency towards chemical conversion for first-order and higher-order reactions. That is, the predicted disinfection in an ideal CSTR will be less than for an equivalent-volume, ideal plug-flow chamber. However, the CSTR assumption is not the most conservative assumption. Recent studies have indicated that in typical over-under baffled contactors, the hydrodynamics of an individual chamber can be worse in terms of reaction efficiency than a single CSTR due to recirculation patterns, dead-volumes, and short circuiting (Kim, Kim, et al. 2010; Kim, Nemlioglu, et al. 2010; Kim, Elovitz, et al. 2010). EPA therefore considers the CSTR assumption inherent in the Extended CSTR method as a reasonable balance between conservatism and pragmatism.

As discussed earlier, the Extended CSTR approach applies to a zone that includes at least three consecutive reactive chambers. Inactivation within each chamber is calculated according to Equation 11-1, exactly as it is for the CSTR chamber, and the sum of the log inactivation values for individual chambers gives the inactivation across the whole zone. The distinction between a CSTR Reactive Chamber and a chamber that is a component of an Extended CSTR zone is the manner in which the value for $C$ is obtained. In the case of the CSTR Reactive Chamber, $C$ is obtained from an actual measurement of the dissolved ozone residual at the exit of the chamber (i.e., $C_{out}$). In contrast, $C$ for a chamber in an Extended CSTR zone is a calculated value. The procedure for calculating $C$ for an Extended CSTR zone is described in this section.

The value of *C* for an Extended CSTR is also calculated using an empirical ozone decay coefficient, *k**, and the ozone residual concentration at the entrance to the zone, $C_{in}$. Equation B-9 shows how to calculate the ozone residual at the effluent of chamber "X" in an Extended CSTR zone:

$$C_X = \frac{C_{in}}{\left[1 + k^* \times \frac{[Volume]_{0-X}}{N_{0-X} \times Q}\right]^{N_{0-X}}} \qquad \text{(B-9)}$$

where:
$k^*$ = Empricial ozone decay coefficient, $min^{-1}$, calculated as described in section B.4.3.1
$C_{in}$ = Calculated ozone residual concentration at the entrance to the Extended CSTR zone, mg/L, calculated as described in section B.4.3.2
$[Volume]_{0-X}$ = Volume, in gallons, from the beginning of the Extended CSTR zone to the effluent of chamber "X"
$N_{0-X}$ = Number of chambers from the beginning of the Extended CSTR zone to the effluent of chamber "X"
$Q$ = Water flow through the contactor, gpm

Equation B-9 describes the Extended-CSTR zone between the first chamber (subscript 0) and chamber X as a series of equal-volume CSTR reactors. This is a simplifying assumption that is based on a balance between ease of implementation and consistency with other provisions within this guidance manual.

Once the values of the ozone residual concentrations at the effluent of each chamber in the Extended CSTR zone are calculated, Equation 11-4 can then be used to calculate the log inactivation achieved across that chamber. The total log inactivation achieved across the entire contactor is equal to the sum of the log inactivation values calculated for each chamber.

$$-\text{Log}\,(I/I_0) = \text{Log}\,(1 + 2.303 \times k_{10} \times C \times \text{HDT}) \qquad \text{Equation 11-4}$$

where:

$-\text{Log}\,(I/I_0)$ = the log inactivation
$k_{10}$ = log base ten inactivation coefficient (L/mg-min)
$C$ = Concentration from Exhibit 11-2 (mg/L)
HDT = Hydraulic detention time (minutes)

The $k_{10}$ can be determined using the equations as presented in Section B.4.2 or calculated from the CT table with the following equation: Log inactivation = $k_{10} \times$ CT.

The values of $k^*$ and $C_{in}$ should be determined every time log inactivation credit is calculated (i.e., at least daily). These parameters are calculated using three measured ozone residuals from three locations within the Extended-CSTR zone.

### B.4.3.1 Determining the Value of *k**

The ozone decay coefficient, *k** is calculated using ozone sample measurements, taken from in-situ sample ports, and an assumed theoretical model of the chamber's hydrodynamics. The Extended CSTR approach assumes that each individual chamber in the Extended zone is as a CSTR, and hence the Extended zone can be modeled as a group of CSTRs in series. It is important to note that the empirical ozone decay coefficient, *k**, will unlikely be the same value as the first-order decay rate constant that would be measured in a batch or plug-flow reactor. Only when the each chamber of the Extended zone behaves as an ideal CSTR and, consequently, the series of chambers acts as CSTRs-in-series, will the empirical *k** be equivalent to the true first-order rate constant. Nonetheless, the simplifying assumption of CSTR hydrodynamics and use of the CSTR-in-series equation (e.g. Equation B.10) enables a reasonable prediction of the ozone concentration at other points in an Extended (Rakness, Najm et al. 2005).

The steps outlined below pertain to a contactor with a minimum of three consecutive chambers with measurable ozone residuals. That is, there should be at least three in-situ sample ports from the Extended CSTR zone with measurable ozone residual. The three ozone residual measurements, $C_1$, $C_2$, and $C_3$, are needed to estimate the value of the ozone decay coefficient, *k**. For example, the Extended CSTR zone in the contactor shown in Exhibit B.5 includes chambers 5 through 10. The ozone residual values at any three chambers in that span can be used to represent $C_1$, $C_2$, and $C_3$ in this analysis. The following steps should be followed to calculate the $k^*$ value:

Step 1 – Use Equation B-10 and residual measurements $C_1$ and $C_2$ to calculate the $k^*$ value representing the ozone decay between locations 1 and 2, $k^*_{1-2}$. (A derivation and explanation of Equation B-10 is presented in Appendix D):

$$k^*_{1-2} = \frac{N_{1-2} \times Q}{[Volume]_{1-2}} \left[ \left( \frac{C_1}{C_2} \right)^{\left( \frac{1}{N_{1-2}} \right)} - 1 \right] \qquad \text{(B-10)}$$

where: $k^*_{1-2}$ = Empirical ozone decay coefficient between sampling locations 1 & 2, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_2$ = Measured ozone residual at location 2, mg/L
$[Volume]_{1-2}$ = Volume between sampling locations 1 and 2, gallons
$N_{1-2}$ = Number of chambers between sampling locations 1 and 2
$Q$ = Water flow through the contactor, gpm

Step 2 – Use residual measurements $C_1$ and $C_3$ along with Equation B-11 to calculate the $k^*$ value representing ozone decay between sampling locations 1 and 3, $k^*_{1-3}$:

$$k^*_{1-3} = \frac{N_{1-3} \times Q}{[Volume]_{1-3}} \left[ \left( \frac{C_1}{C_3} \right)^{\left( \frac{1}{N_{1-3}} \right)} - 1 \right] \quad \text{(B-11)}$$

where: $k^*_{1-3}$ = Empirical ozone decay coefficient between sampling locations 1 & 3, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_3$ = Measured ozone residual at location 3, mg/L
$[Volume]_{1\text{-}3}$ = Volume between sampling locations 1 and 3, gallons
$N_{1\text{-}3}$ = Number of chambers between sampling locations 1 and 3
$Q$ = Water flow through the contactor, gpm

It should be emphasized that sampling location 1 should not be at the entrance to the Extended zone, but should be at least one chamber into the zone. For example, in Exhibit B.4(A), $C_1$ should not be measured at the entrance to chamber 5, since that is the entrance to the Extended zone. Instead, the first Extended CSTR zone sampling location should be located at the effluent of chamber 5, or downstream of that location. Section O.3.2 of Appendix O of the SWTR Guidance Manual provides guidance on the use of in-situ sample ports for direct ozone measurements.

Step 3 – The value of $k^*$ that is to be used in Equation B-1 will be calculated as the average of $k^*_{1-2}$ and $k^*_{1-3}$ as shown in Equation B-12.

$$k^* = \left[ \frac{k^*_{1-2} + k^*_{1-3}}{2} \right] \quad \text{(B-12)}$$

It is normal for the individual values of $k^*_{1-2}$ and $k^*_{1-3}$ to be somewhat different. However, it is recommended that they be within the range of 80 percent to 120 percent of the average $k^*$ value calculated in Step 3. That is,

$$\frac{abs\langle k^* - k_{1-i} \rangle}{k^*} \leq 20\%$$

If the two $k^*$ values do not meet this criterion, the utility may 1) reject the measured residual values and collect new samples until this quality assurance (QA) criterion is met, or 2) select the higher of the two $k^*$ values as the more conservative estimate.

### B.4.3.2 Determining the Value of $C_{in}$

While it is possible to measure the ozone residual at the entrance to the Extended CSTR zone (e.g., an in-situ sample port), it is not recommended that the measured value be used because it is usually higher than the residual predicted by the first-order decay profile (Amy et al., 1997; Carlson et al., 1997; Hoigné and Bader, 1994; Rakness and Hunter, 2000; Rouston et

al., 1998). This phenomenon is commonly attributed to the more rapid initial ozone decay, which is followed by a somewhat slower first-order decay profile. For this reason, the $C_{in}$ representing the ozone decay in the Extended CSTR Zone should be extrapolated using the downstream measured ozone residual values.

The value of $C_{in}$ can be calculated once the value of $k^*$ is estimated from the three residual ozone measurements. Maintaining the assumption of first-order decay rate, and again using the CSTR (or equal-volume CSTR-in-series if there are more than one chamber between sample ports) assumption, Equations B-13 through B-15 can be used to estimate the value of $C_{in}$ from the three measured ozone residual concentrations:

$$C_{in,1} = C_1 \times \left[ 1 + k^* \times \frac{[Volume]_{0-1}}{N_{0-1} \times Q} \right]^{N_{0-1}} \quad \text{(B-13)}$$

$$C_{in,2} = C_2 \times \left[ 1 + k^* \times \frac{[Volume]_{0-2}}{N_{0-2} \times Q} \right]^{N_{0-2}} \quad \text{(B-14)}$$

$$C_{in,3} = C_3 \times \left[ 1 + k^* \times \frac{[Volume]_{0-3}}{N_{0-3} \times Q} \right]^{N_{0-3}} \quad \text{(B-15)}$$

where:
$k^*$ = Ozone first-order decay coefficient, $min^{-1}$
$C_1$ = Measured ozone residual at location 1, mg/L
$C_2$ = Measured ozone residual at location 2, mg/L
$C_3$ = Measured ozone residual at location 3, mg/L
$N_{0\text{-}1}$ = Number of chambers between the entrance to the Extended CSTR Zone and sampling location 1
$N_{0\text{-}2}$ = Number of chambers between the entrance to the Extended- CSTR Zone and sampling location 2
$N_{0\text{-}3}$ = Number of chambers between the entrance to the Extended- CSTR Zone and sampling location 3
$[Volume]_{0-1}$ = Volume, in gallons, between the entrance of the Extended CSTR Zone and sampling location 1
$[Volume]_{0-2}$ = Volume, in gallons, between the entrance of the Extended- CSTR Zone and sampling location 2
$[Volume]_{0-3}$ = Volume, in gallons, between the entrance of the Extended- CSTR Zone and sampling location 3
$Q$ = Water flow through the contactor, gpm

The $C_{in}$ value is then calculated as the average of the three values determined by Equations B-13 through B-15:

$$C_{in} = \left[ \frac{C_{in,1} + C_{in,2} + C_{in,3}}{3} \right]$$

A systematic example of the Extended CSTR approach is presented in section B.4.5.

### B.4.4 Quality Assurance for Extended Method Calculations

The Extended method depends on ozone residual measurements and assumed contactor hydrodynamics in order to predict ozone concentrations through the contactor. This section includes recommended QA controls intended to verify the validity of the residual predictions. Other considerations that have an important impact on characterizing the hydrodynamics and the ozone profile are discussed in Appendices C and E.

The predicted ozone residual concentration, the parameter $C$ in Equation 11-1, encompasses both the hydrodynamic assumption and ozone measurements. The principal QA issues focus on the prediction of the value of $C$. As seen in equation B-1, $C$ depends on the parameters $k^*$ and $C_{in}$. In section B.4.3.1, as part of the discussion on the calculation of $k^*$, it is stipulated that the individual $k^*$ values (i.e., $k^*_{1\text{-}2}$ and $k^*_{1\text{-}3}$) should be within 20 percent of the average value. This QA control is meant to ensure that ozone residual measurements used to calculate the ozone decay profile are consistent with the calculated profile. Since the calculation of $C_{in}$ (Equations B-13 through B-15) depends on $k^*$, as well as the measured ozone concentrations, the QA criteria for $k^*$ is sufficient for $C_{in}$. Therefore, no additional QA criteria are necessary for it.

Finally, one of the most important aspects of any application of a model towards predicting reactor performance is the confirmation of the model's prediction. This is, in essence, "model validation." Appendix O of the SWTR Guidance Manual makes several points to this effect. Ideally, model validation would take the form of measuring the actual disinfection of the *target microorganism*. A more practical alternative is to compare the predicted ozone concentrations to measured values. The general recommendation is that the predicted ozone residual should not be more than 20 percent higher than the measured value. Note that this is a one-sided QA control. In other words, an under-prediction of the ozone residual is acceptable since it results in a conservative CT value. However, an over-prediction by more than 20 percent is not desirable.

The ozone concentration measurements used to calculate $k^*$ and $C_{in}$ cannot be compared to the predicted ozone residuals, since they are interdependent. It is recommended that ozone samples be taken from other sampling locations in the contactor, and those values compared to the calculated C at those locations.

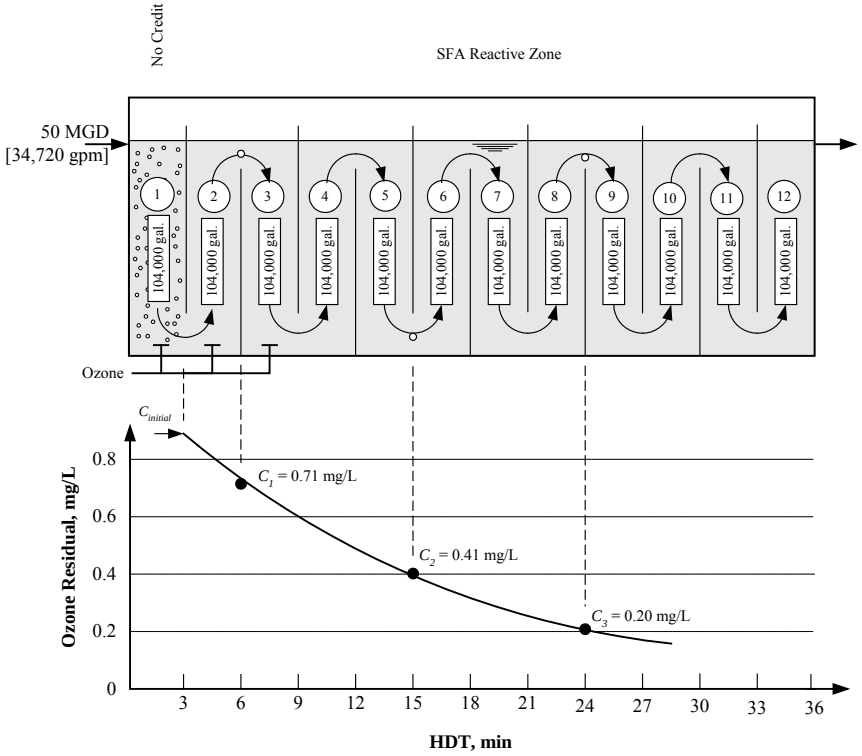
### B.4.5 Examples of Extended Method Application

This section provides examples calculating the log inactivation credits across conventional and non-conventional contactors using the Extended $T_{10}$ method and the Extended CSTR method. The calculations are completed for *Cryptosporidium* log inactivation using $k_{10}$ for *Cryptosporidium*. Calculations for *Giardia* and virus log inactivation are completed similarly by using $k_{10}$ for *Giardia* or virus in place of $k_{10}$ for *Cryptosporidium*.

#### *Example 1 – Conventional Multi-chamber Contactor with In-situ Sample Ports and One Dissolution Chamber*

Exhibit B.5 shows a schematic of a 12-chamber ozone contactor. The contactor is treating 50 MGD of water at a temperature of 20°C. The volumes of the individual chambers are noted on the schematic. Ozone is added to the first chamber only. The bottom graph in Exhibit B.5 shows the values of the ozone residual measured at the effluents of chambers 2, 5, and 8.

## Exhibit B.5 Schematic of the Ozone Contactor and the Measured Ozone Residual Values in Example 1




The goal is to calculate the *Cryptosporidium* inactivation credit across the contactor using the Extended $T_{10}$ and Extended CSTR methods.

### Extended $T_{10}$ Method

**Chamber 1 (First Dissolution Chamber) –** No *Cryptosporidium* inactivation credit is given to the first dissolution chamber.

**Chambers 2 through 12 (Extended $T_{10}$ zone)** – This zone is classified as an Extended zone. The Extended $T_{10}$ zone calculations are applied to determine the log inactivation across each of the 11 chambers. Since the third ozone residual measurement, $C_3$, is above 0.05 mg/L, and it is made at point in the contactor more than 50% of the volume, it is permissible to use the overall $T_{10}$/HDT ratio for each individual chamber. The following steps are implemented:

*Step 1: Calculate $k^*$ value* – The $k^*$ value is calculated as described in section B.3.2.1 using the three ozone residual measurements, $C_1$, $C_2$, and $C_3$ that are shown in Figure B.6. The values of $k^*_{1-2}$ and $k^*_{1-3}$ can be calculated using Equations B-2 and B-3 as follows:

$$k^*_{1-2} = \frac{Q}{[Volume]_{1-2}} \times Ln\left[\frac{C_1}{C_2}\right] = \frac{34{,}720}{3 \times 104{,}000} \times Ln\left[\frac{0.71}{0.41}\right] = 0.0611 \text{ min}^{-1}$$

$$k^*_{1-3} = \frac{Q}{[Volume]_{1-3}} \times Ln\left[\frac{C_1}{C_3}\right] = \frac{34{,}720}{6 \times 104{,}000} \times Ln\left[\frac{0.71}{0.20}\right] = 0.0705 \text{ min}^{-1}$$

The $k^*$ value is then calculated as the average of $k^*_{1-2}$ and $k^*_{1-3}$ as follows:

$$k^* = \left[\frac{k^*_{1-2} + k^*_{1-3}}{2}\right] = \left[\frac{0.0611 + 0.0705}{2}\right] = 0.0658 \text{ min}^{-1}$$

A QA check shows that the values of $k^*_{1-2}$ and $k^*_{1-3}$ are within 7.1 percent of the average $k^*$ value of 0.0658 min$^{-1}$. This value of $k^*$ is within the recommended maximum variability of 20 percent. If this criterion were not met, then the $k^*$ value could be set at 0.0705 min$^{-1}$, which is the higher (i.e., more rapid decay rate) of the two values.

*Step 2: Calculate $C_{in}$ value* – The value of $C_{in}$ is calculated using the approach described in Section B.4.2.2. With the value of $k^*$ calculated at 0.0658 min$^{-1}$, Equations B-5 to B-7 can be used to calculate the $C_{in}$ value as follows:

$$C_{in,1} = C_1 \times \exp\left(k^* \times \frac{[Volume]_{0-1}}{Q}\right) = 0.71 \times \exp\left(0.0658 \times \frac{104{,}000}{34{,}720}\right) = 0.86 \text{ mg/L}$$

$$C_{in,2} = C_2 \times \exp\left(k^* \times \frac{[Volume]_{0-2}}{Q}\right) = 0.41 \times \exp\left(0.0658 \times \frac{4 \times 104{,}000}{34{,}720}\right) = 0.90 \text{ mg/L}$$

$$C_{in,3} = C_3 \times \exp\left(k^* \times \frac{[Volume]_{0-3}}{Q}\right) = 0.20 \times \exp\left(0.0658 \times \frac{7 \times 104{,}000}{34{,}720}\right) = 0.79 \text{ mg/L}$$

Therefore,

$$C_{in} = \left[\frac{C_{in,1} + C_{in,2} + C_{in,3}}{3}\right] = \left[\frac{0.86 + 0.90 + 0.79}{3}\right] = 0.85 \text{ mg/L}$$

*Step 3: Calculate the value of $k_{10}$* – The value of $k_{10}$ for the inactivation of *Cryptosporidium* with ozone at the measured temperature of 20°C can be obtained from equation 11-2 directly and equals 0.2555 L/mg-min.

*Step 4: Calculate the Ozone Residual at the Effluent of Each Chamber* – Knowing the values of $C_{in}$ and $k^*$, the ozone concentration at the effluent of each chamber within the Extended $T_{10}$ zone can be calculated. These values are calculated using Equation B-1:

$$C_X = C_{in} \times Exp\left(-k^* \times \left(\frac{[Volume]_{0-X}}{Q}\right)\right)$$

where $C_X$ is the calculated concentration at a location "X" along the water path through the Extended $T_{10}$ zone. For example, the residual concentration at the effluent of chamber 4, $C_{4,out}$, is calculated as:

$$C_{4,out} = 0.85 \times Exp\left(-0.0658 \times \left(\frac{3 \times 104{,}000}{34{,}720}\right)\right) = 0.47 \text{ mg/L}$$

Note that the Extended $T_{10}$ zone begins at the effluent of Chamber 1, which makes the subscript to [*Volume*] in the equation above depicted as "1-4." Exhibit B.6 lists the calculated residual values for each chamber using the same approach, beginning with chamber 2.

## Exhibit B.6 Application of the Extended $T_{10}$ Method to the Example

| | | |
|---|---|---|
| Vol./Chamber = | 104,000 | gallons |
| Flowrate = | 34,720 | gpm |
| $C_{in}$ = | 0.85 | mg/L |
| $k^*$ = | 0.0658 | $min^{-1}$ |
| $k_{10}$ = | 0.2555 | L/mg-min |
| $T_{10}$/HDT = | 0.65 | |

| (1) | (2) | (3) | (4) | (5) | (6) |
|---|---|---|---|---|---|
| | | **C = $C_{out}$** | | **C = $C_{int}$** | |
| **Chamber** | **HDT from Entrance of Zone HDT, min** | **Calculated $C_{out}$ mg/L** | **Log Inactivation** | **Integrated Residual mg/L** | **Log Inactivation** |
| 2 | 3.0 | 0.70 | 0.35 | 0.77 | 0.38 |
| 3 | 6.0 | 0.57 | 0.29 | 0.63 | 0.32 |
| 4 | 9.0 | 0.47 | 0.23 | 0.52 | 0.26 |
| 5 | 12 | 0.39 | 0.19 | 0.43 | 0.21 |
| 6 | 15 | 0.32 | 0.16 | 0.35 | 0.17 |
| 7 | 18 | 0.26 | 0.13 | 0.29 | 0.14 |
| 8 | 21 | 0.21 | 0.11 | 0.24 | 0.12 |
| 9 | 24 | 0.18 | 0.09 | 0.19 | 0.10 |
| 10 | 27 | 0.14 | 0.07 | 0.16 | 0.08 |
| 11 | 30 | 0.12 | 0.06 | 0.13 | 0.07 |
| 12 | 33 | 0.10 | 0.05 | 0.11 | 0.05 |
| | | Sum = | 1.7 | Sum = | 1.9 |

*Step 4: Calculate Log Inactivation* – To calculate the log inactivation across a chamber using the $T_{10}$ method, the values of *C*, $T_{10}$ and $k_{10}$ are required. The value of $k_{10}$ for the inactivation of *Cryptosporidium* with ozone at the water temperature of 20°C was determined earlier at 0.2555 L/mg-min. Using Exhibit 11.5, the value of *C* used in calculating CT in a reactive chamber can be set equal to $C_{out}$ or to the integrated residual, $C_{int}$. Exhibit B.6 lists the $C_{out}$ and $C_{int}$ values for each chamber in the Extended $T_{10}$ zone, as well as the associated log inactivation using the following equation:

$$Log\ Inactivation = k_{10} \times C \times HDT_{chamber} \times \left( \frac{T_{10}}{HDT} \right)$$

For example, if *C* is set to $C_{out}$, the log inactivation achieved in chamber 4 is calculated as:

$$Log\ Inactivation\big|_4 = k_{10} \times C_{out,4} \times HDT_4 \times \left( \frac{T_{10}}{HDT} \right) = 0.2555 \times 0.47 \times 3 \times 0.65 = 0.23 \text{ logs}$$

On the other hand, if *C* is set to the $C_{int}$, then $C_{int}$ is calculated using the following equation:

$$C_{X,\text{int}} = \frac{\left(C_{X,in} - C_{X,out}\right)}{Ln\left[\frac{C_{X,in}}{C_{X,out}}\right]}$$

For chamber 4, $C_{int}$ is calculated as:

$$C_{4,\text{int}} = \frac{\left(0.57 - 0.47\right)}{Ln\left[\frac{0.57}{0.47}\right]} = 0.52 \text{ mg/L}$$

The log inactivation achieved in chamber 4 is then calculated as:

$$Log\ Inactivation\big|_4 = k_{10} \times C_{out,4} \times HDT_4 \times \left( \frac{T_{10}}{HDT} \right) = 0.2555 \times 0.52 \times 3 \times 0.65 = 0.26 \text{ logs}$$

Column (4) in Exhibit B.6 lists the log inactivation values calculated for chambers 2 through 12 after setting *C* equal to $C_{out}$. Column (6) in Exhibit B.6 lists the log inactivation values calculated for chambers 2 through 12 after setting *C* equal to $C_{int}$. The sum of the log inactivation achieved (totals of Columns 4 and 6 in Exhibit B.6) is 1.7 logs using the $C_{out}$ approach, and 1.9 logs using the $C_{int}$ approach.

**Extended CSTR Method**

**Chamber 1 (First Dissolution Chamber) –** No *Cryptosporidium* inactivation credit is given to the first dissolution chamber.

**Chambers 2 through 12 (Extended CSTR zone)** – This zone is classified as an Extended-CSTR zone. The Extended CSTR calculations (Section 4.3) are applied to determine the log inactivation across each of the eleven chambers. The following steps are implemented

*Step 1: Calculate $k^*$ value* – The $k^*$ value is calculated as described in section B.4.3.2.1 using the three ozone-residual measurements, $C_1$, $C_2$, and $C_3$ that are shown in Figure B.6. The values of $k^*_{1-2}$ and $k^*_{1-3}$ can be calculated using Equations B-2 and B-3 as follows:

$$k^*_{1-2} = \frac{N_{1-2} \times Q}{[Volume]_{1-2}} \left[ \left( \frac{C_1}{C_2} \right)^{\left( \frac{1}{N_{1-2}} \right)} - 1 \right] = \frac{3 \times 34{,}720}{[3 \times 104{,}000]} \left[ \left( \frac{0.71}{0.41} \right)^{\frac{1}{3}} - 1 \right] = 0.0670 \text{ min}^{-1}$$

$$k^*_{1-3} = \frac{N_{1-3} \times Q}{[Volume]_{1-3}} \left[ \left( \frac{C_1}{C_3} \right)^{\left( \frac{1}{N_{1-3}} \right)} - 1 \right] = \frac{6 \times 34{,}720}{[6 \times 104{,}000]} \left[ \left( \frac{0.71}{0.2} \right)^{\frac{1}{6}} - 1 \right] = 0.0785 \text{ min}^{-1}$$

The $k^*$ value is then calculated as the average of $k^*_{1-2}$ and $k^*_{1-3}$ as follows:

$$k^* = \left[ \frac{k^*_{1-2} + k^*_{1-3}}{2} \right] = \left[ \frac{0.0670 + 0.0785}{2} \right] = 0.0728 \text{ min}^{-1}$$

A QA check shows that the values of $k^*_{1-2}$ and $k^*_{1-3}$ are within 8 percent of the average $k^*$ value of 0.0728 min$^{-1}$. This value of $k^*$ is within the recommended maximum variability of 20 percent. If this criterion were not met, then the $k^*$ value could be set at 0.0785 min$^{-1}$, which is the higher (i.e., most rapid decay rate) of the two values.

Also note that the $k^*$ value determined by the Extended T10 method is not the same as that determined by here by the Extended CSTR method. This is to be expected since the two methods use different conceptual hydrodynamic models to determine the ozone profile.

*Step 2: Calculate $C_{in}$ value* – The value of $C_{in}$ is calculated using the approach described in Section 4.2.2. With the value of $k^*$ calculated at 0.0728 min$^{-1}$, Equations B-5 to B-7 can be used to calculate the $C_{in}$ value as follows:

$$C_{in,1} = C_1 \times \left[1 + k^* \times \frac{[Volume]_{0-1}}{N_{0-1} \times Q}\right]^{N_{0-1}} = 0.71 \times \left[1 + 0.0728 \times \frac{[104{,}000]}{1 \times 34{,}720}\right]^{1} = 0.86 \text{ mg/L}$$

$$C_{in,2} = C_2 \times \left[1 + k^* \times \frac{[Volume]_{0-2}}{N_{0-2} \times Q}\right]^{N_{0-2}} = 0.41 \times \left[1 + 0.0728 \times \frac{[4 \times 104{,}000]}{4 \times 34{,}720}\right]^{4} = 0.90 \text{ mg/L}$$

$$C_{in,3} = C_3 \times \left[1 + k^* \times \frac{[Volume]_{0-3}}{N_{0-3} \times Q}\right]^{N_{0-3}} = 0.20 \times \left[1 + 0.0728 \times \frac{[7 \times 104{,}000]}{7 \times 34{,}720}\right]^{7} = 0.80 \text{ mg/L}$$

Therefore,

$$C_{in} = \left[\frac{C_{in,1} + C_{in,2} + C_{in,3}}{3}\right] = \left[\frac{0.86 + 0.90 + 0.80}{3}\right] = 0.85 \text{ mg/L}$$

*Step 3: Calculate the value of $k_{10}$* – The value of $k_{10}$ for the inactivation of *Cryptosporidium* with ozone at the measured temperature of 20°C can be obtained from Equation 11-5 directly and equals 0.2555 L/mg-min. Otherwise the value for $k_{10}$ could be determined using Equation 11-4.

*Step 4: Calculate the Ozone Residual at the Effluent of Each Chamber* – Knowing the values of $C_{in}$ and $k^*$, the ozone concentration at the effluent of each chamber within the Extended CSTR zone can be calculated. These values are calculated using Equation B-1:

$$C_X = \frac{C_{in}}{\left[1 + k^* \times \frac{[Volume]_{0-X}}{N_{0-X} \times Q}\right]^{N_{0-X}}}$$

where $C_X$ is the calculated concentration at the effluent of chamber "X". For example, the residual concentration at the effluent of chamber 4 is calculated as:

$$C_4 = \frac{0.85}{\left[1 + 0.0728 \times \frac{[3 \times 104{,}000]}{3 \times 34{,}720}\right]^{3}} = 0.47 \text{ mg/L}$$

Note that the Extended CSTR zone begins at the effluent of Chamber 1, which makes the subscript to [*Volume*] in the equation above depicted as "1-4". Exhibit B.7 lists the calculated residual values for each chamber using the same approach, beginning with chamber 2.

## Exhibit B.7 Application of the Extended CSTR Method to the Example

| | | |
|---|---|---|
| Vol./Chamber = | 104,000 | gallons |
| Flowrate = | 34,720 | gpm |
| $C_{in}$ = | 0.854 | mg/L |
| $k^*$ = | 0.0728 | $min^{-1}$ |
| $k_{10}$ = | 0.2555 | L/mg-min |

| (1) | (2) | (3) | (4) |
|---|---|---|---|
| **Chamber** | **HDT from Entrance of Zone HDT, min** | **Calculated $C_{out}$ mg/L** | **Log Inactivation** |
| 2 | 3.0 | 0.70 | 0.35 |
| 3 | 6.0 | 0.58 | 0.30 |
| 4 | 9.0 | 0.47 | 0.26 |
| 5 | 12 | 0.39 | 0.23 |
| 6 | 15 | 0.32 | 0.19 |
| 7 | 18 | 0.26 | 0.16 |
| 8 | 21 | 0.21 | 0.14 |
| 9 | 24 | 0.18 | 0.12 |
| 10 | 27 | 0.14 | 0.10 |
| 11 | 30 | 0.12 | 0.08 |
| 12 | 33 | 0.10 | 0.07 |
| | | Sum = | 2.0 |

*Step 4: Calculate Log Inactivation* – Knowing the values of *C*, $k_{10}$, and $k^*$, Equation 11-4 is used to calculate the log inactivation achieved in each chamber in the Extended CSTR Zone:

$$Log-inactivation = Log\left[1 + 2.303\ k_{10}\ C_X\ \frac{[Volume]_X}{Q}\right]$$

where $C_X$ is the effluent residual concentration from Chamber X, $C_{X,out}$ (see Exhibit 11-3), while $[Volume]_X$ is the volume of that chamber. For example, for chamber 4, if the $C_X$ is set at $C_{X,out}$, then the log inactivation achieved in chamber 4 is calculated as:

$$Log-inactivation = Log\left[1 + 2.303 \times 0.2555 \times 0.47 \times \frac{104{,}000}{34{,}720}\right] = 0.26 \text{ logs}$$

Column (4) in Exhibit B.7 lists the log inactivation values calculated for chambers 2 through 12 under this approach. The sum of the log inactivation achieved (total of Column 4 in Exhibit B.6) is 2.0 logs.

Exhibit B.8 shows a schematic of a non-conventional ozone contactor. The contactor is treating 12.5 MGD of water at a temperature of 8°C. The contactor is comprised of a long and narrow channel with no baffle walls. Ozone is added into the raw water line upstream of the contactor. Ozone analyzers are installed at three locations along the length of the contactor. The bottom graph in Exhibit B.8 shows the values of the rolling average ozone residuals reported by the ozone analyzers at the three monitored locations. It is noted that the contactor length:width ratio is 90:12 (7.5:1), while the length:height ratio is 90:9 (10:1), both of which are greater than the minimum desired value of 5:1. The goal is to calculate the *Giardia* inactivation credit across the contactor using the Extended $T_{10}$ and Extended CSTR methods.

Because non-conventional contactors do not possess specific zones, such as the baffle gaps in conventional over-under contactors, where a high degree of mixing can create homogeneity of solute concentrations, they pose additional considerations for measuring representative ozone concentrations at specific HDTs in the contactor. That is, measuring ozone concentrations at the baffle gaps of a conventional contactor is generally regarded as providing a representative average ozone concentration exiting the specific chamber. In contrast, it is not as well understood what the exact hydrodynamics are within the non-conventional contactors, and consequently whether an ozone sample taken at one point along the length of the contactor is representative of the average concentration across the cross-section at that theoretical HDT. The issue of a representative ozone measurement is also true for calculated ozone residuals. Furthermore, since the non-conventional contactor does not have clearly defined chambers, it is necessary to subdivide the Extended zone of non-conventional contactors into an appropriate number of conceptual *segments* such that characteristic ozone concentrations can be calculated and the Extended $CT_{10}$ method applied. The term "segment" is chosen here to avoid confusion with the term "chamber," which may connote an actual physical chamber in a baffled contactor, as opposed to a conceptual or theoretical segment denoted here.

This guidance utilizes a calculation described in numerous chemical reactor design texts (e.g. Levenspiel 1999; Fogler 2005) for defining the hydrodynamics measured by a tracer test in terms of a series of ideal CSTRs. The so-called Tanks-in-Series (sometimes called CSTR-in-series) model, as applied to non-conventional contactors, affords a method for subdividing the single large chamber of the contactor into multiple equal-sized conceptual segments based on an analysis of a tracer test. The calculations necessary are similar to those required for calculating the $T_{10}$ value. Appendix E provides guidance on performing this calculation for both pulse input and step input tracer tests.

The tanks-in-series model is typically used in reaction engineering by employing CSTR-specific reaction equations (e.g. Equations B-11 and B-13) to each of the theoretical tanks. Application of the tanks-in-series calculation here is used only to calculate a theoretical number of segments in which to subdivide the single large chamber. Once the chamber is divided into segments with theoretical HDTs for each, the Extended $T_{10}$ method can be applied accordingly. This calculation is viewed as a pragmatic method to determine a reasonable number of segments. Taken with the discussion regarding the linear extrapolation of the overall $T_{10}$/HDT ratio across all segments (see section B.4.2 above), it is reasonable to establish some recommendation for limiting the extrapolation of $T_{10}$/HDT and calculation of $C_x$ to what could otherwise be an infinite number of conceptual segments.

## Extended $T_{10}$ Method

The entire contactor shown in Exhibit B.8 is treated as an Extended zone. Tracer analysis performed as per Appendix E determined a $T_{10}$/HDT ratio of 0.67 Furthermore, the tanks-in-series tracer analysis determined that the tracer output corresponded to a series of 12 equal-sized segments. The Extended zone, which is 90 ft long, has ozone sample points at approximately 16%, 33% and 50% of the contactor volume. Since the third ozone residual measurement, $C_3$, is above 0.05 mg/L, and it is measured at point in the contactor that is at least 50% of the volume, it is permissible to use the overall $T_{10}$/HDT ratio for each of the 12 theoretical chambers, or segments. The Extended $T_{10}$ zone calculations (Section B.4.2) are applied to determine the log inactivation across each segment, and thus the entire contactor. The following steps are implemented:

*Step 1: Calculate $k^*$ value* – The $k^*$ value is calculated as described in section B.4.2.1 using the three ozone residual measurements, $C_1$, $C_2$, and $C_3$ that are shown in Exhibit B.8. With 12 equal segments, each segment would have a volume of approximately 6059 gallons. Ozone sample ports depicted in Exhibit B.8 are located at approximately the effluents of segments 2, 4, and 6. With a flowrate of 8,680 gpm, values of $k^*_{1-2}$ and $k^*_{1-3}$ can be calculated using Equations B-2 and B-3 as follows:

$$k^*_{1-2} = \frac{Q}{[Volume]_{1-2}} \times Ln\left[\frac{C_1}{C_2}\right] = \frac{8{,}680}{2 \times 6{,}059} \times Ln\left[\frac{0.54}{0.32}\right] = 0.375 \text{ min}^{-1}$$

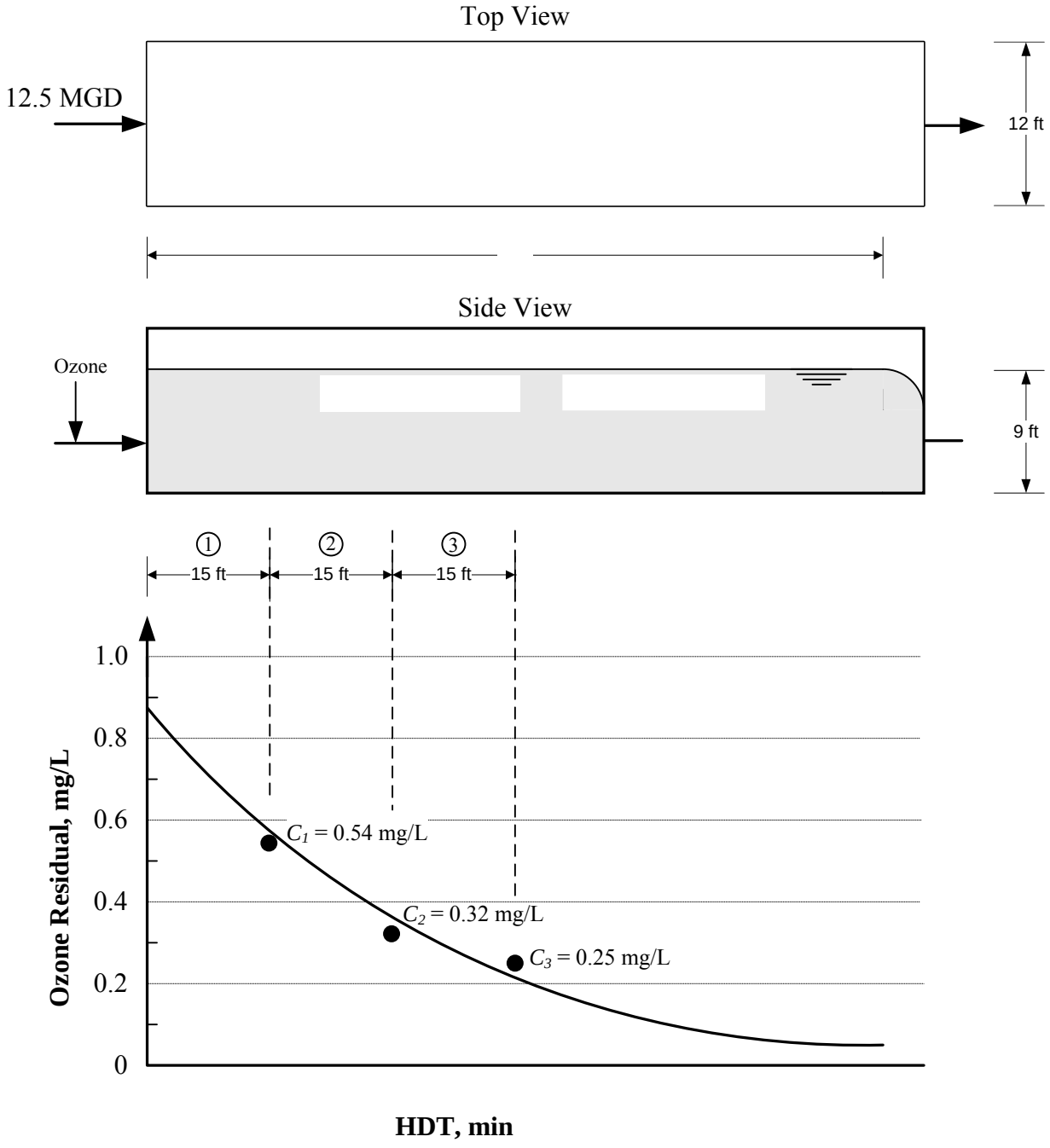
$$k^*_{1-3} = \frac{Q}{[Volume]_{1-3}} \times Ln\left[\frac{C_1}{C_3}\right] = \frac{8{,}680}{4 \times 6{,}059} \times Ln\left[\frac{0.54}{0.25}\right] = 0.276 \text{ min}^{-1}$$

The $k^*$ value is then calculated as the average of $k^*_{1-2}$ and $k^*_{1-3}$ as follows:

$$k^* = \left[\frac{k^*_{1-2} + k^*_{1-3}}{2}\right] = \left[\frac{0.375 + 0.276}{2}\right] = 0.325 \text{ min}^{-1}$$

A QA check shows that the values of $k^*_{1-2}$ and $k^*_{1-3}$ are within 15 percent of the average $k^*$ value of 0.325 min$^{-1}$. This value of $k^*$ is within the recommended maximum variability of 20 percent. If this criterion were not met, then the $k^*$ value could be set at 0.375 min$^{-1}$, which is the higher of the two values.

**Exhibit B.8 Schematic of the Ozone Contactor and the Measured Ozone Residual Values in Example 2**




<u>*Step 2: Calculate $C_{in}$ value*</u> – The value of $C_{in}$ is calculated using the approach described in Section 4.2.2. With the value of $k^*$ calculated at 0.325 $min^{-1}$, Equations B-5 to B-7 can be used to calculate the $C_{in}$ value as follows:

$$C_{in,1} = C_1 \times \exp\left( k^* \times \frac{[Volume]_{0-1}}{Q} \right) = 0.54 \times \exp\left( 0.325 \times \frac{2 \times 6{,}059}{8{,}680} \right) = 0.85 \text{ mg/L}$$

$$C_{in,2} = C_2 \times \exp\left( k^* \times \frac{[Volume]_{0-2}}{Q} \right) = 0.32 \times \exp\left( 0.325 \times \frac{4 \times 6{,}059}{8{,}680} \right) = 0.79 \text{ mg/L}$$

$$C_{in,3} = C_3 \times \exp\left( k^* \times \frac{[Volume]_{0-3}}{Q} \right) = 0.25 \times \exp\left( 0.325 \times \frac{6 \times 6{,}059}{8{,}680} \right) = 0.98 \text{ mg/L}$$

Therefore,

$$C_{in} = \left[ \frac{C_{in,1} + C_{in,2} + C_{in,3}}{3} \right] = \left[ \frac{0.85 + 0.79 + 0.98}{3} \right] = 0.87 \text{ mg/L}$$

*Step 3: Calculate the value of $k_{10}$* – The value of $k_{10}$ for the inactivation of *Giardia* with ozone at the measured temperature of 8°C can be obtained using the equation presented in B.4.2, which states:

$$Giardia\ k_{10} = 1.0380 \times (1.0741)^{Temp}$$

Using the above equation, the $k_{10}$ value for the inactivation of *Giardia* with ozone at 8 °C is calculated at 1.839 L/mg-min.

*Step 4: Calculate the Ozone Residual at the Effluent of Each Segment* – As with a conventional contactor composed of multiple chambers, knowing the values of $C_{in}$ and $k^*$, the ozone concentration at the effluent of each chamber within the Extended $T_{10}$ zone can be calculated. In the case of non-conventional contactors, the tanks-in-series tracer analysis (see Appendix E) is employed to subdivide the single large chamber into multiple theoretical *segments*, and the concentration at the effluent of each segment determined as with conventional contactors. These values are calculated using Equation B-1:

$$C_X = C_{in} \times Exp\left( -k^* \times \left( \frac{[Volume]_{0-X}}{Q} \right) \right)$$

where $C_X$ is the calculated concentration at a location "X" along the water path through the Extended $T_{10}$ zone. For example, the residual concentration at the effluent of the 8th theoretical segment, $C_{8,out}$, is calculated as:

$$C_{8,out} = 0.87 \times Exp\left( -0.325 \times \left( \frac{8 \times 6{,}059}{8{,}680} \right) \right) = 0.14 \text{ mg/L}$$

Note that the Extended $T_{10}$ zone begins at the effluent of segment 1, which makes the subscript to [*Volume*] in the equation above depicted as "1-8". Exhibit B.6 lists the calculated residual

values for each segment using the same approach, beginning with segment 2. In many cases, the ozone sampling ports will be located at HDTs that do not coincide with the HDT of a theoretical segment. However, since values of $C_x$ will be *calculated* for all the theoretical segments, it is problematic that the actual ozone measurements are not used as the characteristic $C_x$ value.

### Exhibit B.9 Application of the Extended $T_{10}$ Method to the Example

| | | |
|---|---|---|
| Vol./Segment = | 6,059 | gallons |
| Flowrate = | 8,680 | gpm |
| $C_{in}$ = | 0.87 | mg/L |
| $k^*$ = | 0.325 | $min^{-1}$ |
| $k_{10}$ = | 1.839 | L/mg-min |
| $T_{10}$/HDT = | 0.67 | |

| (1) | (2) | (3) | (4) |
|---|---|---|---|
| | | **C = $C_{out}$** | |
| **Segment** | **HDT from Entrance of Zone HDT, min** | **Calculated $C_{out}$ mg/L** | **Log Inactivation** |
| 1 | 0.7 | 0.69 | 0.60 |
| 2 | 1.4 | 0.55 | 0.48 |
| 3 | 2.1 | 0.44 | 0.38 |
| 4 | 2.8 | 0.35 | 0.30 |
| 5 | 3.5 | 0.28 | 0.24 |
| 6 | 4.2 | 0.22 | 0.19 |
| 7 | 4.9 | 0.18 | 0.15 |
| 8 | 5.6 | 0.14 | 0.12 |
| 9 | 6.3 | 0.11 | 0.10 |
| 10 | 7.0 | 0.09 | 0.08 |
| 11 | 7.7 | 0.07 | 0.06 |
| 12 | 8.4 | 0.06 | 0.05 |
| | | Sum = | 2.7 |

*Step 4: Calculate Log Inactivation* – To calculate the log inactivation across a segment using the $T_{10}$ method, the values of *C* and $k_{10}$ are required. The value of $k_{10}$ for the inactivation of *Giardia* with ozone at the water temperature of 8 °C was determined earlier at 1.839 L/mg-min. Using Exhibit 11.2, the value of $C_{out}$ is used in calculating CT for each segment in the Extended zone. Exhibit B.9 lists the $C_{out}$ values for each segment in the Extended $T_{10}$ zone, as well as the associated log inactivation using the following equation:

$$Log\ \ Inactivation\ = k_{10} \times C \times HDT_{segment} \times \left( \frac{T_{10}}{HDT} \right)$$

For example, the log inactivation achieved in segment 8 is calculated as:

$$Log\ \ Inactivation\big|_8\ = k_{10} \times C_{out,8} \times HDT_8 \times \left( \frac{T_{10}}{HDT} \right) = 1.839 \times 0.14 \times (5.6 - 4.9) \times\ 0.67\ = 0.12 \text{ logs}$$

Column (4) in Exhibit B.9 lists the log inactivation values calculated for segments 1 through 12 The sum of the log inactivation achieved (total of Column 4 in Exhibit B.9) is 2.7 logs.

**Extended CSTR Method**

The Extended CSTR method is presented in this guidance primarily as a method for systems which choose not to perform a tracer study. For conventional multi-chamber contactors, the utility has the option of using the Extended CSTR method even when a tracer is available. This method may afford a better inactivation credit than the Extended $T_{10}$ method when the $T_{10}$/HDT ratio is considerably low (e.g. below 0.5). However, as explained in section B.4.3, the CSTR assumption may be somewhat conservative with regards to a multi-chamber contactor's hydrodynamics, whose design is to promote plug-flow as opposed to highly mixed flow. In the context of the non-conventional contactor, the guidance described immediately above and in Appendix E uses a tracer study to closely characterize the hydrodynamics in terms of tanks-in-series. In as much as the tanks-in-series analysis calculates the exact number of segments corresponding to the contactor's residence time distribution, there is virtually no conservatism with respect to the hydrodynamic model. There is minor conservatism in rounding down the tanks-in-series calculation to the nearest integer. Additionally, the CSTR assumption inherently leads to a lower reaction efficiency as compared to the complete segregation assumption. Nonetheless, the EPA feels that the overall lack of conservatism related to the hydrodynamic assumption precludes the use of the Extended CSTR method to non-conventional contactors.

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# Appendix C
# Measuring Ozone Residual

Accurate ozone residual data will allow the calculation of correct log-inactivation values and maintain optimized performance. Ozone residual measurements might be inaccurate if sampled or measured incorrectly. Residual measurement Quality Assurance (QA) issues include:

- Configuration of the ozone sample collection lines within the contactor.
- Stability of the indigo trisulfonate reagent when analyzing grab samples.
- Standardization and maintenance of on-line ozone analyzers.

## C.1 Sample Collection

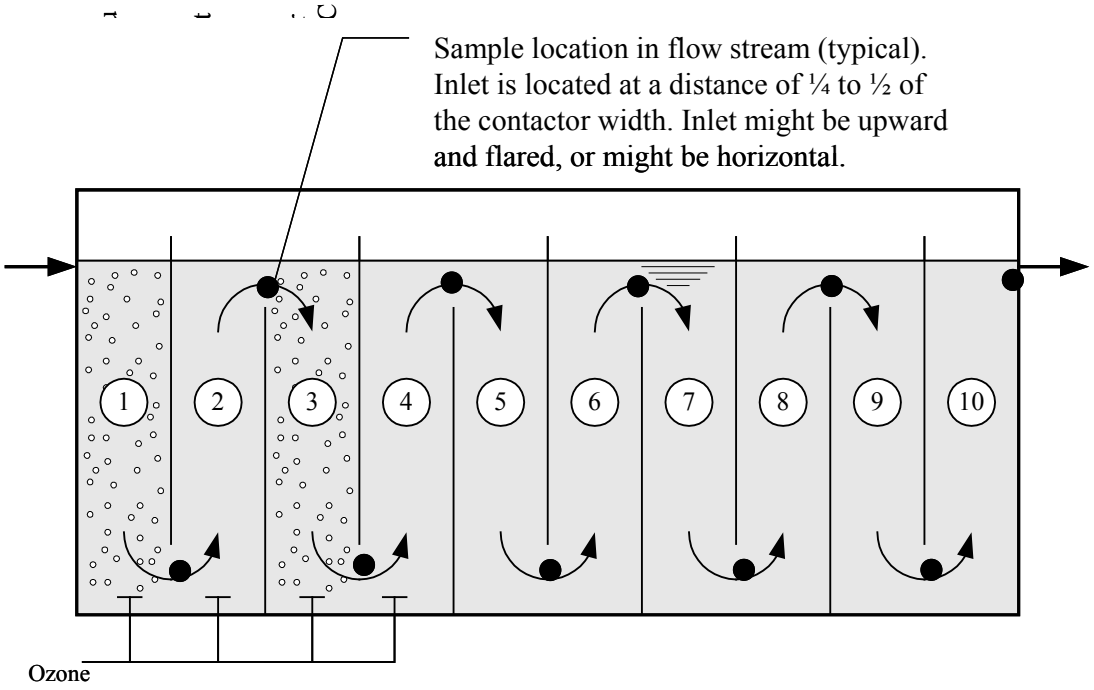
Ozone contactors are sealed vessels that may have a single large chamber, referred to as non-conventional contactors, or multiple chambers (typically separated by walls or baffles with somewhat small openings separating the chambers) referred to as conventional contactors. Water samples from the interior of the contactor are collected via sample lines that penetrate the walls or roof structure of the contactor. Since dissolved ozone decays in water with a half-life ranging from less than a minute to 30 minutes for typical drinking water treatment applications, the ozone profile (the concentration of ozone along the general flow-path of a contactor) will vary significantly depending on the water quality, the method of operation, and water flow conditions (e.g. hydrodynamics and HDT). Consequently, the location, number, and design of sample tubes plays an important role in operating the ozonation process and calculating CT credit. This guidance, and additional information found in the SWTR, provides recommendations for measuring ozone residual including sample ports considerations and analytical methods.

Placement of sample ports depends on the configuration of the contactor – conventional versus non-conventional. For conventional contactors, considerable experience of practitioners, as well as more recent studies (Shiono and Teixeira 2000; Kim, Kim, et al. 2010; Kim, Nemlioglu, et al. 2010; Kim, Elovitz, et al. 2010), suggest that the water flowing through the baffle gap (i.e. exiting one chamber and flowing into the next, see Exhibit C.1) is reasonably well mixed. Whether the water arrives that way from the previous chamber, or slip-streams or "parcels" of different water with greatly varying age distributions arrive simultaneously and mix thoroughly in the constricted region of the baffle gap is not well understood. However, it appears that for typical baffle gaps (i.e. not too wide), placement of the sample tube within the (3-dimensional) center of the gap (¼ to ½ the distance), can afford a representative sample of water exiting the chamber.

Once again, because the rate of ozone decay varies considerably with operating conditions, a separate sample port located at the outlet of each chamber within the contactor allows maximum flexibility for sampling ozone residual over variable operating conditions. Sample ports located at the outlets of diffusion chambers should be placed to ensure the diffusers do not interfere with the collected sample. Gas bubbles might be carried into the sample inlet and

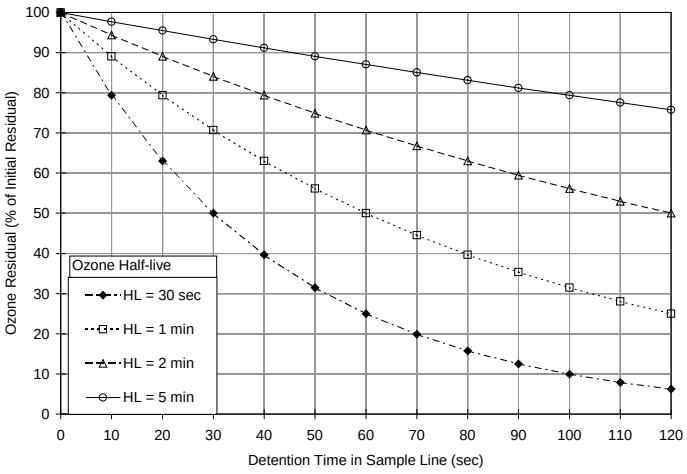
cause errors in the residual measurement. A sample inlet tube that is flared and that is turned either upward or opposite the flow of the water (depending on the location) reduces the potential for entrapment of gas bubbles. However in highly turbid waters, a vertical inlet and flared configuration might result in clogging due to solids deposition inside the line. In these cases a compromise is to position the sample line such that the inlet is horizontal rather than vertical.

**Exhibit C.1 Example Sample Locations in an Over/Under Baffled Bubble Diffuser Contactor**




Minimizing the travel time through the sample line is also important, especially when the ozone decay rate is high (i.e., ozone half-life is short). It is desirable to minimize the travel time so that the ozone decay during travel from the inside of the contactor to end of the tube is <10 percent. Exhibit C.2 shows the relationship between simulated sample line travel time and ozone residual loss for various ozone half-life values. For example, the travel time in the sample line should be less than 10 seconds if the ozone half-life is one minute, in order to maintain the ozone residual loss at or below 10 percent.

## Exhibit C.2 Relationship Between Ozone Residual Loss and Detention Time through the Ozone Sample Line for Various Ozone Half-Life Values




The sample line diameter should be large enough (minimum 3/8$^{th}$ inch inside diameter and preferably ½-in to ¾-inch) to minimize clogging of the line with suspended solids. Sample pipe diameter and flow rate should be selected in order to:

- Maintain consistent flow without plugging.
- Minimize detention time in the sample line.
- Meet flow rate requirements of an on-line analyzer installed at that location.

Gravity flow is all that is necessary to meet sample flow requirements in most locations. In other cases, pumping is necessary. Sample lines might contain some gas bubbles as well as liquid. It is important to ensure that lines are vented in high spots where gas binding might occur. Gaseous ozone in high concentrations is hazardous to breathe. Sample line vents and drains should be directed away from occupied areas. Section O.3.2 of Appendix O of the *Guidance Manual for Compliance with the Filtration and Disinfection Requirements for Public Water Systems Using Surface Water Sources (U.S. EPA 1991)* (commonly referred to as the SWTR Guidance Manual) includes further information regarding direct measurement of dissolved ozone.

This guidance also addresses the use of the Extended $T_{10}$ method for non-conventional contactors. Because these contactors are typically, by definition, dominated by a single large chamber, there are no distinct physical structures (e.g. baffle gaps) that tend to provide a well-mixed environment from which to collect a representative water sample with an estimable HDT. Consequently, the question arises as to how to collect a representative sample, and whether a basic measure of the distance along the longitudinal transect of the theoretical flow-path is a good measure of the HDT associated with a sample port. There is not sufficient knowledge

regarding the range of hydrodynamic conditions that might be present at any point within the array of non-conventional contactor geometries. However, in this guidance, the EPA has taken a pragmatic approach, based on considerable input from stakeholders, that for contactors with sufficiently long chambers (relative to the height and width), the distance along the longitudinal transect is a reasonable proxy for HDT. In addition, water samples representative of the cross-section of the contactor at that HDT can be collected with proper placement of a sample tube.

In this consideration, it is recognized that the inlet and outlet to most non-conventional contactors, regardless of the overall length, are not designed for optimal hydrodynamic efficiency. As a result, the hydrodynamics in the regions of the inlet and outlet may be prone to dead volumes, short-circuiting and large eddy effects. In contrast, if the length-scale of the chamber is large with respect to the cross-sectional dimensions, the hydrodynamics may become more developed, more plug-flow-like and homogeneous (across the cross-section) within the majority of the length of the contactor. Once again, in a pragmatic approach to develop a useful method for calculating CT credit in such contactors, the EPA proposes that representative water (ozone) samples can be collected in the same manner as samples from conventional contactors provided the contactor meets certain geometric criteria. In the context of assuming potentially poor hydrodynamics at the inlet and outlet regions, but more homogenous flow elsewhere, non-conventional contactors should have high length-to-width (L:W) and length-to-height (L:H) ratios.

To employ the Extended $T_{10}$ method, a minimum of three sample ports is needed. Because of the potential to have considerable inlet/outlet effects, placement of the sample tubes is important, and consideration should be given to placement in the areas believed to have more developed, steady-state flow. The EPA suggests the installation of additional ports to allow flexibility in monitoring and control. Moreover, the EPA suggests an assessment of the short-term variability in the ozone measurement from any sample port to help determine if the hydrodynamics in that region are irregular (unsteady and non-ideal) or relatively stabile (more plug-flow-like). For example, if multiple ozone samples collected over a short time-span during steady operating conditions demonstrate high variability (e.g. > 20%) in ozone concentration, then this may indicate that the hydrodynamics of the contactor in that region are not conducive to affording a representative ozone sample.

All other considerations regarding sample port design follow those of conventional contactors.

## C.2 Ozone Residual Measurement

Ozone residual should be determined using the Indigo Method (Standard Methods 4500-Ozone – 20$^{th}$ Edition 1998) when analyzing grab samples. The method assumes that high-purity reagents are used. Since the publication of the 20$^{th}$ Edition, several reports (Gordon et al. 2000a and 2000b, Rakness et al. 2001, Rakness and Hunter 2001, and Rakness et al. 2002) have been published discussing a potential biasing in the Indigo Method. The potential biasing involves the value of the so-called "sensitivity factor," *f*, as defined in the Standard Method. In short, these

reports suggest that the actual sensitivity factor might be lower than the Standard Method's value, and hence the calculated ozone concentration will be undervalued.

The Standard method's proportionality constant, *f*, (0.42 L $mg^{-1}cm^{-1}$) that is used to calculate the ozone residual is based on an indigo trisulfonate molar absorbance, $\varepsilon$, of 20,000 $M^{-1}$ $cm^{-1}$. These recent reports suggest that *f* may not be constant and may depend on:

- The source and age of the neat indigo trisulfonate solid.
- The age and handling of the indigo stock solution that is prepared as part of the method.

Briefly, these reports indicate that, due to either of the above aspects, *f* can be substantially lower than 0.42 L $mg^{-1}cm^{-1}$. In other words, the molar absorbance can be much lower than 20,000 $M^{-1}cm^{-1}$. Gordon et al. (2000a and 2000b), Rakness et al. (2001), Rakness and Hunter (2001), and Rakness et al. (2002) reported that the apparent molar absorbance of some indigo stock solutions might be as low as 11,000 $M^{-1}cm^{-1}$, and in an extreme case 6,000 $M^{-1}cm^{-1}$. The authors suggest that the ramifications of applying an *f* value of 0.42 L $mg^{-1}cm^{-1}$ when the solution has a lower true *f* value are the underestimation of the ozone concentration. EPA does not have information that these issues are resolved at the time of issuing the guidance manual.

The gravimetric indigo trisulfonate method is fairly easy to apply in the field and is accurate. It should be noted that the method described herein is somewhat different than the 20th Edition of Standard Methods in that the volume of both the blank and the samples are determined gravimetrically. The procedural steps include:

- Prepare indigo stock solution as described in Standard Methods.
- Prepare Reagent II solution (for ozone residuals greater than 0.05 mg/L), as described in Standard Methods.
- Prepare flasks for sampling.
- Clean, dry and label several 125 mL Erlenmeyer flasks (enough for each sample plus one blank).
- Obtain the tare weight of each flask.
- Add 10.0 mL of Reagent II solution to each flask.
- Add approximately 90 mL of distilled water to one or two flasks and use these flasks as the blank (i.e., use value from one blank or average of values from two blanks).
- Collect ozone sample.
- Thoroughly flush sample line to be used.

- Do not run sample down the side of the flask, as this will cause ozone off-gassing.
- Fill flask with sample, gently swirling flask until a light blue color remains. Do not bleach completely or the residual value will be incorrect.
- Wipe-dry the outside of sample and blank flasks.
- Weigh sample and blank flasks.
- Total weight for sample is tare weight of flask plus 10 mL indigo plus added sample.
- Total weight for blank is tare weight of flask plus 10 mL indigo plus added distilled water.
- Prepare the spectrophotometer for measuring absorbance.
- Identify the cell path length (e.g., 1 cm, 5 cm, etc.).
- Set the wavelength to 600 nanometers.
- Measure absorbance of blank and samples within four hours.
- Follow instructions for spectrophotometer concerning zeroing the instrument.
- Record absorbance of each sample and each blank.
- Complete calculations – see example below.

Example:

A 10 mL aliquot of Reagent II solution was added to a 125 mL Erlenmeyer flask that was used for the blank. The flask had a tare weight of 83.62 g. The final weight of the flask, plus the 10 mL aliquot of reagent, plus the added distilled water was 179.77 g. The total volume of the 10 mL Reagent II aliquot plus added distilled water was determined by subtracting the bottle's tare weight from the total weight, assuming that 1 mL of liquid weighs 1 g (96.15 mL = [179.77 g – 83.62 g] * 1 mL / 1 g).

The spectrophotometer had a path length of 1 cm. The absorbance reading of the gravimetric blank was measured as 0.234 $cm^{-1}$ at wavelength of 600 nm. This reading must be corrected for the difference in the volume of the blank used in order to check the quality of the reagent. The calculated absorbance of a 1:100 blank dilution can be determined using Equation C-1. In this case, the 1:100 absorbance value was 0.225 $cm^{-1}$, which is greater than or equal to 0.225 $cm^{-1}$. This means that the indigo trisulfonate solution was considered acceptable.

$$\frac{\left(\frac{\text{Absorbance}}{\text{Path Length}}\right)}{100\text{ mL}} \times \text{Volume of Blank} \;=\; \text{Absorbance in cm}^{-1}\ @\,100\text{ mL} \qquad \text{(C-1)}$$

$$\frac{\left(\frac{0.234}{1\text{ cm}}\right)}{100\text{ mL}} \times 96.15\text{ mL} \;=\; 0.225\text{ cm}^{-1}$$

The 125 mL flask that was used for the ozone sample had a tare weight of 94.10 g. Sample water was directed into the 10 mL of Reagent II solution until a light blue color remained. The final weight of the flask, plus the 10 mL aliquot plus the sample, was 167.39 g. The absorbance reading at a path length of 1 cm was 0.159. The volume of the water sample was 63.29 mL (63.29 mL = [167.39 g – 94.10 g – 10 g] * 1 mL / 1 g). The ozone residual was calculated using Equation C-2, which resulted in a value of 0.41 mg/L.

$$\text{mg/L} = \frac{(A_B \times V_B) - (A_S \times V_T)}{f \times V_S \times b} \qquad \text{(C-2)}$$

where $A_B$ = absorbance of the blank (as measured, not as corrected by equation C-1)

$A_S$ = absorbance of the sample

$V_B$ = volume of the blank plus indigo, mL

$V_T$ = total volume of the sample plus indigo, mL

$V_S$ = volume of the sample (total weight – tare weight – 10)

$f = 0.42$

b = path length of cell, cm

$$\frac{(0.234 \times 96.15) - (0.159 \times 73.29)}{0.42 \times 63.29 \times 1} \;=\; 0.41\text{ mg/L}$$

## C.3 On-line Ozone Residual Analyzer Calibration

On-line ozone residual analyzers are available that can continuously monitor ozone residual in the water. This makes it possible to automate the disinfection credit calculation using the plant's computer-control system. However, the analyzers must be maintained properly and their calibration must be checked periodically so that readings match grab-sample results that are based on the indigo trisulfonate procedure. Generally, probe-type monitor readings tend to drift downward over time due to weakening of the electrolyte solution. Calibration checks should be

conducted regularly, such as at least once per week. This section describes a calibration check protocol which involves collecting grab-samples and analyzer readings simultaneously and comparing the values.

The calibration check should consist of collecting at least three, and preferably five, ozone residual grab samples and corresponding analyzer readings. The following calibration protocol has been used successfully at operating ozone facilities.

- Collect three to five grab-sample ozone residuals. Obtain an analyzer reading while the grab sample is being collected. Wait 15 seconds to 30 seconds between each pair of grab sample and analyzer reading.

- Measure the ozone residual concentration in the grab samples using the indigo trisulfonate method.

- Calculate the average grab-sample ozone residual value and the average analyzer ozone residual value.

- Compare the average of the on-line analyzer to that of the indigo grab-samples. The average of the on-line analyzer should not deviate more than 10 percent or 0.05 mg/L (which ever is largest) from the grab-sample average. If the average of the on-line analyzer deviates more than this, then adjust the meter reading per the manufacturer's instructions. Note that this QA control is two-sided. It is especially important that the on-line analyzer not record more than 10 percent or 0.05 mg/L greater than the grab samples. However, a negative deviation (negative bias), while not effecting public health, may also be useful as an indication of a malfunctioning unit.

- Allow the analyzer to stabilize for a period of 30 minutes after adjusting the meter reading and repeat steps 1 through 4 until the difference calculated in step 4 is <10 percent of the grab-sample average and <0.05 mg/L.

## C.4 References

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# Appendix D

# Derivation of Extended CSTR Equations

The discussion presented in the document used some key equations and relied on specific assumptions. In this appendix, one key equation is derived, and one key assumption is discussed and justified.

## D.1 Derivation of the Equation Used to Calculate k*

In Appendix B, Equation B-2 expressed the value of $k^*$ between two points 1 and 2 as shown by Equation D-1:

$$k^*_{1-2} = \frac{N_{1-2} \times Q}{[Volume]_{1-2}} \left[ \left( \frac{C_1}{C_2} \right)^{\left( \frac{1}{N_{1-2}} \right)} - 1 \right] \tag{D-1}$$

Equation D-1 is a transformation from the equation of first-order decay across a series of $N$ equal-size continuous stirred tank reactors (CSTRs):

$$\left( \frac{C_2}{C_1} \right) = \frac{1}{\left[ 1 + k^*_{1-2} \left( \frac{HDT}{N_{1-2}} \right) \right]^{N_{1-2}}} \tag{D-2}$$

The derivation of this equation can be found in many reference texts on modeling chemical reactors (e.g., Froment et al., 1990, Levenspiel 1999). Since hydraulic detention time (HDT) is equal to the volume between locations 1 and 2, $[Volume]_{1\text{-}2}$, divided by the flowrate, $Q$, then Equation D-2 is transformed to Equation D-3:

$$\left( \frac{C_2}{C_1} \right) = \frac{1}{\left[ 1 + k^*_{1-2} \left( \frac{[Volume]_{1-2}}{Q \times N_{1-2}} \right) \right]^{N_{1-2}}} \tag{D-3}$$

Therefore,

$$\left[ 1 + k^*_{1-2} \left( \frac{[Volume]_{1-2}}{Q \times N_{1-2}} \right) \right]^{N_{1-2}} = \left( \frac{C_1}{C_2} \right) \tag{D-4}$$

then,

$$\left[1 + k^*_{1-2}\left(\frac{[Volume]_{1-2}}{Q \times N_{1-2}}\right)\right] = \left(\frac{C_1}{C_2}\right)^{\frac{1}{N_{1-2}}} \tag{D-5}$$

then,

$$k^*_{1-2}\left(\frac{[Volume]_{1-2}}{Q \times N_{1-2}}\right) = \left[\left(\frac{C_1}{C_2}\right)^{\frac{1}{N_{1-2}}} - 1\right] \tag{D-6}$$

and then,

$$k^*_{1-2} = \frac{Q \times N_{1-2}}{[Volume]_{1-2}}\left[\left(\frac{C_1}{C_2}\right)^{\frac{1}{N_{1-2}}} - 1\right] \tag{D-7}$$

As noted, Equation D-2 is based on the fundamental assumption that the hydrodynamic profile through the volume separating locations 1 and 2 can be approximated by a series of *N* equal-size CSTRs. If equal-size chambers separate locations 1 and 2, then each chamber is somewhat conservatively assumed to be an ideal CSTR, with HDT = [*Volume*]/Q, and the value of *N* in the above derivation is set equal to the number of chambers between locations 1 and 2. However, it was recognized that not all ozone contactors are configured with equal-size chambers in series. It is possible to treat each chamber as its own CSTR and have a series of unequal-size CSTRs. An expression of $C_2/C_1$ similar to that shown in Equation D-2 is still possible. For example, if locations 1 and 2 were separated by three CSTRs with HDT values of $HDT_a$, $HDT_b$, and $HDT_c$, the ratio of $C_2/C_1$ for a first-order decay reaction can still be expressed as:

$$\left(\frac{C_2}{C_1}\right) = \frac{1}{\left[1 + k^*_{1-2}\left(HDT_a\right)\right]} \times \frac{1}{\left[1 + k^*_{1-2}\left(HDT_b\right)\right]} \times \frac{1}{\left[1 + k^*_{1-2}\left(HDT_c\right)\right]} \tag{D-8}$$

Or in general terms,

$$\left(\frac{C_2}{C_1}\right) = \prod_i \frac{1}{\left[1 + k^*(HDT)_i\right]} \tag{D-9}$$

Unfortunately, it is not possible to transform Equation D-9 to derive a simple linear expression of $k^*$ as a function of the other measured parameters when the number of chambers is greater than three. To maintain a singular methodology for any number of chambers, and to

allow the calculation to be performed in conventional spreadsheets and plant computer control systems, a compromise was to assume equal-volume CSTRs. With this assumption, Equation D-1 is used to calculate the value of $k^*$ between two sampling locations regardless of the number and sizes of chambers between the two locations.

The simplifying assumption of equal-size CSTRs for calculating $k^*$ is non-conservative relative to a $k^*$ value calculated by allowing for unequal-sized chambers. That is, for first-order ozone decay reaction, unequal-sized CSTR reactors in series would be the least efficient (ideal) reactor configuration for promoting ozone decay. Hence, calculating $k^*$ based on equation D-9 gives the largest, or most conservative, value of $k^*$. The model of equal-sized CSTR reactors in series is a more efficient configuration for promoting ozone decay. Hence, calculating $k^*$ from Equation D-1 (based on equation D-2) gives a less conservative estimate of $k^*$. To take the comparison to the opposite extreme, calculating $k^*$ based on a plug-flow assumption (e.g., Equation 4-7) gives the smallest, or a non-conservative, estimate of $k^*$.

The impact of the simplifying equal-sized CSTR assumption on the estimate of $k^*$ and $C_{in}$ involves several considerations. The first issue is the quantitative difference between the most conservative estimate, based on Equation D-9, and the recommended approach based on Equation D-2. This is essentially an issue of what chemical and hydrodynamic conditions affect the efficiency of the ozone decay reaction. This is a somewhat complex issue dependent on the reaction rate (represented by the Damköhler I Number, $D_{a1}$ [$D_{a1}= k^* \times HDT$]), the number of chambers considered, and the disparity in volumes among the unequal-sized chambers. In principle, as the reaction rate increases, the number of chambers approaches two (the minimum), and the volume differences among the chambers increases, the difference between the reaction efficiencies of the two reactor configurations increases. Some situations could result in approximately 30% differences between $k^*$ values. Other situations could results in negligible differences. Because of the many factors involved, it is difficult to establish qualitative rules for all possible cases. However, the utility and the primacy agency may consider further analysis for contactors with 2-3 chambers with a large volume difference and a large $D_{a1}$.

The second, and perhaps overriding, issue concerning the impact of the simplifying assumption is whether or not it still provides a certain element of conservatism over the true contactor performance. That is, an actual contactor with unequal-sized chambers might have reasonably good hydrodynamics such that even the equal-size CSTR assumption is conservative. This too, however, is very system specific, and is a difficult issue to resolve due to the numerous factors involved.

# Appendix E
# Tracer Test Data Development & Analysis

## E.1 Overview & Quality Assurance

Tracer test data are required to implement the Extended $T_{10}$ method. The tracer test is conducted only once, and its results are applicable as long as the contactor geometry remains unchanged. Tracer tests can be conducted in the same manner as those described in Appendix C of the Surface Water Treatment Rule Guidance Manual (SWTR Guidance Manual) (USEPA 1991). Either a "step input" or a "pulse input" tracer test could be used. The step input tracer test consists of applying a constant dosage of a conservative chemical (tracer) at the head of the ozone contactor and measuring the concentration of the tracer chemical at the outlet of the contactor at selected time intervals, while maintaining constant flowrate through the contactor. The pulse input tracer test (also called a slug dose test) consists of a rapid injection of a specific mass of a tracer chemical at the head of the ozone contactor over a very short period of time, and then measuring the tracer concentration at the outlet of the contactor at selected time intervals, again while maintaining a relatively constant flowrate through the contactor.

Appendix C of the SWTR Guidance Manual recommends that at least four tracers be performed at a range of expected operational flowrates. This Guidance supports the earlier recommendation and further *strongly* recommends that at least two tracer tests be conducted at the lowest and highest expected operational flowrates. This recommendation is based on possibility that hydrodynamic efficiency of the contactor could vary as a function of flowrate. For example, in terms of dispersion, it could be expected that the overall dispersion in a contactor could vary with the flowrate. For the ozone-*Cryptosporidium* reaction system, increasing dispersion leads to an effective decrease in efficiency of the relevant reactions in the contactor. Therefore, it is important to consider the effect of flowrate on the hydrodynamic efficiency. Once all the tracer tests have been evaluated, it is recommended that the utility use the tracer data that demonstrates the greatest "spread" (i.e. dispersion or variance) in the tracer. This is a measure of conservatism applied to the Extended $T_{10}$ approach, and perhaps more important for the newly developed approach for non-conventional contactors. The added safety factor it provides depends strongly on the specific contactor geometry and the range of the expected flowrates.

Systems considering tracer studies should contact their state regulatory agencies regarding the use of tracer chemicals. Commonly used tracer chemicals are fluoride or lithium ions. Fluoride might be added as sodium fluoride (NaF) or as fluosilicic acid ($F_6H_2Si$). Lithium is typically added as lithium chloride (LiCl). At the time of preparation of this document, only sodium fluoride and fluosilicic acid were NSF-certified additives for drinking water. As such, a fluoride-based chemical has been the tracer of choice in many applications.

Lithium chloride is not yet NSF-approved, but some State regulatory agencies allow the use of lithium for tracer testing because of the following three primary advantages it has over fluoride:

1. Lithium is present at very low background concentrations (about 5 to 10 ppb) in most natural waters compared to the fluoride background concentration, which might range from 0.2 to 0.5 mg/L.

2. Lithium can be analyzed reliably at concentrations as low as 5 ppb, which is much lower than the typical minimum-reporting limit for fluoride (which is about 0.1 mg/L).

3. There is no health-based limit for lithium in drinking water. The fluoride limit is 4 mg/L.

The range of lithium concentration during a tracer test is typically between 5 and 250 ppb, which is a 50-fold range. The range of fluoride concentration in the tracer test is between about 0.5-mg/L to 3.5 mg/L, which is a 5-fold range. A broader range provides for a better resolution in the tracer test results. The collection of reliable tracer test data is important for accurate assessment of the variance as well as $T_{10}$ values. The Quality Assurance criteria outlined below are applicable for obtaining high-quality tracer test results. Once completed, follow the steps concerning preparing test results for calculation of the variance of the data.

1. At least two tracer tests should be performed for each contactor. The tracer tests should be conducted at the lowest and highest expected flowrates through the contactor. The expected flowrate is that which the utility plans on operating the system. If there are multiple identical parallel contactors, tracer tests can be conducted on one contactor and applied to the other identical contactors. Of the RTDs developed from the tracer tests, it is recommended that, for further use in the Extended $T_{10}$ method, the utility use the RTD demonstrating the greatest tracer spread.

2. During the tracer test, the flow rate through the contactor should remain as constant as possible, with the maximum or minimum value remaining within 90 percent to 110 percent of the average flowrate during the test. That is, the flow rates should be ± 10 percent of the set flow rate.

3. The tracer test should be conducted over a minimum period of three hydraulic detention times (HDT) of the contactor. For example, if the HDT of the contactor is eight (8) minutes, then the minimum test duration (during which effluent samples are being collected) is 24 minutes.

4. The number of samples collected during a tracer test should be maximized to generate the most accurate estimate of the RTD as possible. There are practical limitations to the number of samples that can be taken and analyzed. However, it is recommended that a minimum of thirty samples (30) should be collected during a tracer test. Samples can be collected at unequal time intervals. Sampling density should be focused on the inflection and peak (for pulse-input) portions of the breakthrough curve to minimize the numerical errors due to approximations during the RTD calculations. This would benefit the utility by decreasing the estimated spread of the tracer and hence the inefficiency of the contactor hydrodynamics.

5. The background concentration of the tracer chemical upstream of the point of tracer injection should be measured over the duration of the tracer test. The number of background samples collected should be no less than 20 percent of the number of effluent samples collected during the test.

6. For pulse input tests, the total mass of the tracer ion or chemical recovered should be calculated and should be between 85 percent and 115 percent of the mass added. It should be emphasized that this refers to the tracer ion being measured (i.e., $Li^+$ or $F^-$) and not the chemical being added (i.e., LiCl or NaF). Therefore, the mass of $Li^+$ or $F^-$ added is calculated and then compared to the total mass of $Li^+$ or $F^-$ recovered from the contactor effluent over the course of the tracer test.

7. For step-input tests, the average tracer ion or chemical concentration in the samples collected during the last 10 percent of the sampling period should be between 85 percent and 115 percent of the tracer dose being added. It is also noted here that the concentration referenced is that of the tracer ion being measured (such as $Li^+$ or $F^-$), and not the tracer chemical being added (i.e., LiCl or NaF).

## E.2 Development & Analysis of Tracer Test Data for Variance Calculation

The tracer test characterizes the hydrodynamic patterns inside a flow-through contactor. Details concerning tracer tests are described in Appendices C and O of the SWTR Guidance Manual, in Teefy & Singer (1990), and Teefy (1996). The two types of tracer tests - *step* tracer test and *pulse* (or slug) tracer test - differ in how the test is conducted, but they result in nearly the same final set of values that are required for the implementation of the extended $T_{10}$ approach. The guidance outlined here generally follows precedence set in Appendices C and O of the SWTR Guidance Manual. In particular, Appendix C describes the technique for spreadsheet-style numerical integration of tracer data using *right rectangle rule* integration step (described on page C-21, Appendix C simply as the "rectangle rule"). Appendix O in turn describes a technique for numerical differentiation of the tracer data using the *forward differentiation* technique. These two techniques take slightly different approaches to step-wise treatment of the data; an aspect that is apparent only for treatment of the step dose data.

The tracer data analysis presented here outlines the procedure for transforming the tracer data to a residence time distribution (RTD), or exit age distribution $E$. In addition, the procedures allow for calculation of the mean residence time ($t_m$) (which is defined as the first-moment of the RTD) and the variance of the RTD (also called the second moment about the mean of the RTD). Finally, calculations are outlined for determining the cumulative distribution function $F$ (also called $F$-curve, which is useful in determining the $T_{10}$ value.

### E.2.1 Pulse-Input Tracer Test Data Development & Analysis

An example of a pulse input tracer study test and the RTD analysis results are shown in Exhibit E.1. The tracer test corresponds to the Extended $T_{10}$ example shown in Appendix B, section B.4.5, Exhibits B.8 and B.9 for the non-conventional contactor. The tracer test was conducted at a flow rate that resulted in a theoretical hydraulic detention time (HDT) of 8.4 minutes. This HDT value was calculated as the total volume (in gallons) of the main large chamber shown in Exhibit B.8 divided by the water flow rate during the tracer test (in gpm).

In this example, a 5-gallon solution containing 10 lbs of fluoride ($F^-$) was injected as quickly as possible (<15 seconds) into the influent water flow to the contactor. Samples were collected frequently from the feed water upstream of the point of tracer injection (background samples) and from the contactor effluent. The background fluoride concentration was monitored throughout the test duration and the average was calculated at 0.20 mg/L. The HDT of the water volume between the tracer addition point and the effluent tracer sampling location was 8.35 minutes. The contactor effluent samples were collected every minute for 25 minutes.

The first four columns in Exhibit E.1 represent the actual tracer test results. The subsequent columns include calculated values based on the tracer test results. Columns 5 through 10 contain information that is required for the Extended $T_{10}$ calculation for the non-conventional contactors. Column 11 contains additional information. A description of the contents and calculations for each column is presented below:

Column 1: Datapoint counter.

Column 2: Time of tracer chemical sample collection from the start of the test.

Column 3: Background tracer chemical concentration measured in the influent water during the test.

Column 4: Tracer chemical concentration measured at the effluent of the contactor during the test.

Column 5: Average background tracer chemical concentration calculated as the average of all the background values listed in Column 3.

Column 6: "Effective Concentration", which refers to the tracer concentration measured in the effluent of the contactor (Column 4) minus the average background tracer concentration (Column 5). The value of the effective concentration must be positive. At the beginning and at the end of the step-tracer test, the value of the effective concentration may be calculated as a negative value because the effluent tracer concentration may be slightly lower than the background concentration due to minor analytical errors. This is the case, for example, for the first two time points with measured concentrations of 0.15 and 0.19 mg/L. In such cases, the value of effective concentration must be set to zero instead of a negative value.

Column 7: The values in Column 7 are the product of the effective concentration (Column 6) and the preceding timestep (i.e. $t_i - t_{i-1}$). For example, the effective concentration at datapoint #8 (2.27 mg/L) is multiplied by the difference between the elapsed time for datapoint #8 (7 minutes) and the elapsed time for datapoint #7 (6 minutes).

All the values Column 7 are summed at the bottom Column 7. This sum calculates the area under the pulse response curve according the equation

$$area = \int C dt \cong \sum_i C_i \Delta t_i$$

where $\Delta t_i = t_i - t_{i-1}$.
It has units of [mass × time/volume] and is used to calculate the total mass recovered during the tracer test. This is examined further below.

This summation is an example of the right rectangle rule for integration established in Appendix C of the SWTR Guidance Manual. Note, that due to the process of using the previous timestep, the first cell in this column is blank (or "0").

Column 8: The values in Column 8 are a simple multiplication of the time (Column 2) and the value in Column 7.

All the values Column 8 are summed at the bottom Column 8. This sum represents the integral

$$sum = \int tCdt \cong \sum_i t_i C_i \Delta t_i$$

where $\Delta t_i = t_i - t_{i-1}$

In addition, this value is used in the calculation of the mean residence time, $t_m$, by dividing the sum of Column 8 (128.27) by the sum of Column 7 (14.29).

$$t_m = \frac{\int tCdt}{\int Cdt} \cong \frac{sum\ column\ 8}{sum\ column\ 7}$$

In this example, a value of 8.97 minutes is calculated. This compares well with the theoretical HDT of 8.35 minutes. Differences may arise due to experimental error in conducting the tracer test or numerical dispersion related to the quality and time-resolution of the tracer data.

Column 9: The values in Column 9 represent the values for the residence time distribution (RTD) or exit age distribution *E*. These values are calculated by dividing the effective concentration (Column 6) by the sum of the values of Column 7 (i.e. 14.29, shown at the bottom of Column 7). For example, the value for *E* for data point #11 is calculated by dividing the concentration (1.81; Column 6) by 14.29.

Column 10: The values in Column 10 are calculated as the product of several terms according to the equation

$$(t_i - t_m)^2 E_i \Delta t_i$$

where $\Delta t_i = t_i - t_{i-1}$, and $t_m$ is taken as the value calculated at the bottom of the Exhibit. Note, one should use the calculated value of the mean residence time rather than the theoretical HDT.

Once the values for each term in Column 10 are calculated, the sum of all values is made at the bottom of Column 10. $\int (t - t_m)^2 Edt \cong \sum (t_i - t_m)^2 E_i \Delta t_i$

The summed value at the bottom of Column 10 represents the variance of the RTD about the mean, typically denoted by the term $\sigma^2$. The significance of this value is discussed below.

Note, this summation is again an example of the right rectangle established in Appendix C of the SWTR Guidance Manual. Due to the process of using the previous timestep, the first cell in this column is blank (or "0").

Column 11: Finally, the values in Column 11 represent the *F*-curve. They are calculated according to the equation

$$F(t_i) = \int_i E dt \cong \sum_i E_i \Delta t_i$$

The value for each datapoint is calculated by multiplying the value in Column 10 by the preceding time step (i.e.$<t_i = t_i - t_{i-1}$) and then adding this value to the preceding value of F (i.e. in the cell above). For example, for datapoint #12,

$$F_{12} = E_{12}\Delta t_{12} + F_{11} = 0.09(11-10) + 0.76 = 0.85$$

The *F*-curve is useful for determining the $T_{10}$ value as described the Appendix O of the SWTR Guidance Manual.

## Exhibit E.1 Example of Pulse-Tracer Test Results & RTD Analysis

**theoretical HDT =** 8.35 min
**Flow Rate =** 12.5 MGD

| | | | % Recovered |
|---|---|---|---|
| **Mass Added =** | 10 | lbs | |
| **Mass Recovered ($t_m$-basis) =** | 9.3 | lbs | 93% |
| **Mass Recovered (HDT-basis) =** | 8.6 | lbs | 86% |

| (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) | (9) | (10) | (11) |
|---|---|---|---|---|---|---|---|---|---|---|
| **Tracer Test Data:** | | | | **Avg.** | | | | | | |
| **Data Point** | **Time min** | **Background Conc. mg/L** | **Conc. mg/L** | **Background Conc. mg/L** | **Effective Conc. mg/L** | **$C_i\Delta t$** | **$t_iC_i\Delta t$** | **E(t)** | **$(t-t_m)^2xE_i\Delta t$** | **$F(t_i)$** |
| 1 | 0 | 0.200 | 0.150 | 0.20 | 0.00 | | | 0.00 | | |
| 2 | 1 | | 0.190 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 3 | 2 | 0.230 | 0.200 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 4 | 3 | | 0.203 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 | 4 | 0.210 | 0.293 | 0.20 | 0.09 | 0.09 | 0.36 | 0.01 | 0.16 | 0.01 |
| 6 | 5 | | 0.793 | 0.20 | 0.59 | 0.59 | 2.95 | 0.04 | 0.65 | 0.05 |
| 7 | 6 | 0.220 | 1.700 | 0.20 | 1.50 | 1.50 | 8.98 | 0.10 | 0.93 | 0.15 |
| 8 | 7 | | 2.469 | 0.20 | 2.27 | 2.27 | 15.86 | 0.16 | 0.62 | 0.31 |
| 9 | 8 | 0.180 | 2.530 | 0.20 | 2.33 | 2.33 | 18.62 | 0.16 | 0.15 | 0.47 |
| 10 | 9 | | 2.475 | 0.20 | 2.27 | 2.27 | 20.45 | 0.16 | 0.00 | 0.63 |
| 11 | 10 | 0.190 | 2.0150 | 0.20 | 1.81 | 1.81 | 18.12 | 0.13 | 0.13 | 0.76 |
| 12 | 11 | | 1.470 | 0.20 | 1.27 | 1.27 | 13.94 | 0.09 | 0.36 | 0.85 |
| 13 | 12 | 0.200 | 1.0931 | 0.20 | 0.89 | 0.89 | 10.68 | 0.06 | 0.57 | 0.91 |
| 14 | 13 | | 0.780 | 0.20 | 0.58 | 0.58 | 7.50 | 0.04 | 0.65 | 0.95 |
| 15 | 14 | 0.200 | 0.490 | 0.20 | 0.29 | 0.29 | 4.02 | 0.02 | 0.51 | 0.97 |
| 16 | 15 | | 0.380 | 0.20 | 0.18 | 0.18 | 2.65 | 0.01 | 0.45 | 0.98 |
| 17 | 16 | 0.200 | 0.312 | 0.20 | 0.11 | 0.11 | 1.74 | 0.01 | 0.38 | 0.99 |
| 18 | 17 | | 0.262 | 0.20 | 0.06 | 0.06 | 1.00 | 0.00 | 0.27 | 0.99 |
| 19 | 18 | 0.220 | 0.250 | 0.20 | 0.05 | 0.05 | 0.84 | 0.00 | 0.27 | 1.00 |
| 20 | 19 | | 0.218 | 0.20 | 0.02 | 0.02 | 0.29 | 0.00 | 0.11 | 1.00 |
| 21 | 20 | 0.200 | 0.210 | 0.20 | 0.01 | 0.01 | 0.13 | 0.00 | 0.06 | 1.00 |
| 22 | 21 | | 0.210 | 0.20 | 0.01 | 0.01 | 0.15 | 0.00 | 0.07 | 1.00 |
| 23 | 22 | 0.190 | 0.203 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.00 |
| 24 | 23 | | 0.195 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.00 |
| 25 | 24 | 0.200 | 0.201 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.00 |
| 26 | 25 | | 0.200 | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.00 |
| | | | | | note, all values must be ≥0 | 14.29 | 128.27 | | 6.32 | |

| | |
|---|---|
| $t_m$ = | **8.97** |
| $\sigma^2$ = | **6.32** |
| $N$ = | **12.7** |
| $\sigma_\theta^2$ = | **0.079** |

In Exhibit E.1, the summations at the bottom of specific columns are the information needed to implement the Extended $T_{10}$ method for non-conventional contactors. The principal calculation needed is the equivalent number of tanks-in-series represented by the RTD. According to the tanks-in-series theory, the number of theoretical number of tanks, *N*, is determined according to

$$N = \frac{t_m^2}{\sigma^2}$$

As shown in Exhibit E.1,

$$N = \frac{t_m^2}{\sigma^2} = \frac{(8.97)^2}{6.32} = 12.7$$

Therefore, the tracer test determined that the contactor has a hydrodynamic character similar to a series of 12.7 CSTRs in series. Implementation of the Extended $T_{10}$ method rounds the value of *N* down to the nearest integer, and uses this value for the number of segments for subdividing the non-conventional contactor. In this example, the Extended $T_{10}$ calculation would use 12 segments.

Exhibit E.1 contains additional calculations that are not necessary for implementing the Extended $T_{10}$ method, but are needed for quality control of the tracer test, or for further general information concerning the contactor hydrodynamics.

The mass of the tracer chemical recovered is calculated according to:

$$Tracer\ Mass,\ lbs = 0.005792 \times Q \times t_m \times \sum C_i \Delta t_i$$

where: Q = water flow rate during the test, MGD

$t_m$ = mean residence time determined by the RTD analysis, minutes

$\sum C_i \Delta t_i$ = total sum of the area under the tracer response curve, calculated as the sum of the terms in Column 7.

0.005792 = conversion factor, which is 8.34 lbs/gallon divided-by 1440 min/day

The mass of tracer chemical recovered was calculated at 8.6 lbs using the above equation, which is 93 percent of the tracer mass added (10 lbs). This value is within the acceptable range of 85 percent – 115 percent. For comparison, the mass of tracer recovered if the calculation is based on HDT instead of $t_m$ is 8.6 lbs for a 86 percent recovery.

Exhibit E.1 also shows the value for the dimensionless variance denoted by the term, $\sigma_\theta^2$, which is the dimensionless form of the variance, $\sigma^2$. The variance represents the second moment abut the mean of the RTD. It is calculated according to

$$\sigma_\theta^2 = \frac{\sigma^2}{t_m^2}$$

which is the inverse of the equation for determining *N*, such that

$$\sigma_\theta^2 = \frac{1}{N}$$

Aside from an alternate form of calculating *N*, the value of $\sigma_\theta^2$ is useful for comparing the results of different tracer studies. For example, a higher value of $\sigma_\theta^2$ measured at a tracer study of a different flowrate suggests that there is greater mixing in the contactor under that flowrate. When more than one tracer test is performed, such as at various flowrates as

recommended, the conservative recommendation is to use the RTD data set associated with the greatest variance, $\sigma_\theta^2$.

Finally, Column 11 shows the calculation of the *F*-curve which is one method for determining the $T_{10}$ value for the contactor. The $T_{10}$ value for this tracer study appears to be between 5 and 6 minutes. When interpolated properly, and divided by the mean residence time $t_m$, the $T_{10}$ ratio is obtained.

### E.2.2 Step-Input Tracer Data Development & Analysis

An example of a step-input tracer test data and the RTD analysis results are shown in Exhibit E.2. As with the pulse test above the step tracer test was conducted in the non-conventional contactor shown for the Extended $T_{10}$ example in Appendix B (section B.4.5, Exhibits B.8 and B.9). This tracer test was also conducted at a flow rate that resulted in a theoretical hydraulic detention time (HDT) of 8.35 minutes. This theoretical HDT value was calculated as the total volume (in gallons) of the main large chamber shown in Exhibit B.8 divided by the water flow rate during the tracer test (in gpm). The chemical feed solution concentration and flowrate achieved a theoretical tracer concentration of 1.7 mg/L. This tracer dose value is denoted as $C_o$ in the header of Exhibit E.2.

## Exhibit E.2 Example of Step-Tracer Test Results and RTD Analysis

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | **$C_o$ =** | 1.7 | mg/L | | | |
| | | | | | **Average Final Effective Conc. =** | 1.54 | mg/L | | | |
| | | | | | **% Steady State =** | 90% | | | | |
| (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) | (9) | (10) | (11) |
| **Tracer Test Data:** | | | | **Avg.** | | | | | | |
| **Data** | **Time** | **Background Conc.** | **Conc.** | **Background Conc.** | **Effective Conc.** | | | | | |
| **Point** | **min** | **mg/L** | **mg/L** | **mg/L** | **mg/L** | **$<C_i$** | **$F(t_i) = C_i/C_{avg\_max}$** | **$t_i<C_i$** | **$E_i = <F_i/<t_i$** | **$(t-t_m)^2xE_i$ <t** |
| 0 | 0 | 0.20 | 0.20 | 0.20 | 0.00 | | 0.000 | | 0.0000 | |
| 1 | 1 | | 0.20 | 0.20 | 0.00 | 0.00 | 0.000 | 0.00 | 0.0000 | 0 |
| 2 | 2 | 0.25 | 0.20 | 0.20 | 0.00 | 0.00 | 0.000 | 0.00 | 0.0000 | 0 |
| 3 | 3 | | 0.20 | 0.20 | 0.00 | 0.00 | 0.000 | 0.00 | 0.0044 | 0.1580832 |
| 4 | 4 | 0.22 | 0.21 | 0.20 | 0.01 | 0.01 | 0.004 | 0.03 | 0.0397 | 0.9870264 |
| 5 | 5 | | 0.27 | 0.20 | 0.07 | 0.06 | 0.044 | 0.30 | 0.1017 | 1.6171139 |
| 6 | 6 | 0.18 | 0.43 | 0.20 | 0.22 | 0.16 | 0.146 | 0.94 | 0.1563 | 1.3955149 |
| 7 | 7 | | 0.67 | 0.20 | 0.46 | 0.24 | 0.302 | 1.68 | 0.1747 | 0.6902496 |
| 8 | 8 | 0.21 | 0.94 | 0.20 | 0.73 | 0.27 | 0.477 | 2.15 | 0.1582 | 0.1544181 |
| 9 | 9 | | 1.18 | 0.20 | 0.98 | 0.24 | 0.635 | 2.19 | 0.1252 | 1.811E-05 |
| 10 | 10 | 0.20 | 1.37 | 0.20 | 1.17 | 0.19 | 0.760 | 1.92 | 0.0907 | 0.092923 |
| 11 | 11 | | 1.51 | 0.20 | 1.31 | 0.14 | 0.851 | 1.53 | 0.0599 | 0.2424492 |
| 12 | 12 | 0.19 | 1.60 | 0.20 | 1.40 | 0.09 | 0.911 | 1.10 | 0.0378 | 0.3425397 |
| 13 | 13 | | 1.66 | 0.20 | 1.46 | 0.06 | 0.948 | 0.75 | 0.0228 | 0.3667422 |
| 14 | 14 | 0.18 | 1.70 | 0.20 | 1.49 | 0.04 | 0.971 | 0.49 | 0.0125 | 0.3130403 |
| 15 | 15 | | 1.72 | 0.20 | 1.51 | 0.02 | 0.984 | 0.29 | 0.0066 | 0.2383962 |
| 16 | 16 | 0.21 | 1.73 | 0.20 | 1.52 | 0.01 | 0.990 | 0.16 | 0.0046 | 0.2238288 |
| 17 | 17 | | 1.73 | 0.20 | 1.53 | 0.01 | 0.995 | 0.12 | 0.0020 | 0.1252382 |
| 18 | 18 | 0.20 | 1.74 | 0.20 | 1.53 | 0.00 | 0.997 | 0.05 | 0.0013 | 0.1056345 |
| 19 | 19 | | 1.74 | 0.20 | 1.53 | 0.00 | 0.998 | 0.04 | 0.0018 | 0.1786096 |
| 20 | 20 | 0.20 | 1.74 | 0.20 | 1.54 | 0.00 | 1.000 | 0.05 | -0.0011 | -0.137209 |
| 21 | 21 | | 1.74 | 0.20 | 1.54 | 0.00 | 0.999 | -0.04 | 0.0018 | 0.2571887 |
| 22 | 22 | 0.20 | 1.74 | 0.20 | 1.54 | 0.00 | 1.001 | 0.06 | -0.0011 | -0.191685 |
| 23 | 23 | | 1.74 | 0.20 | 1.54 | 0.00 | 0.999 | -0.04 | 0.0006 | 0.1111397 |
| 24 | 24 | 0.20 | 1.74 | 0.20 | 1.54 | 0.00 | 1.000 | 0.02 | 0.0000 | 0 |
| 25 | 25 | | 1.74 | 0.20 | 1.54 | 0.00 | 1.000 | 0.00 | | |
| | | | | | | 1.54 | note, all values must be ≥ 0 | 13.81 | | 7.27 |
| | | | | | | | | | **$t_m$ =** | **8.99** |
| | | | | | | | | | **$\sigma^2$ =** | **7.27** |
| | | | | | | | | | **N =** | **11.1** |
| | | | | | | | | | **$\sigma_\theta^2$ =** | **0.090** |

The first four columns in Exhibit E.2 represent the actual tracer test results. The subsequent columns include calculated values based on the tracer test results and are required for the Extended $T_{10}$ calculation for the non-conventional contactors. Column 11 contains additional information. A description of the contents and calculations for each column is presented below:

Column 1: Datapoint counter.

Column 2: Time of tracer chemical sample collection from the start of the test.

Column 3: Background tracer chemical concentration measured in the influent water during the test.

Column 4: Tracer chemical concentration measured at the effluent of the contactor during the test.

Column 5: Average background tracer chemical concentration calculated as the average of all the background values listed in Column 3.

Column 6: "Effective Concentration", which refers to the tracer concentration measured in the effluent of the contactor (Column 4) minus the average background tracer concentration (Column 5). The value of the effective concentration must be positive. At the beginning of the step-tracer test, the value of the effective concentration may be calculated as a negative value because the effluent tracer concentration may be slightly lower than the background concentration due to minor analytical errors. This is the case, for example, for the first two time points with measured concentrations of 0.15 and 0.19 mg/L. In such cases, the value of effective concentration must be set to zero instead of a negative value.

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Column 7: The values in Column 7 are calculated as the change between the effective concentration (Column 6) at this timestep and the value of the effective concentration of the preceding time step (i.e. $<C_i = C_i - C_{i-1}$). For example, to calculate the <C for datapoint #16, the effective concentration at datapoint #15 (1.49 mg/L) is subtracted from the concentration at datapoint #16 (1.51) to obtain 0.02 mg/L.

All the values Column 7 are summed at the bottom Column 7. This sum calculates a value for the maximum concentration reached at the end of the tracer experiment. This value (1.54 mg/L) is shown at the top of the Exhibit as "Average Final Effective Conc." This value is often determined from averaging the last few data points in the Column 7. This value is compared to the theoretical chemical dose ($C_o$) of 1.7 mg/L. This result is discussed below in terms of quality control.

This summation is an example of the right rectangle rule for integration, established in Appendix C of the SWTR Guidance Manual, for the term

$$\int dC \cong \sum_i \Delta C_i$$

where $C_i = C_i - C_{i-1}$.

Note, that due to the process of using the previous timestep, the first cell in this column is blank (or "0").

Column 8: Column 8 determines the values for the *F*-curve, or the cumulative distribution function, *F*, calculated as the ratio of the effective concentration (Column 6) to the calculated tracer chemical dose (1.54 mg/L) determined at the bottom of Column 7. It is noted that all the *F* values must be positive. The value of *F* might be returned as negative at the beginning of the step tracer test, because the effluent tracer concentration is slightly lower than the background concentration due to minor analytical errors. In such cases, the value of *F* must be set to zero instead of the negative value. The F-curve can be used for calculating the $T_{10}$ value. It is also used further for calculating the *E*-curve (Column 10).

Column 9: The values in Column 9 are the product between the time at that datapoint (Column 2) and the change in the effective concentration (Column 6) at this timestep and the value of the effective concentration of the <u>preceding</u> timestep (i.e. $<C_i = C_i - C_{i-1}$). This is also equivalent to multiplying the value in Column 2 and that in Column 7. For example, to calculate the $t<C$ for datapoint #16, the effective concentration at datapoint #15 (1.49 mg/L) is subtracted from the concentration at datapoint #16 (1.51) to obtain 0.02 mg/L, and that value is multiplied by the time for datapoint #16 (15 min) to obtain 0.29 for Column 9.

All the values Column 9 are summed at the bottom Column 9, and the value is used further in the calculation of the mean residence time, $t_m$, according to

$$t_m = \frac{\int tdC}{\int dC} \cong \frac{\sum t_i \Delta C_i}{\int \Delta C_i} = \frac{sum\ column\ 9}{sum\ column\ 7}$$

In this example, a value of 8.99 minutes is calculated. This compares well with the theoretical HDT of 8.35 minutes. Differences may arise due to experimental error in conducting the tracer test or numerical dispersion related to the quality and time-resolution of the tracer data.

As before, this summation is an example of the right rectangle rule for integration established in Appendix C of the SWTR Guidance Manual. Note, that due to the process of using the previous timestep, the first cell in this column is blank (or "0").

Column 10: Column 10 calculates the values for the residence time distribution (RTD) or exit age distribution *E*. These values are calculated by a process of numerical differentiation based on the relationship

$$E = \frac{dF}{dt}$$

In accord with the procedure demonstrated in Appendix O of the SWTR (page O.2-19), a forward differentiation is used according to

$$E_i = \frac{\Delta F_i}{\Delta t_i} \approx \frac{F_{i+1} - F_i}{t_{i+1} - t_i}$$

As noted, this forward differentiation maintains a slight conservatism with regard to determining the *E* function. Because the process is a forward process, the last value in the column is blank (or "0").

As an example, to calculate the *E* value for datapoint #5,

$$E_5 = \frac{F_6 - F_5}{t_6 - t_5} = \frac{0.044 - 0.004}{5 - 4} = 0.04$$

The *E* function is used further in Column 11.

Column 11: The values in Column 10 are calculated as the product of several terms according to the equation

$(t_i - t_m)^2 E_i \Delta t_i$

where $\Delta t_i = t_i - t_{i-1}$, and $t_m$ is taken as the value calculated at the bottom of the Exhibit. Note, one should use the calculated value of the mean residence, $t_m$, time rather than the theoretical HDT.

Once the values for each term in Column 11 are calculated, the sum of all values is made at the bottom of Column 11. $\int (t - t_m)^2 E dt \cong \sum (t_i - t_m)^2 E_i \Delta t_i$

The summed value at the bottom of Column 11 represents the variance of the RTD about the mean, typically denoted by the term $\sigma^2$. The significance of this value is discussed below.

Note, this summation is again an example of the right rectangle established in Appendix C of the SWTR Guidance Manual. Due to the process of using the previous timestep, the first cell in this column is blank (or "0").

In Exhibit E.2, the summations at the bottom of specific columns are the information needed to implement the Extended $T_{10}$ method for non-conventional contactors. The principal calculation needed is the equivalent number of tanks-in-series represented by the RTD. According to the tanks-in-series theory, the number of theoretical number of tanks, *N*, is determined according to

$$N = \frac{t_m^2}{\sigma^2}$$

As shown in the example above,

$$N = \frac{t_m^2}{\sigma^2} = \frac{(8.99)^2}{7.27} = 11.1$$

Therefore, the tracer test determined that the contactor has a hydrodynamic character similar to a series of 11.1 CSTRs in series. Implementation of the Extended $T_{10}$ method rounds

the value of *N* down to the nearest integer, and uses this value for the number of segments for subdividing the non-conventional contactor. In this example, the Extended $T_{10}$ calculation would use 11 segments.

It is interesting to note that the results of the step dose tracer test analyzed in Exhibit E.2 differed somewhat from the results of the pulse dose tracer test analyzed previously in Exhibit E.1. Some of this difference may be attributed to basic errors in dosing and analytical determinations in the tracer tests. In addition, the step dose tracer test involves a numerical differentiation (Column 10, step dose) that is typically less accurate than numerical integration (used several times in each analysis). This could be a source for numeric dispersion which leads to higher estimated hydrodynamic variance. This is not to imply that one procedure is better than another. There are advantages and disadvantages to both dose methods. Details of those issues can be found in Bellamy, Finch et al (1998) and the previous referenced sources.

Exhibit E.2 contains additional calculations that are not necessary for implementing the Extended $T_{10}$ method, but are needed for quality control of the tracer test, or for further general information concerning the contactor hydrodynamics.

The average effective concentration during the last 10 percent of the test duration (ca. 3 minutes) was 1.54 mg/L. That is also the maximum concentration calculated according to Column 7. That maximum was within was 90 percent of the dose of 1.7 mg/L added to the influent water. This value is within the acceptable range of 85 percent to 115 percent.

Exhibit E.2 also shows the value for the dimensionless variance denoted by the term, $\sigma_\theta^2$, which is the dimensionless form of the variance, $\sigma^2$. The variance represents the second moment abut the mean of the RTD. It is calculated according to

$$\sigma_\theta^2 = \frac{\sigma^2}{t_m^2}$$

which is the inverse of the equation for determining *N*, such that

$$\sigma_\theta^2 = \frac{1}{N}$$

Aside from an alternate form of calculating *N*, the value of $\sigma_\theta^2$ is useful for comparing the results of different tracer studies. For example, a higher value of $\sigma_\theta^2$ measured at a tracer study of a different flowrate suggests that there is greater mixing in the contactor under that flowrate. When more than one tracer test is performed, such as at various flowrates as recommended, the conservative recommendation is to use the RTD data set associated with the greatest variance, $\sigma_\theta^2$.

## E.3 References

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Fogler, H. S., *Elements of Chemical Reaction Engineering*. 4th ed.; Prentice Hall: Upper Saddle River, NJ, 2005.

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Teefy, S. 1996. *Tracer Studies in Water Treatment Facilities: A Protocol and Case Studies.* Final Report. American Water Works Association Research Foundation. American Water Works Association, Denver, CO.

Teefy, S. and P. Singer. 1990. Performance and Analysis of Tracer Tests to Determine Compliance of a Disinfection Scheme with the SWTR. *Journal AWWA*, 82(12):88-89.

Wen, C. Y.; Fan, L. T., *Models for Flow Systems and Chemical Reactors*. Decker: New York, 1975.

# Appendix F
# Watershed Control Best Management Practices (BMPs) and Case Studies

This appendix provides a list of programmatic resources and guidance available to assist systems in building partnerships and implementing watershed protection activities. Examples of partnerships and possible control measures for different sources are summarized in Chapter 2, Section 2.4.3.2; this appendix provides further detail to the control measures described in chapter 2.

## F.1 Regulatory and Other Management Strategies

For systems in watersheds where most of the land is privately owned, land use regulations may be the best way to control pollution, especially in heavily developed or growing areas. Examples of possible regulations include septic system requirements, zoning ordinances specifying minimum lot sizes or low-impact development, limits on discharge from wastewater treatment plants and other facilities, pet waste cleanup ordinances, and requirements for permits for certain land uses. Your ability to regulate land use will depend on the authority granted to your municipality by the state, the ownership of your system (public or private), and the support of your local government and the public. Regulatory authority, steps for designing a regulation that can withstand lawsuits, and types of land use regulations are described in the paragraphs below.

### F.1.1 Determining Authority to Regulate

Where a water system is privately owned, it may be necessary to ask the cooperation of the local government to get source water regulations passed. For a municipal water system whose watershed is located entirely within the municipality, issuing zoning or land use ordinances should be less of a hurdle. The ability of a municipality to pass a land use ordinance or other law to help reduce contamination may depend on the authority the state grants to the local government in the state constitution or through legislation, although states normally do not interfere with the actual land use and zoning rules (AWWARF 1991). States generally permit zoning for the purposes of protecting public health or general welfare. However, some states may prevent local governments from passing laws that are more stringent than state law or that conflict with state laws. State laws in other states may prevent municipal governments from passing certain local laws that are not expressly permitted elsewhere in state law.

If the watershed or the area of influence on water quality extends throughout several municipalities, it can be difficult to standardize watershed control practices throughout the watershed. The legal framework used will depend on who has jurisdiction over land use in the watershed and on the authority of the water system (AWWARF 1991). New York State law, for instance, authorizes municipalities to draft watershed regulations, which are then approved and adopted by the state. This gives the municipalities the authority to enforce the watershed rules within their watersheds even if the watershed is outside municipal boundaries. For instance, New

York City sets water quality standards, land use restrictions, and approves wastewater treatment plant designs in its watersheds in upstate New York. The City of Syracuse conducts watershed inspections on Skaneateles Lake, its source of supply, several miles outside of Syracuse. Both of these systems are filtration avoidance systems, so it is especially important that they have some control over areas outside their jurisdictions.

The Metropolitan District Commission, although not a PWS, was created by the state of Massachusetts and is authorized to promulgate and enforce watershed protection regulations in watersheds used by the Massachusetts Water Resources Authority to provide water to the Boston metropolitan area. Some watersheds which extend across state boundaries have governing bodies authorized by Congress. The formation of the Tahoe Regional Planning Agency was the result of a compact between the states of California and Nevada and was approved by Congress. The agency is authorized to pass ordinances, including source water protection rules that regulate land use in the area around Lake Tahoe.

County governments in some states may have some zoning authority and may be able to assist with enforcement of some regulations affecting source water (e.g., septic systems). In most cases where watersheds cross jurisdictions, however, public water systems (PWSs) will not have regulatory or enforcement authority. PWSs in this situation should work with other local governments, PWSs and agencies in their watersheds to sign memoranda of agreement or understanding, in which each entity agrees to meet certain standards or implement certain practices.

The City of New York signed a memorandum of agreement in 1997 with the state of New York, U.S. Environmental Protection Agency (EPA), and 79 municipalities within its watersheds. The agreement calls for the creation of local and regional watershed protection programs and, for New York City, funding for water quality and infrastructure improvement projects in upstate New York. Other cities, such as Salem and The Dalles, both in Oregon, have signed memoranda of understanding with the U.S. Forest Service, which owns most of the land in the cities' watersheds. These memoranda define the management responsibilities of each PWS and the Forest Service.

## F.1.2 Zoning

This section describes the steps you should follow to make a zoning law that is likely to withstand a legal challenge. Basically, it is important to make sure the appropriate procedures are followed and that the law has sufficient scientific basis (AWWA 1999). First, be sure you have the authority to regulate, especially if you are proposing something besides a simple zoning law. Make sure the rule is specific enough; if a map of an overlay district is not drawn to a small enough scale, it may be difficult to tell which properties are affected. Comply with all administrative procedure requirements, such as notifying the public of the proposed changes and holding a public hearing; failure to do so is the most common reason for rules being revoked.

Follow substantive due process, which means that the regulation should promote the municipality's public health goals. In practice, this means the ordinance should conform to the objectives of the watershed control program plan. The plan should contain enough data to illustrate how the ordinance will affect water quality.

Ordinances should also be designed to withstand a takings lawsuit (AWWA 1999). The Fifth Amendment to the U.S. Constitution states that private property may not be taken for public use without just compensation. Any physical invasion without consent is usually considered a taking, even if the landowner retains ownership of the land. Installation of a monitoring well or stream gauge without consent is an example of a taking.

In addition, ordinances that fail to advance a legitimate government interest or deny a landowner economically viable use of his land can be viewed as takings, even if the landowner retains full ownership (AWWA 1999). The first criterion means that there should be a need for the ordinance; for example, if a planned development's storm sewers and wastewater treatment plant will discharge into an area outside a municipality's wellhead protection area, the municipality cannot cite impacts on the drinking water as a factor in the decision to restrict development without compensating landowners. Under the second criterion, if property values decrease but still retain some value (e.g., due to a decrease in permitted building density), the ordinance does not result in a taking. A regulation that restricts all development would probably be considered a taking. In keeping with these two criteria, the effect of an ordinance should be proportional to the predicted impact of development. Thus, if a municipality determines that half-acre zoning is sufficient to protect a drinking water source, it may not zone for five acres.

To prevent takings claims, the municipality should show the need for the regulation and a connection between the ordinance and the expected result (AWWA 1999). This proof should be based on a scientific analysis beginning with an accurate delineation of the watershed or wellhead protection area/recharge area. A zoning district based on an arbitrary fixed radius around a well or lake would probably be considered insufficient in court unless it is characterized as an interim boundary. A court challenge could claim that such a district protects an area that does not contribute to the watershed *or* that land that is part of the watershed is not being protected (failing to advance the government's interests).

Following the delineation, determine the impact the regulation will have by mapping current and projected residential, commercial, and industrial development under current zoning requirements. Then map current and projected development for existing regulations and for the proposed ordinance, and determine the potential pollutant load under each scenario (AWWA 1999). You may not be able to determine *Cryptosporidium* loading if you have not monitored, but there may be data available on fecal coliform bacteria from different sources in your watershed (e.g., agriculture, septic system failure, pets and wildlife). If your PWS has not collected such data, other local agencies, such as sewer authorities, non-profit groups, universities, or planning commissions, as well as the U.S. Geological Survey, may have water quality data. Water quality models can help you determine pollutant loading. This buildout

analysis will help you show how your proposed ordinance advances a legitimate government interest and how the effect of the ordinance is proportional to the impact of land use in your watershed.

*Types of Ordinances*

Watershed ordinances usually apply within an overlay district, which may be the area of influence you determined for your watershed control plan. All existing zoning or land use regulations apply within that area, but additional requirements apply within the overlay district. Following are some land use ordinances you may wish to consider:

- Large-lot or low-density zoning. Unless lots are very large (such lots can use septic systems and wells), large-lot zoning may be inefficient, as it increases costs for sewer, water, and road development. This type of zoning also may go against affordable housing requirements. However, it may be useful in agricultural areas for preserving rural character and preventing subdivision of farms.

- Limits on certain types of land use except by special permit. Such ordinances should specify criteria for granting special permits and designate an authority that may grant permits. The authority should present findings that back up its decision to grant the permit. Special permits are granted for a particular lot, not for the owner of that lot.

- Impact fees. The regulating authority must be sure it has authority to impose such fees. Impact fees collected can be used to pay for mitigation of pollution caused by development, e.g., for preventing runoff or buying land elsewhere in the watershed. Fees should be proportional to the impact and the cost of mitigation, and the purpose of the fees should be specified in the regulation. A disadvantage to impact fees is that they may in some cases be considered taxes, and local government's authority to impose taxes may be limited. Fees are more likely to withstand challenge if they are framed as optional services provided to the developer (i.e., the developer can choose not to develop) and if the fees are set aside for the PWS or stormwater utility rather than put into general funds.

- Submission and approval of a watershed protection plan or impact study as a condition for development of a subdivision or apartment complex. This type of ordinance requires that watershed protection plan or stormwater control be implemented before a building certificate of compliance is issued. Plans should be required to designate the party responsible for maintaining stormwater facilities after construction is complete.

- Performance standards. A performance standard permits development but limits impact of the development. For example, the regulation could specify that permits require that the pollutant loading rate of the development is no more than a certain percentage of the pre-development loading rate of the area. This would require enforcement or monitoring to make sure the development continues to comply. In its permit application, the

developer would also be required to list mitigation steps it would take if it exceeded the pollutant loading requirements.

Most zoning ordinances have grandfather clauses that allow nonconforming land uses to continue. Ordinances may also allow the zoning authority to grant variances if the topography or size of a lot make it difficult to comply with a zoning requirement.

Examples of source water protection ordinances can be found at: http://www.epa.gov/owow/nps/ordinance/osm7.htm.

### F.1.3 Land Acquisition and Conservation Easements

Acquisition of watershed land by the utility or its affiliated jurisdiction is often the most effective approach to protecting the water source. Landowners usually consider acquisition as fair, since it compensates them for their property while protecting the watercourses nearby. Land conservation has also been found to provide multiple benefits aside from controlling pathogen contamination, such as flood control, limited recreational use, and the protection of historic and environmental resources. EPA's Drinking Water State Revolving Fund (SRF) allows a percentage of the fund to be set aside for land acquisition associated with watershed protection. Note that some states may not allow SRF funds to be used in this manner.

Several organizations exist that can help systems purchase watershed land to protect drinking water quality, especially when government agencies are unable to move quickly enough to buy land when it becomes available. The Trust for Public Land (http://www.tpl.org) and small local land trusts and conservancies can facilitate the land acquisition process. Trusts can buy and hold land from multiple landowners on behalf of a water system until the system can assemble funding to purchase it from the trust. Trusts may also maintain land ownership themselves. The Trust for Public Land also can assist with development of financing strategies for land purchases.

Trusts also can work with landowners to buy or have landowners donate conservation easements. An easement is a legal document that permanently limits the development of a piece of land, even after the land is sold or otherwise changes ownership. The landowner selling or donating the easement specifies the development restrictions to apply to the land. The law varies from state to state, but the owner of the easement (the government agency or land trust) has the authority to determine if the requirements of the easement are being followed. If not, the owner of the easement make take legal action. Easements donated to government agencies or to land trusts may be eligible for tax deductions. See http://www.landtrust.org/ProtectingLand/EasementInfo.htm for frequently asked questions about easements and for an example of a model easement for use in the state of Michigan. The Land Trust Alliance (http://www.lta.org), a trade organization for land trusts, has published handbooks on designing and managing conservation easement programs.

Other government agencies, such as the U.S. Forest Service or state natural resource departments, may be able to buy parcels in your watershed if you are unable to afford to purchase all the land that needs to be protected.

## F.2 Addressing Point Sources

### F.2.1 Concentrated Animal Feeding Operations

Some animal feeding operations (AFOs) may be considered concentrated animal feeding operations (CAFOs) if they have more than a specified number of animals and/or if they discharge pollutants into navigable waters through a manmade ditch or other device or if they discharge directly into waters of the United States. Possible sources of pollutants at CAFOs include runoff that flows through feedlots; failure of pumps, pipes, or retaining walls of manure storage lagoons; runoff from areas where manure is applied to the soil; and direct contact of animals with surface water. CAFOs are located primarily in the South and Midwest, but the number of such facilities is increasing as farms consolidate their operations.

EPA recently issued a rule that changed the requirements on CAFOs that must apply for National (or state) Pollutant Discharge Elimination System (NPDES or SPDES) permits (U.S. EPA 2008). CAFOs that discharge or propose to discharge must apply for NPDES permits and submit Nutrient Management Plans (NMPs) at the time that they submit a permit application. Permitting authorities are required to review the NMPs and provide the public with an opportunity for meaningful public review and comment. The permitting authorities are required to incorporate the terms of NMPs as NPDES permit conditions. If an unpermitted CAFO can certify to the permitting authority that they do not discharge or propose to discharge, then they are not required to apply to an NPDES permit. In the final rule, EPA provided clarification on how operators should evaluate whether they discharge or propose to discharge. The CAFO owner or operator must determine on a case-by-case basis whether the CAFO will discharge based on the CAFO design, construction, operation, and maintenance.

Many CAFOs do not currently have permits due to limited state resources for compliance (medium and small AFOs may be designated as CAFOs only by state or regional staff after onsite inspection). For CAFOs (and other NPDES permittees) that do have individual permits, you may want to attend the public hearing required as part of the permit renewal process, especially if you have any concerns about the adequacy of the existing permit requirements to prevent *Cryptosporidium* or other drinking water contamination.

### F.2.2 Wastewater Treatment Plants

All wastewater treatment plants in the United States are required to provide secondary treatment (primary treatment consists of sedimentation, while in secondary treatment, aeration

provides oxygen to bacteria that take in nutrients and digest organic material) (U.S. EPA 2001b). Most plants are also required to disinfect their effluent before discharging. However, conventional chlorine disinfection may be ineffective against *Cryptosporidium*.

Some wastewater treatment facilities are beginning to implement treatment similar to that used for drinking water treatment. The Robbins Plant of the Upper Occoquan Sewerage Authority in Centreville, Virginia, discharges into a stream that feeds into a reservoir in northern Virginia. Following secondary treatment using activated sludge, the facility provides other treatment, including clarification, multimedia filtration, and disinfection (U.S. EPA 2000a). The Cole Pollution Control Plant in Fairfax County, Virginia, which discharges into a creek flowing into the Potomac River, also uses advanced treatment, including chlorine disinfection, filtration, and dechlorination (Fairfax County 2001).

PWSs should identify all wastewater treatment plants in their watersheds and determine what their permit effluent limits are and whether the limits are being met. Some of this information may already be available through the source water assessment program. PWSs may wish to work with the wastewater utilities and appropriate government agencies to get them to voluntarily upgrade the treatment provided. PWSs with the appropriate legal authority may wish to require wastewater plants to use certain technologies. An example might be switching from chlorine to ozone or ultraviolet radiation disinfection before discharging.

### F.2.3 Combined Sewer Overflows

Combined sewer overflows (CSOs) are most common in older cities in the northeastern and midwestern United States and can be a significant contributor of *Cryptosporidium* to urban watersheds.

There are three major structural solutions to the problem of CSOs. The first is to separate combined sewers into sanitary and storm sewers, where sanitary sewers flow to the wastewater treatment plant and storm sewers release to surface water. This separation may cause the unwanted side effect of increasing overall contamination due to the fact that storm water is no longer being treated. For example, separating sewers resulted in only an estimated 45-percent reduction in fecal coliform removal in a bay in Boston (Metcalf and Eddy 1994, cited in U.S. EPA 1999c). Separating sewers is also very expensive and often impractical. The second option is to increase the capacity of the wastewater treatment plant so that it is able to treat combined sewage from most storms. The third, very expensive solution is to build aboveground open or covered retention basins or to construct underground storage facilities for combined sewage to hold the sewage until the storm has passed and can be treated without overloading the plant. The Metropolitan Water Reclamation District in Cook County, Illinois, chose the third option, building 109 miles of tunnel up to 35 feet wide and several underground reservoirs underneath Chicago and its suburbs, with most funding from EPA (MWRD 1999). In addition to reducing CSOs, the tunnels eliminated flooding that had previously affected the area due to its flat

topography. The project also eliminated the need for individual municipalities to implement their own CSO programs.

CSOs are not regulated directly under their own program, but EPA has a CSO control policy (U.S. EPA 1994) which encourages minor improvements to optimize CSO operation, and CSO management may be written into NPDES or SPDES permits. The CSO policy also encourages development of long term control plans for each CSO system; such plans would require significant construction, and few utilities have drafted or implemented them yet. Planned construction projects can be included as control measures in watershed control plans. PWSs should determine the extent of the CSO programs in place in municipalities within their watersheds. They may be able to work with other utilities to address overflow sites of particular concern. Many municipalities with CSOs made major structural changes to their systems in the 1980s and 1990s; current improvements are more likely to involve streamlining operation and management.

Many large cities have already addressed a significant portion of their CSOs, but there are additional smaller steps they can take to reduce the amount of sewage released during a wet weather event. These include maximizing in-line storage (storage available in the sewer pipes themselves) through regular inspection and removal of obstructions and sediment, installation and maintenance of flow regulators, upgrading pumping capacity (assuming the treatment plant can handle the increased volume); raising weirs at CSO outfalls; and installing computerized sensors to control flow during storms. Stormwater Best Management Practices (BMPs) can also reduce the impact of CSOs.

Additionally, reducing inflow (entry of storm water into the combined sewers) and infiltration (entry of storm water through cracks and manholes) is important. Inflow can be reduced by disconnecting roof drains and sump pumps from sewers, restricting flow into storm drains, and constructing storm water detention ponds and infiltration devices. If overflow events can be reduced, it may be possible to eliminate some outfalls. Some sewer systems also have installed some treatment of CSOs including disinfection and screening; this treatment may be required as part of a NPDES permit.

### F.2.4 Sanitary Sewer Overflows

Sanitary sewer systems normally feed into wastewater treatment plants but can still cause water quality problems. Sanitary sewer overflows (SSOs) occur when untreated and mostly undiluted sewage backs up into basements, streets, and surface water. SSOs discharging to surface water are prohibited under the Clean Water Act. Insufficient maintenance and capacity and illegal connections are some of the primary causes of SSOs. Many sanitary sewers are subject to inflow and infiltration, just as combined sewers are, caused by cracks in pipes or bad connections to service lines. They may receive water they were not designed to receive, such as storm water from roof drains that should be connected to storm sewers, or wastewater from new

developments that did not exist when the wastewater treatment plant was designed. SSOs can be reduced by cleaning and maintaining the sewer system; reducing inflow and infiltration by repairing leaking or broken service lines; increasing sewer, pumping, and/or wastewater treatment plant capacity; and constructing storage for excess wastewater (U.S. EPA 2001c).

EPA and the states will continue to address these problems using various aspects of the capacity, management, operation and maintenance (CMOM) concept. The CMOM concept encourages the use of self-assessments and pro-active correction of system deficiencies to avoid further deterioration of the sanitary sewer infrastructure and resultant SSOs. In some cases, EPA and the states will use a combination of administrative and civil judicial enforcement action to achieve these goals (U.S. EPA 2009).

### F.2.5 Municipal Separate Storm Sewer Systems

Municipal separate storm sewer systems (MS4s) in areas with populations of more than 100,000 are also required to obtain NPDES permits. Information on storm sewer outfall locations, volume discharged, conventional pollutant loads, and existence of illicit discharges is submitted as part of the permit application process (U.S. EPA 1996). In addition, these MS4s must develop management plans addressing items such as outfall monitoring, structural and nonstructural BMPs to be implemented, and identification and elimination of illicit discharges. Illicit discharges to MS4s include any non-stormwater discharges, such as discharges that should be connected to sanitary sewers (e.g., water from sinks, floor drains, and occasionally toilets), illegal dumping of sewage from recreational vehicles, sanitary sewer overflow backing up through manhole covers into storm drains, effluent from failing septic systems, water from sump pumps, etc.

Small MS4s (serving areas with populations of less than 100,000) are subject to NPDES permit requirements if they are located in urbanized areas as determined by the Bureau of the Census. Some small MS4s in urbanized areas may be eligible for waivers from the NPDES requirement. Those MS4s subject to NPDES permits must implement control measures in six areas, including a plan for eliminating illicit discharges (U.S. EPA 2000b).

PWSs should work with all MS4 utilities in the area of influence to gather existing information about storm water contamination. MS4 utilities may need to install or retrofit structural BMPs, such as retention ponds, to reduce contamination. Most studies of structural stormwater BMPs focus on nutrient or sediment removal, so almost no information is available on *Cryptosporidium* removal, and limited information is available on bacterial removal. However, a few studies of bacteria in structural BMPs show that bacteria survive for weeks to months in retention pond sediments and natural lake environments. In addition, other studies showed higher bacteria levels in retention pond effluent than in influent. This suggests that stormwater pond sediments resuspended during storms can be a source of pathogens (Schueler 1999).

## F.3 What BMPs Can Help Alleviate Nonpoint Sources?

The following sections describe BMPs for agricultural, forestry, and urban sources of *Cryptosporidium*. Your watershed control program plan must discuss how these or any other BMPs you choose will be implemented in the area of influence. EPA Section 319 grants and Clean Water State Revolving Fund (CWSRF) loans can be used for nonpoint sources and watershed management purposes.

### F.3.1 Agricultural BMPs

#### F.3.1.1 Management Programs

The U.S. Department of Agriculture (USDA) (2000) recommends a multiple-barrier approach to controlling pathogen transport and proliferation on farms and in agricultural watersheds. It recommends the following control points:

- Preventing initial infection by controlling pathogen import to the farm.
- Controlling the reproduction and spread of the pathogen throughout the farm.
- Managing waste.
- Controlling pathogen export from the farm.

These control points should not be treated separately. For example, waste management affects reproduction and spread of the pathogen if feed becomes contaminated with waste. Waste management is also related to pathogen export; composting can kill *Cryptosporidium* oocysts before they leave the farm.

BMPs that can reduce pathogen loading include composting, waste management (manure storage and land application), grazing management, feedlot runoff diversion, and buffer or filter strips. PWSs should work with their local soil and water conservation districts and agricultural or cooperative extensions, which can help farmers design and implement pollution management plans and BMPs. Details about these conservation practices are provided in the USDA Natural Resources Conservation Service's (NRCS) National Handbook of Conservation Practices (NRCS 1999) at http://www.ftw.nrcs.usda.gov/nhcp_2.html.

Management strategies designed to minimize direct livestock contamination of surface water with *Cryptosporidium* should focus primarily on young animals (those less than 3 months old) and their waste, since calves are more likely to shed *Cryptosporidium*. Efforts should also focus on cow herds as a whole when calves are present.

Several NRCS programs provide technical assistance to farmers and subsidize the cost of implementing BMPs. These include Agricultural Management Assistance, the Environmental Quality Incentives Program, the Conservation Reserve Program, and the Conservation Reserve Enhancement Program (see www.nrcs.usda.gov/programs). The last two programs also pay farmers rent on erodible cropland taken out of production. More information is available at http://www.fsa.usda.gov/dafp/cepd/crpinfo.htm. The 2002 Farm Bill increased funding for these programs and created new ones as well. For example, the new Conservation Security Program will recognize and reward farmers who are leaders in environmental management.

### F.3.1.2 Composting

Composting can effectively reduce pathogen concentrations. Temperatures greater than 55 degrees Celsius (131E F) can be easily attained and maintained long enough to inactivate most oocysts (Blewett 1989). To reliably achieve *Cryptosporidium* inactivation, however, the entire waste mass should be uniformly treated and there should be no cold spots. Intense management may be needed to completely mix the composted material.

A study was conducted to determine the effectiveness of compost piles in inactivating *Cryptosporidium* oocysts. Four compost piles were used. Two compost piles consisted of manure while the other two compost piles consisted of surface soil. Each compost pile was injected with two million oocysts in an aqueous suspension. Every two to four weeks, *Cryptosporidium* oocysts were extracted and tested from both sets compost piles. Both experiments show that inactivation of *Cryptosporidium* oocysts occurred after 40 days of composting. However, the compost pile with manure fared slightly better after 150 days (Jenkins et al. 1999).

### F.3.1.3 Buffer Strips

Buffer strips, or filter strips, provide a buffer between the area of manure application or grazing and adjacent streams or lakes. Filter strips have been studied primarily with regard to their effectiveness at sediment and nutrient removal. Nutrient removal has been shown to be extremely variable, while agricultural grass filter strips consistently remove 65 percent or more of sediment (Ohio State University Extension undated). How sediment removal relates to *Cryptosporidium* removal is not known. *Cryptosporidium* often adsorbs to suspended material the size of clay and silt particles, which is the type of sediment that is likely to pass through the filter strip, especially at high flow velocities.

Few studies have evaluated the ability of buffer strips to remove *Cryptosporidium*. However, one study found that grass filter strips with slopes of 20 percent or less and widths of at least 3 meters resulted in removal of 1 to 3 log (90 to 99.9 percent) during mild to moderate precipitation (Atwill et al. 2002). More data are available on removal of bacteria. Moore et al. (1988) reviewed the work of several investigators and concluded that vegetative filters are most

reliable at removing bacteria at high concentrations from waste effluent. Bacterial populations in runoff from buffer areas seem to equilibrate at approximately 104 to 105 organisms per 100 milliliters, regardless of experimental conditions. For this reason, USDA (2000) recommends that buffers and filter strips be considered secondary practices for pathogen control and be used in conjunction with other source, proliferation, and waste treatment and control measures to form an integrated, comprehensive pathogen management system.

The NRCS encourages the use of riparian forest buffers of at least 35 to 100 feet (depending on floodplain width) for stream restoration purposes but recommends additional width in high sediment and animal waste application areas. Grass filter strips may be added upgradient of the forest buffers or may be used alone. The NRCS (1999) recommends grass filter strip widths of at least 20 feet, but width should be determined based on the slopes of the strip and the field being drained, the area being drained, the erosion rate, sediment grain size distribution, runoff volume, and the vegetation in the strip. Filter strips should follow contours as much as possible to promote sheet flow. The area being drained should have a slope of less than 10 percent. Grazing should not generally be permitted within the filter strip. Maintenance activities should include mowing to prevent woody growth, inspection after storm events, repair of any gullies, reseeding of disturbed areas, and any other steps needed to maintain overland sheet flow.

Vegetated buffer strips were tested to see if they were effective at removing *Cryptosporidium* during rainfall rates of 15 or 40 mm/h for four hours. Buffers were set on a slope of 5 to 20 percent and soil textures consisted of silty clay, loam, or sandy loam.

It was found that vegetated buffer strips consisting of sandy loam or higher soil bulk densities had a 1 to 2 log reduction/m. Buffers consisting of silty clay, loam, or lower bulk densities had a 2 to 3 log reduction/m. Also, it was found that vegetated buffer strip made of similar soils removed at least 99.9 percent of *Cryptosporidium* oocysts from agricultural runoff when slopes were less than or equal to 20 percent and had a length of at least 3 meters (Atwill et al. 2002).

### F.3.1.4 Grazing Management

Managed grazing can be cheaper and less environmentally damaging than confined feeding and unmanaged grazing. It decreases feed, herbicide, equipment, and fertilizer costs, while reducing erosion and increasing runoff infiltration and manure decomposition rates (Ohio State University Extension, undated). In managed, or rotational, grazing, a sustainable number of cattle or other livestock graze for a limited time (usually 2-3 days) on each pasture before being rotated to the next pasture. This allows vegetation regrowth and prevents overgrazing, which can contribute to erosion and runoff, and helps distribute manure evenly over the grazed area. It also prevents soil compaction, thereby increasing infiltration. One of the best ways to prevent surface water contamination during rotational grazing is to prevent grazing along streams (through

fencing and use of a buffer strip) and to provide alternative water sources for livestock. Providing water in each paddock can increase the number of cattle the pasture is able to support. Even where rotational grazing is not used, surface water contamination can be reduced by keeping cattle, especially calves, out of streams.

Vermont wanted to reduce the concentration of *E. coli*, fecal streptococcus, and fecal coliform bacteria as well as total phosphorus found in the Lake Champlain Basin Watershed. It was assumed that a significant concentration of bacteria originated from nonpoint sources located in agricultural lands. To reduce the bacterial concentration entering the watershed, Vermont decided to improve animal waste management with dairy cows. In nearby areas, pastures containing dairy cows near streams and streambanks were found. It was believed that these pastures were one of the major sources of contamination due to bacterial excretion and streambank erosion caused by dairy cows. Minimizing erosion along streambanks allows for healthy vegetation, which will help filter nutrients. To prevent dairy cows from getting near or in streams and streambanks, bridges were constructed across streams. Fences along streambanks were also constructed to keep dairy cows from eroding streambanks.

When construction was complete, the watershed was monitored for three years. Exhibit F-1 represents the average reduction in concentration for the specific contaminant.

**Exhibit F-1 Average Reduction of Specific Contaminant**

| | |
|---|---|
| *E. coli* | 29% |
| Fecal streptococcus | 40% |
| Fecal coliform bacteria | 38% |
| Total phosphorus | 15% |

These numbers show that the construction of bridges and fences had a significant impact on the reduction of bacterial and total phosphorous concentrations. Only minimal fence maintenance was required. Therefore, keeping dairy cows away from streams and streambanks may significantly reduce bacterial and phosphorus concentration in the watershed with minimal hassles (U.S. EPA 2002c).

### F.3.1.5 Manure Storage

Manure storage facilities allow farmers to wait until field conditions are more suitable for land application. Without manure storage facilities, farmers must distribute manure on adjacent fields daily. However, weather conditions are not always appropriate for manure application. During the winter, for example, frozen soil conditions allow *Cryptosporidium* oocysts to be washed into watercourses, and oocysts survive longer at cold temperatures.

Manure storage facilities should be designed to prevent discharge through leaching or runoff. They should be lined and, if possible, covered. Facilities that are not covered should be designed to contain precipitation and runoff from a 25-year 24-hour storm. Storage areas should have embankments to prevent overflow and collapse of the storage facility and to divert runoff from outside the facility from contamination. Facilities should be sited outside of flood plains. Manure should be stored for a time period sufficient for microorganisms to die off.

### F.3.1.6 Land Application of Manure

Several precautions taken in manure application can prevent runoff from entering surface water, reducing the likelihood of *Cryptosporidium* contamination. Buffer strips should be situated between the water body and area of manure application. Manure should not be applied to frozen ground or before predicted rainfall. Manure should not be applied near tile drains or dry wells or to land subject to flooding. If soil is dry and cracked, fields should be tilled before application. Soil and manure should be tested for nutrient levels, and the application rate should be tailored to the soil and specific crop needs. To minimize runoff, waste should be injected (injection creates holes 6-14 inches deep and does not turn soil over) or applied to the surface and then plowed under. Applying manure to land with crop residue or new crops rather than bare soil also minimizes erosion. Surface application without plowing under may be acceptable if conditions are warm and dry this enables significant pathogen die-off (Vendrall et al. 1997) by exposure to ultraviolet light (UV) light and desiccation. The Agricultural Waste Management Field Handbook (NRCS 1992), Chapter 5, Table 5-3 contains a detailed review of restricting features that should be considered during manure spreading.

For pastures to be used for grazing, waste should be stored for at least 60 days and then applied at least 30 days before the scheduled grazing period, to avoid infection of the animals. Use of these areas for grazing should be limited to mature animals. Manure spreading on pastures used for grazing or on hayfields should take place when minimal amounts of vegetation are present, just after harvesting or grazing. This allows sunlight and desiccation to destroy the most pathogens and reduces the chance of pathogen adherence to the forage.

Critical source areas are defined as saturated areas that can expand and contract rapidly, based on soil, hydrological, and slope characteristics (Gburek and Poinke 1995). These areas are dominated by saturated overland flow and rapidly respond to subsurface flow. Therefore, watershed managers should identify the boundaries of potential saturated areas and ensure that waste is only applied outside of those boundaries to minimize *Cryptosporidium* oocyst runoff. Some tools have been developed to delineate critical source areas (e.g., Cornell Soil Moisture Routing Model; Frankenberger 1999). Less detailed delineations can also be made using information such as soil drainage class, flooding frequency, wetland mapping, areas of concentrated flow, and aerial photo interpretations.

#### F.3.1.7 Feedlot Runoff Diversion

Clean roof and surface water can be diverted away from feedlots to a drainage system that is independent of a farm's waste management system (Ohio State University Extension 1992). All roofs that could contribute to feedlot runoff should have gutters, downspouts, and outlets that discharge away from the feedlot. Berms around the feedlot can divert surface runoff. Diverting clean water before it drains into the feedlot can significantly reduce the amount of wastewater that needs to be managed. Runoff within the feedlot should be contained and treated in the waste management system for the lot.

### F.3.2 Forestry BMPs

Forestry practices are not likely to significantly contribute to *Cryptosporidium* sources, since wildlife levels decrease or, at most, remain constant after logging. However, logging can cause increased erosion, leading to increased runoff and making it more likely that *Cryptosporidium* present in wildlife will reach the source water. In addition, logging can cause elevated sediment levels, resulting in high turbidity, which affects water treatment efficiency.

Filter strips, where ground cover is maintained around lakes, permanent and intermittent streams, and wetlands, help trap sediment. Filter strip width should increase with slope of the area being logged. Streamside or riparian management zones are intended to stabilize stream banks and maintain shade over streams to minimize water temperature fluctuations. Streamside management zones and filter strips often overlap, but limited logging is often permitted within streamside management zones (NRCS 1999).

Logging roads should be constructed to minimize runoff through proper grading and drainage. Road runoff should be diverted away from streams and prevented from channelizing. Loggers should minimize soil disturbance and compaction on skid trails, the trails used to drag logs to trucks for loading (U.S. EPA 2002a).

### F.3.3 Urban/Suburban BMPs

Urban/Suburban BMPs can reduce burden on sewage infrastructure and address CSOs and non-point sources. See http://www.epa.gov/owm/mtb/mtbfact.htm for fact sheets on technologies and BMPs municipalities can use to reduce contamination from wastewater and stormwater.

### F.3.3.1 Buffer Zones

For watersheds in urban areas, buffer zones help to protect development on the floodplain from being damaged when the water is high, as well as protect the stream from the effects of the development.

The utility, municipality, or cooperating jurisdictions may acquire land bordering the reservoir and/or its tributaries. Alternatively, buffers can be required by zoning ordinances, conservation easements, or subdivision regulations. Buffer zones can be fixed width or variable width. In a fixed-width zone, the buffer zone encompasses a certain distance from the stream bank or some other hydrological reference point (e.g., the high water mark of a stream). The widths of fixed buffer zones vary considerably among water sources, frequently ranging from 50 feet to 250 feet of buffer from the stream edge. Another form of buffer zone, the variable-width buffer, can vary in width depending on the hydrological sensitivity, stream size, and character of the land adjacent to the watercourse.

Considerations for developing local buffer requirements are the size and location of the stream, the nature of existing or potential development, and the financial and political feasibility of establishing protected zones around the streams and reservoir of the watershed. Although buffer zones have been found to trap fecal waste (Coyne and Blevins 1995, Young et al. 1980), the extent to which they reduce *Cryptosporidium* loading is not well understood. For this reason, buffer zones should be used to augment, rather than replace, other watershed management practices to help protect overall source water quality.

Buffer zones should be routinely inspected to ensure that sources of contamination have not been introduced to the area and that the buffer is being maintained (e.g., that buffers are kept unmowed). Watershed managers should also be aware of storm sewers and culverts that may be draining into the waterways and bypassing the buffer zones altogether.

### F.3.3.2 Dry Detention Basins

Dry detention basins temporarily store stormwater runoff and release the water slowly to allow for settling of particulates and the reduction of peak flows. These structures hold a certain amount of water from a storm and release the water through a controlled outlet over a specified time period based on design criteria. Most basins dry out completely between storm events. The major failure of these basins is that some are not designed or maintained properly, resulting in too slow a release of water to empty the basin before the next storm. If the basin remains partially full, only a portion of the design runoff volume from the next storm will be retained. With inadequate detention, pollutants are not removed from the runoff. Dry detention basins also risk the possibility of resuspension of pathogens from the basin sediments if hydraulic retention times are compromised by poor design or failure to keep the outlets open.

### F.3.3.3 Infiltration Devices

Infiltration devices remove pathogens and particles by adsorption onto soil particles and filtration as the water moves through the soil to the ground water. Infiltration devices include infiltration basins, infiltration trenches, and dry wells (NALMS 2000). Properly designed devices can reproduce hydrological conditions that existed before urban development, and provide ground water recharge and control of peak storm water flows. In order for them to function effectively, infiltration devices must be used only where the soil is porous and can readily absorb storm water at an adequate rate. An advantage of infiltration devices over many other urban BMPs is that they provide significant ground water recharge in areas with a high percentage of impervious surface.

### F.3.3.4 Sand Filters

Sand filters can be used to treat storm water runoff from large buildings and parking lots. As the name implies, storm water is filtered through beds of sand, which may be located above ground in self-contained beds, or can be installed underground in trenches or concrete boxes. Underground sand filters can be installed in urban settings where space is restricted and the filters are not visible. Pathogens and particles are removed by filtering storm water through approximately 18 inches of sand. Above-ground filters may be preceded by grassed filter strips or swales to pre-treat the incoming storm water and prevent the sand filters from clogging.

Sand filters are often more expensive to construct than infiltration trenches (NALMS 2000). They do not provide a significant amount of storm water detention, and their ability to remove pathogens is limited. They require little maintenance; the sand surface should be raked and a few inches of dirty sand on the filter surface should be removed and replaced periodically, so that the filters do not clog.

### F.3.3.5 Wet Retention Ponds

Wet retention ponds maintain a permanent pool of water that is augmented by storm water runoff. The ponds fill with storm water, which they slowly release over several days until the pond returns to its normal depth. Ponds can effectively reduce suspended particles and, presumably, some pathogens, by settling and biological decomposition. There is concern, however, that ponds attract wildlife that may contribute additional fecal pollution to the water, rather than reducing contamination. Bacteria may also survive in pond sediment.

Many people find wet ponds aesthetically pleasing, and welcome their use for storm water control. Some maintenance of the ponds is required in order for them to continue to function effectively and to avoid nuisance odors and insect problems. Wetland plants should be

periodically harvested, and the pond inlets and outlets should be kept clear so that flow is not impeded. Wet ponds can be an appealing play area for children, so safety measures should also be taken to restrict access.

### F.3.3.6 Constructed Wetlands

Constructed subsurface flow wetlands (where wetland plants are not submerged) can reduce *Cryptosporidium* and bacteria concentrations in wastewater (Thurston et al. 2001). Subsurface flow prevents the public from coming into contact with wastewater and prevents mosquitos problems. Wetlands may also be useful for treating storm water or other polluted water. However, the matrix material of a constructed subsurface flow wetland (gravel is often used) may provide an environment for bacterial growth, and animals living in the wetlands may contribute microbes to the effluent (Thurston et al. 2001). Animals are probably less significant than they would be in a free water surface wetland. The growth of bacteria in the wetland medium is both positive and negative; bacteria that help break down materials in wastewater are more plentiful, but fecal coliform also can survive in such environments. Constructed wetlands are relatively inexpensive and are often used on small scales to treat water at small facilities such as schools, apartment complexes, and parks (U.S. EPA 2000c).

A wetland was constructed in Tucson, Arizona to help remove *Cryptosporidium* from the secondary sewage effluent. The wetland had a maximum depth of 1.4 meter, length of 61 meters, and width of 8.2 meters. The wetland was designed to have a retention time of approximately four days with an average flow rate of 58 liters/minute. The wetland was planted with cattail, bulrush, black willow, and cottonwood. It was found that the wetland effectively removed 64.2 percent of *Cryptosporidium* oocysts from the secondary sewage effluent (Thurston et al. 2001).

Two wetlands were constructed to determine if they could effectively remove *Cryptosporidium* from untreated domestic wastewater. One wetland was planted with bulrush and the other wetland was made of bare sand. The influent domestic water flowed directly into two setting tanks in series. Then the flow split into the two wetlands in parallel. The wetlands' detention time was 1 to 2 days. The results of this study showed that both planted and unplanted wetlands removed about 90 percent *Cryptosporidium* oocysts. Slightly more oocysts were removed in the planted wetland. The test shows that planted and unplanted wetlands are effective in removing *Cryptosporidium* oocysts (Quinez-Daz et al. 2001).

### F.3.3.7 Runoff Diversion

As with feedlot runoff diversion, structures can be installed in more urban settings to divert clean water flow before it reaches a contamination source. Structures that channel runoff away from contamination sources include stormwater conveyances such as swales, gutters, channels, drains, and sewers. Graded surfaces can also be used to re-direct sheet flow, and

diversion dikes or berms can be installed to route sheet flow around areas that are being protected from runoff.

### F.3.3.8 Pet Waste Management

Municipalities can implement pet waste management programs to encourage pet owners to properly collect and dispose of their animal's waste. Many communities have pet waste ordinances that require pet owners to clean up after their pets on public property or anywhere outside their own yards; however, compliance is limited, and enforcement is usually not a priority. In addition, most ordinances do not require pet owners to clean up pet waste in their own yards (this problem can usually be addressed, though only reactively, through nuisance or pet neglect laws). Some communities have ordinances that govern the cleanup process by requiring disposal of pet waste with regular trash, burial, or flushing it down the toilet. Enforcement of these ordinances with fines for noncompliance is probably the best way to increase compliance.

To increase public awareness about pet waste, you can distribute educational materials through emails, letters, public service announcements, and signs. Posting is the most common outreach strategy for managing pet waste. Pet waste stations containing waste receptacles for public use are another popular solution. Public works departments have also formed voluntary commitment and partnership programs with pet owners and local pet stores in the community to promote good pet waste management.

### F.3.3.9 Water Conservation

Water conservation is usually presented as a practice that can help preserve the amount of water available for use, especially in times of drought. However, water conservation can also decrease the amount of wastewater and stormwater generated, thereby protecting the *quality* of the water supply (U.S. EPA 2002b). Use of low-flow toilets and showerheads, for example, can allow wastewater treatment plants to treat wastewater from more customers without having to increase capacity, reducing the occurrence of combined or sanitary sewer overflows. The reduced load on wastewater treatment plants can also decrease the need for rate increases. Reducing lawn watering decreases the amount of runoff entering storm sewers, combined sewers, and surface water.

### F.3.3.10 Low Impact Development

Low impact development, or better site design, is a watershed practice that reduces pollutant loads, conserves natural areas, saves money, and increases property values (Center for Watershed Protection 1999). A fundamentally different approach to residential and commercial development, site design tries to reduce the amount of impervious cover, increase natural lands

set aside for conservation, and use pervious areas for more effective stormwater treatment. Low impact development involves changing traditional practices for residential street and parking lot design, lot development, and conservation of natural areas. Some specific steps for better site design include the following (Center for Watershed Protection 1999):

- Design residential streets based on the minimum width needed to support travel lanes, on-street parking, and emergency and maintenance vehicle access. For example, a street with single family houses with driveways does not need two lanes for parking. Construct sidewalks on only one side of the street.

- Minimize the number of cul-de-sacs. Where cul-de-sacs are built, place landscaped islands to reduce their impervious cover.

- Advocate open space or cluster design subdivisions on smaller lots.

- Reduce imperviousness by promoting alternative driveway surfaces and shared driveways that connect two or more homes together. Reduce driveway length by allowing decreased front setbacks.

- Direct rooftop runoff to pervious areas such as yards, open channels, or vegetated areas rather than the roadway and stormwater sewers. Better yet, install open vegetated channels instead of storm sewers.

- Reduce the imperviousness and size of parking lots by minimizing stall dimensions, incorporating efficient parking lanes, and using pervious materials in the spillover parking areas where possible. Use lower parking ratios where possible (e.g., where mass transit is available and codes permit).

- Provide stormwater treatment for parking lot runoff using bioretention areas, filter strips, and/or other practices.

- Create a naturally vegetated buffer system along all perennial streams that encompasses critical environmental features such as the 100-year floodplain, steep slopes, and wetlands.

- Clearing and grading of forests and native vegetation at a site should be limited to the minimum amount needed to build lots, allow access, and provide fire protection. Specify a party legally responsible for maintaining the vegetated area.

Some aspects of low impact development may be prohibited outright under traditional zoning and development regulations, so low impact development practices may need to be codified. Where such practices remain voluntary or require exemptions from existing regulations, water systems should work with local planners to encourage the switch to better site design.

### F.3.3.11 Septic Systems

Failing septic systems can be a major source of microbial contamination in a watershed. Poor placement of leachfields can feed partially treated waste directly into a drinking water source. Poorly constructed percolation systems may allow wastewater to escape before it has been properly treated. Failing systems can result in clogging and overflow of waste onto land or into surface water.

Most septic system regulations require construction permits and an inspection before the system begins operating, but few require any follow-up. Where failing systems are a serious problem or are close to a drinking water source, however, some municipalities have maintenance or inspection requirements. For example, the Portland (Maine) Water District requires permits for all septic systems within 200 feet of Sebago Lake, its primary source (U.S. EPA 1999a). These septic systems are subject to regular inspection and may face stricter design requirements than systems outside the boundary. Portland also has the authority to inspect systems within 1,000 feet of Sebago Lake tributaries. Similarly, the Onondaga County Water Authority in New York visually inspects every septic system in the water system annually. Every three years each septic system is subject to a dye tracer test. Enforcement cases are referred to the county health department (U.S. EPA 1999a).

Although water systems rarely have enforcement authority over septic systems, they should work closely with the local regulatory authority to ensure that septic system codes are being properly enforced and to strengthen codes where necessary. Utilities should also encourage residents with septic systems in the watershed to understand their systems and the proper maintenance that their systems require. Programs run by many state cooperative extensions provide educational material and checklists for septic system owners about proper siting and maintenance. Utilities may also want to encourage residents to hook up to a sanitary sewer system where feasible. CWSRF loans, USDA Rural Utilities Service funds, and Department of Housing and Urban Development Community Development Block Grants are available for septic system rehabilitation or replacement. Individual homeowners may be eligible for some of these loans (U.S. EPA 1999b). Some of these funds may also be used to build centralized wastewater treatment.

### F.3.3.12 Wildlife BMPs

Steps taken to prevent wildlife from contaminating source water vary with the source and type of wildlife. Some reservoirs and lakes employ boats with noisemakers to scare seagulls and geese away. Many systems with control of the land around their reservoirs place fences on the water's edge to keep out larger land animals and humans. To keep geese from feeding along the river bank just upstream from one of its intakes, the Philadelphia Water Department planted a riparian buffer and wildflower meadow and conducted a public education program to prevent people from feeding the geese (Philadelphia Water Department 2003).

## F. 4 Case Studies of Existing Watershed Control Programs

Many types of systems can benefit from a watershed control program. This section contains case studies of watershed control programs in place at different PWSs around the United States. These studies show how systems of different sizes and source water types and with varying regulatory authority have adopted watershed control programs to fit their specific needs. This section also describes advantages and disadvantages of implementing a watershed control program.

As shown by the case studies below, successful watershed control programs will vary significantly in their approach to source protection. The systems in the case studies did not focus specifically on *Cryptosporidium* but on controlling microbial point and non-point sources and other contaminants. However, many of the elements noted in these case studies may be useful in watershed control programs addressing *Cryptosporidium.* However, since each watershed is different, the appropriate watershed control program (WCP) plan for each watershed will also be different. Some PWSs may need to develop efforts or measures completely different than those outlined in these examples. Furthermore, some of the approaches outlined in the referenced examples may not be suitable for other watersheds due to different site-specific conditions, and hence may not be used by PWSs developing a successful state-approved WCP for these watersheds.

For more case studies, see the following sources:

- Case Studies of Source Water Protection (U.S. EPA 2005a; http://cfpub.epa.gov/safewater/sourcewater/sourcewater.cfm?action=Case_Studies).

- Section 319 Nonpoint Success Stories (U.S. EPA 2005b; www.epa.gov/owow/nps/Success319).

- Watershed Success Stories – Applying the Principles and Spirit of the Clean Water Action Plan (U.S. EPA 2000d; http://nepis.epa.gov/EPA/html/DLwait.htm?url=/Exe/ZyNET.exe/20001P7N.PDF?ZyActionP=PDF&Client=EPA&Index=2000 Thru 2005&File=D%3A%5CZYFILES%5CINDEX%20DATA%5C00THRU05%5CTXT%5C00000001%5C20001P7N.txt&Query=%E2%80%A2%09Watershed%20Success%20Stories%20%E2%80%93%20Applying%20the%20Principles%20and%20Spirit%20of%20the%20Clean%20Water%20Action%20Plan&SearchMethod=3&FuzzyDegree=0&User=ANONYMOUS&Password=anonymous&QField=pubnumber%5E%22800R00003%22&UseQField=pubnumber&IntQFieldOp=1&ExtQFieldOp=1&Docs= ).

- Protecting Sources of Drinking Water: Selected Case Studies in Watershed Management (U.S. EPA 1999a; http://www.epa.gov/safewater/sourcewater/pubs/swpcases.pdf).

*Burlington, Vermont*
*Medium Surface Water PWS, Watershed Located in Multiple Jurisdictions*

The City of Burlington has a population of 40,000 and is located on the shore of Lake Champlain, a 120-mile long, 12-mile wide lake that is the source of drinking water for the city and other municipalities. In such a large watershed with multiple landowners, it is difficult to control activities that affect water quality. Burlington addresses microbial pollution through a combination of land use control, reduction in combined sewer overflow, watershed restoration, and outreach.

Through Act 250, the state of Vermont regulates land use near lake shores and rivers, accounting for new wastewater treatment plants and sewer systems, timber management, impervious surface area, water withdrawal by ski areas for snowmaking, and other issues. To address combined sewer overflow problems that were affecting Lake Champlain water quality, the city increased the capacity of its main wastewater treatment plant and extended the outfall far into the lake to dilute the effluent. The city separated the sanitary and storm sewers at its smaller plants. Two streams feeding into the lake that suffer from poor water quality are currently undergoing restoration, including retrofitting of existing storm water detention ponds, channel stabilization to prevent erosion, and outreach to change pet waste management, lawn care, and other practices (U.S. EPA 2001a).

*Manchester, New Hampshire*
*Large Surface Water System Where State Plays an Active Role*

The City of Manchester receives its water from Lake Massabesic, which is located approximately three miles east of downtown Manchester. Management of the water supply is primarily under the jurisdiction of the Manchester Water Works. The lake has a surface area of about 2,500 acres and a gross storage capacity of nearly 15 billion gallons. For more than 120 years, this reservoir has served Manchester and five other communities. The Lake Massabesic water supply is supplemented by Tower Hill Pond, which has a gross storage capacity of 1.3 billion gallons. Manchester controls microbial pollution by restricting land use in the portions of the watershed controlled by the water works and the state.

The watershed area for the supply covers 42 square miles with over 25 percent owned and managed by the New Hampshire Department of Environmental Services (NHDES). The NHDES monitors these areas and controls recreational use through regulations posted in the surrounding area, which are enforced by a staff of watershed patrol officers. These regulations strictly prohibit such activities as waste disposal, horseback riding, boating, or any other activity that would immediately or indirectly endanger the surface water quality. Other areas of the watershed are primarily monitored by the Manchester Water Works and have regulated levels of outdoor recreation. Activities such as mountain biking or the establishment of docks and moorings are subject to review and permitting by this agency. Parts of Lake Massabesic closest to the intake are closed to all activity.

The NHDES has provided funding to the Manchester Water Works for the protection of its watershed, specifically funding the installation of a storm water treatment facility and a project to address erosion and sedimentation. DES also provided funding for emergency planning, wellhead protection management plans, drainage mapping, storm water best management practices, and public outreach and education. The source of all this funding was the source water protection-related set-asides from the Drinking Water State Revolving Fund (U.S. EPA 2001b).

*Springfield, Missouri*
*Large GWUDI and Surface Water System with Rapidly Urbanizing Watershed*

Springfield is a city of approximately 150,000 residents located in southwestern Missouri. Much of Springfield's bedrock is limestone and dolomite, and karst features are very pronounced. There are numerous streams, springs, and large concentrations of sinkholes in the area. The city's drinking water is provided by City Utilities of Springfield, a municipally-owned utility. The city uses a combination of springs, wells, reservoirs, and the James River to supply its daily demand of approximately 30 mgd.

The three primary threats to Springfield's water quality that have been identified by its watershed committee are: 1) urbanization in the watershed; 2) wastewater treatment in suburban and rural areas, which consists primarily of septic systems on karst terrain; and 3) agriculture, especially animal waste from concentrated beef and dairy cattle operations. Agricultural and urban BMPs are the primary methods used to address microbial contaminants.

In 1984 a citizen-based Watershed Management Coordinating Committee was established to guide and oversee water protection efforts. The group later incorporated as a non-profit organization and renamed itself the Watershed Committee of the Ozarks. The committee's operating budget is provided by Greene County (in which much of the watershed lies), the City of Springfield (containing the bulk of the water users), and City Utilities (U.S. EPA 2001c).

In 2001, the Committee hosted a workshop on conservation development and better site design for Springfield and Greene County planning and zoning staff members, hosted a workshop on agricultural BMPs for farmers, helped local developers incorporate stormwater BMPs and better site design into their developments, and helped local farmers install alternative watering facilities. The Committee currently has grants under Section 319 of the Clean Water Act to restore several of the area's watersheds. One of these grants involves a study of the current and future loading rates of sediment and nutrients and future construction of a wetland or forebay to treat runoff from the Valley Water Mill watershed as it enters the reservoir. Another project for the Little Sac River Watershed, which provides 85 percent of Springfield's water, has just gotten underway (Watershed Committee of the Ozarks 2001).

## F.5 References

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# Appendix G

## Review Criteria for Use by States When Reviewing Watershed Control (WSC) Program Plans

| LT2 WSC Requirement | Assessment Criteria | Addressed in Sufficient Detail? |
|---|---|---|
| **Watershed Control Program Plan** | | |
| * | Does the plan specifically address potential and existing *Cryptosporidium* sources in the watershed? | |
| * | Have the proposed actions in the plan been clearly defined and sufficiently addressed? | |
| * | Does the plan explain how the actions described are expected to contribute to specified goals? | |
| | Does the plan prioritize its proposed efforts? Does it define short-term and long-term actions and prioritize them? | |
| * | Does the plan include, in detail, what other resources will be required to implement the watershed control measures? Does it identify the source(s) of those resources? | |
| **Review of Potential Sources** | | |
| * | Has the area of influence been delineated in appropriate detail, taking into consideration available information about *Cryptosporidium* fate, transport and local hydrogeological characteristics? Have sensitive areas been identified? | |
| | Is the scale of the delineation appropriate for the watershed plan? Does it provide a level of detail sufficient for effective decisions to be made? | |
| | Has the intake location been identified relative to the water body? | |
| | Is any information available about time of travel in the watershed? | |
| * | Does it seem that all activities within the watershed that could result in *Cryptosporidium* contamination of the water supply have been identified and located? | |
| * | Have contaminant sources been located and described relative to the drinking water source intake location? | |
| | Have the likelihood and timing of releases of contamination been addressed? | |

| LT2 WSC Requirement | Assessment Criteria | Addressed in Sufficient Detail? |
|---|---|---|
| * | Are there permitted wastewater discharges (NPDES) of concern? If there are wastewater treatment plants in the area of influence, systems should include information about their size, discharge quantity, and whether there has been any recent significant noncompliance with permit conditions. | |
| * | Are sludge disposal areas identified and characterized? Are there any locations in the watershed where biosolids have been applied? Have they been identified? When in the year are they applied? | |
| * | Have stormwater discharges been located? Are there any discharges directly into the surface water supply? | |
| * | Have septic systems been identified and located? Is information available about their age, condition, design, and siting? | |
| | Has land use been characterized and mapped? Are areas subject to zoning requirements or changes in zoning? | |
| * | If land uses in the watershed include agriculture, have the types of farming been identified? Are feedlots located? Are fields where manure is spread identified? | |
| * | Have Concentrated Animal Feeding Operations (CAFOs) been identified and located? | |
| * | Have natural sources of *Cryptosporidium* been identified and located? | |
| * | Have recreational areas (e.g., campgrounds, trailer parks) been identified and located? | |
| | Has any on-site landfilling, land treating, or surface impounding of waste other than landscape waste or construction and demolition debris taken place, and will such circumstances continue? | |
| | Does the analysis address the effectiveness of physical barriers (e.g., geology, hydraulic conditions, intake structure and location) at preventing the movement of contaminants to the drinking water source? | |

| LT2 WSC Requirement | Assessment Criteria | Addressed in Sufficient Detail? |
|---|---|---|
| * | Have tributaries or areas of the reservoir with evidence of high levels of microbial contamination been identified? If so where are they located relative to the intake? | |
| | If *Cryptosporidium* monitoring data exist for the watershed, have results been addressed and discussed? | |
| * | Have recreational uses of the surface water supply been identified? Has the effect of those uses on *Cryptosporidium* loading been addressed? | |
| * | Are there portions of the watershed with high percentages of impervious surfaces which might lead to increased stormwater runoff? | |
| | Is water quality monitoring and assessment information (305(b) Report available? | |
| | Have existing best management practices or controls been identified and located? | |
| | Is there any information available about the effectiveness of current pollution prevention activities? | |
| **Potential Control Measures to Control Cryptosporidium Contamination** | | |
| * | Do the control measures proposed specifically address the reduction of *Cryptosporidium* contamination? | |
| * | Would the implementation of the proposed control measures take place in areas where there would be an impact on *Cryptosporidium* loading into the water supply? | |
| | If the proposed control measures are ongoing, has the utility explained how they would be sustained? | |
| * | Is the water utility in a position where it could implement the control measures itself, or would other parties be responsible? | |
| * | Are there implementation agreements between the utility and other parties responsible for implementation? | |
| | How does the utility track control measures implemented by other parties? | |

| LT2 WSC Requirement | Assessment Criteria | Addressed in Sufficient Detail? |
|---|---|---|
| | Has the water system responded adequately to concerns expressed about the source or watershed area in past inspections and sanitary surveys? | |