Abstract

MARK A. MANZIONE. Effluent Chlorination Health Effects and Policies (Under the direction of Prof. JAMES C. LAMB, III).

Evaluation of recent literature regarding the adverse effects and the benefits of effluent chlorination for disinfection indicates that for most cases, secondary effluents should continue to be disinfected. Chlorine continues to be the most inexpensive and reliable method of disinfection. The adverse effects to aquatic ecosystems and the formation of possibly hazardous reaction products are, at the present, apparently minor or feasibly mitigable.

EFFLUENT CHLORINATION HEALTH EFFECTS AND POLICIES

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I. INTRODUCTION

Chlorination of the effluent from sewage treatment plants has been widely practiced in the United States. Chlorination is the method of disinfection with the best established reliability, ease of operation, and lowest cost. In the United States, it is the method used in over 99% of the wastewater treatment plants which disinfect, accounting for about two-thirds of total municipal wastewater flow [Maxted,1983; Virginia Disinfection Task Force (VDTF),1984]. It appears that as the nation's wastewater treatment capacity expands, the use of chlorination increases at equal or greater rates.

Requirements for disinfection of wastewater reflect concern for protecting public health, but vary considerably from state to state since the primary benefits of disinfection in preventing the transmission of waterborne disease do not apply uniformly to all locations at all times. In areas where there is high dilution of the effluent, only seasonal recreational uses, or even no downstream uses involving human contact, the benefits of disinfection may be non-existent or may not justify the expense and possible adverse effects.

Disinfection of sewage treatment plant effluent reduces the chances of transmission of infectious disease by reducing microbial populations , since the discharge of viable pathogens into the environment can constitute a hazard to human health. If pathogens are discharged into sewers, survive through treatment plants, and are released, people may ingest them from waters used for potable water sources, seafood growth, or swimming. Users of waters receiving discharges may be directly exposed to contamination, as in swimming, and therefore depend on the protection provided by sewage treatment or they may be protected by further treatment, as in most drinking water systems. Although downstream drinking water system withdrawers do not rely upon upstream dischargers' disinfection, they do nevertheless benefit from a reduced disease risk due to lower pathogen load on their protective treatment.

There are, however, adverse secondary effects accompanying chlorination. These are mainly the toxicity of residuals to aquatic life and the possible formation of chlorinated organic compounds harmful to human health. Attention in recent years to the harmful effects on aquatic life by various chlorine compounds has led more than two-thirds of the states to establish water quality standards and criteria that limit chlorine discharges [VDTF, 1984]. Also, discovery of the formation of trihalomethanes(THMs) in drinking water from reaction of chlorine with trace organic compounds has caused alarm. Chloroform, the most common THM, is suspected of being a carcinogen based on extrapolation from studies of animals given unusually high doses of chloroform. The potential effects on human health of lifelong consumption of trace amounts of chlorinated organic compounds in drinking water has prompted concern about their presence in chlorinated wastewaters. Partly due to concern about adverse effects, over half the states are currently reviewing their disinfection or chlorine residual regulations [VDTF, 1984]. Ideally, all adverse effects should be evaluated and considered along with all expected benefits of a particular wastewater disinfection policy.

Rational policy for regulation of sewage treatment plant effluent disinfection requires determining the best solution for a problem which involves a tradeoff between two potential public health risks:

A. Requiring disinfection on a broad scale maximizes 1) the protection against transmission of infectious waterborne disease, but also 2) the production of whatever chlorinated organics are formed and the potential health risk imposed on downstream users.

B. Allowing the reduction or elimination of chlorination lowers the operating costs and may minimize formation of chlorination byproducts, but presents the possibility of higher risk of waterborne infection.

A rational evaluation of the benefits and adverse effects of effluent chlorination would require, in theory at least, knowledge of the quality of waters, effects of chlorination on them, and the exposure and effects on the environment and humans. Unfortunately, some of these sets of data, such as scientific evidence of the actual long-term human exposures and effects of halogenated organic micropollutants, are unlikely to ever be available because of restrictions on experimentation with humans.

Other needed sets of information, such as estimates of the risk of waterborne disease, involve major variability and uncertainty. Altogether, the present state of knowledge about the benefits and detriments of effluent chlorination is fair to poor, but available information can and should be evaluated and taken into consideration when examining effluent disinfection policy.

II. OBJECTIVES

The objective of this report is to present and assess information about the benefits and detriments of effluent chlorination. Specifically, the following questions relating to effluent chlorination will be addressed through evaluation of recent publications on the subject.

1. What is the pathogen content of wastewater treatment plant effluents ?
2. What is the impact of chlorination on the pathogen content of those effluents?
3. What are the benefits provided by effluent disinfection in preventing waterborne disease?
4. What chlorinated byproducts are formed during chlorination of secondary treatment plant effluent?
5. What are their potential adverse impacts on human health?
6. What are their demonstrated adverse impacts on human health?
7. What damages to aquatic life result from effluent chlorination?
8. What have been the policies in regulatory agencies with respect to effluent chlorination?
9. What are the recent trends in these regulatory policies?
10. What alternatives to chlorination of effluent are available, including modified chlorination, no disinfection, and alternative disinfectants?

III. PATHOGENS IN WASTEWATER TREATMENT PLANT EFFLUENTS

A. Pathogens in sewage

In areas served by sewerage systems, nearly all human wastes are discharged into the sewage which consequently contains any pathogenic organisms excreted. The extent and likelihood of new infections depends, among other factors, on the survival of the pathogens and subsequent exposure of humans to them. Disinfection of wastewater effluent is intended to reduce the chances of pathogens surviving into water to which humans will be exposed, thereby lowering the risk of transmitting infectious disease.

Large numbers of microorganisms are present in sewage, their concentrations and types varying considerably with time and between communities. Reported bacterial isolates and concentrations from a recent field study of several wastewater treatment plants are shown in Table 1 [Sorber,1980]. The presence of pathogenic microorganisms in sewage depends particularly on the disease rates in the contributing community. Most places in the United States currently have a very low incidence of infectious disease and therefore the sewage will usually receive few pathogens. When an enteric disease occurs in a community, that pathogen will be present in the sewage in quantities roughly proportional to the number of infected persons [Gerba, 1983].

Dangerous waterborne infectious diseases are rare but present in this country, and the associated pathogens are therefore present , though not widespread, in U.S. sewage. The

Table 1. Quantities of Viable Bacteria Measured at Three Wastewater Treatment Facilities.

bacteria type	concentration cfu/100 ml		
	Pleasanton(a)	Portland(b)	Chicago(b)
Citrobacter	$<5.0\times10^{4}$[c]	6.6×10^{3}	$<3.0\times10^{2}$[c]
Clostridium	$2.8\mathrm{X}10^{2}$	---------	1.5×10^{3}
Enterobacter	3.0×10^{6}	5.0×10^{4}	2.0×10^{4}
Escherichia	1.0×10^{6}[d]	6.7×10^{3}	$<3.0\times10^{2}$[c]
Klebsiella	6.0×10^{6}	3.7×10^{4}	1.0×10^{5}
Leptospira	4.6×10^{3}	---------	2.4×10^{3}
Mycobacterium	7.0×10^{4}	1.3×10^{6}	1.3×10^{5}
Providencia	1.0×10^{6}	$<3.3\times10^{3}$	$<3.0\times10^{2}$[c]
Serratia	$<5.0\times10^{4}$[c]	6.6×10^{3}	$<3.0\times10^{2}$[c]
Staphylococcus	3.0×10^{5}	3.3×10^{2}	2.0×10^{5}
Fecal coliform	1.0×10^{7}	5.3×10^{4}	1.0×10^{5}
Total coliform	1.1×10^{8}	1.8×10^{5}	3.7×10^{6}
Total plate count	5.8×10^{8}	4.8×10^{8}	8.2×10^{8}

a =ponded secondary effluent
b =aeration basin
c =none detected
cfu=colony forming units
[Sorber, 1980]

waterborne scourges such as typhoid and cholera that were major concerns for earlier sanitary engineering are not currently significant problems in the United States, but are, however, still present. Typhoid in the U.S. progressively decreased from 1900-1960 then leveled during the late 1960's to a current incidence of 0.2-0.3 cases reported per 100,000 general population per year; about half of these typhoid cases are acquired during travel outside the country [Hornick, 1983]. Rarer than typhoid in the Americas, cholera has been present once recently in the United States. Thirteen cases, all caused by the eltor strain, were identified in coastal Louisiana in 1978. At that time, the same strain of Vibrio cholerae was isolated from shellfish and crabs in several local coastal marshes [Carpenter, 1983]. It is important to see here that a policy including effluent disinfection as a public health protective measure should be based on the potential biological hazard of sewage-contaminated waters, not on just the hazard presented by current conditions.

The hazard of biologically contaminated water is routinely measured by surrogates (indicators). Discovery, identification, and enumeration of various actual pathogenic microorganisms is not commonly done because of insufficient techniques for some pathogens, expense of doing vast quantities of laboratory tests, and availability of better methods for evaluating the microbiological risk. Standard counts of indicator organisms which are assumed to be proportionally representative of overall microbiological hazard are used instead. The concepts of using indicator organisms have been much discussed [Cabelli, 1978, 1982a, 1983; Dudley,1976; Hendricks,1978; Pipes,1978].

There are problems with using bacterial indicators for other than routine screening. One result of the assumptions necessary for the use of indicator organisms is the addition of uncertainty to the assessment of the risk of infectious disease associated with wastewater effluents. Estimating the probability of an occurrence by sampling its frequency becomes difficult and less precise when that event is rare, as is the case with waterborne disease in the United States. The potential risk of a particular disease being transmitted by sewage would be underestimated if that risk is estimated on the basis of analysis of sewage which currently contained no pathogens of that type. For any particular disease, the sewage from a community where there is no actively infected person discharging to that system will not contain pathogens of that type. This is a source of uncertainty in estimates of risk regarding waterborne disease in the United States and the effects on that risk of various protective practices such as effluent chlorination.

The bacterial pathogens most commonly associated with sewage include species of Salmonella, Campylobacter, Shigella, and Vibrio (Table 2). Among the 1700 identified types of Salmonella are those responsible for typhoid and paratyphoid fevers. Some other Salmonella species cause gastroenteritis, as do Yersinia and Campylobacter. Fecal coliforms, present in great quantities, are predominantly non-pathogenic but some strains of E. coli have been found to cause severe diarrhea [Dupont, 1971].

Shigellosis is present in the U.S. at a reported incidence

Table 2. Pathogenic Organisms in Sewage.

Group	Pathogen	Disease caused
Bacteria	*Salmonella* (1700 types)	Typhoid, paratyphoid, salmonellosis
	Shigella (4 spp.)	Bacillary dysentery
	Enteropathogenic *E. coli*	Gastroenteritis
	Yersinia enterocolitica	Gastroenteritis
	Campylobacter jejuni	Gastroenteritis
	Vibrio cholerae	Cholera
	Leptospira	Leptospirosis
Protozoa	*Entamoeba histolytica*	Amebic dysentery, liver abcess, colonid ulceration
	Giardia lamblia	Diarrhea, malabsorption
	Balantidium coli	Mild diarrhea, colonic ulceration
Helminths	*Ascaris lumbricoides* (round worm)	Ascariasis
	Ancyclostoma duodenale (Hookworm)	Anemia
	Necator americanus (Hookworm)	Anemia
	Taenia saginata (Tapeworm)	Taeniasis
Viruses	*Hepatitis A virus*	Infectious hepatitis
	Coxsackie virus, Norwalk types, etc.	Gastroenteritis

[Gerba, 1983; Dienstag, 1976; Murphy, 1979] of 8-10 cases per 100,000 population per year [Center for Disease control (CDC), 1982]. Shigellosis is spread primarily by human contact, but indirect transmission has been shown in 25 foodborne or waterborne outbreaks documented in the U.S. between 1964-1968. Four of the twelve water related shigellosis outbreaks were associated with swimming in small contaminated freshwater lakes [CDC, 1982].

Of the common protozoa which may be found in wastewater, only three species are significant in the transmission of disease to humans: Endamoeba histolytica, Giardia lamblia, and Balantidium coli. Significant waterborne outbreaks of illnesses due to E. histolytica and B. coli have not been reported recently in the U.S. [Gerba, 1983], but it is estimated that 1-3% of the United States population is infected with these organisms [Juniper, 1983a]. The possibility of infection with these protozoans from domestic sewage in the United States is, therefore, significant.

There has been a significant recent rise in the United States of the number of reported waterborne outbreaks of Giardia enteritis (giardiasis), which has been reported for about 10,000 cases in the U.S. since 1971 [CDC, 1982; Craun, 1979]. Giardia lamblia, a protozoan, is usually fecally-orally transmitted: by contamination of food, hand to mouth, or via drinking water where the cysts are resistant to common doses of chlorine (<3.0 mg/l) [Dykes, 1980]. Sewage may be a major carrier of the organisms since levels of Giardia cysts in feces from infected persons can be as high as 10^6 cysts per gram. Though it is primarily spread by human-human fecal oral vectors, giardiasis has been found to not always be transmitted from humans to humans only. Evidence has indicated that beavers in an upland watershed may serve as a reservoir for Giardia [Dykes, 1980]. Even though such an outbreak of giardiasis originating from a non-human source would not be prevented by reducing pathogen concentrations at a different source (such as sewage effluent), the benefit of effluent disinfection in reducing the risk of wastewater transmission of giardiasis remains valid.

Helminth parasites are present in the United States population, but the reported incidence of disease due to these agents has been low for the last few decades [Gerba, 1983].

Enteric viruses (those fecally excreted by and pathogenic to humans) are also excreted in widely varying amounts in different places but generally in lesser numbers than pathogenic bacteria [Irving, 1981; Hanson, 1973]. The numbers of virus that are measured depend, in addition to the wastewater source, on the detection technique employed [Sorber, 1980]. More than one hundred strains have been isolated from sewage and most are in six categories: polio, hepatitis, coxsackie, adeno, echo, and reoviruses. Mean total virus concentrations isolated from raw sewage run the range from 150 infectious units per liter (IU/l) to 15,000 IU/l, with 90-100% of samples positive [Irving, 1981]. Enteric viruses isolated from samples taken from a sewage-polluted river have been identified generally as the same types as those found in sewage effluents [Metcalf, 1968]. Only the Hepatitis-A virus (HAV) has been clearly shown to cause waterborne viral diseases [Dienstag, 1976; Kruse, 1971; Mason, 1962].

Recently, however, Norwalk type viruses have been implicated as the cause of one large gastroenteritis outbreak [Murphy, 1979]. With the current United States incidence of Hepatitis A at about 30,000 clinical cases per year [CDC, 1982], and an estimated ratio of inapparent infections to clinical cases of 10:1 [Hanson, 1973], there could easily be 300,000 people in the U.S. each year infected with and excreting Hepatitis A virus. Each infected person excretes 10,000 to 100,000 infectious doses per gram of feces [Metcalf & Eddy, 1972]. Even after dilution, raw sewage HAV quantities may be very high if infected persons are discharging to that sanitary system.

B. Removal by conventional treatment

Sewage treatment achieves both a decrease in numbers of bacteria and, of major importance, a change in the kinds present. Microorganisms pathogenic within the human body generally do not multiply in the wastewater environment. Although pathogenic bacteria are not absolutely eliminated by treatment, the effect is to greatly reduce their numbers, replacing them with saprophytic varieties [Carlson, 1943]. Removals of various pathogens that are accomplished by sedimentation, trickling filters and activated sludge are shown in Table 3.

Helminth ova settle readily and are removed to the primary sludge [Cram, 1943; Kabler, 1959]. Protozoan cysts, though, are not so extensively removed during primary sewage treatment. Cysts of *E. histolytica* have been shown to pass through primary settling and trickling filter or activated sludge processes irrespective of those processes' BOD removal efficiency, but are significantly removed in secondary clarification or sand filtration [Cram, 1943].

Table 3. Removal of Various Organisms by Conventional Wastewater Treatment.

Treatment	Agent	Removal (%)	Test System
Plain Sedimentation	Viruses:		
	Polio 1	0	bench
		to 69	plant
	Polio 1,2,3	0-12	plant
	Enterovirus	10	plant
	Adenovirus	30	plant
	Reovirus	5	plant
	Parasites:		
	Beef tapeworm ova	50	bench
	E. histolytica cysts	0 to *	plant
	Bacteria:		
	Mycobacterium tuberculosis	50	plant
	Coliform	27-96	bench
Trickling filters	Viruses:		
	Coxsackie A9	94	bench
	Echovirus 12	83	bench
	Polio 1	85	bench
	Mixed (natural)	* to 69	plant
	Parasites:		
	Beef tapeworm ova	18-30	bench
	Ascaris ova	70-76	plant
	E. histolytica cysts	*	plant
		90-99.9	bench
	Bacteria:		
	Mycobacterium tuberculosis	45	plant
	S. typhosa	72	plant
	Coliform	98	plant
	Ps. aeruginosa	+74	plant
	Cl. perfringens	92	plant
Activated sludge	Viruses:		
	Coxsackie A9	96-99	bench
	Polio 1	79-94	bench
	Mixed (native)	53-71	plant
	Polio 1, 2, 3	76-90	plant
	Enterovirus	92	plant
	Adenovirus	81	plant
	Reovirus	27	plant
	Parasites:		
	Beef tapeworm ova	0	bench
	Ascaris ova		
	E histolytica cysts	*	plant
	Bacteria:		
	Salmonella typhosa	86-99	bench
	Vibrio cholera	96-100	bench
	Mycobacterium tuberculosis	90+	bench
	Coliform	97	bench
	Fecal Streptococci	96	bench

*Incomplete removal

[Irving, 1981; Kabler, 1959; Sorber, 1980]

Many processes, both in the treatment plant and in the receiving waters, accomplish reduction of pathogen concentration. The pathogen concentrations expected in the effluent are important for evaluating the disinfection process, and the overall pathogen reduction is important for evaluating the risk of infectious waterborne disease. For certain pathogens, though, such as Mycobacteria, disinfection appears to be the only reliable process for their removal [Heukelekian, 1956].

Effluent disinfection is a process solely intended for reducing the concentration of viable microrganisms, but many other natural and artificial processes act to affect the microbiological character of discharged wastewater. All wastewater treatment processes which reduce the concentration of pathogenic micro-organisms contribute to the overall reduction of the infectious disease hazard. The removal of various organisms by conventional treatment processes precedes effluent disinfection. Removal or inactivation of pathogens that occurs after discharge, but prior to human exposure to the water, affects the need for disinfection of the effluent.

There are many conditions in natural surface waters which help to inactivate microorganisms. One of these is the effect of sunlight. Light at the wavelength of sunlight has been shown to increase the die-off rate for viruses and E. coli [Kapuscinski, 1983]. Inactivation rates under light for three types of bacteriophage and for E. Coli were found to be one order of magnitude faster than for organisms kept in the dark. Of particular interest was that the die-off of E. Coli under conditions of ambient sunlight was greater than that of bacteriophage virus under identical conditions, indicating that the former would not be a valid surrogate for measuring the presence of the latter in open waters.

Virus inactivation (loss of infectivity) in natural waters is exponential and appears to be influenced primarily by temperature [O'Brien, 1977]. The influence of temperature is such that the 1-log inactivation of coxsackie and poilioviruses that occured in Rio Grande water at 25 ^{o}C over 19-25 hours took more than twice as long when chilled to 5 ^{o}C. In the normal river warmth of 23-27 ^{o}C, inactivation of 2 logs occurred in 2-3 days and 3 logs in 3-4 days [O'Brien, 1977]. In the waters of streams and rivers, the significant inactivation of viruses usually takes several days and is usually slower than the inactivation occuring in saline waters. In natural estuarine water a mean 3 log reduction of various initial virus titers occurred in 2-3 days. Comparable inactivation of viruses in freshwater required from three to more than fourteen days depending on the type of virus [Hurst, 1980].

Competition and predation by other microorganisms present in the treatment or receiving waters is a factor contributing to reduced pathogen survival. Experiments have shown greater decreases in the numbers of foreign bacteria when the full natural microbiotic community was present. When indigenous protozoans were filtered out of estuarine water, die-off of coliform populations became negligible compared to the 3 log reduction in 5 days in unfiltered water [Enzinger,1976]. Another investigation also found that inhibition of protozoans allowed E.

coli to maintain populations 3 logs greater than in natural estuarine water, or alternatively, required 4 days longer for die-off to reach the same levels [McCambridge]. In the latter study with natural complete estuarine water, predacious protozoa exerted their major influence on E. coli destruction during the first two days. The former study established protozoan-positive sample by seeding one milliliter of fresh bay water into 50 mls of sterilized bay water and observed greater protozoan predation between days two and four. It has been noted but not investigated that the predacious destruction of foreign bacteria is apparently greater in marine waters than in freshwater systems [Enzinger, 1976; McCambridge, 1980].

Removal of pathogens by conventional primary and secondary sewage treatment processes is not consistently sufficient enough to accomplish the task of disinfection. Processes such as activated sludge or trickling filters cannot be relied upon to achieve more than a 1 to 2 log reduction in pathogen concentrations. Considering the high numbers of pathogens which can be present in sewage, this is a reduction which, by itself, provides insufficient assurance of protection from the possibility of infection and disease to persons who are exposed to effluent downstream.

IV. WASTEWATER CHLORINATION AND ITS EFFECTS ON PATHOGENS AND OTHER MICROORGANISMS

Chlorination has been and continues to be the preferred method for disinfection of wastewater effluents. According to a 1980 EPA survey, 62% of the total municipal wastewater flow in the United States is chlorinated. It was found that the practice of effluent disinfection is increasing in the U.S. and that at least 90% of the time the preferred method is chlorination [Maxted 1983].

Primary among the several reasons why chlorine is the predominant method for wastewater disinfection is that chlorine is the most cost-efficient method of reliably destroying microorganisms in water. Also, operation of chlorination is generally simpler than other disinfection methods, and there has been extensive experience with it. Successful experience with chlorination has also shown that its effectiveness can be easily approximated by measuring the contact time and residual concentration of chlorine in the effluent, rather than by bioassay [White 1972].

Use of chlorine compounds in the treatment of sewage preceded their use in potable water. Disinfection with chlorine for the purpose of controlling disease transmission was done as early as 1879 when, in England, calcium hypochlorite was applied to typhoid feces before discharge to a sewer [American Public Health association (APHA), 1934]. In the U.S. from 1890-1910 several attempts at sewage purification with hypochlorite solutions, generated onsite by electrolysis of brine, were done. Several studies on disinfection of raw sewage and trickling filter effluent by application of hypochlorite powders and solutions were done in the U.S. and Europe during this period. The practice of wastewater chlorination in the United States grew concurrently with that of chlorination of water supplies, beginning in about 1910 [Race,1918; Thoman,1958]. Developments in chemical manufacture during WWI made available cheap elemental chlorine (Cl_2 gas) which has since been the economically preferable form for large scale water or wastewater disinfection [APHA,1934; White 1972, 1978].

Chlorine has been used in sewage treatment plants for many purposes. Chlorination of influent, or at other points, for odor control, has been done. A strong oxidant, chlorine will help remove reduced species such as sulfides or ammonia. Chlorine has been applied to effluents to reduce or delay BOD in the receiving water [APHA, 1934]. This report is concerned only with effluent chlorination for the purpose of disinfection.

In the U.S. each year about 10.5 million tons of chlorine are manufactured, most of which is used in chemical manufacture and pulp/paper industries. An estimated 3-4% of the total generated, or 630-840 million pounds, is used for sanitary purposes -- including drinking water and wastewater treatment, swimming pools, household use, cooling water biofouling control, and food processing water [White, 1972]. Though estimates vary, the total amount of chlorine that is used for disinfection and then released to the environment is large. For example, according to Maryland statistics, the use of chlorine for disinfection in that state could contribute to the Chesapeake Bay, assuming no degradation, 27 million lbs./year of chlorine via municipal wastewater treatment plants, and 2.2 million lbs./year from power plant cooling water anti-fouling chlorination [Kopperman, 1978]. Roughly 1% of manufactured chlorine, or about 200 million pounds per year, is used for wastewater chlorination [Jolley, 1975].

The use of chlorine in sewage treatment plants in the United States has accelerated during the middle part of this century. From 1910 when 22 plants used chlorine in treating the wastes from an estimated 0.12% of the nation's population, the practice has grown faster than sewage treatment capacity [Laubusch, 1958; Maxted, 1983; Thoman, 1958]. The steady increase in the use of chlorine from 4% of surveyed plants in 1910, grew to 18% in 1934, passed 49% of all plants in 1957, and now is practiced at over 60% of all U.S. wastewater treatment facilities [Laubusch, 1958; Maxted, 1983; Thoman, 1958].

According to a survey by the Water Pollution Control Federation done in 1979 of over 2500 municipal wastewater treatment plants, of the 740 responding, 80% disinfect their effluent, 20% do not [WPCF 1980]. Of the nearly 600 plants which practiced disinfection, nearly all did so by chlorination (1 plant reported using chlorine dioxide), and of these, less than 5% followed with any dechlorination process. The median dose range for chlorine was 3-6 mg/l, at 58% of the plants [WPCF, 1980].

A. Chemistry of chlorine in wastewater

Chlorine disinfection efficiency and the effect and fate of discharged chlorine residuals depend on many factors. Primary among these is the chemistry of chlorine in water.

Chlorine gas added to water rapidly hydrolyzes to form hypochlorous acid (HOCl) and hydrochloric acid.

$$Cl_2 + H_2O \rightarrow HOCl + H^+ + Cl^-$$

Half of the chlorine applied becomes the non-disinfectant chloride ion (Cl^-). Hypochlorous acid exists in equilibrium with hypochlorite ions (OCl^-).

$$HOCl \Longleftrightarrow OCl^- + H^+$$

The equilibrium relative quantities of HOCl and OCl^- depend on pH, with HOCl predominating at low pH, equal amounts at pH = pKa =7.5, and OCl^- predominating at higher pH. The sum amount of HOCl and OCl^- is called free residual chlorine, and can be put into water by addition of chlorine gas or hypochlorite compounds such as NaOCl.

Free chlorine in wastewater will first react with any easily oxidized species that are present, such as sulfide, nitrite, and other reduced compounds. After that demand is satisfied, combined available chlorine (CAC) is formed as chlorine combines with ammonia in a weight ratio of close to 5:1 to form monochloramine. Municipal wastewaters receiving secondary treatment contain significant amounts of ammonia (NH_4^+ or NH_3, pka=9.3). Even wastewater which has undergone nitrification of most of the ammonia to nitrate will still contain some ammonia [Snoeyink, 1974].

$$\begin{matrix} NH_4^+ & & HOCl & & & & H_3O^+ \text{ or} \\ \text{or} & + & \text{or} & \longrightarrow & NH_2Cl & + & H_2O \text{ or} \\ NH_3 & & OCl^- & & & & OH^- \end{matrix}$$

As more chlorine is applied to convert the ammonia nitrogen (5 mg/l Cl_2 for each 1 mg/l NH_3-N), dichloramine begins to form and decompose, resulting in a decline of available combined chlorine until free residual chlorine begins to be established at the "breakpoint" (Figure 1).

$$NH_2Cl + HOCl \longrightarrow NHCl_2 + H_2O$$

$$2H_2O + 2NHCl_2 \longrightarrow N_2 + 4Cl^- + 2H_3O^+$$

The rate of formation of monochloramine varies with pH because the speciation of the reactants varies with pH. Monochloramine formation is very fast in the pH range 7.5-9.3, in which the reactants are predominantly hypochlorous acid (HOCl) and ammonium (NH_4^+) [Lietzke, 1978]. At pH>7, essentially only monochloramine is stably produced; at pH below 7 dichloramine will become increasingly present; and nitrogen trichloride (NCl_3) may be significant at pH<4. Overall, at wastewater of typical pH, the oxidation of ammonia nitrogen to monochloramine and dichloramine will consume about 10 times as much free chlorine by weight before allowing a free available residual [Snoeyink, 1974]. Some

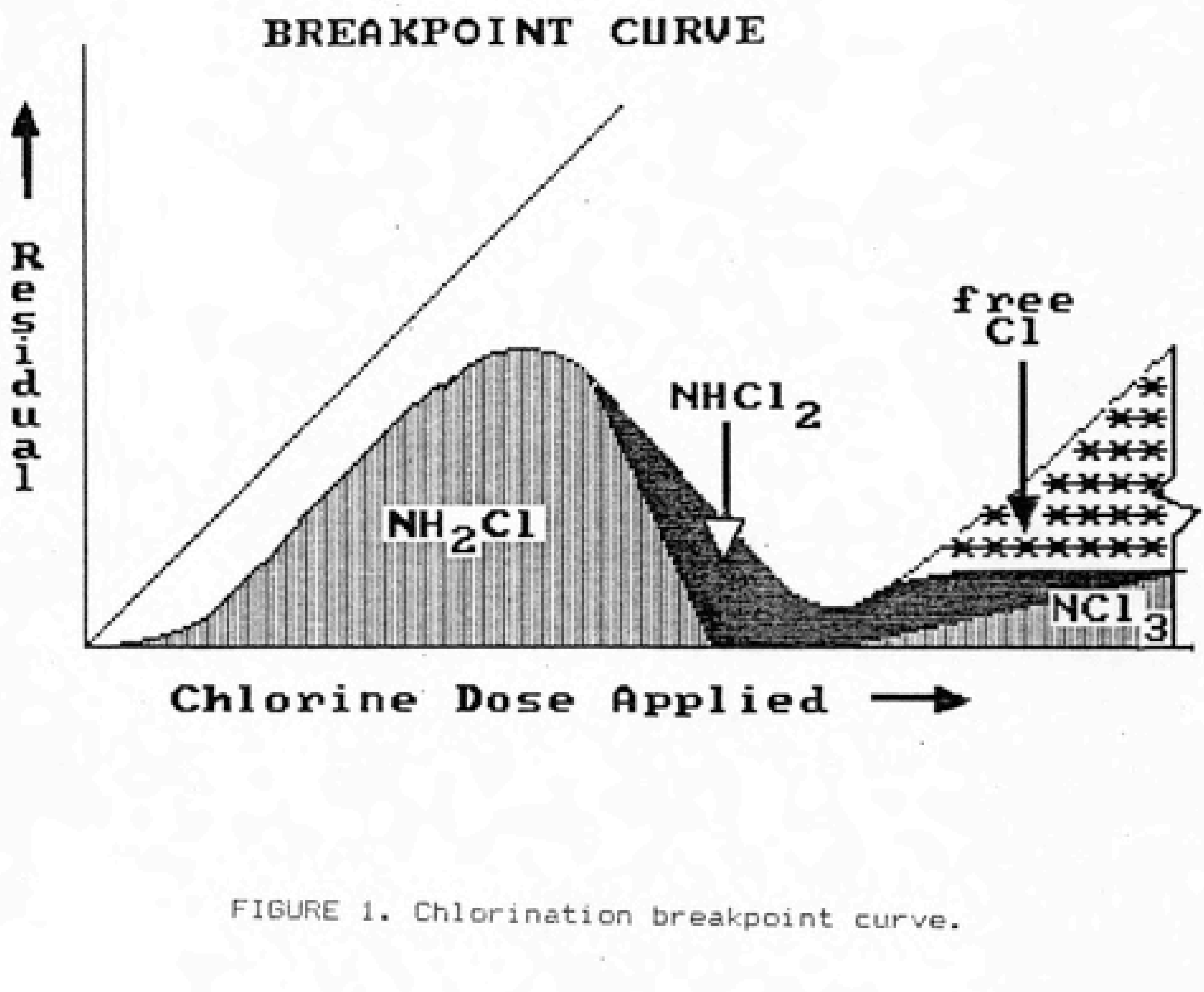

FIGURE 1. Chlorination breakpoint curve.

free chlorine is consumed in oxidizing N_2 - nitrogen to nitrate, thereby delaying the breakpoint slightly further [Saunier, 1979].

$$N_2 + 6HOCl \Longrightarrow 2NO_3^- + 6Cl^- + 6H^+$$

Most wastewater chlorine doses are far below that needed to reach the breakpoint and create a free residual. The actual distribution of chlorine forms in the effluent will vary, depending on the dose, pH, and which compounds present the demand. In nearly all chlorinated secondary effluents the predominant residuals form of chlorine is monochloramine.

In addition to the production of several inorganic species, various reactions may occur with organic material in the water that form chlorinated organic compounds. These may include trihalomethanes, chlorophenols, chlorinated amino acids, and organic chloramines. The formation and significance of chlorinated organic compounds to human health will be addressed in Section VI.

Chlorinated effluents which are discharged to saline waters involve some additional reactions. Brackish estuarine waters are diluted seawater, and therefore contain bromide (Br^-), a halogen not found in most fresh waters. In full strength seawater, there is 65-70 mg/l bromide. In estuarine water that is only 1% seawater, for example, there will be about 0.7 mg/l bromide concentration, which is of the same order or greater than typical discharged chlorine residuals.

Because chlorine is a stronger oxidizing agent than bromine, hypochlorite will react with bromide to produce hypobromite, reducing the chlorine to chloride.

$$Br^- + HOCl \longrightarrow HOBr + Cl^-$$

Though monochloramine is less reactive than hypochlorite, there is evidence that bromamines are formed from reaction of bromide with monochloramine and/or hypobromite with ammonia. Similar to chlorine, a set of bromamines, bromides, and brominated organics may form [Johnson, 1975; Scott, 1983].

B. Effects of effluent chlorination on microorganisms

Chlorine's effect on microorganisms depends on the nature, distribution, and concentration of the organisms and of the chlorine, on the pH, temperature, and other characteristics of the water, and on mixing and time of contact. The manageable variables are 1) the nature and concentration of the disinfectant, 2) mixing of water, and 3) assured time of contact.

Under the ideal conditions of:

1. no interfering substances in the water
2. disinfectant chemical composition doesn't change
3. disinfectant concentration doesn't change
4. disinfectant and target microorganisms are uniformly dispersed;

the rate of disinfection can be modeled as a function of contact time, disinfectant concentration, and temperature.

As the time of contact between disinfectant and organisms is longer, more destruction can occur. This important factor in the efficency of disinfection is described as a first order rate equation known as Chick's Law. Applying chemical reaction principles to the study of disinfection, Chick found that, with excess disinfectant, the death rate of anthrax cells, dN/dt, was proportional to N, the number remaining [Chick, 1908].

$$-dN/dt = KN$$

where: N=number of cells

t=time

K=rate constant

Or, for the period t and having begun with N_o cells,

$$N/N_o = e^{-kt}$$

Chemical disinfection proceeds more rapidly at warmer temperature. This relation basically follows the Arrhenius relationship, and was noticed by Chick. To achieve equal extent of disinfection at lower temperature, T_1 (absolute), requires a longer contact time, t_1.

$$\log \frac{t_1}{t_2} = \frac{T_2 - T_1}{T_2 T_1}$$

Chick also noticed variations in the first order kinetics. In one set of experiments this was attributed to varying susceptibility within a species, where she found that younger B. paratyphosa were more resistant than older cells. For lower disinfectant doses, a logarithmic relation between disinfection rate and disinfectant concentration, c, was found. Watson later used Chick's data to define a second order expression [Watson, 1908]:

$$N \log c + \log t = K = \text{rate constant}$$

Or, in the exponential form [Trussell, 1977]:

$$N/N_o = 10^{-kct}$$

It must be kept in mind, however, that the ideal conditions are not met, especially in wastewater disinfection. Particulates, especially aggregates, shield microorganisms from exposure to disinfectant [Culp, 1978]. Ammonia and other chlorine-demanding materials react with chlorine to reduce available disinfectant concentrations and/or convert it to less effective forms. In addition to the intraspecies variation in organism susceptibility pointed out by Chick, the various types of target pathogens are quite different in their resistance to disinfection. Certain organisms such as mycobacteria, amebic cysts, and some enteric viruses, have been found to be significantly more resistant to chlorine disinfection than others, such as coliform bacteria [Burns,1967; Dudley,1976; Hendricks, 1978].

Thorough mixing of disinfectant with wastewater is of great importance because the process seeks reductions extending over several orders of magnitude. Experiments have shown that thorough initial mixing, rapid or slow, of chlorine with wastewater gave consistently better disinfection efficiency of coliforms (MPN) than with no initial mixing [Eliassen, 1948]. A rapid initial mix was also found to be necessary for efficient virus inactivation in wastewater because the viral disinfection was accomplished in the first few moments when the added chlorine

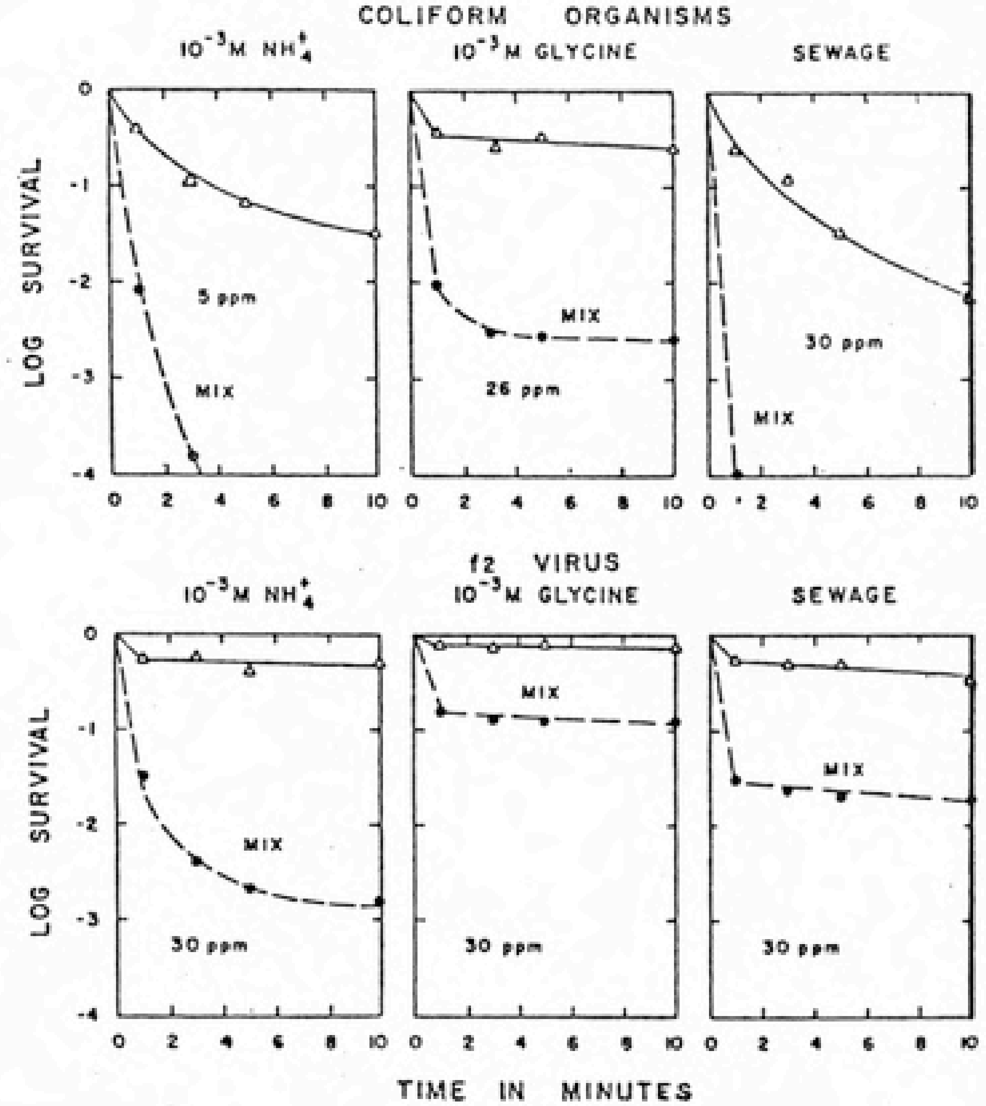

FIGURE 2. Bactericidal and viricidal effects of rapid mixing [Krusé].

was still in the more effective free form [Kruse, 1971]. (Figure 2). The importance of uniform dispersion of disinfectant is illustrated by a hypothesized contact chamber in which 2% of the flow is short-circuiting enough to only receive a 50% reduction in microorganism concentration The maximum overall disinfection that could then be achieved is a two log (99%) reduction in microorganism concentration - generally insufficient for the numbers of microorganisms in sewage effluents.

Inefficiency of chlorination facilities seems to be a widespread problem [Sepp, 1981; Trussell, 1977], and results in insufficient disinfection or overapplication of chlorine since the operators's primary control is the chemical dose. Sepp's study of six California treatment plants whose normally applied doses were from 6 to 47 mg/l chlorine showed that an optimized pilot plant at each site improved disinfection efficiency. The process improvements consisted of rapid mix, direct automatic control of dosage by residual monitoring, and plug flow contact chamber design. At all plants the disinfection process was improved, with up to 50% less chlorine used [Sepp, 1981].

Since the physical, chemical, and biological character of wastewaters are so varied, definitive conclusions regarding chlorine disinfection effectiveness in wastewater are not possible. There are, however, reviews of the factors influencing disinfection, and many experiments with chlorine and chloramine disinfection, mostly with clean water [Brodtman, 1979; Mancini, 1978; National Research Council (NRC), 1977; Olivieri, 1983]. Data for disinfection in demand-free systems with controlled chlorine speciation indicates that the relative microbial inactivation efficiencies (time * concentration product for a given viability reduction) of hypochlorous acid, hypochlorite ion, and monochloramine are on the order of 1, 10 and 1000 [NRC, 1977; Olivieri, 1983; White, 1972].

Sewage disinfection is different from potable water chlorination, though. As discussed above, many influential factors vary over wide ranges, some with effects that make several orders of magnitude difference in the numbers of microorganisms surviving, such as the effect of rapid mixing.

Effluent chlorination can and does routinely provide excellent disinfection. In Figure 3 data are shown for chlorination of water containing ammonia at concentrations similar to those found in secondary effluent; the disinfection is a four log reduction. Note, however, that the same dose in a glycine solution had an inconsequential effect. Figure 4 shows the chlorination of the same two solutions, inoculated this time with virus and dosed with 20 ppm chlorine. The greater resistance to chlorine of viruses compared to coliforms is evident. There is no epidemiological indication though, that United States wastewater disinfection practices (usually based on coliform indicators) allows significant risk of waterborne viral disease [Kruse, 1971].

Practical and experimental chlorination of secondary effluents has demonstrated the process' efficiency at meeting effluent coliform standards. This efficiency at various typical assured mean contact times (30,15, and 5 minutes) and rapidly mixed doses (10,5, and 2 mg/l) is illustrated in Figure 5. Each of these results is the mean of eight experiments run on five

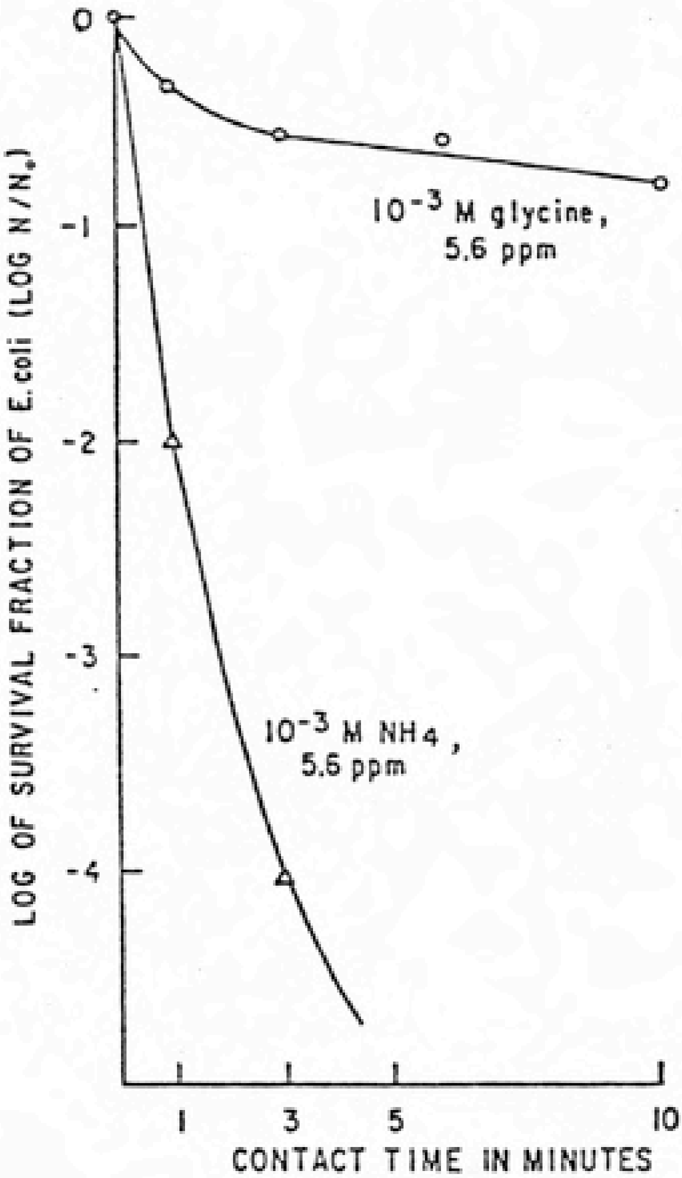

FIGURE 3. Chlorination of waters containing ammonia and glycine and their effect on the rate of disinfection for coliforms, dose 5.6 ppm [Olivieri].

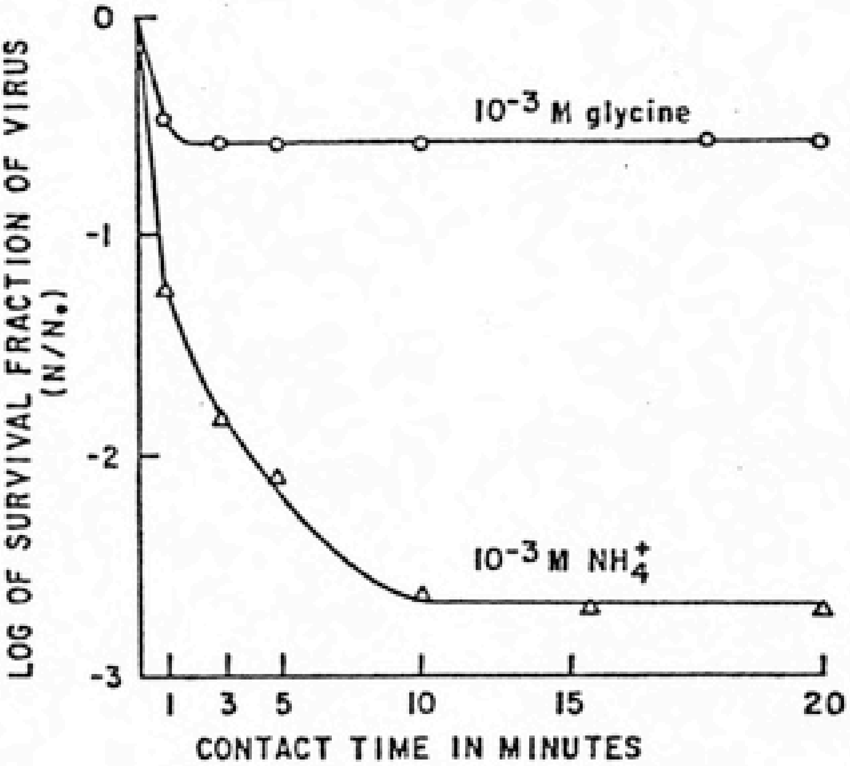

FIGURE 4. Chlorination of waters containing ammonia and glycine and their effect on the rate of disinfection for viruses, dose 20 ppm [Olivieri].

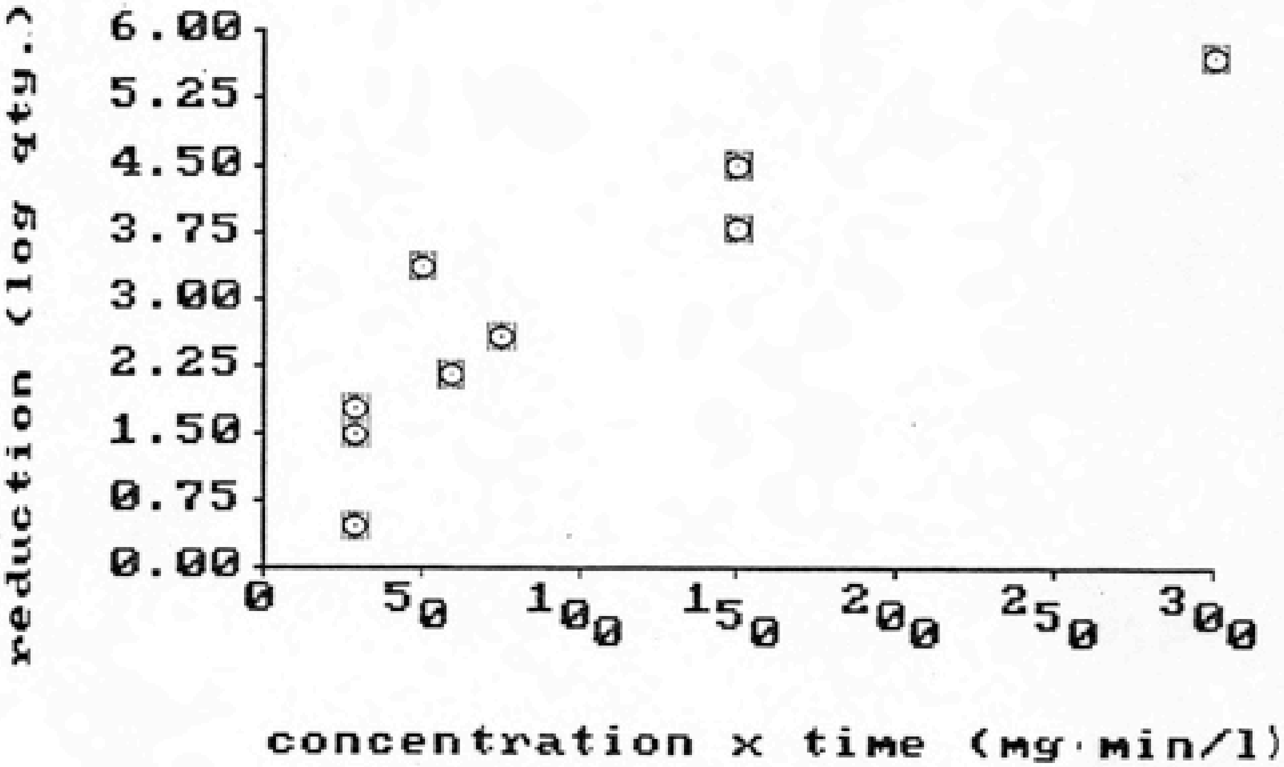

FIGURE 5. Measured reduction in coliform concentrations for various chlorine doses and contact times [Aieta].

different days over a two week period during which the wastewater characteristics were rather stable: T= 24 ^{o}C, COD 23-29 mg/l, pH 6.7-7.4, and NH_3-N 27-34 mg/l [Aieta, 1980].

Nitrified effluents have been shown to require more chlorine to achieve the same level of disinfection than typical secondary effluents containing moderate amounts of ammonia [Sepp, 1981; Dhaliwhal, 1983; Gasser, 1984; White, 1981]. For example, filtered nitrified wastewater at San Jose, California, was found to require application of 17 mg/l chlorine to reduce coliforms to a level of 2.2/100 ml (MPN). This 17 mg/l applied chlorine, after 49 minutes contact, left 9 mg/l residual chlorine, about half free and half combined. This wastewater had trace amounts of ammonia (<0.1 mg/l), and 1.3-2.3 mg/l organic nitrogen. However, when 2 mg/l ammonia was added prior to chlorination, an application of only 12 mg/l chlorine was sufficient to achieve the required level of 2.2 coliforms/100 ml; the resultant 7 mg/l residual was about 3/4 monochloramine and 1/4 dichloramine [White, 1980]. White attributed this phenomenon to the chloramines having greater disinfection efficiency than the chlorine species formed with low NH_3-N levels.

Regrowth of bacteria populations after being damaged by chlorination has been known for over 70 years [Race, 1918]. This recovery has been investigated in the laboratory and found to be helped by growth in hospitable media [Camper, 1979]. Observed regrowth of coliforms in wastewater effluents in the field and laboratory has been shown to be inversely proportional to residual chlorine and the numbers of coliforms [Graham, 1983;

Hulka, 1973; Silvey, 1974; Shuval, 1973]. In Shuval's study, fecal coliforms in discharged effluents generally did not exhibit regrowth as much as total coliforms. Regrowth did not always occur, and when it did, it never exceeded 2 logs of population. These observations saw an average 5 log reduction of coliforms due to chlorination disinfection followd by a mean regrowth, after 3 days in a storage reservoir, of 1 log [Shuval, 1973].

Because residual chlorine is a factor holding down the regrowth of coliforms, absence of any residual due to dechlorination allows more regrowth. Indicated bacteria aftergrowth following dechlorination is shown in Figure 6 [Chen, 1981]. As with the aftergrowth observed in effluents discharged to rivers and ponds, this aftergrowth following dechlorination recovers about one third of the logarithmic population reduction accomplished by disinfection. Aftergrowth of indicator bacteria can occur to even greater extents, but this does not imply that significant regrowth of populations of pathogens occurs outside of hosts [Shuval, 1973].

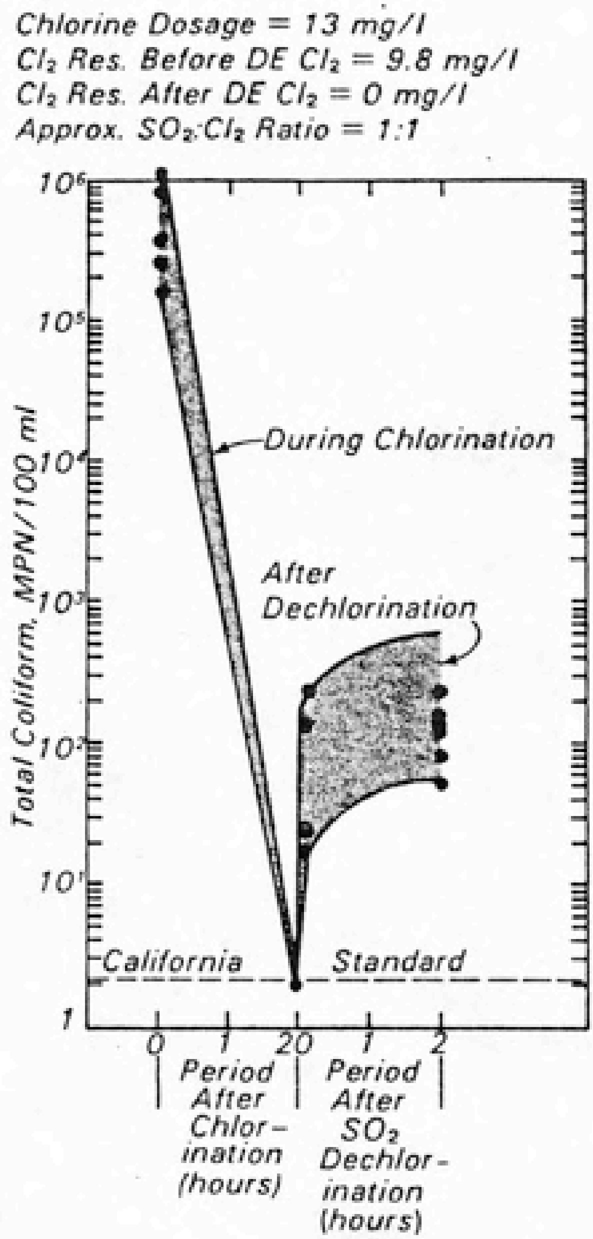

FIGURE 6. Measured reduction and regrowth of bacterial concentrations during chlorination and dechlorination [Chen].

V. THE NEED FOR AND BENEFITS OF PATHOGEN REDUCTION

Disinfection of wastewater effluents serves to protect public health. There are other purposes and effects of chlorination of wastewater effluents, such as discouraging odor or reducing the effluent BOD, but this report is concerned with chlorination of effluents for disinfection.

Maladies that are transmitted through wastewater and water systems can be caused by infectious microorganisms or by other contaminants in water. This section addresses the hazard of infectious disease and factors of exposure to its agents -- pathogenic organisms.

Where either the pathogenic hazard or the likelihood of exposure is low, the direct protective effect of disinfection is of reduced importance. As shown in Chapter III, pathogens and other microorganisms are physically removed from wastewater in varying degrees by most conventional treatment processes. Pathogens also die away or are destroyed in significant numbers during treatment and after discharge in natural waters. The specific goal of the unit process, wastewater effluent disinfection, is to kill any pathogens in the wastewater before release to the environment and, thus, to reduce the risk of transmission of infectious waterborne disease. Methods of disinfection which successfully kill pathogens do not distinguish them from non-pathogenic microorganisms and so destroy, in varying efficiencies, any microorganisms present in the water. The accomplishment of the task of disinfection is usually evaluated by measuring the concentrations of viable coliform bacteria, either total or the fecal variety, that are in the wastewater or receiving water.

Risk of human exposure to pathogens released in wastewater effluents occurs during subsequent intake or contact with receiving waters. Diseases associated with contact with contaminated open waters (water-contact disease) are often categorized separately from those associated with the ingestion of contaminated water (water-borne disease) [McJunkin, 1982]. This division serves to consider separately the risks of various modes of exposure and to plan effective interventions for breaking the disease transmission cycle [McJunkin, 1982]. For consideration of the health effects of effluent chlorination, public exposure to wastewater effluents can be divided into two categories as follows: 1) occupational or recreational contact and 2) the consumption of seafood taken from contaminated waters. Both depend upon that water quality for biological safety. Subsequent removal of water from the receiving waters, however, includes an opportunity for further treatment, and this opportunity must be considered together with disposal treatment before assessing the infectious hazard of wastewater effluents.

A. Hazard of water related infections

Intake of sufficient quantity of viable pathogens to incur infection depends on the amount of contaminated water ingested, the concentration of viable pathogens in that water, and the number of that type of pathogen which constitute the infective dose.

Pathogen concentration is the one of the three infection risk factors mentioned above which is under the purview of water quality management. An individual's exposure via consumption or contact and that person's susceptibility to infection by pathogens passing from wastewater systems are both factors that are not within the control of those wastewater systems.

This is not to imply that destruction of pathogens is the only or best way to combat disease. For instance, people's susceptibility to infection and illness can be altered by means of vaccinations. For typhoid fever, vaccination effectiveness has been known for over 80 years. The effect of acquired immunity is seen in that in the areas of the world where typhoid is today endemic the highest incidence is in children. Adults in non-endemic areas such as the United States, however, are also less likely to have had subclinical infection and acquired immunity; and, therefore, the population is potentially more susceptible in such areas [Hornick, 1983].

Infective doses vary for the different agents of disease and among exposed individuals. The impact of a chemical poison is a function of type, time, and concentration of exposure. For some infectious agents, however, one viable organism may be sufficient to establish infection [Koprowski, 1955; Rentdorff, 1954]. Other infectious maladies seem to get established in normal people only upon ingestion of massive numbers of the pathogen. The virulence of a pathogen is a widely varying probability of its survival through the unfavorable conditions of the gastric tract and hostile immune defenses.

Exposure to a pathogen may result in establishing an infection, and an infection may cause illness. An individual's defenses against microbial infection present a large but variable probability of destroying a microorganism and preventing its multiplication within the body. A microorganism may multiply within the body without producing overt or debilitating symptoms. Such inapparent infections can be important in the further transmission of infections. Disease results when the infection causes observable abnormality. The quantity of a type of microorganism that presents sufficient likelihood of establishing propagation is called the infective dose.

The establishment of an infection in a person can be inferred by measuring greater quantities of microorganism in the tissue or excretions than were originally inoculated. For enteric infections this would be shown by more of the infectious agent being present in the stool than was ingested. Serological measurement of the host producing antibodies against the agent also indicates infection.

Since the establishment of infection is not necessarily the same event as the occurence of disease, the dose of infectious agent sufficiently likely to cause disease is termed the pathogenic dose and can be quite different from the infectious dose. For some waterborne disease, a general ratio of one percent of infections resulting in observable disease has been employed [Pipes, 1978].

Pathogenesis, the progression of an infection to a disease, is not clearly dose-dependent, but the prerequisite factor of exposure leading to infection does carry a probability that is greater with increased dosage [Dupont, 1971, 1972; Hornick, 1970;

Katz, 1967; Koprowski, 1955; Rentdorff, 1954].

Experimentation on prisoner volunteers isolated and fed Endamaeba coli cysts or Giardia lamblia cysts in various doses via gelatin capsules or drinking water has implied low infective doses in the range of 1-10 cysts for those organisms [Rentdorff, 1954]. Results are shown in Tables 4 and 5.

Certain strains of enteropathogenic Escherichia coli have been tested in prisoner volunteers and found to cause severe diarrhea in a majority of men who ingested doses of 10^{6}-10^{8} bacteria [Dupont, 1971].

Experiments with healthy adult Americans who developed fever after being dosed with viable Salmonella typhosa have indicated that the pathogenic dose for typhoid fever is in the range of 10^{7}-10^{8} cells [Hornick, 1970]. See Table 6.

While the human response to typhoid appears to be dose related, the response to Shigella ingestion appears to be less directly a function of quantity of cells ingested [Dupont, 1972]. See Table 7.

The only human experimental studies of infection from ingested viruses is with attenuated vaccine polioviruses, but a rough indication may be inferred of possible infectious dose for other types of viruses. Adults fed attenuated poliovirus in capsules were found to be susceptible to infection at doses as small as 2 plaque forming units (PFU) [Koprowski, 1955]. (The quantity of virus expressed as plaque forming units (PFU) applies to tissue culture and is not directly the quantity of viruses, since viruses agglomerate or adsorb onto particles.) This positive dose-response relation is shown in Table 8. A statistically based

Table 4. Experimental Results of _Endamoeba coli_ Cysts Ingested in Capsule or Drinking Water and Consequent Infection Rate.

Approx. quantity of cysts	Infection Rate	Percentage
0	0/15	0
1	1/8	12.5
10	3/10	30
100	2/4	50
1,000	0/2	0
10,000	2/2	100
All doses	8/26	31.1

[Rentdorff, 1954]

Table 5. Experimental Results of _Giardia lamblia_ Cysts Ingested in Capsules or Drinking Water and Consequent Infection Rates.

Approx. quantity of cysts	Infection rate	Percentage
0	0/21	0
1	0/5	0
10	2/2	100
25	6/20	30
100	2/2	100
10,000	3/3	100
100,000	3/3	100
300,000	3/3	100
1,000,000	2/2	100
All doses	21/40	53

[Rentdorff, 1954]

Table 6. Experimental Results of *Salmonella typhosa* Ingested and Consequent Disease Rate.

Approx. quantity S. Typhosa	Disease Rate	Percentage
10^3	0/14	0
10^5	32/116	28
10^7	16/32	50
10^8	8/9	89
10^9	40/42	95

[Hornick, 1970]

Table 7. Experimental Results of *Shigella flexneri 2a* Ingested and Consequent Infection and Disease Rates.

Approx. dose of cells	Disease Rate	%	Infection Rate	%
180	9/36	25	9/36	25
5000	28/49	57	33/49	67
10^4	52/88	59	66/87	76
10^5	14/24	58	15/24	63

[Dupont, 1972]

Table 8. Experimental Results of Attenuated Poliovirus Ingested in Capsules by Adults and Consequent Infection Rate.

Dose PFU	Infection Rate	Percent
0.2	0/2	0
2	2/3	67
20	4/4	100
200	4/4	100

[Koprowski, 1955]

Table 9. Experimental Results of Attenuated Poliovirus Ingested by Infants and Consequent Infection Rates.

Dose (TCD-50 units)	Infection Rate	Percent
1	3/10	30
2.5	3/9	33
10	2/3	67

[Katz, 1967]

measure of virus quantity sufficient to infect 50% of tissue cultures inoculated, the TCD_{50}, was used to experimentally relate infection response to ingested poliovirus dose in infants for the study shown in Table 9 [Katz, 1967].

Different susceptibility among individuals has been shown and complicates measurement of disease incidence but these individual variations can be averaged by observation of large populations. This differing response is due to the variance of survival of pathogens in the human body's gastric tract and immune defense system and differences thereof among individuals. The human gastric environment is normally very hostile to ingested microorganisms, operating at a pH of about 2.0 . Age, nutrition, and other variables of physical condition all affect an individual's susceptibility to disease. Immune responses to enteric virus infections appear to often provide that individual with a lifelong resistance for that type of virus [Shuval, 1984]. Immunity against enteric bacteria is less lasting and there seems to be little protection provided by immune responses to protozoans [Cliver, 1980]. Differing exposure histories among individuals, along with varying physiological condition and genetically defined response abilities, can present major variations in human responses to infective hazards.

Because of this varying reaction among persons exposed to the same hazard, additional uncertainty enters any attempt to relate an estimate of average risk associated with certain conditions to a particular individual. For purposes of assessing widespread impacts on public health, epidemiological studies of whole populations are appropriate and are able to include an averaged susceptibility without defining the actual frequency distribution of pathogen-host activity. For estimating the dose, however, this frequency distribution can be defined [Pipes, 1977].

B. Exposure to pathogens

One of the classical modes of infectious disease transmission is the fecal-oral route via consumption of contaminated water. Exposure routes relevant to assessment of risk associated with wastewater effluents also include ingestion and contact during recreational use of receiving waters, and consumption of seafood taken from contaminated waters. (Inhalation of micro-organisms lifted in wastewater aerosols presents a risk that is significant only locally and occupationally [Majeti, 1981].) Water-based sanitation systems emptying into sources of drinking water comprise a potential major circuit for spread of fecally-orally transmitted infectious diseases. Most of these diseases are enteric [McJunkin, 1982].

When considering the health risks associated with wastewater discharge, water related diseases classified as water contact disease are also of concern. Occupational or recreational exposure to contaminated waters can lead to many types of illness. Common water contact illnesses include inflammations of the ear (otitis), sinuses (sinusitis), eyes (conjunctivitis), and infection of any exposed wound or abrasion [McJunkin].

Categorization of water related diseases according to location and mode of water use is of particular relevance for considering the health protection benefits of wastewater effluent disinfection because it is important to consider the multiple mitigating conditions which apply to the various exposures to effluent discharges. Dilution, natural die-off, and/or intervening treatment alter the health hazard of discharged wastewater that is subsequently used by humans. Varying probable exposures to the hazard must also be included in an assessment of the health risk.

The various opportunities for exposure to a microbiological hazard can, for the purposes of assessing wastewater disinfection, be separated according to whether risk depends solely on the quality in receiving waters or whether deficiencies can feasibly be alleviated by further treatment.

Often it is uneconomical or impossible to obtain satisfactorily plentiful and pure potable water that needs no further treatment. Millions of persons in the U.S. are served by supplies from surface waters, all of which have some potential for upstream contamination.

Where drinking water is taken from contaminated sources, intervening treatment effectively serves the basic need for biological purity. The effectiveness of water treatment in the U.S. in protecting the health of consumers is well proven. Coagulation, sedimentation, filtration, and disinfection are the basic processes which provide clear, clean, safe drinking water. A significant reduction in measured waterborne disease in the U.S. during the early 20th century accompanied the advent of modern municipal water purification. For typhoid alone, the average five-year death rate dropped by 65% in American cities which installed filtration for their water supply system [McJunkin, 1982].

Although water treatment in the U.S. is often extensive and quite sufficient for providing pure water from impure sources, analysis of the chain of risks of contaminant transmission illustrates the benefit of controlling discharges upstream. The pathogen concentration in contaminated water poses an infectious hazard of a magnitude which is proportional to that concentration. This is because the chance of ingesting a pathogen is greater when it is present in greater numbers, and because the risk of infection (prerequisite for disease) increases with the number of pathogens ingested.

There are small but real chances for entry of contaminated water into a distribution system [NRC, 1982]. Distribution systems and treatment deficiencies in community water systems were the proximate causes of 34% of the outbreaks and 60% of the cases of waterborne disease reported by the U.S. Center for Disease Control for 1980 [CDC, 1982]. Since the resulting risk is proportional to amount of contaminated water, the degree of contamination, and the amount and degree of exposure, reduction of pathogen concentration in receiving waters that are used downstream reduces the hazard and, thus, the risk attendant to accidental potable water exposures. Although the vast majority of persons in the U.S. who drink water taken from an impure source are protected by effective water treatment systems, some persons regularly consume insufficiently treated contaminated water. Small non-community water supplies account for most of the outbreaks of waterborne disease reported in the United

States--64% of the reported outbreaks [CDC, 1982;Craun, 1981].

Because the incidence of serious waterborne disease in the United States is a small fraction of the incidence in the past or in other parts of the world today [McJunkin, 1982]; generally low incidence and multiple public health safeguards preclude determination of the benefits of one safeguard or prediction of the effects of removing that safeguard.

Giardiasis is currently the most common identified cause of reported waterborne disease in the United States [Craun 1979, 1981], and epidemiologic studies suggest that drinking untreated surface water is the most important factor in endemic Giardia infection in the United States [Craun 1979]. Infection with Giardia is often asymptomatic and, therefore, often undiagnosed. The estimated incidence of giardia infection is 4% in the U.S. general population [Juniper 1983b], and therefore may be excreted by 4% of the general population. Giardia lamblia cysts are apparently resistant to normal drinking water chlorination (<3mg/l) and inadequate or no filtration was the blamed deficiency for over 10,000 cases in the four largest recent epidemics in the United States [Dykes, 1980; Craun, 1981]. In most of the 24 documented Giardiasis outbreaks in the U.S. there has been little or no bacterial contamination reported in the water [Craun, 1979]. A major investigation following a large epidemic of giardiasis in Camus, Washington, discovered that failed filters allowed giardia to pass from the creek sources into the distribution system. Isolations from animals captured in the watershed implicated beavers as a reservoir of Giardia

[Dykes, 1980].

Infectious hepatitis (Hepatitis A) is the viral pathogen most known to be transmitted via water [Craun 1973,1981; Kruse, 1971]. Among the many routes of transmission of Hepatitis-A, it is estimated that much less than 5% is water related [CDC,1982; Hutzler, 1980]. In the U.S., the incidence of Hepatitis-A has dropped from 28 cases/100,000 population/year in 1970 to 13 cases/100,000 popn./yr. in 1980 and the mortality rate for those clinical cases implicating fecal-oral transmission is less than 1% [CDC, 1982]. Attention regarding water related transmission of hepatitis is specifically on shellfish contamination because this has been the demonstrated route. Shellfish contamination is covered in a later section of this report.

Most waterborne disease outbreaks in the U.S. are of undetermined etiology [Craun, 1981]. Since 1971, the EPA Health Effects Research Laboratory (HERL) and the CDC have cooperated on the surveillance of waterborne disease in U.S. [Craun, 1981]. Reports of outbreaks have increased since 1971. This is attributable to the increased effort and is illustrated by the fact that Pennsylvania, with its extensive active investigation system by local and state public health officials, contributed 21% of all reported waterborne disease outbreaks for the period [Craun, 1981]. By comparison, Pennsylvania's portion of the U.S. population was 5.5% for the period.

Reported outbreaks, by virtue of attracting public health officials' attention, are unusual and acute events. The major attributed causes of outbreaks in potable water supplies were contamination of municipal distribution systems (primarily as a result of cross connections or backsiphonage), use of untreated contaminated groundwater, or deficiencies in treatment processes [Craun 1973, 1981].

The ultimate source of contamination, and the portion of illness transmitted via drinking water in the U.S. today that might be caused by pathogen content of wastewater effluents is partially reflected in the fact that 9% of all waterborne disease outbreaks reported 1971-1978 occurred in systems using untreated surface water [Craun, 1981]. This category of "untreated" surface water includes giardiasis outbreaks where the water was chlorinated but not filtered.

Of the 22 largest disease outbreaks associated with water supplies (accounting for 73% of the total illnesses) six were in systems drawn from surface water sources. No indication of upstream dischargers is given, as blame was placed on water systems [Craun, 1981]. See Table 10.

The two largest outbreaks in systems using surface water were giardiasis blamed on inadequate or no filtration. Since these protozoan cysts are resistant to destruction by chlorination at common drinking water doses, filtration is a more effective means of removal; but has been neglected at some places where the water source was thought to be pure [Craun, 1979; Dykes, 1980]. Since 1978 Colorado regulations require filtration of surface water source supplies.

An important consideration for the safety of reused water, and therefore also affecting the need for sewage effluent disinfection, is the treatment that reused water will or could

TABLE 10. The Six Largest Reported Outbreaks in Systems Using Surface Water, U.S. 1971-1978.

Year	Place	Cases	Etiology	Deficiency
1978	Vail, CO	5000	Giardiasis	Inadequate filtration
1974	Rome, NY	4800	Giardiasis	Surface water disinfection only
1978	Bennington VT	3000	Campylobacter	Inadequate disinfection
1977	Berlin, N.H.	750	Giardiasis	Inadequate filtration
1976	Camas, WA	600	Giardiasis	Inadequate filtration
1975	Shasta Lake, CA	900	Acute Gastro-enteritis	No filtration & inoperative wastewater disinfection

[Craun, 1981]

receive. There is this opportunity to treat water prior to use in potable water supplies, industrial food processing, or agricultural irrigation. The feasibility, reliability, and efficiency of such treatment must be assessed and included for a comprehensive evaluation of effluent disinfection. For instance, the reuse of surface waters for irrigation is unlikely to include any biological-purification treatment because of the high cost of such treatment, but such use is also likely to be done only in dry regions where no better water is available. The same rationale regarding risk transmittal and compounding of risk reduction that applied to microbiological concentrations in receiving waters which are used downstream for drinking water supplies also applies to these other withdrawals.

For drinking water, disinfection before use is economical and easy. However, where ingestion may occur without interceding treatment, the only means of intervention is to control the pathogen concentration in that water. Seafood harvesting or contact recreation in water receiving wastewater effluents incurs the risk associated with any pathogens which may be present. Hazard reduction by effluent disinfection is therefore important for water that may subsequently be used as is and where is, because in these cases wastewater treatment is the most feasible opportunity to interdict water-borne transmission of disease by reducing the pathogen load in water.

Of vital concern regarding seafood harvesting from contaminated waters are the circumstances when that food will be eaten without cooking, such as is often done with oysters. Adult shellfish growing in an estuary receiving wastewater effluents are more likely than finfish to become contaminated because of their fixed location and filter feeding, by which they concentrate microorganisms from the surrounding waters.

An important public health protection is the testing and certification of shellfish growing waters. Closure of chronically contaminated beds to commercial harvesting does not completely cover their hazard, though. Private harvesting and harvesting adjacent to closed zones have been the source of shellfish implicated in two serious outbreaks [Dienstag, 1976; Mason, 1962].

There is compelling epidemiological evidence that associates some common-source outbreaks of hepatitis with the victims' eating of raw or undercooked bivalve molluscs taken from contaminated waters [Dienstag, 1976; Mason, 1962]. However, of the reported Hepatitis-A cases in the United States for 1980, only 16% were epidemiologically associated with previous consumption of shellfish [CDC, 1982]. There is an approximate thirty day incubation period for Hepatitis-A, making it difficult to trace the etiology of most cases [Dienstag,1976; Mason,1962]. For the outbreaks traced to biologically contaminated shellfish there was found to have been much gastroenteritis (an illness of indeterminate etiology and often unreported) among persons partaking of shellfish from these sources [Mason,1962; Murphy,1979; Dienstag,1976]. In a major Hepatitis outbreak (in Mississippi) the implicated oysters traced to several suppliers and private harvesting were all from the same bay, 1-4 miles from a sewage outfall bypassing the city's treatment plant during enlargement construction [Mason, 1962]. Bacterial levels measured in oysters sampled from suppliers during outbreaks when there was indication of fecal contamination are shown in Table 11.

A massive outbreak in Australia of gastroenteritis from eating raw oysters was associated with Norwalk virus [Murphy 1979]. The virus was identified in 39% of fecal specimens and manifested as antibody response in 75% of the victims. At present there is no technique to identify Norwalk virus in water. 73% of the oysters from the implicated estuary were found to be excessively fecally contaminated during the outbreak, while only 28% of the samples from oysters actually causing the illness had high bacteria levels [Murphy, 1979]. This is attributable to the shellfish's differing ability to eliminate bacteria and viruses. Live shellfish will purify themselves when removed from contaminated water and placed in purer water. Bacterial levels in the bivalves lag by 24-48 hours the levels in the ambient water. Viruses are eliminated (e.g. 3 log reduction) over a significantly longer period (about 120 hours) [Hedstrom, 1964]. This process is inhibited by presence of chlorine at concentration as low as 0.2 ppm because adult shellfish respond to low concentrations of chlorine by ceasing pumping. Therefore, chlorine can be used for decontamination of the exterior of harvested shellfish, but alternative disinfectants must be used for purification water for the entire shellfish. Filtration or ultra-violet disinfection are used for purifying the depuration water at most such facililties in the United States [Blogoslowski, 1980]. Shellfish taken from some estuaries are required by law to be depurated for two days in disinfected water

TABLE 11. Bacterial Contamination of Oysters From Suppliers Sampled During Associated Outbreaks.

Outbreak, Location	Group	Quantities	Method
Norwalk virus, New South Wales	fecal coliforms:	270-1720/100g	SPC
	E. coli:	270-930/100g	SPC
Hepatitis A, Pascagoula	coliforms	4900-24000/100ml	MPN

[Hedstrom, 1964]

[Murphy, 1979; Blogoslowski, 1980].

Exposure to pathogens in surface waters during direct contact and the consequent risk of infectious disease has been known for a long time. In this century in the United States, prevention of contamination of bathing waters by chlorinating sewage effluents was begun in 1923 at Cleveland, Ohio, where discharges to Lake Erie were chlorinated [APHA, 1934]. Chlorine applications averaged 8-9 ppm.

There is some epidemiological evidence of the benefit of pathogen concentration reduction in waters used for bathing. A two year study of several New York City area beaches showed that illness rates were higher among beachgoers who immersed themselves as compared to those who didn't bathe in the contaminated water (total coliform MPN=1213/100 ml, fecal coliform MPN=565/100 ml) [Cabelli, 1979]. At a relatively unpolluted beach (total coliform MPN=43.2/100 ml, fecal coliform=28.4/100 ml) the same comparison showed no significant difference in subsequent illness rate [Cabelli, 1979]. The inference is that immersion in contaminated water is unhealthy.

Coliforms are evidently not the best correlated indicator organism for recreational water quality. In a study of 5400 swimmers and 2300 controls that found a significant correlation between post-swimming illness rates and concentration of bacterial indicators in the water, enterococci correlated well; while the correlation between illnesses and E. coli densities was not consistent [Ktsanes, 1981]. Other studies have also found enterococci concentrations to be a better indication of recreational water quality than coliforms [Cabelli, 1982 B].

Attempts at quantitative estimates of recreational exposures and consequent risk involve great uncertainty [Haas, 1983 B]. Ingestion of contaminated water is the usually assumed mode of pathogen intake and quantities of water ingested during an average swimming experience are assumed to be on the order of 10 ml [Dudley, 1976] to 100 ml [Haas, 1983 A]. One estimate of the benefit of wastewater disinfection in reducing the risk to swimmers of viral illness (assuming that disinfection provided a 1 log reduction in viruses from a concentration of 257 pfu/l and swimmers ingest 100 ml of water) suggested an absolute risk of viral illness equal to 6.3×10^{-4} /person/event for non-disinfected water and 6.3×10^{-5} /person/event for disinfection [Haas, 1983 A]. The risk differential (a quantity suggestive of the benefit of disinfection) is 5.67×10^{-4}. Assuming a use rate of swimming events/person/year (0.924 in this case), the following equation could be used to estimate the additional cases of viral illness in a population swimming at a certain area that might result from ceasing effluent disinfection.

(population)(use rate)(risk differential) = (additional cases)

Using this method, Haas concluded that relaxation of disinfection requirements in Illinois would not significantly increase the risk of viral illness for any individual, but would however result in about 2700 additional cases of viral illness from recreational exposure in the state per year [Haas, 1983 A].

Swimming-associated illness can originate from many sources, even though association of illness rates with swimming in waters which receive municipal effluents has been made [Cabelli, 1982 B; Rosenberg, 1976]. One large study that showed an association between rate of swimming activity and enteroviral illness included swimming pool facilities and natural water bodies, thus suggesting a general health risk due to swimming [D'Alessio, 1980]. On the other hand, a strong suggestion of the risk from uncontrolled pathogen concentrations in effluents is given in a study where 31 of 45 cases in an outbreak of Shigellosis in Iowa were traced to swimming in a river area 8 km downstream from a wastewater treatment plant [Rosenberg, 1976]. At the swimming area the measured fecal coliform concentration were about 17,500/100 ml and during that same month the treatment plant had been discharging water with up to 1.2×10^{7} fecal coliforms/100 ml. A cause-effect relation cannot be stated, but considering the waterborne transmission route of Shigella and the low infective doses (10-100), the association between the outbreak and the wastewater effluent is suggested [Rosenberg, 1976].

VI. POTENTIAL ADVERSE EFFECTS OF EFFLUENT CHLORINATION ON HUMAN HEALTH

Disease is caused also by non-infectious agents, such as toxic or carcinogenic chemicals which have been discharged to the environment. Recent research has shown that there may be certain compounds formed during chlorination that are potentially harmful to humans.

A. Chlorinated compounds in wastewater treatment plant effluents

Alarm at the discovery of halogenated organics formation resulting from chlorination of drinking water [NRC, 1977, 1980; Symons, 1975] has prompted some re-examination of wastewater chlorination practice [Jolley, 1978, 1980, 1983].

Halogenated organics are of special interest because they do not occur naturally in aquatic systems and some are generally considered to be toxic, mutagenic, and carcinogenic. Concern about this group of compounds is manifested in the fact that more than half of the EPA designated priority pollutants are halogenated organics [Young, 1980]. Halogenated organic compounds vary in their health effects, occurrence, and notoriety. Specific analysis has identified and characterized numerous organic compounds in drinking waters, polluted surface waters, and wastewaters. (Water pollutants are measured as collective or surrogate parameters such as biochemical oxygen demand (BOD) or total organic halogen (TOX), or are measured more directly with specific analyses such as gas chromatography/ mass spectrometry (GC/MS) analysis). GC analysis of soluble organics extracted from the secondary effluent at a plant which was treating wastewater that was 80% domestic / 20% industrial found the soluble organics to be 40-45% humic substances, 20-25% proteins, 12-15% anionic detergents, 10-12% carbohydrates, 7-10% ether extractables, and 1-2% tannins [Rebhun, 1971]. As a result of chlorination of water that contains organics, some chlorinated organic compounds are formed.

The chlorination of surface water supplies high in humics has been shown to produce elevated levels of trihalomethanes (THMs) [Symons, 1975]. However, the presence of ammonia or amino groups results in chloramine formation and retards reaction of chlorine with soluble organics and other compounds, making chlorination of these organics less likely [Murphy, 1975]. Chloramines have a much lower oxidation ability than free available chlorine species, but will, however, combine with organic compounds by substitution reactions if given long enough contact times (about 10 times as long as with free chlorine) [Murphy, 1975].

Chlorine added to a typical secondary effluent, at a dosage slightly below the breakpoint (20-40 mg/l Cl) has been shown to eventually produce up to 300 ug/l TOX (after 24 hours) [Chow, 1981]. The same dose in a highly nitrified, filtered effluent (achieving some free available chlorine residual) produced about 700 ug/l TOX after the same time period. Included in the measurement of total organic halogen are trihalomethanes, which in these experiments comprised from 5-20% of the TOX by weight, the higher portions after longer reaction times. The amount of chlorine converted to organic halogen in the long-term near-breakpoint chlorination of secondary effluent was on the order of 1% of the chlorine applied [Chow, 1981].

Only minor portions of all halogenated micropollutants are now amenable to identification [Jekel, 1980]. For instance, after experimental superchlorination (Cl:C molar ratio=4) of fulvic acid isolated from lake water, the four principal identified reaction products accounted for only 14% of the weight of original organic material and 53% of the TOX [Christman, 1983].

Samples of secondary effluent taken to the laboratory, filtered, and superchlorinated (1500-2000 mg/l continuously applied over one hour) at a low pH (2-3) have yielded high amounts of several chlorinated compounds [Glaze, 1975]. Of an estimated 3000-4000 ug/l TOX, thirty-two compounds accounting for 780 ug/l were identified [Glaze, 1975]. Results of this analysis are shown in Table 12.

Chlorination of domestic wastewater effluents at more normal lower dosage levels has been shown to still produce stable chlorinated organics from about 1% of the chlorine applied [Jolley 1975, 1982]. This yield was in the same range for both primary effluent chlorinated to a 1 mg/l combined residual for 15 minutes and for a secondary effluent chlorinated to 0.5 mg/l for 30 minutes [Jolley 1975]. Longer contact times slightly increased the yield. Specific analyses of chlorine-containing compounds in these effluents are shown in Table 13.

Chlorinated organic compounds appearing in the effluent of wastewater treatment plants do not necessarily originate from application of chlorine at the plant. Chlorinated organics appear in the influent to treatment plants, even ones receiving no industrial discharges. For example, in studies on the effect of disinfection on organics at a 1.5 MGD municipal tertiary

Table 12. Specific Analysis of Chlorinated Organics in Wastewater Effluent and Estimated Concentrations.

Compound name	Concentration (ug/l)
Chloroform	--
Dibromochloromethane	--
Dichlorobutane	27
3-chloro-2-methylbut-1-ene	285
Chlorocyclohexane	20
Chloroalkyl acetate	--
O-dichlorobenzene	10
Tetrachloroacetone	11
P-dichlorobenzene	10
Chloroethylbenzene	21
Pentachloroacetone	30
Hexachloroacetone	30
Trichlorobenzene	--
Dichloroethyl benzene	20
N-methyl-trichloraniline	10
Dichloromethoxytoluene	32
Trichloromethylstyrene	10
Trichloroethyl benzene	12
Dichloro-a-methyl benzyl alcohol	10
Dichloro-bis(ethoxy)benzene	30
Trichloro-a-methyl benzyl alcohol	25
Tetrachlorophenol	30
Trichloro-a-methyl benzyl alcohol	50
Tetrachloromethoxytoluene	40
Dichloroaniline derivative	13
Dichloroaromatic derivative	15
Dichloroacetate derivative	20

[Glaze, 1975]

Table 13. Tentative Identifications and Concentrations of Chlorine-Containing Constitutents in a Chlorinated Secondary Effluent.

Compound name	Concentration (ug/l)
5-Chlorouracil	4.3
15-Chlorouridine	1.7
8-Chlorocaffeine	1.7
6-Chloroguanine	0.9
8-Chloroxanthine	1.5
2-Chlorobenzoic acid	0.26
5-Chlorosalicylic acid	0.24
4-Chloromandelic acid	1.1
2-Chlorophenol	1.7
4-Chlorophenylacetic acid	0.38
4-Chlorobenzoic acid	1.1
4-Chlorophenol	0.69
3-Chlorobenzoic acid	0.62
3-Chlorophenol	0.51
4-Chlororesorcinol	1.2
3-Chloro-4-hydroxybenzoic acid	1.3
4-Chloro-3-methylphenol	1.5

[Jolley, 1975]

treatment plant (activated sludge, biological nitrification, filtration) which had no known industrial wastewater contribution, several chlorinated organic compounds were found [Chappell, 1981]. This domestic sewage did, however, include some previously chlorinated sewage from a nearby national park facility. Volatile compounds identified in the treatment plant effluent, prior to disinfection, are shown in Table 14.

Volatile chlorinated organic compounds entering municipal wastewater treatment plants appear to be significantly removed during treatment. The concentrations of volatile chlorinated compounds measured at the influent and effluent, before and after chlorination, are shown in Table 15 [USEPA Task Force, 1976].

Large metropolitan sewer systems are likely to receive some amounts of halogenated discharges. In primary effluent from the several major municipal water systems in the Los Angeles area, on average, there were found 10% of the 113 priority-pollutant trace organics at levels above 10 ug/l [Young, 1980]. Of these, chloroform (the major THM formed from water chlorination) was measured at concentrations ranging from <10 to 64ug/l with a concentration averaging right at the median of those measured [Young, 1980]. The content of chlorinated organics in wastewater effluents from sources other than disinfection chlorination will likely vary considerably, especially for various mixtures of domestic and industrial wastewaters.

Chlorinated organic compounds appearing in waters which receive chlorinated municipal effluents may originate at other sources. Among the potential industrial sources of aqueous halogenated micropollutants are chlorination to prevent fouling

Table 14. Volatile Organic Compounds Identified in Treatment Plant Effluent, Prior to Disinfection.

Compound name	Concentration (ppb)
Chloroform	0.2
1,2-Dichloroethane	a
Carbon tetrachloride	a
n-hexane	a
Bromodichloromethane	a
Trichloroetylene	a
Dimethyl disulfide	a
Toluene	0.5
Tetrachloroethylene	1.0
p-xylene	.01
Styrene	.01
o-xylene	.01

a= compound identified but concentration not determined

[Chappell, 1981]

TABLE 15. Some Volatile Chlorinated Organic Compounds in Water at Sewage Treatment Plants.

Compound[a]	Concentration (ug/l) Influent before Treatment	Effluent before Chlorination	Effluent after Chlorination
Methyl chloride	8.2	2.9	3.4
Chloroform	9.3	7.1	12.1
1,1,1-Trichloroethane	16.5	9.0	8.5
1,1,2-Trichloroethylene	40.4	8.6	9.8
1,1,2,2-Tetrachloroethylene	6.2	3.9	4.2
Dichlorobenzenes	10.6	5.6	6.3
Trichlorobenzenes	66.9	56.7	56.9

[a] All confirmed by GC-MS

[USEPA Task Force Report, 1976]

of thermo-electric power plant cooling waters and wastewater from bleached pulp/paper mills. Of the chlorine manufactured in the U.S., about 15%, or 3.6 billion pounds per year is used in the pulp/paper industry for bleaching [Leach, 1979; White, 1972]. Most of this chlorine ends up as chloride in effluent wastewater, but recent experiments indicate that up to 10% of the applied chlorine is incorporated in nonvolatile organic compounds dissolved from the pulp [Leach, 1979]. Volatile chlorinated organics are also formed in large amounts. For example, chloroform measured in pulp mill effluent averaged 110 ug/l even after an estimated 94% reduction during biological treatment [Claeys]. In some receiving waters, then, the contribution of chlorine and chlorine reaction products from municipal wastewater disinfection may be comparatively minor.

B. Potential health effects of chlorination products

Reaction products possibly formed as a result of wastewater chlorination are of very uncertain composition and concentration because of widely varying chlorine application rates, wastewater composition, contact time, pH, temperature and other conditions. Dissipation and decompostion after discharge also affect the products of effluent chlorination.

Despite significant uncertainties regarding the occurrence, identification, measurement, and persistence of chlorination reaction products, an evaluation of their potential risk to public health will be made by considering chloroform as a representative. It should, however, be realized that chloroform generally represents about only 15% of total organohalogens, that the remaining compounds will likely behave differently, may present potential health risks which are less than, similar to, or greater than chloroform, and that synergistic effects may occur. For these reasons the data relating to chloroform only cover part of the potential health risks involved. An estimated average chloroform production of 9 ug/l will be used, based on an EPA gross average from 25 plants' secondary effluents where chlorination caused an average measured increase of chloroform from 5-14 ug/l [USEPA 1979 B]. First, the exposure of the American public to this assumed level of chloroform in effluents will be roughly estimated, then the effects of these exposures will be assessed.

Potential exposures to chlorinated effluents can be classified according to mode of water use. Ingestion via drinking water taken from surface water sources, contact during aquatic recreation, and ingestion via seafood harvested from receiving waters will be evaluated in turn.

Human ingestion of effluent chlorination reaction products via drinking water from systems that draw upon surface waters which receive wastewater effluents appears to be not significant at this time. According to the National Organics Reconnaissance Survey for Halogenated Organics, the raw water from surface sources contained no or very low concentrations of THMs [Symons, 1975]. Chlorinated organics appearing in significant amounts in surface waters which are used as a source by drinking water systems will likely be from multiple or obscure sources. However, as awareness of the many micropollutants, from many sources, increases, the effort by waterworks to remove them by treatment to acceptable levels will increase significantly [NRC, 1980, 1982]. Therefore, hypothetical exposures to possible wastewater chlorination products will be sketched here.

Recreational exposure to chloroform in receiving waters could be via inhalation, skin absorption, or ingestion [USEPA 1979 A]. Chloroform in water at very dilute concentrations follows Henry's Law, such that the partial pressure of chloroform in the gas phase is proportional to that in solution. Under standard pressure, at a temperature of 25 ^{o}C, and assuming a breathing rate of 6m^{3}/hour, a person breathing undisturbed air overlying water containing 9 ug/l chloroform would inhale an estimated 10 ug of chloroform vapor per hour [USEPA 1979 A]. Government estimates of total recreational use of open waters for boating, fishing, swimming, and waterskiing, when aggregated and divided by the U.S. population suggest an average recreational contact with open waters of about 43 hours/person/year [USEPA 1979 A]. So, if there were 9 ug/l additional chlorine in all recreational waters, the average annual inhalation of chloroform from this source might be 430 ug. Estimates of skin absorption of chloroform from immersion in water containing 9 ug/l, assuming chloroform is as easily absorbed through the skin as ethyl ether, suggest skin absorption effective doses may be of the same order as inhalation [USEPA 1979 A]. If swimmers ingested 100 ml/hour, and all swimmers swam in water containing 9 ug/l chloroform, then the average annual exposure to chloroform by this route would be two orders of magnitude greater than inhalation and skin absorption [USEPA 1979 A].

For ingestion of halogenated micropollutants via consumption of seafood grown in contaminated waters, the possibilities of bioconcentration, bioaccumulation, or biomagnification are of concern. Bioconcentration in this case is the incorporation of halogenated organics from the water into the tissues of organisms such that those compounds are at higher concentrations in the tissues than in the water. Bioconcentration which is not reversed would be bioaccumulation. Multiple steps of bioaccumulation in a food chain present a case of biomagnification. Bioconcentration of chlorination products has been shown to occur in shellfish. Bromoform, a potential product of chlorine discharged to saline waters, has been shown to slightly bioconcentrate in oysters, but reversibly [Scott, 1983]. Bioaccumulation and biomagnification that has been shown for some chlorinated organics, such as PCBs or various chlorinated hydrocarbon pesticides, does not directly apply to the compounds that may be formed during effluent chlorination [Kopperman, 1978; Scott, 1983]. Bioaccumulation of effluent chlorination products may occur, but cannot be fully evaluated until more is known about the identity of compounds that may be formed.

The great uncertainties regarding estimates of exposure to possible chlorination reaction products are complemented by uncertainties in evaluation of the hazard of these compounds. Assessment of the effects requires, at least in theory, a knowledge of the identity of the compounds and their effects on humans at actual exposure levels. Neither of these sets of data is now fully available. The effects of long-term exposures are important because that is when low concentrations will be most manifested. Long-term exposures and effects on humans are difficult to discern, register, or analyze retrospectively from available data, and experimental exposures of large samples of humans to hazardous compounds is not feasible.

Animal studies on the long term effects of chlorination reaction products, in particular chloroform, all depend on extrapolation from high doses [Gruener,1978; Jorgenson,1980; Moore,1981; NRC,1980; OECD,1982]. A notable study is the 1976 National Cancer Institute bioassay upon whose results chloroform was declared an animal carcinogen [Christman,1983]. In this study, chloroform dissolved in corn oil was administered by gavage to rats and mice at two dose levels five times per week. Dose levels of 90 or 180 mg/kg body weight were given to male <u>rats</u> for 78 weeks; the female rats received higher doses of 125 or 250 mg/kg for the first 22 weeks and the same dose as the males thereafter. After 111 weeks the rats were sacrificed and a statistically significant incidence of kidney epithelial tumors was found in the males but not in the females. The male <u>mice</u> first received doses of 100 or 200 mg/kg and the females 200 or 400 mg/kg. After 18 weeks, the doses were raised by 50% for the males and by 25% for the females. Highly significant increases in hepatocellular carcinoma were found in both sexes [OECD, 1982]. Note that the doses used in this study, as in others, were extremely high.

Some other studies of the long-term effects of chloroform exposure have been inconclusive. For instance, in a 90 day study of rats and mice given drinking water with 200, 400, 600, 900

1800 and 2700 ppm chloroform in it, the initial loss of appetite and refusal to drink the water resulted in weight loss in some groups that led to better short-term survival rates directly proportional to chloroform dosages [Jorgenson, 1980]. The final results, though, gave no significant dose-related effects.

While there are no epidemiological studies dealing with the carcinogenicity of chloroform per se, there have been epidemiological studies of consumers of chlorinated drinking water. There appears to be a weak, but statistically significant, risk of cancer of the bladder from the consumption of water from chlorinated supplies [Cantor, 1982]. However, the potential error from confounding factors such as smoking or diet is large and undetermined, since no such information was available on the people studied [Cantor, 1982].

Assessment of the health effects of chloroform using standard International Agency for Research on Cancer criteria applied to the evidence for chloroform concludes that the evidence supports categorizing chloroform as carcinogenic in laboratory animals at very high dosages, but does not support categorizing chloroform as carcinogenic to people [USEPA, 1984].

VII. ADVERSE EFFECTS OF EFFLUENT CHLORINATION ON AQUATIC ECOSYSTEMS

A. Toxicity of residuals

Because disinfection of wastewater is based on a strong concern for protecting people from the health risks associated with microorganisms in sewage, little attention was given until the past 10 to 15 years to the adverse effects that routine use or overuse of chlorine has on the environment. Chlorination sufficient to disinfect, as indicated by suitably reduced fecal coliform levels, typically produces chlorine residuals of several tenths of a mg/l or more. Such residual chlorine levels are greater than those which have been found to be toxic to some aquatic animals. Thus, within the discharge plume of a wastewater treatment plant that is disinfecting with chlorine, aquatic life may be inhibited or damaged.

The residual chlorine discharged from most secondary wastewater treatment plants which disinfect with chlorine is composed predominantly of chloramines. Only if chlorine is applied in amounts greater than ten times the weight concentration of ammonia nitrogen will there be free chlorine residual species. Below this ratio, chlorine combines with ammonia that is present to form a combined available chlorine residual which is predominantly monochloramine and is maximum at application rates of about 5 mg/l Cl_2 per 1 mg/l NH_3-N. Most secondary effluent chlorination operation is represented by the initial portion of the breakpoint curve shown in figure 1.

A few wastewater plants intentionally chlorinate to beyond the breakpoint, mostly with highly nitrified effluent and to meet stringent coliform regulations, but these also usually dechlorinate before discharge [White, 1972, 1978]. Thermal-electric power generating plants require cooling, and usually draw and discharge vast amounts of surface water to do so. To preserve the heat exchange efficiency of the equipment, slimes or other fouling biological growths are discouraged by intermittant disinfection with chlorine. Chlorine is used for biofouling control at 90% of power plants in the United States [White,1972]. Typical practice is 1-2 mg/l chlorine for 20-30 minutes two or three times/day. The discharged chlorine residuals tend to be less than those from wastewater treatment plants, but contain a higher portion of free chlorine species [Hall,1981; Hollod,1982; White, 1972].

Discharged chlorine residuals decrease as a result of reactions and dissipations. Available free chlorine species are less stable than combined forms, either entering the air, reacting with any of many reducing agents in the receiving water, or decomposing to chloride. Monochloramine, the most common residual chlorine species in chlorinated municipal secondary effluents, dissipates more slowly than free available chlorine. In a study on the impact of chlorinated secondary effluent on receiving river water quality, Lee, et al., found that the fate of the predominantly monochloramine residual in a muddy river (60 NTU) at $10^{o}C$ was 60% volatilization, 28% reaction due to oxidation demand, and 12% photoinduced decay for a one order of magnitude decrease that occurred during twenty hours instream [Lee,1982]. It was determined that dissipation of residual chlorine was first order for each of these mechanisms and that in the river the chlorine dissipated twice as fast in the summer as in the winter [Lee,1982].

The toxicity of chlorine residuals to many species of aquatic life has been demonstrated in laboratory and field, and has been summarized [Brungs,1973; Mattice,1976]. Other components of wastewater, such as ammonia and suspended solids, have also been implicated as toxic to aquatic life [Esvelt,1973; Garber,1980]. Aquatic toxicity depends on the levels of residual chlorine remaining in the discharge and on the relative amounts of free and combined chlorine species. However, the toxicity of free chlorine and chloramines are of the same order, and measurements of the total residual chlorine (TRC) are reasonable for defining aquatic toxicity [Brungs,1973; Mattice,1976; Wolfe, 1984].

Acute toxicity to fish and other aquatic animals increases with time of exposure. Most tests for acute toxicity on aquatic organisms are done for 96 hours because the concentration-effect vs. time curve often appears generally flat at and beyond 96 hours [Stephan, 1980]. For the conservative protection of the aquatic life, minimum effect levels are taken at the long-term exposures.

Though wastewater chlorination produces a variety of chlorine compounds, the inorganic chloramines (of which NH_2Cl predominates over $NHCl_2$ or NCl_3) are thought to be among the most toxic forms of combined chlorine [Zillich,1972; Wolfe,1984]. Concentrations of monochloramine as low as 0.01 mg/l have been found to cause a 50% mortality in oyster larvae exposed for 30 minutes, and similar low LD-50 values for chloramine on other aquatic invertebrates have been demonstrated [Wolfe,1984]. After 24 hour exposures, common warm water fish such as sunfish, catfish, and minnows succumbed to a few tenths of mg/l NH_2Cl. Other species were affected when exposed to lower concentrations. Monochloramine levels of 0.06-0.08 mg/l were lethal to freshwater trout [Zillich, 1972]; 0.043-0.085 mg/l total residual chlorine (TRC) significantly reduced reproduction in fathead minnows and 0.16-0.21 mg/l killed half the samples of the same species [Zillich,1972; Arthur,1975]. Other findings of the toxicity of chlorine residuals discharged in wastewater effluents are listed in Table 16. Overall, a conservative no-effect threshold level for continuous chronic exposure to wastewater chlorine residuals is 0.01 mg/l TRC [Zillich, 1972; Brungs, 1973; Canada, 1978; Wolfe, 1984].

Chlorine residuals seem to be similarly, or slightly less, toxic to common estuarine and marine organisms than to freshwater organisms [Bellanca, 1977]. The conservative no-effect threshold is about 0.02 mg/l TRC for the most sensitive saltwater organisms [Mattice, 1976].

Available data indicates that, for certain pollutants in certain waters, some species of aquatic animals are over 6000 times more sensitive than other species [Stephan, 1980; Wolfe, 1984]. Other materials have shown an interspecies range of sensitivity of not over 30 [Stephan, 1980]. Sensitivity differences of over 2000 occur for most monochloramine concentrations sufficient for median lethality of aquatic fish and invertebrates; half of a group of pike perch fry survived 20

TABLE 16. Toxicity of Wastewater Effluent Chlorine Residuals to Aquatic Animals.

Species Common Name	Exposure Concentration (mg/l)	Exposure Duration (min)	Effect
Invertebrates:			
Water flea	0.002	20160	decreased reproduction
Scud	0.054	161280	decreased survival
"	0.019	201600	decreased reproduction
"	0.135	43200	no effect
"	0.900	1440	50% mortality
Crayfish	0.780	10800	50% mortality
Caddisfly	0.550	10080	" "
Stonefly	0.480	4320	" "
Operculate snail	> 0.810	20160	" "
Pulmonate snail	> 0.810	20160	" "
Fish:			
Coho salmon	0.230	720	" "
Rainbow trout	0.020	7200	" "
" "	0.014	5760	" "
" "	0.029	5760	" "
Brook trout	0.360	720	" "
White sucker	0.248	720	" "
Fathead minnow	0.185	720	" "
" "	0.110	100800	no spawning
Largemouth bass	0.494	1440	50% mortality
Yellow perch	0.365	720	" "
Walleye	0.267	720	" "

[Arthur, 1975; Mattice, 1976]

mg/l NH_2Cl for 24 hours, compared to 50% mortality of oyster larvae in only 0.01 mg/l for 30 min [Wolfe, 1984].

B. Mitigating factors

Water quality guidelines which are based on concentrations selected so that they will not have any impact on sensitive aquatic organisms receiving chronic, life-time exposure are overlooking several mitigating factors. First, the concentrations of contaminants in natural water are not constant over time or space due to variations of stream flow volume and mixing, pollutant persistence, and discharged concentrations. Second, because higher aquatic animals are motile and others are free-floating, only attached forms such as adult molluscs would likely remain chronically in areas of high concentration. Third, the community of organisms naturally existing in receiving waters is very site specific and not simply a function of water composition. These factors may mitigate aquatic toxicity of wastewater effluents minimally or significantly depending on the receiving water body, time of year, treatment facility and operation.

Free-swimming fish have been found to detect and avoid chlorine residuals at concentrations well below toxic levels. Rainbow and brook trout have been found to select against free chlorine residuals as low as 0.001-0.01 mg/l (compared with 0.1 LD-50). Several other species have shown avoidance behavior for monochloramine concentrations that were less than one quarter of their species' LD-50 [Morgan, 1980].

Most adult shellfish respond to low concentrations of chlorine by ceasing pumping. When they "clam-up," they stop feeding and growing, and can withstand conditions as high as 10 ppm chlorine for 30 days. This avoidance behavior reduces the impact on adult shellfish of temporally variable chlorine residuals.

After cessation of effluent chlorination, the fish communities in streams receiving secondary wastewater treatment plant discharges have been observed to increase in quantity and species diversity [Paller, 1983]. The facilities at these sites were properly functioning secondary treatment plants treating domestic wastewater and operating within their design capacity. Non-disinfected secondary effluents present only slight toxicity to aquatic organisms, even if undiluted [Arthur, 1975; USEPA Task Force, 1976].

Poorly designed chlorine contact chambers also can contribute to damage of aquatic life. An operator usually has control over only one factor of the disinfection process-- the chemical application rate. If the facility is inefficient, the chlorine application rate may be made higher to try to achieve sufficient disinfection. These larger applications will subsequently create greater chlorine residuals with their consequent hazard to aquatic life. Optimization of chlorination facility design can reduce aquatic damage by reducing chlorine residuals [Sepp, 1981].

One method to avoid possible damage to the ecosystem of receiving waters is to dechlorinate the wastewater prior to discharge. Adverse effects of residual chlorine on survival and growth are eliminated, and no undesirable side effects are produced, when residual chlorine is neutralized by proper dechlorination with sulfur dioxide or sodium thiosulfate [Arthur, 1975; Ward, 1980; Zillich, 1972]. Dechlorination by application of these chemicals, or by other processes, presents additional expense. Estimates of the cost of dechlorination processes are included in Section IX.

C. Bioaccumulation

Chlorine can combine with a wide variety of organic compounds, and is a component of over three quarters of the EPA listed organic priority pollutants, many of which enter municipal plants which treat combined domestic and industrial wastewaters [Young, 1980]. Chlorination of effluents has shown the potential for formation of chlorinated organics [USEPA Task Force, 1976; Jolley, 1975] including, under certain conditions such as low pH, chlorinated aromatics if petroleum hydrocarbons are present in the effluent [Glaze, 1975].

Certain chlorinated organic compounds, such as the chlorinated aromatic hydrocarbon pesticides, are extremely toxic and have been found to bioaccumulate in aquatic animals and be concentrated up the food chain. Research and experience with chlorinated pesticides, herbicides, and PCBs indicates that biomagnification of chlorinated amino acids, carboxylic acids, or phenols would not occur to any large extent whereas chlorinated aromatic hydrocarbons would. Apparently, the contribution of effluent chlorination to the load of chlorinated aromatic hydrocarbons in the environment is relatively small compared to other sources [Young, 1980; Canada, 1978].

Bioaccumulation of bromoform ($CHBr_3$), a more common product of chlorination of saline waters, has been found to occur in oysters to a small extent. When chlorine is added to brackish water (with salinities above 5 parts per thousand), bromoform is the dominant trihalomethane produced. Estuarine waters (salinity 24-30 ppt) dosed with 1 mg/l chlorine resulted in 0.25 mg/l TRC and 19-31 ug/l bromoform. Dechlorination after 1.1 minutes reduced bromoform formation to a range of 15-22 ug/l. Oysters growing in these waters were found to have accumulated 70-130 ug/g (wet weight) $CHBr_3$ from the 19-31 ug/l bromoform water, and 20-40 ug/g in the dechlorinated water. The maximum concentration factor between water and meat was about four. In all cases, depuration in unchlorinated seawater reduced the body burdens of bromoform rapidly, with none detected after 48 hours [Scott, 1983]. Thus, bioconcentration of trihalomethanes appears to be minor and temporary.

VIII. RECENT REGULATORY POLICY

Prior to 1972 and implementation of the U.S. Federal Clean Water Act (P.L. 92-500), most effluent disinfection requirements were based on water quality criteria of the states. These criteria varied from state to state, and in some states chlorination practice was seasonal or site-specific depending on whether the probability of public exposure was low. In 1958, about three-quarters of the states had some regulation or recommendation relative to wastewater treatment plant chlorination [Laubusch, 1958]. Of these, 31 had adopted the recommendation of the 1952 Ten States Standards for a minimum of 2.0 ppm chlorine residual in the effluent [Laubusch, 1958].

In August 1973, U.S. Environmental Protection Agency regulations following P.L. 92-500 essentially required disinfection at most wastewater treatment plants by setting specific limits on fecal coliform concentrations in effluents. The fecal coliform limits (200/100 ml monthly average, 400/100 ml weekly average) were low enough to generally require practice of effluent disinfection at all municipal wastewater plants [Hais, 1984]. During the next four years, most, if not all, states were induced by the scheme of P.L. 92-500 to implement water pollution control regulations similar or more stringent than federal requirements, including standards which required disinfection. In July, 1976, the USEPA removed the federal limitations on fecal coliforms and since then has left to the states the responsibility for regulations regarding disinfection [Hais, 1984].

areas, and 23/100 ml for confined recreational waters, all assuming effluents are diluted greater than 100 to 1 [White, 1978]. Some other California standards are very strict. The state standard for chlorine residual in receiving waters for maximum daily, instantaneous, and six month average is .002 mg/l [Garber, 1980]. Also, the bacteriological standard for discharges into ephemeral streams and other areas where dilution is low is a median MPN of coliform to not exceed 2.2/100 ml. To achieve this essentially coliform-free effluent necessitates some type of tertiary treatment and/or severe disinfection, such as filtration or nitrification and 10 to 25 mg/l chlorine to produce a free residual [White, 1978].

Recent concern regarding the potential effects of wastewater chlorination has led to much review of policies regarding wastewater disinfection. Over half of the states report that they are now reviewing their policies [VDTF, 1984]. For example, the Illinois Pollution Control Board in 1981 proposed that the effluent fecal coliform standard of <400/100 ml be applied only to discharges within 20 miles of bathing beaches, potable water supply intakes, lakes, or another state, and that any water quality coliform standards be deleted [Haas, 1983 A].

Canadian disinfection policy and practice varies from province to province. As in the U.S., chlorination is the predominant method. The Canadians generally require disinfection wherever wastewater effluents may present risk to public health. Four provinces set requirements for disinfection on a case-by-case basis, one has seasonal requirements, four require it year-round, two generally don't require it at all, and one insists only upon process availability for emergency [Canada, 1978].

In western Europe, routine effluent disinfection is generally not practiced. The few reported uses of chlorine in wastewater include disinfection for certain shellfish-growing or bathing areas, hospital sewage, emergency situations, and, in Italy for discharges to urban water supply sources [Canada, 1978].

IX DISINFECTION OPTIONS

The most common method of disinfecting wastewater has been, and continues to be, chlorination [Maxted, 1983; Thoman, 1958; White, 1972, 1978]. Where there is minimal hazard to the public health from wastewater effluents, there is often no disinfection practiced [VDTF, 1984]. Where chlorine residuals in effluents present problems, dechlorination prior to discharge is a viable process option which reduces the residual and mitigates the adverse effects discussed earlier [Chen, 1981]. There are several other methods which have received more attention recently [Bossart, 1983; Gould, 1981; Haas, 1982; Venosa, 1983]. These disinfectants include ozone, ultraviolet radiation, chlorine dioxide, and bromine chloride. Other disinfection methods, used infrequently for wastewater, are extreme pH, gamma irradiation, heat, and application of iodine. Alternative disinfectants appear to be increasing in popularity, but all of them currently are more expensive. This higher cost and the widespread commitment to chlorine are obstacles to fast adoption.

Ozone and ultraviolet (UV) light appear to be the most promising competitive alternatives to chlorine [Englebrecht, 1983; Severin, 1980; Venosa, 1983]. Each has been shown to be effective at disinfection, and each has potential problems such as process design and control, and, for ozone, possible toxic residual oxidation products [Englebrecht, 1983; Johnson et al., 1983; Legube, 1980; Nebel, 1973; Venosa, 1983].

Evaluation of the alternative disinfection processes and their environmental impacts and health effects is beyond the scope of this report. However, since the important decision factor of economic cost has not been presented in this report yet, and because this factor is an obstacle to implementation of new processes, available cost estimates will be presented for disinfection options. With the exception of the two options of chlorination and no disinfection, much of the cost information must be considered tentative. Because of the relatively rare use of alternative methods, no large data base regarding their cost is available. Additionally, continuing developments will affect the relative and absolute costs of the various options.

Simple chlorination is by most estimates still the cheapest method of achieving a given degree of disinfection. The itemized cost of chlorination for various size facilities, as reported by a survey of almost 600 plants, is shown in Figure 7 [WPCF, 1980]. At small plants, however, ultraviolet disinfection appears to have become cost competitive with chlorination, (these estimates of relative costs vary with assumptions such as electricity and chemical prices, capital amortization, and disinfection efficiency) [Severin, 1980].

A summary of costs for various disinfection options is presented in Table 17 [USEPA Task Force,1976]. Caution should be used in comparing various methods, though, since these estimates of relative cost are based on many assumptions, some of which are rather dubious, such as that all chemical disinfectants are equivalent at an 8 mg/l dose [USEPA Task Force, 1976].

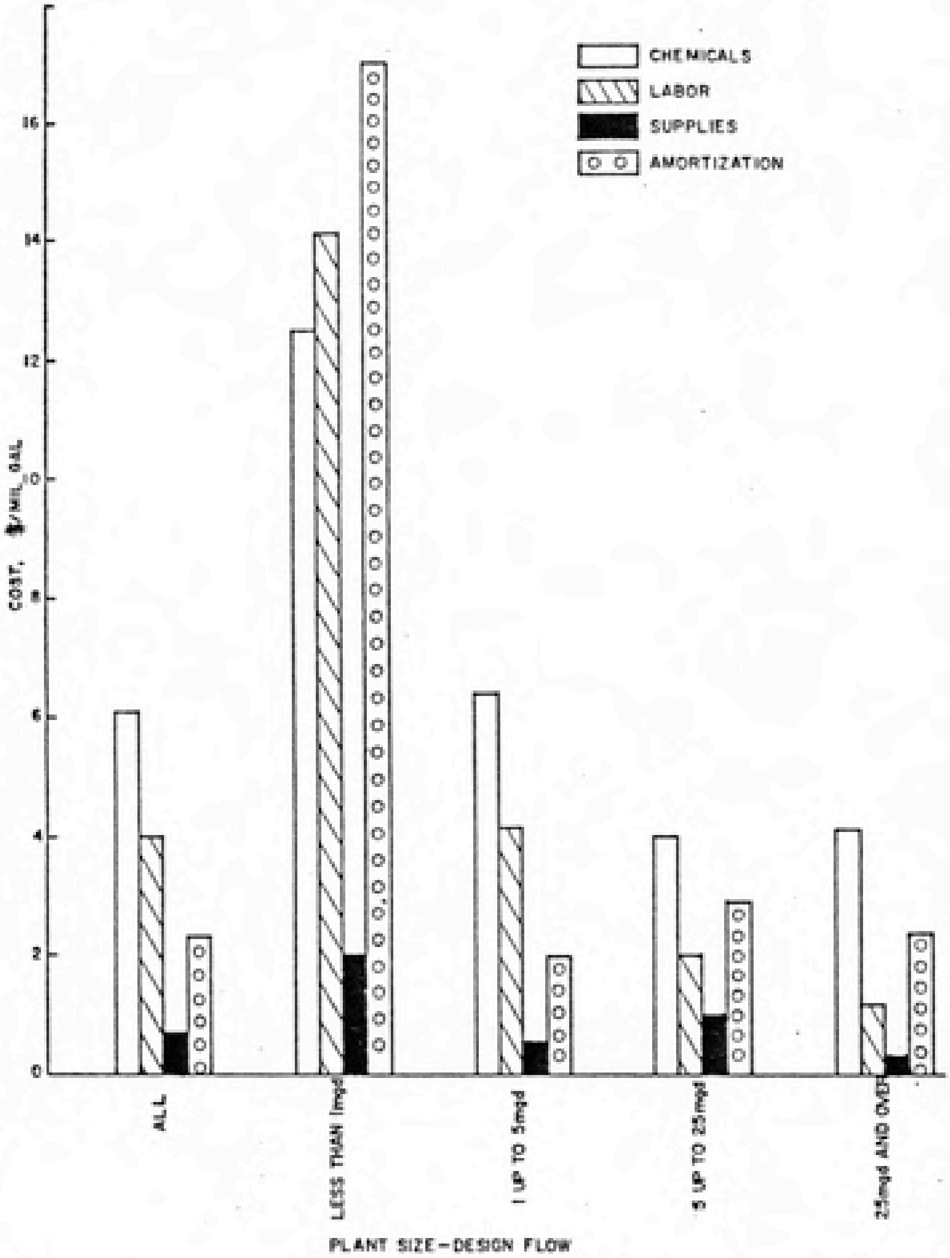

FIGURE 7. Itemized economic costs of chlorination, the prevalent method of wastewater disinfection [WPCF, 1980].

Table 17. Economic Cost Comparisons of Various Disinfection Options.

Process	Disinfection cost (¢/Kgal.)		
size of facility:	1 MGD	10 MGD	100 MGD
No disinfection	0	0	0
Chlorine	3.49	1.42	0.70
Chlorine/SO2 dechl'n.	4.37	1.75	0.89
Chlorine/SO2 dechl'n + aeration	7.66	2.39	1.19
Chlorine/carbon dechl'n.	19.00	8.60	3.28
Ozone/ from air	7.31	4.02	2.84
Ozone/ from oxygen *	7.15	3.49	2.36
Ultraviolet *	4.19	2.70	2.27
Bromine chloride	4.52	3.04	2.65
for comparison:			
Activated sludge	55.90	20.20	14.00

* costs for tertiary treatment not included.
All figures are cents per thousand gallons, U.S.

[USEPA Task Force Report, 1976]

X. DISCUSSION

Recent attention and policy trends regarding chlorination of wastewater treatment plant effluents reflect concern for the adverse environmental impacts and health effects of chlorination [Bossart, 1983; Gould, 1981; Haas, 1982; Jolley, 1978, 1980, 1983; VDTF, 1984]. These concerns have advanced on two major fronts during the last dozen years. The first seeks to avoid adverse impacts on wildlife and the natural environment, and has moved forward along with the significant effort to clean the nation's water by controlling sources of pollution (e.g. P.L.92-500). In particular, the adverse impact of effluent chlorination which has been attacked on this front is the toxicity of chlorine residuals to aquatic life in receiving waters. A second strike against effluent chlorination enlists concerns about the possible adverse human health effects resulting from chlorine applied to water. These concerns arise from discoveries that chlorine combines with organic materials that are present in waters and forms a number of halogenated organic compounds, which are suspected to present a hazard to human health.

These adverse effects have been demonstrated under certain conditions. Some species of fish and other aquatic and marine animals have been shown to be adversely affected by chlorinated wastewater effluents discharged in their water and by chlorine residuals concentrations much lower than those commonly discharged. Also, of the many halogenated organic compounds that have been formed from reaction of chlorine with materials commonly found in wastewater effluents, some have been shown to cause adverse effects in laboratory animals exposed to high concentrations for long periods.

However, the conditions under which adverse effects have been demonstrated are not necessarily or generally the same conditions found at most applications of chlorine for effluent disinfection. Severe toxicities of chlorine residuals to aquatic life have been demonstrated for certain life stages of certain species when held captive and exposed. However, other organisms have been shown to tolerate much higher exposures, so the actual impact on aquatic life will depend in part on the character and condition of the native biological community of the particular receiving waters. Dilution, mixing, dissipation of residuals, and the opportunity to avoid plumes of undesirable high chlorine concentrations in the receiving water body are some of the mitigating factors which may spare aquatic animals from damage. As a result of these variables, adverse impacts of effluent chlorination on aquatic ecosystems are likely to be significant only at some places and times. For these cases, the toxicity of chlorine residuals can be effectively avoided at a reasonable cost by dechlorinating the effluent prior to discharge.

The extreme conditions under which the human health hazard of chlorination reaction products have been demonstrated are far from the conditions of reality. Even at exposure levels several orders of magnitude higher than those to which humans could be expected to be exposed to from chlorinated effluents, the laboratory animal studies did not consistently show adverse effects. There is indication, however, that chlorination causes the formation of halogenated organics which are not yet specifically identified and are therefore an uncertain potential hazard.

Nevertheless, prudence directs that suspected and potential adverse effects, even if unproven, be avoided within the latitude in which benefits are reasonably obtained. The primary benefit of, and reason for, disinfecting wastewaters is ostensibly to reduce the actual or potential numbers of pathogens present in wastewaters. This benefit is difficult to demonstrate because it is one of prevention of something which is generally prevented - sewage source illness. Outbreaks of waterborne disease associated with recreational exposure to, and shellfish consumption from, waters indicated to be fecally contaminated suggest the connection between the pathogen content of wastewater effluents and the incidence of disease among persons exposed to those effluents. Disinfection of sewage effluents is a significant public health protective measure where the waters which receive the effluents are used for swimming, food harvesting, or drinking water sources.

It should be kept in mind, though, that use of chlorine as the method of disinfection is not necessary for obtaining the benefits of pathogen reduction, since other methods of disinfection can be effective. These other methods may avoid the adverse effects of chlorination, but at the present they are not widely used because of chlorination's lower cost, established reliability, and ease of operation.

XI CONCLUSIONS

1. Under current conditions in the U.S., it is not feasible to measure with certainty the benefits of effluent chlorination or the risk to human health posed by cessation of effluent disinfection.

2. Discharge of chlorinated effluents can under some conditions damage aquatic ecosystems. This damage can be feasibly avoided by practice of dechlorination.

3. There is not now sufficient evidence to show that chlorination reaction products at estimated exposure levels present a significant hazard to human health, as compared with the significant potential hazard to public health from infectious waterborne disease.

XII RECOMMENDATIONS

Evaluation of wastewater chlorination should take into consideration all the estimated adverse effects and expected benefits of disinfection under the conditions presented. Because the adverse effects as we currently understand them are not sufficient to offset the inferred benefits of effluent chlorination in most cases, most effluent chlorination practice need not be discontinued on the basis of the information we now have.

In situations where the adverse effects are judged to be significant, alternative disinfection methods are available at generally higher cost. Where the adverse effect of concern is the toxicity to aquatic life of discharged chlorine residuals, dechlorination with sulfur dioxide is a remedy of demonstrated efficacy and reasonable cost. Development of understanding of the adverse effects to humans of wastewater chlorination has far to go. If and where the potential health risk to humans is judged to be significant, and effluent disinfection is called for, alternative technologies such as ultraviolet irradiation and ozonation are available.

In situations where the expected exposure to effluents is minimal, as with high effluent dilution, seasonal, or no downstream use, the need for disinfection is also minimal and the practice may reasonably be curtailed.

APPENDIX A: SURVEY OF STATES' WASTEWATER DISINFECTION POLICIES

from

Virginia Disinfection Task Force, "Draft Report to State Water Control Board," Richmond, (Feb. 1984).

provided by
Cal Sawyer, acting director, BWE
Department of Health
Commonwealth of Virginia

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements*	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Alabama	Water Supply:gm2000FC/100 ml Coastal Primary Contact:gm 100FC/200ml Other Primary Contact: gm 200FC/100 ml Shellfish: gm 70TC/100 ml Fish & Wildlife:gm1000FC/100ml	All dischargers must disinfect year round.	0.5 mg/l TRC max. No FC limits.	None	0.5 mg/l max. TRC	None	No
Arizona	Primary Contact:gm200FC/100 ml Agricultural:gm1000 FC/100 ml (Note:"Wastewater dominated" streams have standards for enteric virus, Ascaris eggs, entamoeba, taeniarhynchus eggs.)	All dischargers must disinfect year round.	Based on bacteriological standards.	None	None	None	No
Arkansas	Class AA:gm 200FC/100 ml Class A: gm 200FC/100 ml (April 1 - Oct. 1) Class B: gm 1000 FC/100 ml	(proposed) -Disinfection required into water supplies and primary contact recreaction waters. -Disinfection not required to intermittent streams (unless necessary to protect uses) -Seasonal disinfection may be allowed.	Based on bacteriological standards.	None	None	2 ozone 2 UV several dechorinate (4% of all STPs)	Yes, protection of cold water fisheries
Connecticut	Water Supply: gm20FC/100 ml Primary contact:gm200FC/100ml Secondary contact: gm 1000FC/100 ml Shellfish:gm 20FC/100 ml	Allow seasonal disinfection for listed stream segments (May 1-Oct. 1)	gm 200 FC/100 ml TRC range of 0.5-3.0 mg/l	None	TRC range of 0.5-3.0 mg/l	None	Yes
Delaware	Water Supply:gm 200FC/100ml Primary contact: gm200FC/100 ml Secondary contact:gm 770FC/100 ml Shellfish:gm 70TC/100 ml	All dischargers must disinfect year round.	gm 200FC/100 ml max. 1000TC/100 ml (shellfish areas only)	0.01(Cr) Undectable for shellfish areas.	1.0-4.0 mg/l TRC Further studies may be required to determine chlorine impact.	1 dechlorination facility in construction	Yes
District of Columbia	Primary Contact:gm200FC/100 ml Secondary Contact: gm 1000FC/100 ml	All dischargers must disinfect year round.	gm 200FC/100 ml	Aquatic life: 0.02	Dechlorination for STP	None	Yes, conducting several studies
Florida	Water Supply:gm200FC/100 ml Shellfish:gm 14FC/100 ml Recreation: gm200FC/100 ml	All dischargers must disinfect year round.	Based on bacteriological standards.	0.01 (Cr.)	None for NPDES permit State permit may require permittee evaluate the need for dechlorination.	1 ozone none dechlorinate (out of 4400 facilities)	No

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Georgia	Water Supply:gm 1000FC/100 ml Coastal Recreation:gm100FC/100 ml Other Recreation:gm200FC/100ml Aquatic Life:gm1000FC/100 ml Agricultural:gm5000FC/100 ml	Most dischargers disinfect year round. Pond systems not providing secondary treatment exempt.	gm 200 FC/100 ml (no limits for ponds unless drinking water supply)	None	None	1 dechlorinate (trout stream)	No
Hawaii	Recreation:gm 200FC/100 ml	All inland and nearshore discharges must disinfect year round.	gm 200 FC/100 ml	None	None	None	No
Idaho	Primary Contact:gm 50FC/100 ml (May 1-Sept. 30) Secondary Contact: gm 200FC/100 ml	All dischargers must disinfect year round.	gm 200 FC/100 ml	None	Discharge requirement that stream values not exceed 0.002 mg/l for cold water and 0.01 mg/l for warmwater	-2 UV -4 dechlorinate	No
Illinois	Proposed: no standards	(Proposed) -Year round for water supplies -seasonal (May-Sept.) within 20 miles of licensed bathing beach -no requirements elsewhere	Max. 400 FC/100 ml TRC range of 0.2-0.75mg/l	None	TRC range of 0.2-0.75 mg/l	None	Yes, see proposed disinfection standards and requirements
Indiana	Water supply:max5000TC/100 ml Primary Contact:gm 200FC/100 ml Secondary Contact: gm 1000FC/100 ml	Disinfection required year round only if necessary to protect public health. Most facilities only disinfect from April 1 - Oct. 31.	TRC range of 0.5-1.0 mg/l for facilities serving > 10,000 people, gm 200FC/100 ml also.	None	TRC range 0.5 -1.0 mg/l	2 ozone by late 1983 (250 MGD) 2 dechlorinate in near future	Yes
Iowa	Primary Contact:gm 200FC/100ml (April 1-Oct. 31) Secondary Contact: gm 2000FC/100 ml (April 1-Oct. 31)	Dischargers must disinfect seasonally except those discharging to non-classified streams.	None	None	None	2 UV	Yes
Kansas	Primary Contact: gm 200 FC/100 ml Secondary Contact: 2000 FC/100 ml Water supply and aquatic life may fall into either category.	Generally, disinfection only required in neighborhood streams in urban areas. Out of 425 STPs Only 50 disinfect.	gm 200FC/100 ml	None	-no set procedure -may limit TRC on case-by-case basis	2UV	No

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/1)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Kentucky	Water Supply:gm 2000FC/100 ml Primary Contact:gm 200FC/100 ml (May 1-Oct. 31) Secondary Contact:5000 FC in 10% of samples (May 1 - Oct. 31)	All dischargers must disinfect year round.	gm 200 FC/100 ml	Warmwater:0.01(Cr.) Coldwater:0.002 (Cr.)	None	2 ozone	No
Maine	Class A:max. 20FC/100 ml Class B-1:max. 60FC/100ml Class B-2:max. 200FC/100 ml Class C:max 1000FC/100 ml Shellfish:gm 14FC/100 ml Class SC:gm 200 FC/100 ml Class SB-1:gm 50 FC/100 ml Class SB-2:gm 100FC/100 ml	(Experimental) year round disinfection required to water supply and shell fish areas. Seasonal disinfection required above recreational areas (April 15-Oct. 15)	Class A:gm 20FC/ml ClassB-1,B-2: gm 60FC/100 ml Class C & D: gm 200 FC/100 ml Tidal: gm 15 FC/100 ml	None	1.0 mg/1 max.	4 UV	Yes, may make experimental program permanent.
Maryland	Recreation and Aquatic Life: gm 200 FC/100 ml Shellfish: gm 14 FC/100 ml	All dischargers must disinfect year round.	gm 200 FC/100 ml (except gm 14 FC/100 ml for discharges to shellfish areas)	Recreation trout: 0.002 (ST) Native Trout: Use of chlorine compounds prohibited (2 exceptions)	Mass balance calculation to achieve 0.01 mg/1 in waters other than recreational trout (0.002); if no mass balance problem, establish max limit of 0.5 mg/1	Out of 442 POTWs -150 dechlorinate -13 UV 1 ozone 7 maintain < 0.5 mg/1 without dechlorination	Yes, may allow seasonal or no disinfection on case-by-case basis
Massachusetts	Class A: gm 50TC/100 ml Class B: gm 200TC/100 ml Class C: gm 1000TC/100 ml Class SA: 70 TC/100 ml Class SB: median 700 TC/100 ml Class SC: gm 1000 FC/100 ml	Allow seasonal disinfection (generally April 15-Oct. 15) 80% of STPs disinfect on a seasonal basis	-Min. 1.0 mg/1 at 15 minutes contact -Max. 1.5 mg/1 at point of discharge	None	-1.0 mg/1 min. at 15 min. contact points -1.5 mg/1 max. at discharge point	1 UV no dechlorination (out of 122 STPs)	Yes
Michigan	Primary Contact: gm 200 FC/100 ml All other: gm 1000 FC/100 ml	Allow seasonal disinfection (May 15-Oct. 15) for all dischargers	Based on Bacteriological standards.	None	-max. of 0.5 mg/1 -use mass balance 7Q10 and TRC in-stream value of 0.024 mg/1 (dechlorination not required until 1988)	1 dechlorinates 1 ozone	Yes

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Minnesota	Fisheries and Recreation: gm 200FC/100 ml (March 1 - Oct. 31) Limited Resource Value: gm 1000FC/100 ml (May 1 - Oct. 31) Water Supply: gm 200FC/100 ml	Stabilization ponds not required to disinfect. Others must seasonally unless they are within 25 miles of water intake where they must disinfect year round.	gm 200FC/100 ml	0.005(ST) for waters classified for fisheries and recreation	Discharges which disinfect with chlorine and discharge to waters classified for fisheries and recreation must dechlorinate.	6-8 dechlorinate (several more in planning stages) -several UV -several ozone	No
Mississippi	Water Supply: gm 2000FC/100 ml Recreation: gm 200 FC/100 ml Shellfish: gm 14FC/100 ml Fish and Wildlife: gm 2000 FC/100 ml Ephemeral Streams: no limit (can be assigned on a case-by-case basis)	Generally, all dischargers must disinfect year round. Disinfection may not be required to ephemeral streams or streams with very high dilution.	-gm 200 FC/100 ml -TRC range of 0.1 -1.0 mg/l	None	TRC range of 0.1 -1.0 mg/l	None (some planned on Gulf Coast)	Yes, possibly eliminating disinfection except above water supplies, high recreation use, etc.
Missouri	Primary Recreation: gm 200FC/100 ml	Disinfection only required to primary streams (April 1 -Oct. 31), losing streams, and through densely populated areas. (out of 2800 facilities, only 275 disinfect)	gm 200FC/100 ml	warmwater: 0.01 (ST) cold water: 0.002(ST)	Dechlorination required except where: (a) into an unclassified stream $\geq$ 1 mile from classified stream (b) where 7Q10 $\geq$ 50 x STP Q	1 ozone several UV	No
Montana	Class A - closed, A-1: gm 200FC/100 ml Class B-1,2,3; C-1,2,3, and E: WT $\geq$ 60°F: gm 200FC/100 ml WT < 60°F: FC only limited when necessary to protect human health	Allow seasonal disinfection (April 1 - Oct. 31) for certain dischargers (except above water supply)	Based on bacteriological standards.	None	-Mass Balance calculation using 7Q10 and EPA Red Book criteria -Max. of 0.5 mg/l		No

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Nebraska	Primary Contact: gm 200FC/100 ml Secondary Contact: gm 1000FC/100 ml	Seasonal (April 1 - Sept. 30) allowed for dischargers to primary contact and "urban" streams. Not required for secondary contact streams (only 14% of STPs disinfect)	Based on bacteriological standards.	Cold water and high quality streams only: 0.01	Based on water quality standards.	None	Yes, thorough review underway.
New Hampshire	Water Supply: 50TC/100 ml Primary Contact: 240TC/100 ml Secondary Contact: mean of 1000 TC/100 ml Shellfish: 70TC/100 ml	Seasonal disinfection allowed except to shellfish waters and water supplies. (Recreational activities must also be considered) Season: April 1-Oct. 31	240 TC/100 ml	None	None	None	- - -
New Jersey	Freshwater: gm 200 FC/100 ml Shellfish: gm 70 TC/100 ml Primary Contact Tidal: gm 770 FC/100 ml Secondary Contact Tidal: gm 1500 FC/100 ml Coastal Ocean: gm 50 FC/100ml Deep Ocean: gm 200 FC/100 ml	Year round disinfection required for most discharges	gm 200 FC/100 ml	Freshwater: 0.003 (Cr.) Tidal & Coastal: 0.01 (Cr.)	Max. TRC limits established on case-by-case basis generally on small sensitive streams (Limit is often "undectable")	≃ 1% of STPs	Yes, looking at Broadening application of TRC standard and eliminating seasonal disinfection allowance.
New Mexico	Water Supply, Primary Contact: gm 100 FC/100 ml Secondary Contact: gm 1000 FC/100 ml	All dischargers must disinfect year round.	Max. 500 FC/100 ml	None	None	1 alternative	Yes, will include strict TRC limits to cold water fisheries
New York	Water Supply: gm 200FC/100 ml Primary Contact: gm 200 FC/100 ml (seasonal) Secondary Contact: gm 2000 FC/100 ml (seasonal) Shellfish: gm 70 TC/100 ml	-year round to water supply & shellfish -seasonal for primary contact recreation -disinfection not allowed into all other unless there is a demonstrated actual health need.	-gm 200 FC/100 ml -May include operational requirements on case-by-case basis.	None	Water quality values of 0.05 (warm water) and 0.005 (cold water) are used as a max. value on case-by-case basis	-Very Rare	No

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
North Carolina	Water Supply: gm 1000 FC/100 ml Primary Contact: gm 200 FC/100 ml (May 1 - Sept. 30) Fish & Wildlife: gm 1000 FC/ 100 ml Shellfish: gm 70 TC/100 ml	Disinfection required on year round basis unless discharge is diluted ≥ 67:1 and not above water supply, primary contact, or shellfish.	gm 1000 FC/100 ml	Trout Waters: 0.002 (ST)	Apply chlorine limits to discharges to trout waters only.	3 UV 3 ozone 18 dechlorinate (out of 2400 discharges)	Yes, considering limiting disinfection requirements and adding TRC standard to all streams.
North Dakota	All waters: 200 FC/100 ml (May 1 - Sept. 30)	All but one facility required to meet FC limits year round (Note: Vast majority of STPs are lagoons which meet FC limits without disinfection)	gm 200 FC/100 ml	0.01 (ST)	Generally, 0.5 mg/l max. This value can be increased or decreased based on mass balance calculation.	None. (Note: only 10 STPs employ "conventional disinfection", 300 lagoon systems do not.)	No
Ohio	Bathing Waters (Lifeguard): gm 200 FC/100 ml Primary Contact: gm 1000 FC/ 100 ml Secondary Contact: Max 5000 FC/100 ml for 10% of samples (uses change with seasons)	Seasonal Disinfection Allowed (March 1 - Oct. 31)	Based on bacteriological standards.	0.002 (ST)	-TRC range of 0.2 - 0.7 mg/l -Dechlorination may be required based on site-specific situation	1 dechlorinates 2 UV 1 ozone	Yes, established a task force that endorsed existing policies.
Oklahoma	Water Supply: gm 200 FC/100 ml Primary Contact: gm 200 FC/ 100 ml (May 1 - Oct. 1) Secondary Contact: no numbers specified	-year round to water supplies -seasonal to primary contact waters -no requirements elsewhere	gm 200 FC/100 ml	None	None	1 UV None dechlorinate (out of 200-300)	Yes
Pennsylvania	Swimming Season (May 1 - Sept. 30): gm 200 FC/100 ml Remainder of year: gm 2000 FC/100 ml	As needed to meet bacteriological standards. Seasonal allowed. Year round required in Delaware River Basin.	Based on bacteriological standards.	None	None	None	Yes, disinfection requirements

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Puerto Rico	Water Supply: gm 2000 FC/100 ml Primary Contact: gm 200 FC/100 ml Secondary Contact: gm 2000 FC/100 ml Pristine Coastal Waters: gm 70 FC/100 ml Shellfish: gm 70 TC/100 ml	All dischargers must disinfect year round.	Based on bacteriological standards.	None	Max. 0.5 mg/l	None	Yes, disinfection requirements
South Carolina	Shellfish: median 70 TC/100 ml Primary Contact, Aquatic Life: gm 200 FC/100 ml Secondary Contact: gm 1000 FC/100 ml	All dischargers must disinfect year round.	gm 200 FC/100 ml	None	None	<1% dechlorinate or use alternate disinfectants	No
South Dakota	Water Supply: gm 5000 FC/100 ml Primary Contact: gm 200 FC/100 ml (May 1 -Sept. 30) Secondary Contact: gm 2000 FC/100 ml (May 1 -Sept. 30)	Seasonal disinfection allowed to primary and secondary contact waters -Disinfection not required elsewhere	Based on bacteriological standards.	0.02 (Cr.)	Mass balance calculation using instream criteria.	Approximately 10% of STPs chlorinate/dechlorinate	No
Tennessee	Water Supply & Aquatic Life: gm 1000 FC/100 ml Recreation: gm 200 FC/100 ml	All dischargers must disinfect year round.	Based on bacteriological standards.	None	-Mass balance calculation using 0.5 mg/l instream -Allow variance from dechlorination unless stream of unusually high quality (then give max. 0.1 mg/l)	20 dechlorinate 1 ozone	Yes
Texas	Primary Recreation: gm 200 FC/100 ml Secondary Recreation: gm 2000 FC/100 ml Shellfish: gm 70 TC/100 ml	Disinfection required unless total residence time at STP>21 days	Min. 1.0 mg/l TRC after 20 minutes contact (at peak flow)	None	Min. of 1.0 mg/l TRC after 20 minutes (at peak flow)	None	No
Utah	Water Supply: gm 2000 FC/100ml Primary Contact: gm 200 FC/100 ml Secondary Contact: gm 2000 FC/100 ml	All dischargers must disinfect year round.	-now, gm 200 FC/100 ml -by 1985, gm 20 FC/100 ml	cold water: 0.002(ST) warmwater: 0.01 (ST)	Mass balance calculation	1 UV 1 dechorinates 15 more require dechlorination	No
Vermont	Water Supply:max 100 TC/100 ml Primary Contact:max 200 FC/100 ml Secondary Contact: max 1000FC/100 ml	All dischargers must disinfect year round.	Max. 200 FC/100 ml TRC range of 0-4.0 mg/l	None	TRC range of 0-4.0 mg/l	4 UV 4 dechlorinate 4 dechlorination in design (out of 80 POTWs)	Yes, considering allowing seasonal disinfection and reducing TRC max.

RESULTS OF NATIONAL DISINFECTION SURVEY NO. 1

Political Jurisdiction	Bacteriological Standards	Disinfection Requirements	Permitting Procedure Regarding Disinfection	Standard or Criteria for Chlorine (mg/l)	Permitted Procedure Regarding Chlorine	Utilization of Dechlorination or Alternative Disinfection	Are you reviewing your disinfection or chlorine regulations?
Washington	Class AA: gm 50FC/100 ml(fresh) gm 14FC/100 ml(marine) Class A: gm 100 FC/100 ml(fresh) gm 14 FC/100 ml(marine) Class B: gm 200 FC/100 ml(fresh) gm 100 FC/100 ml(marine) Class C: gm 200 FC/100 ml(marine) Lakes: gm 50 FC/100 ml	All dischargers must disinfect year round.	gm 200 FC/100 ml	None	Not normally limited unless there are receiving stream problems. Then limits are included to achieve 0.002 mg/l (freshwater) and 0.005 mg/l (marine) outside dilution zone.	4 dechlorinate seasonally 4 dechlorinate year round 1 ozone	Yes, revising "Criteria for Sewage Works Design"
West Virginia	All waters: gm 200 FC/100 ml	All dischargers must disinfect year round.	gm 200 FC/100 ml	Troutwaters: 0.002(ST) Other: 0.01 (ST) (standards don't apply to wet weather streams	Use 7Q10, STP Q and 0.5 mg/l TRC. If problems exists, dechlorination or alternatives must be provided (except wet weather streams)	3 dechlorinate 3 UV	Yes, looking at reliability of UV and need for disinfection
Wisconsin	All waters: gm 200 FC/100 ml	Disinfection of all discharges required except: (1)stabilization ponds (unless short circuiting) (2)where costs exceed benefits from disinfection of secondary or higher quality effluents.	None	None	-Dechlorination or alternatives required when discharging to trout streams. -Mass balance using 0.14 mg/l TRC -0.5 mg/l max.	2-3 UV ca.20 dechlorination or UV facilities in various stages of design or construction	Yes, established a disinfection committee
Wyoming	Water Supply & Intermittent Streams: gm 200 FC/100 ml Primary Contact: gm 200 FC/100 ml (May 1 - Sept. 30) Secondary Contact: gm 1000 FC/100 ml (May 1 - Sept. 30)	-year round to water supplies and intermittent streams. -seasonal for discharges to primary and secondary contact waters	Based on bacteriological standards.	Cold water:0.002(ST) Warmwater:0.01 (ST)	-Mass balance calculation -intermittent streams exempt	<1% of all STPs	No

Legend:

gm = geometric mean
am = arithmetic mean
ST = standard
Cr = criteria
UV = ultraviolet radiation
FC = fecal coliform
TC = total coliform
TRC = total residual chlorine
WT = water temperature
7Q10 = 7-day, 10-year low flow

Does not include:
AL, CO, CA, LA, NV,
OR, RI, VA
Does include:
PR, DC

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