Appendix 12. Risk reduction technologies potentially supporting interbasin water transfers

Water diversions envisioned under the Flood Control Act of 1944 nearly 60 years ago markedly differ from those being considered as options under water management plans currently motivating this study. And, given the concerns related to biota transfers initially voiced by the International Joint Commission (IJC) in the late 1970's, control systems have been considered that minimize risks of biota transfers associated with water diversions between Missouri River source waters and the receiving systems, Red River of the North. Implementing interbasin water transfers in full compliance with the Safe Water Drinking Act (1974) as amended (including amendments promulgated in 1996) would bring to resource management discussions a system of control technologies that are effectively risk reduction tools. Within the context of risk reduction, these technologies as integrated components of control systems serving to implement interbasin water diversions are presently considered in this risk characterization. While these tools serve to reduce risks, in this case related to biota transfers, there are attendant uncertainties that must also be considered within an adaptive management context, and these uncertainties have also been considered briefly in the following sections.

Water treatment control systems as a risk reduction tool

Fifty to one-hundred years ago, waterborne diseases such as typhoid and cholera reached epidemic proportions in American cities (see, e.g., Percevial et al 2004), and the development of technologies to treat drinking water was one of the major public health advances in the 20th century. During that time interval, various control technologies have been developed to assure water disinfection is achieved and limited as a major factor in disease outbreaks and epidemics. These control technologies range from chemical and physicochemical treatments (e.g., chlorination and chloramination to ozonation) to physical barriers acting as filters (e.g., pressure-driven membrane technologies), each capable of reducing risks of biota transfers associated with interbasin water diversions (see Letterman 1999). These technologies may be used singly or in combination in control systems designed to meet user specifications, yet regardless configuration, the systems themselves present collateral risks that must be considered in any adaptive resource management plan, e.g., chemical treatments such as chlorination may yield unintended byproducts which may pose risks consequent to interaction with naturally-occurring materials in the water (see, e.g., Percival et al 2004, Letterman 1999).

Chemical treatments: Chlorination[1] and Chloramination

Disinfection in water treatment is required by the Surface Water Treatment Rule of 1990 and subsequent regulations (see, e.g., http://www.epa.gov/OGWDW/mdbp/ieswtr.html) which mandates effective disinfection through (1) filtration pre-treatment of source waters followed by (2) inactivation of organisms such as bacteria and viruses by disinfectants such as chlorine, chlorine dioxide, or ozone, and (3) as applicable, treatment requirements for waterborne pathogens, e.g., *Cryptosporidium* spp. in addition to meet existing requirements for *G. lamblia* and viruses.

Water is disinfected, not sterilized, in the water treatment process. As such, disinfection generally occurs as a two-step process wherein (1) particulate matter is removed by conventional filtration to reduce turbidity in source waters and thus, reduce "habitat" for viruses and bacteria adsorbed to particulate material, and then (2) pathogenic microorganisms are inactivated, e.g., by chemical treatments (such as chlorination and chloramination), physicochemical treatments (such as ozonation), or removed through physical treatments (such as membrane filtration; see, e.g., Letterman 1999 for overview of water treatment process; see also Mallevialle et al 1996, Duranceau 2001, Schippers et al 2004 for discussions of pressure-driven membrane systems).

Chlorination has been synonymous with disinfection, since the chemical's use as an agent for disinfection increased in the US over the past 100 years (see, e.g., Letterman 1999, and http://www.awwa.org/Advocacy/learn/info/HistoryofDrinkingWater.cfm last accessed December 8, 2004). Initially, chlorine was available for use only as hydrated lime, chloride of lime, or as bleaching powder, and in 1894 chlorine was first used as a disinfectant on a plant scale basis in Brewster, New York (see http://www.nywea.org/clearwaters/321011.html last accessed December 8, 2004). In Europe, chlorine gas was first used as a disinfectant in drinking water in 1903, and the first full scale chlorine installation at a drinking water plant in the United States occurred in 1908 (Pontius 1990). Numerous technologies were developed for the chlorination of drinking water, and in October, 1914 the Department of the Treasury enacted the first set of standards that required the use of disinfection for drinking water. These standards called for a maximum level of bacterial concentration of 2 coliforms per 100 millilters, and because

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[1]Updated and revised source materials originally developed by Eric Emenheiser, Chris Forstner, Matt Curtis & Erik Johnston, Becky Cheadle, Mark Smith, Daniel Gallagher, Virginia Polytechnic Institute and State University, Blacksburg, Virginia. Last updated December 8, 2004.

chlorination was the main disinfectant at the time, these standards dramatically increased the number of treatment plants using chlorine (White 1999). Then, in 1925 new drinking water standards were enacted that reduced the maximum permissible limit of coliforms from 2 to 1 coliform per 100 millilters. Advancements in chlorination as a primary means of disinfection occurred throughout the first half of the 20th century, but in 1972 the first published report of disinfection by-products (DBPs) in drinking water was published (EPA 1972), and focused on DBPs resulting from chlorination of source water polluted with organic chemicals (White 1999).

Much of the process of chlorination relies on technology developed in the 1950's and 1960's (see White 1999 and earlier editions of this reference). Although the tools for chlorination have continued to be refined, few innovations have been made recently. Other disinfection technologies have been developed (e.g., ozonation, UV irradiation), but chlorine remains widely used as a disinfectant throughout the US because of its low cost, ability to form a residual, and its effectiveness at low concentrations.

Chlorine gas, when exposed to water reacts readily to form hypochlorus acid, HOCl, and hydrochloric acid. $Cl_2 + H_2O \rightarrow HOCl + HCl$, and if pH exceeds 8, the hypochlorus acid will dissociate to yield hypochlorite ion, $HOCl^-$ which dissociates, H^+ and OCl^-. If pH is much less than 7, then HOCl will not dissociate. If ammonia is present in the system, hypochlorus acid will react to form one three types of chloramines depending on the pH, temperature, and reaction time. Monochloramine and dichloramine are formed in the pH range of 4.5 to 8.5, with monochloramine most common at pH > 8. When pH < 4.5, the most common form of chloramine is trichloramine. Chloramines are bacteriocidal, but are less effective as disinfectants for viruses.

Overall, chlorine presents numerous advantages for disinfection, including the chemical's ease of application and residual presence in the distribution system, its effectiveness at low concentrations, and its relatively simple conversion to chloramines which also provide strong residual effects with limited DBPs. From an engineering cost perspective, chlorine is a relatively inexpensive disinfecting agent. Despite these advantages, chlorine has "down side" characteristics that must be managed, if it is selected as a disinfection agent of choice. Chlorine reacts with organic materials in source waters, effectively reducing its concentration while creating trihalomethanes (THMs) and DBPs, compounds that may become health risks in drinking water distribution systems. More importantly from the perspective of its role as a disinfection chemical, chlorine provides poor disinfection levels for *Cryptosporidium* spp. and other microorganisms characterized by chlorine-resistant stages in their life history (e.g., spore formation; see Appendix

3B). For target organisms such as *Cryptosporidium* spp., filtration provides an alternative disinfection method used singly or in conjunction with chlorination (see, e.g., Schippers et al 2004, Duranceau 2001, Mallevialle et al 1996).

Water Disinfection with chlorine dioxide

Chlorine dioxide (ClO_2) has found increased use in drinking water treatment, since it is as good as, or better as chlorine as a disinfectant (see White 1999). From a water treatment perspective, chlorine dioxide is a good oxidant, reducing iron, manganese, sulfur compounds, and odor-causing organic substances in raw waters. The chemical's increased use, however, stems in part from its use as a pre-oxidant, since chlorine dioxide does not as readily chlorinate organic compounds in source waters. In addition to the chemical's reduced reactivity with natural organic matter (NOM) or organic pollutants to form trihalomethanes (THMs) or other chlorinated byproducts, chlorine dioxide has also found favor in water treatment, because ClO_2 will not oxidize bromide (Br^-) to bromate (BrO_3). Hypobromous acid (HOBr) can also form brominated DBPs in reactions with NOM. Regardless of the source of bromate, this constituent will be regulated at 0.010 mg/L by the Disinfectant-Disinfection By-Product (D-DBP) Rule, owing it the chemical's health risks (EPA 2001). As a disinfectant, ClO_2 is as good or better than chlorine for the inactivation of *Giardia* and is better than either chlorine or chloramines for the inactivation of *Cryptosporidium* (see Letterman 1999). While C*t values will vary depending on system design, comparative C*t values for chlorine, chloramines, ClO_2, and O_3 are summarized in Table 1 to illustrate range of disinfection realized under various technologies (see Connell 1996, Haas 1999 and White 1999 for discussion).

Table 1. Examples of C*t Values (mg/L * min) for various disinfectants (Connell 1996, Haas 1999, White 1999).

Indicator	Chloramines	Chlorine	Chlorine Dioxide	Ozone
Giardia 0.5 log inactivation pH 6-9, 5˚C	340-380	15-50	4.0-6.0	0.3-0.6
Viruses 2 log inactivation pH 6-9, 5˚C	825-900	4-7	5.0-6.0	0.6-0.9
Cryptosporidium pH 7, 25˚C	7200 2 log inactivation	7200 1 log inactivation	78 1 log inactivation	5 - 10 2 log inactivation

In contrast to chlorine, chlorine dioxide does not react as readily with organic constituents in source waters; hence, chlorinated by-products such as THMs are reduced in the post-processing stream. For drinking water treatment, typical ClO_2 treatments have been targeted at less than 1.5 mg/L, given the maxmium daily residue load (MDRL) for finished-water concentrations of ClO_2 0.8 mg/L. By-products of chlorine dioxide include chlorite ion (ClO_2^-) and chlorate ion (ClO_3^-), which have been linked to potential adverse health effects, and subject to regulatory levels mandated by Stage 1 Disinfectant/Disinfection By-Products (D/DBP) Rule. Maximum contaminant level (MCL) for ClO_2^- is 1.0 mg/L, with no ClO_3^- MCL yet proposed (EPA 2001a).

Treatment with chloramine. Chloramines are the product of chloride reacting with ammonia, as noted in the brief discussion of chlorine earlier in this section. Chloramines, particularly monochloramine, have also been used as disinfectants since the 1930's, and its use in drinking water disinfections is an increasingly common standard practice among water utilities (see also Haas 1999). In part, the increasing use of chloramine in water treatment stems from chlorine's disadvantages.

While chloramine is a weaker disinfectant than chlorine, it is more stable in water solutions under operating pH and the chemical's benefits as a disinfectant are available over longer periods of a system's operation. Chloramines primary use in water treatment is as a secondary disinfectant, since it helps maintain a disinfectant residual in the distribution system. Chloramine also provides the following benefits:

- Chloramine is not as reactive as chlorine with organic material in water, thereby producing substantially lower concentrations of DBPs such as THMs and haloacetic acids (HAAs) which have associated adverse health effects at high levels.

- Because the chloramine residual is more stable and longer lasting than free chlorine, it provides better protection against bacterial regrowth in systems with large storage tanks and dead-end water mains.

- Like chlorine, chloramine effectively controls the formation of biofilms within the distribution system. Controlling biofilms reduces microbial habitat in distribution systems, which reduces concentrations of coliforms and other microorganisms, and helps reduce biofilm-induced corrosion of pipes.

● Because chloramine does not tend to react with organic compounds, many systems will experience less incidence of taste and odor complaints when using chloramine.

In addition to these technical advantages of chloramine, many drinking water utilities in the US have switched to chloramine as their disinfectant residual, since regulatory limits for THMs in drinking water have been lowered with promulgation of the Stage I Disinfection Byproducts Rule and subsequent administrative targets for lowering standards of DBPs (see EPA 2001a for a quick reference, or EPA 2001b).

Membrane filtration

Membrane filtration technology has been increasingly applied to water treatment problems, where the range of membrane technologies have become efficient and safe water treatment alternatives. Water treatment systems singly dependent on membrane filtration, or incorporating membrane technology within a multiple-treatment process, yield product waters of consistent quality that meets or exceeds water quality standards, especially with respect to disinfection (see, e.g., Schippers et al 2004). Membrane separation technology removes substances largely based on size and shape, with pore size and particle-size exclusion typically measured in nanometers (nm, or 10^{-9} meters), Angstroms (Å, or 10^{-10} meters), or molecular weight (MW, often times expressed as units, D for daltons). A range of membranes have been developed with mass transfer properties and pore sizes such that ionic, molecular and organic substances measuring 1-1000 Å (MW between 100 and 500,000) are removed or rejected (Table 2).

As a "stand-alone" water treatment technology, membrane filtration is a physical process that may require little or no chemical treatment, depending on the choice of membrane device selected. Unlike ultraviolet radiation technology, membrane filtration allows not only the removal of pathogens, bacteria and viruses likely to be found in the water drawn, but also to reduces color, turbidity and mineral content. While various types of filtration are available, three general types are briefly considered: microfiltration, ultrafiltration, and nanofiltration (Figure 1; graphic after AWWA).

Table 2. Comparative rejection values for range of membrane technologies (modified from Water Technology, Latham. NY; see Mallevialle 1996, Duranceau 2001).

Species	RO	Loose RO	NF	UF
Sodium Chloride, NaCl	99%	70-95%	0-50%*	0%
Sodium Sulfate, Na_2SO_4	99%	80-95%	99%	0%
Calcium Chloride, $CaCl_2$	99%	80-95%	0-60%	0%
Magnesium Sulfate, $MgSO_4$	>99%	95-98%	>99%	0%
Sulfuric Acid, H_2SO_4	98%	80-90%	0%	0%
Hydrochloric Acid, HCl	90%	70-85%	0%	0%
Fructose, MW 180	>99%	>99%	>99%	0%
Sucrose, MW 360	>99%	>99%	>99%	0%
Humic Acid	>99%	>99%	>99%	0%
Viruses	99.99%	99.99%	99.99%	99%
Proteins	99.99%	99.99%	99.99%	99%
Bacteria	99.99%	99.99%	99.99%	99%

*values reflect controlled conditions where "0%" rejection is valid for a 30,000 parts per million (ppm) solutions occur as mixture of ions. Actual rejection of defined 30,000 ppm solution ranges from 5-15%, with the higher value valid for dilute solutions. Actual rejection will vary depending on the composition of the feed and membrane characteristics.

Microfiltration is characterized as a solid-liquid separation process with a molecular weight cut off between between 0.1 µm and 10 µm (Figure 1). Microfiltration reduces the passage of suspended particles, high-molecular weight lipids and fats, macromolecules, bacteria and protozoa (although *Cryptosporidium* spp. and *Giardia* spp. may not be removed completely). It is frequently used for the production of drinking water and waste water treatment.

Ultrafiltration allows the filtration of smaller particles than microfiltration with a molecular weight cut off between between 0.01 µm (micrometers, 10^{-6} meters) and 0.1 µm, which effectively excludes all protozoa, bacteria and virus particles, as well as most proteins and high molecular weight organic compounds (Figure 1). Ultrafiltration is finding widespread use for a variety of applications such as producing drinking water, treating waste water and treating

process water (e.g., discharges from agricultural, biotechnology, petrochemical, municipal waste streams).

Nanofiltration provides the greatest filtration capacity of the membrane technologies, with pore sizes less than 10 nm (Figure 1). As such, nanofiltration not only excludes those constituents separated by ultrafiltration, but also limits passage of divalent ions, dissolved organic material and sugars. Given the membranes characteristic molecular-weight cut off, nanofiltration provides for partial demineralisation, which tends to yield potable water from slightly brackish water or humic-stained surface water.

FIGURE 1   Removal characteristics and operating pressures of pressure-driven membranes
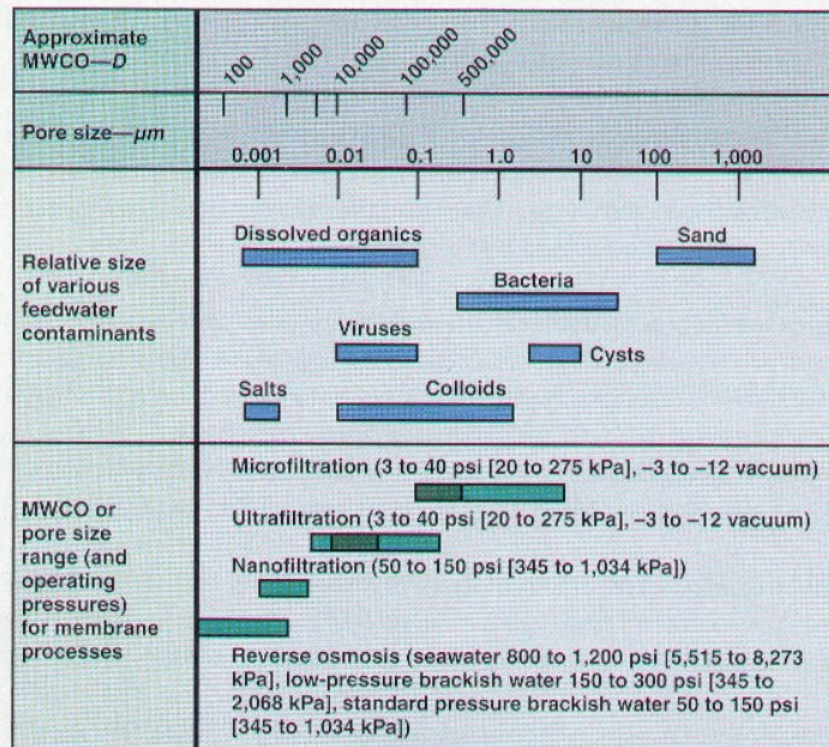
Darker-shaded regions indicate the range of commercially marketed hollow-fiber low-pressure membrane products for potable water applications. MWCO—molecular weight cutoff

(Source: American Water Works Association)

Although not considered withing the context of alternatives pertinent to the current focus on reducing risks associated with interbasin biota transfer, reverse osmosis technologies have

numerous applications in industry, and it use in water treatment processes is most often associated with desalination projects. As a membrane technology, reverse osmosis allows for the separation of suspended solids and ionic species. A key difference between reverse osmosis and other filtration membranes lies in membrane pore size, with reverse osmosis membrane pores measuring between 1 to 15 Å (Angstroms). Membrane technologies are usually operated according to a cross-flow mode where source or "raw" water enters the treatment system via an inlet, product or filtered water (permeate) exists the system via an outlet, and rejected water (concentrate) is discharged through another outlet. Dead-end filtration modes are also designed as required for specific operations; in dead-end modes, rejected water is "recycled" to achieve higher rates of product recovery (see Malleviallen1996, Duranceau 2001).

Chemical treatment, membrane filtration, and disinfection of water

Chlorine disinfection has been effective in treating drinking water supplies in the US for most of the 20th century, especially with respect to protecting drinking water resources from bacterial and viral contamination (see Haas 1999 for brief history). Water-borne diseases such as typhoid fever (*Salmonella typhi*), cholera (*Vibrio cholerae*) and bacillary dysentery (*Shigella dysenteriae*) have been eliminated or dramatically reduced to sporadic outbreaks infrequently associated with failed treatment facilities in the developed countries. Three primary chemical agents are used in chlorine disinfection are free chlorine, chloramine, and chlorine dioxide. The disinfection capacity for these chemicals, however, reflects chemical properties of chloride that necessarily contribute to "trade offs" associated with the chemical's use. Chlorine is highly reactive, and when added to water in its various forms, disinfection occurs but in the process, chlorine reacts with other chemicals, including organic constituents, present in the water. These chemicals generally enter the water supply through natural plant and soil breakdown, or through anthropogenic chemicals such as agricultural or industrial chemicals released to the environment.

When chlorine reacts with organic constituents in source water, DBPs such as THMs (e.g., chloroform) and HAAs. Other disinfection chemicals also generate DBPs, but given chlorine's long record in water treatment, more is known regarding DBPs resulting from chlorination than by other disinfectants. Many DBPs resulting from chlorination have been characterized as health risks, e.g., animal studies with DBPs at high exposure concentration have been characterized by increased incidence of cancer, although mechanisms leading to these pathological responses is poorly understood (see Regli et al 1994).

Given the health risks associated with DBPs, rather than use chlorine gas in the chlorination process, chloramine or chlorine dioxide may serve as alternative sources for chlorine disinfection. While these sources produce fewer DBPs than chlorine, these chemicals also have their attendant risks and technical shortcomings. For example, as a disinfectant, chloramine is not as strong as chlorine, and disinfection with chlorine dioxide produces its own DBPs (White 1999). Currently, DBPs occur in US drinking water at very low concentrations, and unequivocal links between exposures to DBPs at low concentrations and adverse health risks associated with these exposures is relatively poorly characterized (Regli et al 1994, Singer 1999). Under the authority of the Safe Drinking Water Act (SDWA), EPA has regulated acceptable levels of some DBPs since 1979. And, under the Safe Drinking Water Act Amendments of 1996, EPA has been charged to implement its initial Disinfectant and Disinfectant-By-Product (DDPB) rule and Interim Enhanced Surface Water Treatment Rule (IESWTR), both originally proposed in 1994.

The continued need for chemical disinfection of drinking water and the attendant risks associated with DBPs illustrates the trade-offs between benefits and risks associated with chlorination, given the characterization that microbial pathogens consistently present greater risks in drinking water sources than do DBPs resulting from chemical treatments such as chlorination (see, e.g., Regli et al 1994). Although most regulatory attention has focused on DBPs of chlorine, other disinfection processes also produce byproducts when chemicals used in treatment react with organic or inorganic constituents in raw water, e.g., ozonation has been used effectively in water disinfection, but bromate ion is a DBP resulting from ozonation of water high bromide. Bromate is being evaluated by EPA during the Stage I rule developed to support the SDWA as amended 1996.

Cryptosporidium. Another factor that is likely to impact the choice of primary disinfectants by utilities is the need to address disinfection focused on *Cryptosporidium*. Chlorine is not very effective in treating *Cryptosporidium*, but adequate filtration appears to be the best protection from the disease agent. In some areas with poor water quality, however, a combination of disinfection technologies may be necessary to provide disinfection of *Cryptosporidium* and other protozoan, bacterial, and viral agents of waterborne disease (see, e.g., Percival et al 2004). Chlorine dioxide and ozone have been shown to be relatively effective for inactivating *Cryptosporidium*, yet the literature suggests that these disinfectants can be even more effective when used sequentially with various membrane technologies (see, e.g., White 1999, Letterman 1999, Schippers et al 2004).

Groundwater Disinfection. According to the EPA there are over 150,000 groundwater systems in the US (see http://www.epa.gov/safewater/ last accessed December 8, 2004). The "Groundwater Rule" is intended to address microbial contamination of groundwaters (see http://www.epa.gov/fedrgstr/EPA-WATER/2000/May/Day-10/w10763.htm last accessed December 8, 2004). Groundwater, although filtered by natural processes, is often susceptible to microbial contamination, especially in rural communities, and source waters may need disinfection as part of the treatment process. Increasingly, sources of drinking water dependent on groundwater have been found vulnerable to microbial contamination, including indication of fecal contamination from tests for total coliform bacteria, *E. coli*, coliphage and human viruses (NRC 2003). Drinking water derived from groundwater sources has also been the source of nearly half of all waterborne disease outbreaks in the US, and with an increasing reliance on groundwater, inadequate disinfection of groundwater and untreated groundwater will continue to dominate as sources of waterborne disease outbreaks in the US (see CDC 2004; Figure 2).

FIGURE 2 Drinking water-associated outbreaks, by etiologic agent, water system, water source, and deficiency — United States, 2001–2002
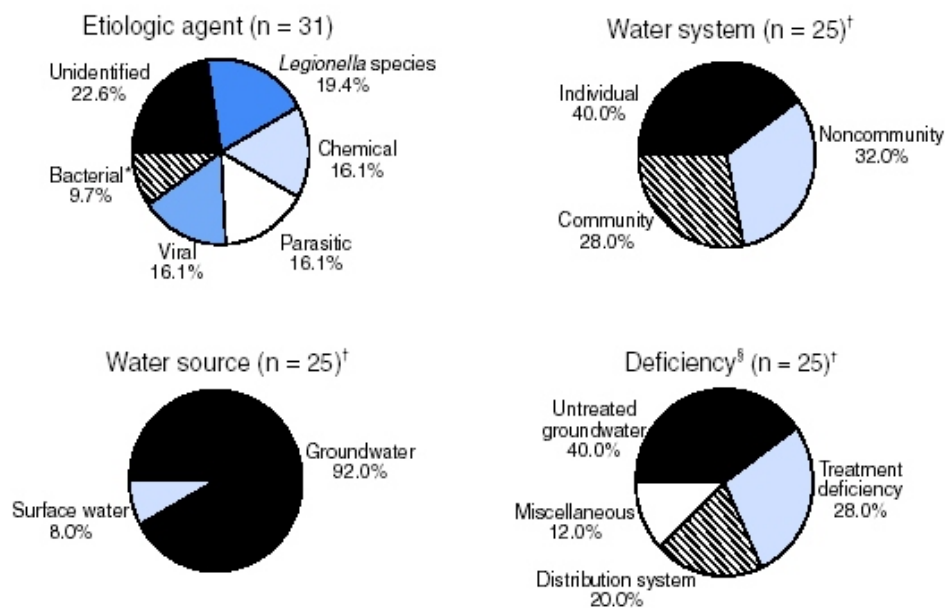
* Other than *Legionella* species.
[†] Excludes outbreaks attributed to *Legionella* species.
[§] No outbreaks were attributed to untreated surface water.

(Source: Centers for Disease Control and Prevention)

[2]**UV Disinfection of Drinking Water**

In view of the "Best Available Technologies" being considered as part of the Groundwater Rule, brief summaries of UV and ozone disinfection technologies are considered in the following sections. As noted many times in past reports, the problem of unsafe drinking water is not an isolated technical problem, but interrelated to the problems of adequate water supply, community education in public hygiene, access to sanitation, and effective and safe disposal of human and animal wastes (see, e.g., Percival et al 2004).

Use of ultraviolet (UV) light to disinfect water of waterborne pathogens relies on the germicidal properties of a narrow range of the UV spectrum. Provided dosage is sufficient (e.g., exposure duration long enough to yield desired C*t values), UV wavelengths ranging from 240 to 280 nanometers (nm) deactivate, or effectively kill, microorganisms by damaging their DNA so as to prevent DNA repair. The organism, if not killed, is unable to replicate and thrive (see, e.g., McKey et al 2001, Jacangelo et al 2002). UV dose measured in microwatt-seconds per square centimeter is the product of UV intensity and exposure time, and exposures for 90% kill of most bacteria and viruses range from 2,000 to 8,000 μW-s/cm^2, and for disinfection targeted on *Giardia* spp., *Cryptosporidium* spp., and other large cysts and parasites, UV doses are an order of magnitude greater (approximately 60,000-80,000 μW-s/cm^2; see, e.g., McKey et al 2001, Jacangelo et al 2002). Most UV disinfection systems use a low-pressure or medium-pressure mercury vapor lamp and expose water to UV by pumping the water around a sleeve within which the UV lamp is supported. UV systems can also be coupled with a pre-filter to remove larger organisms that would otherwise pass through the UV system unaffected. The pre-filter also clarifies the water to improve light transmittance; therefore, UV dose is achieved throughout the entire water column. Proper handling and storage of UV-treated waters are a critical part of any UV treatment system. UV treatment alone offers no residual disinfection, and if bacteria are not killed as a result of UV exposure, the organisms may be able to repair their DNA and reactivate in a few days when exposed to visible light (see, e.g., Mara and Horan 2003).

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[2]Edited and updated from original source material prepared by Ashok Gadgil, Anushka Drescher, David Greene, Peter Miller, Cynthia Motau, and Frank Stevens and published by Lawrence Berkeley Laboratory Report Number LBNL 40360 (1997).

[3]**Disinfection with Ozone**

Ozone is a form of oxygen with the molecular formula O_3 which forms when O_2 or clean dry air is exposed to a powerful electric current. In nature, it forms in the upper atmosphere when lightning passes through the air. As a disinfectant, ozone is highly unstable and decomposes to O_2 and O^- shortly after its formation; O^- is the highly decomposition product responsible for ozone's disinfectant properties. As a highly reactive species, ozone is a powerful oxidant and one of the most powerful disinfectants available in water treatment.

Ozone was discovered late in in 1783, and by the close of the close of the 19th century, ozone was used for water disinfection in Europe. By 1980, there were over 1100 water treatment facilities using ozonation, with most of these facilities in Europe (see, e.g., Letterman 1999, White 1999). In the 1930's, several water treatment plants in New York, Pennsylvania, and Indiana experimented with ozone, but only recently has the use of ozone gained attention in the US. Although few water treatment facilities relied on ozonation in 1987 (Haas 1999), a few years later nearly 40 water treatment plants in the US were equipped with ozonation facilities. Water disinfection in the US currently remains more heavily dependent on less expensive disinfectants such as free chlorine, chlorine dioxide, and chloramines (see, Letterman 1999). However, recent legislation such as the Safe Drinking Water Act Amendments of 1996 and its associated rules or proposed rules (e.g., Surface Water Treatment Rule, Groundwater Rule) place more strict rules on both the range and amount of disinfection needed and the concentrations of DBPs allowed in drinking water. As a consequence, the use of free chlorine and other common disinfectants may become less cost-effective, and ozone disinfection has gained much attention, given the increasing awareness of public health risks associated with agents of waterborne disease and ozone to effectively inactivate *Cryptosporidium* and *Giardia* (see, Regli et al 1994, White 1999).

Disinfection is achieved when ozone reacts with source water upon entering a "reaction chamber" designed to allow for sufficient contact time, wherein ozone decays to form oxygen molecules and free hydroxyl radicals (Haas 1999). Both ozone and the highly reactive free radicals generated in its decomposition serve as strong oxidants, attacking organic molecules, including pollutants such as herbicides and pesticides, and biological compounds such as proteins,

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[3]Updated and edited from original material prepared by Rob Dunham, Hong He & Ken Woodard, Daniel Gallagher, Stacie Kramer and Susanna Leung, Virginia Technological Institute and State University, Blacksburg, VA.

carbohydrates and humic acids, at double-bonded carbons to damage and destroy critical components of organisms found in the water. Contact times are relatively short duration, but highly effective at disinfection (Table 3), since the free hydroxyl is very reactive compared to other disinfectants (see White 1999).

Table 3. Summary of ranges of C*t-values for 99% inactivation of various waterborne disease agents by various disinfectants at 5°C (see White 1999)

Organism	Free Chlorine pH 6-7	Preformed Chloramine pH 8-9	Chlorine Dioxide pH 6-7	Ozone
E.coli	0.034-0.05	95-180	0.4-0.75	0.02
Polio I	0.1-2.5	768-3740	0.2-6.7	0.1-0.2
Rotavirus	0.01-0.05	3806-6476	0.2-2.1	0.006-0.06
Phage F2	0.08-0.18	---	---	---
Giardia lamblia cysts	47-150	---	---	0.5-0.6
Giardia muris cysts	36-630	1400	7.2-18.5	1.8-2.0
Cryptosporidium parvum	7200	7200	79	5-10

Precautions must be taken to assure that ozone and its highly reactive free radicals are eliminated from product water following disinfection, given the highly reactive decomposition products of ozone.

From an engineering perspective, ozone has its own set of benefits and risks as a disinfectant (Table 4). Currently, the primary drawbacks to ozonation are capital costs relative to alternative chlorination systems, and the increased cost of maintenance and operation over conventional water treatment facilities (see, e.g., http://www.awwa.org/Advocacy/ pressroom/ozone.cfm; see also, EPRI 1996, 1999; http://www.epri.com/attachments/ 285588_WaterImperative.pdf). As with UV disinfection, ozonation does not yield disinfectant residuals in the water distribution system post-treatment; hence, long-term effectiveness of ozonation may also limit is adoption, since bacterial regrowth may occur in the distribution system following treatment, if bacteriocidal effects were not achieved in the ozonation process.

Table 4. Simplified summary of selected benefits and costs associated with ozone disinfection of source water.

Benefits	Costs
Extremely powerful disinfectant	Expensive option
Does not form trihalomethanes	Can form other hazardous disinfection-by-products such as bromate
Requires relatively short contact time	Requires high level of technology
Reduces taste, odor, and color in water by oxidizing the algae and humic material which causes these problems	Requires another disinfectant to achieve residual disinfection levels
Forms microfloc upon contact therefore improving coagulation and reducing the required coagulant dose	Unstable - must be generated on-site
- Can improve filtration rates. With improved coagulation, more material settles in the sedimentation basin. Hence, less material reaches the filters and the filters can be run longer before backwashing.	Climate control needed to maintain solubility
- Environmentally friendly - decays back to oxygen	Not widely used in US

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