Disinfection of Wastewater Effluents for Reuse Purposes

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Abstract

Many pathogens are continuously present in wastewater as a result of being excreted in large numbers by infected people. Some pathogens multiply in wastewater treatment plants and may survive conventional treatments including filtration.

Domestic wastewater, commonly have an average of $1x10^7$ TC counts/100 ml. A well designed, maintained, and operated treatment plant will achieve 99% reduction yielding $1x10^5$ TC counts/100ml which is far higher than any even most relaxed reuse standard. Therefore, disinfection is an important step not only to prevent water-borne diseases but also to insure sustainable reuse applications utilizing effluents as a reliable water source.

In this study UV radiation, chlorine, and ozone, as wastewater disinfectants, will be presented. Their effectiveness, efficiency, economics will be analyzed. The aim is to provide a logical base for selection of a proper disinfectant based on these factors.

Introduction

Domestic wastewater influents commonly have about 1 x 10^7 TC/100 ml, and well designed and operated treatment plants will achieve an average 99% reduction in total coliform counts. This will yield effluent counts of approximately 1 x 10^5 TC/100 ml which is still far from the standards set by legislators. FC has been reported to count generally 33% to 17% of total coliform counts (Humenick, 1975). But it differs greatly up to 95% or as little as 10% and is related very much to variations influenced by many environmental factors (Aulicino *et al.,* 1996).

Disinfection is probably the most important way of limiting the spread of water-borne diseases which originate in wastewater discharges. Disinfection aim is not sterilisation which requires complete microbial destruction The disinfection process can be chemical (e.g. Cl_2 and O_3), physical (e.g. UV and heat), even mechanical (fine screens and ultra filtration) or a combination of these (Metcalf and Eddy, 1990).

Today, chlorine and its derivatives, UVR, and ozone are the most widely used for wastewater disinfections. Other means have a very limited application in this field. There is a trend to change chlorine compounds to the other methods due to formation of organochlorine compounds. Selecting the most feasible method depends on many factors, such as the cost, quality and quantity of disinfected effluents, its origin (domestic or industrial) and the purpose of disinfection (to be reused or discharged).

The recognition of the significance of disinfection was done already by early workers. In 1909, E. Phelps noted, (as cited by Olivieri, 1979): "Disinfection of sewage will someday be regarded as an integral part of its purification and as a necessary measure for the protection of the community."

2. Disinfection with chlorine

Chlorine has been and still is the most commonly used disinfectant in water and wastewater practices (Olivieri, 1979; Metcalf and Eddy, 1990; Ellis, 1991).

Chlorine as an oxidating agent has a strong affinity to combine with nitrogenous substances in sewage. Its use as a disinfectant is usually desired for two reasons: for its low cost and its residual as a protective measure (which may be fit for drinking water). It has the disadvantages of being stable and extremely toxic to aquatic life (Sollo *et al.*, 1975; Stringer *et al.*, 1975). Chlorine also reacts with organic matter from decaying material to form chlorinated organics which are thought to have toxic, mutagenic, and carcinogenic properties. Other disadvantages expressed by different authors include increased BOD, COD, and TOC values of chlorinated effluents (El-Rehaili, 1995). This was attributed to modifications of dissolved part of OM by chlorine.

Chlorine has also shown virucidal and cysticidal shortcomings (Olivieri, 1979). For example, in studies of inactivation of *Entamoebic histolytica*, chlorine was found to be a poor cysticide especially the combined form (Stringer *et al.*, 1975; Dychdala, 1977). Other protozoa, namely the *Giardia* and *Cryptosporidium* species are also seen very chlorine resistant.

The germicidal effectiveness of chlorine is largely dependent on the concentration available and form of chlorine and other factors. These factors may include the effect of pH, temperature, organic matter, hardness, presence of catalysts, and most importantly chemical form of chlorine. As pH increases, antimicrobial activity will decrease drastically. It has the greatest influence on chlorine effectiveness. The rise of temperature also decreases the effectiveness by 50 - 60% for 10 C rise, which leads to an increase of the contact time by 2 - 3 folds (Dychdala, 1977). Other disinfectants, such as peroxyacetac acid (PAA) have been found to be more effective than chlorine in hot climatic conditions (Baldry *et al.*, 1994)

OM consumes available chlorine and reduces its capacity. Total hardness of Mg and Ca up to 400 mg/l does not have any effects on chlorine efficiency, but catalysts such as Cu, Co, and Ni reduce it. More stable chlorine solutions are obtained, by a low chlorine concentration, absence or low concentration of Cu, Co, Ni, and other catalysts and high alkalinity (Dychdala, 1977). Low temperature, and storage in dark containers to avoid UV light would be needed.

3. Disinfection with ozone

Ozone is generated using electric discharge (corona), electrodialysis of perchloric acid, or UV lamps. The latter being less practiced and only limited to small scale use (Rosen, 1972). Ozone is about 15 times more soluble in water than oxygen at standard temperature and pressure (Venosa and Opatken, 1979; Farooq and Bari, 1989; Ellis, 1991).

When ozone is applied to aqueous solution, it decomposes to (Rosen, 1972; Venosa, 1972; Masten and Davies, 1994).

Principal ozone decomposition products in aqueous solution are molecular oxygen and the highly reactive free radicals namely, $HO_2^{\cdot}$, $OH^{\cdot}$, and $H^{\cdot}$ which are believed to play a significant role in ozone disinfection mechanism (Rosen, 1972). Hydroxyl radical $OH^{\cdot}$ is very important because of its high oxidation potential, yet it is a non-selective agent. It has very short life of about 10 seconds in alkaline solution (Masten and Davies, 1994). Its effectiveness is reduced in the presence of scavangers such as carbonates, bicarbonates, and natural organic matter as part of it will react to form carbonate radicals. The reaction of the carbonate and bicarbonate radicals with OM is more selective and proceed at lower rate than the parent radical OH^{-}. Since O_3 is a non-selective strong oxidant, portions of the O_3 transferred dose is consumed by the presence of impurities. Ozonation of low quality effluents will require more contact time and eventually consume more ozone than relatively purified effluents (Venosa, 1972; Labatiuk *et al.,* 1992).

Several studies suggest that ozone disinfection effectiveness is pH dependent. The effectiveness of O_3 against *Giardia* cysts decreases as the pH increases from 7 to 9 (Ellis, 1991). On the contrary, Diaper (1975) compared disinfection efficiencies of Cl_2 and O_3 against bacteriophages and found that only the efficiency of Cl_2 is pH dependent which is in disagreement with other reported results.

Ahmad and Farooq (1984) demonstrated that the bubble size of ozone has a direct relation to the inactivation of microbes in the system. After studying effects of different bubble sizes in the system disinfecting secondary effluents they, concluded that a decreased bubble size gave a higher solubility and higher microbial inactivation. It was concluded that systems generating bubbles with a size of less than 0.1 cm in diameter had 32 times more contact value than a 1.0 cm bubbles (at constant flow rate). Venosa (1972) concluded that for a temperature less than 10 C the concentration of O_3 required was doubled to achieve same the bacteriological level at a higher temperature (i.e., 0.1 mg/l and 0.05 mg/l respectively). This may suggest that contrary to chlorine, O_3 is most effective at high temperatures (e.g. hot climates).

The most pronounced disadvantage of ozone disinfection is its high operation costs. Since ozone generation is energy dependent the high cost of production is considered to be one of the main limitations for its use (Masten and Davies, 1994; Ellis, 1991). About 20 Kwh is needed for each 1 kg of ozone produced (Venosa, 1972). It is not economical to treat water containing high organic matter because of the higher demand exerted by oxidised compounds (Venosa, 1972; Rosen *et al.,* 1975; Glaze, 1987). This fact might make it a less attractive alternative as an effluent disinfectant. Costs of ozonation systems for water and wastewater treatment plants are capital intensive. It can not be stored, therefore must be produced on site continuously. Operation and maintenance costs are generally high but they vary from site to site.

The oxidation potentials are -3.06, -2.07, -1.49, -1.36, and -0.75 for OH, O_3, HOCL, Cl_2, and NH_2Cl which reflect their disinfection power (Kinman, 1972). The theory of ozonation and the mode of action of microbial destruction is discussed in details elsewhere (Venosa, 1972; Rosen, 1972; Venosa and Opatken, 1979; Farooq and Ahmad, 1989)

4. UV disinfection of wastewater

General background

The short wave section between 200 and 290 nm (UVC) is the most microbiocidal region and is referred to as the abiotic region with an intensity peak at 254 nm which is in the region of maximum germicidal effects (Cheremisinoff and Young, 1975; Harris, 1986; White *et al.*,1986; Scheible, 1987; Wolfe, 1990; Nieuwstad *et al.*, 1991). UVC (or UV for short) does not occur naturally at the earth surface because the atmospheric ozone layer filters out sunlight radiation with wave length less than 290 nm. UV with a wave length of 254 nm can be produced by passing an electrical current of suitable voltage between two electrodes in a sealed tube containing an inert gas together with small amount of Hg (Cheremisinoff and Young, 1975; Ellis, 1991).

UV was firstly employed for small-scale drinking water disinfection in 1916 but it was abandoned by 1930 with the advent of chlorine of some technical problems and the high cost (Wolfe, 1990). Nowadays, after technical improvements of UV reactors and lamp performance, UV systems have been re-evaluated as a reliable water and wastewater disinfection method due to its many advantages over other disinfectants especially after the recognition of the by-products of chlorine, toxicity of its derivatives and the high cost of ozone (Scheible and Bassell, 1979; Severin *et al.*,1984; Havelaar *et al.*, 1990; Wolfe, 1990; Ellis, 1991; Cairns *et al.*, 1995).

Its application to marginal water quality such as secondary effluents has recently increased dramatically after hardware development. Recent studies report an increased use of UV disinfection for secondary effluents as a competitive alternative to both chlorine and ozone (Scheible and Bassell, 1979; White *et al.*, 1986; Thampi, 1990; Dizer *et al.*, 1993).

One of the first full-scale evaluations of UV disinfection was done by Scheible and co-workers in 1979, which concluded that UV is a feasible alternative to chlorine. This study resulted in the approvement of UV disinfection by USEPA treatment technology. In many countries, UV disinfection of secondary effluents has been accepted and increasingly licensed by concerned authorities. Approximately 2000 water and wastewater treatment plants are using UV disinfection technology in Europe and almost 1000 water and wastewater plants in the USA and Canada (Wolfe, 1990; Cairns *et al.*,1995).

Surprisingly, the use of UV disinfection in wastewater treatment plants has not only been adopted for new plants but also for retrofitting existing chlorination systems. Harris *et al.* (1987) stated that a short time exposure of UV systems negate the need for large contact tanks and makes it possible to retrofit most of UV systems within an existing chlorination contact tank. Maarschalkerweerd *et al.* (1990) reported that the number of UV systems in USA have increased six folds during the period of 1986 - 1990. The majority of these plants have been retrofitted from chlorination systems. The capital cost of retrofitting proved to be favourable to the costs of chlorination especially if dechlorination is required (Miller, 1994).

A case study indicated that a privately owned wastewater treatment plant was required by local authorities to reduce its chlorine concentration in effluent by the installment of a dechlorination system. Since the aim was to limit the effluent chlorine, the plant's management found that installation of a UV system was a cost effective alternative to the different solutions proposed. As a result, the UV system was retrofitted and the chlorination system was cancelled (Levin, 1991).

4.1 UV systems

Wastewater effluents can be exposed to UV rays via two general methods, either open channel or closed vessel (Czarnecka-Nieminska, 1985; Harris, 1986). In the open channel systems, UV lamps are suspended over the wastewater flow in shallow trays at a distance of 10 - 15 cm above water level. Despite being easily maintained, the open channel systems are not popularly used because the water level in the exposure trays must be kept critically levelled as a thin film (1 - 2.5 cm). Exposure chambers utilising immersed lamps are more effective for the disinfection of secondary effluents than the suspended-over exposure tray systems (Scheible and Bassell, 1979). In addition, these systems pose less occupational exposure hazard to eye and skin than the suspended-over system. Details of these systems are shown elsewhere (e.g. Jackson, 1994).

The closed vessel systems are becoming more popular and are subdivided into two types: either annular or coaxial systems. In the former, the effluent passes through the disinfection chamber containing the UV lamps. In the latter, the effluent flows in a tube to be irradiated by UV lamps surrounding the conduit where the whole system is enclosed in box with a high reflection degree such as stainless steel (Harris *et al.,* 1987; Scheible, 1987; Kreft *et al.,* 1986; Thampi, 1990).

There are two types of material used to encase UV lamps, either polytetrafluoroethylene (Teflon) or quartz sleeve materials. The former is used in coaxial system while the latter is employed in annular configurations. Both materials (quartz and Teflon) absorb some UV energy and decrease intensity (White *et al.,* 1986). Quartz allows 70 - 90% UV output to be transmitted (Cheremisinoff and Young, 1975). Other studies rated the quartz sleeve to transmit as high as 90 - 95% of the UV light produced (Harris *et al.,* 1987) whereas Teflon will transmit 70 - 85% of UVR generated.

Annular quartz systems are reported to be the most commonly used (Harris, 1986). They are easily maintained and can be expanded (Scheible and Bassell, 1979). Their disadvantages include build up of scale, fragility and difficulties to seal and clean (Czarnecka-Nieminska, 1985; Harris, 1986). Cheremisinoff and Young (1975) reported that they require protection against the temperature difference between the lamp and the effluent. In the second system, non-reactive non-wetting Teflon is used which is easily cleaned due to its low friction coefficient (Harris *et al.,* 1987). For configuration of the two systems, one can refer to Harris *et al.,*(1987).

In some instances, both systems suffer from short circuiting, thus reducing exposure time (Kreft *et al.,* 1986; Harris, 1986). These problems are considered minor as the energy absorbed by the material is very small and the improved closed vessels can ensure very limited short circuiting or dead spots. The major operational problem with UV systems, especially closed-vessel units, is the build up of coatings on the lamp surface which could dramatically reduce its effectiveness, regardless of its age (Cheremisinoff and Young, 1975; Ellis, 1991). The solution to this short coming is the cleaning which can be done by several methods. In physiochemical cleaning, a pressure spray is used with the addition of detergents or acids. This first method is recommended by EPA because it is simple and cheap (Bohn, 1991). System may need to be kept out of service while being cleaned. Secondly, ultrasonic device can be installed inside the unit to disturb deposition. In the third method, mechanical cleaning, mechanical wipers utilising rubber rings are used in quartz systems (Cheremisinoff and Young, 1975).

Types of lamps

Generally the lamps have either hot cathodes which are used when a low intensity (I) is required (e.g. disinfection of air) and cold cathodes which have a longer life and are relatively not affected by the number of starts (Ellis, 1991). Most of the energy is used to produce light at WL of 254 nm and these lamps operate at room temperature with optimum temperature of 40 C (Czarnecka-Nieminska, 1985).

The type of lamp employed has an important role in disinfection efficiency. Therefore, lamp selection has been the subject of many studies. Low pressure mercury lamps (LPML) and medium pressure mercury lamps (MPML) are the most widely used (Havelaar *et al.,* 1990). The term pressure refers to the internal pressure inside the tube. The LPML systems operate at low pressure (10^{-3}-10^{-2} torr), MPMLs operate at medium pressure (around 760 torr or 1 atmosphere), and HPML operate at high pressure (10^{3}-10^{4} torr) (Cairns, 1996).

Monochromatic lamps emit a single intensity (I) while for polychromatic lamps, the total I emitted is the sum of I for each WL (Cairns,1996). In monochromatic LPML, a narrow band of UVR is produced with a peak near 253.7 nm (i.e., single wave length emission) (Labatiuk *et al.,* 1992; Cairns, 1995). The full efficiency will be reached after a "burn in" period of about 100 hours (Bohn, 1991), which could mean newly started systems might not be fully efficient. On contrast, in polychromatic MPML, the WL produced is boarder (200 - 400 nm) but with an overall energy output greater than LPML (Ellis, 1991). These lamps emit about 40% of its light in that favourable wave length at an internal temperature of between 600 - 800 °C (Cairns, 1996). Both lamp types produce enough UV dose to inactivate the majority of microbes (Cairns *et al.,* 1995). Both systems perform equally well but each has its advantages in different applications (Wolfe, 1990).

MPMLs have a greater treatment capacity (about 25 folds) than LPMLs because of their greater I (10-20 times higher than LPML) (Wolfe, 1990). It is reported to be even as higher as 50-80 times more than LPML (Cairns, 1996). Therefore, they require much more smaller space which is a spectacular operational advantage over the LPMLs (Havelaar *et al.,* 1990; Ellis, 1991). Although they consume more energy than LPML, it is argued that the reduced cost of capital and lamp replacement compared to LPMLs compensate for their increased energy cost (Havelaar *et al.,* 1990). However, there are some contradictions in the literature about choosing a suitable system. For example, it is reported that LPML types are predominantly used because they are more energy efficient than MPMLs (Havelaar *et al.,* 1990; Cairns *et al.,* 1995). Energy loss by heat in LPML is lesser than in MPML and dose (D) assay may be easier due to the emission of monochromatic radiation compared to polychromatic in MPML. Nevertheless, MPML has been the choice of most wastewater treatment plants in North America, probably due to its compact size (Bohn, 1991). A single LPML produces an average 8800 uW/cm² at 65% transitivity, while MPML produces an average of 120,000 µW/cm² at the same transitivity (White *et al.,* 1986).

Even after the recent development of LPML, some disadvantages remain unsolved, such as the need for a large number of lamps and the large maintenance costs encountered (average one cleaning per week due to fouling) (Cairns and McGee, 1996). The authors added that collimated-beam studies show that MPML obtained a 3 logs higher bacterial kill than an equivalent unit length of LPML. From a case study in an actual wastewater treatment plant, Cairns and McGee (1996) reported that 400-500 LPMLs were replaced by only 32 MPMLs which required much less equipment than the previous LPML system, smaller space, and lower maintenance costs. This is a remarkable advantage of the MPMLs. It seems that LP systems are suitable for small installations (< 10,000 m^3d) while MP will serve better for larger installations (> 10,000 m^3d) because of space requirements (Cairns, 1996).

The average life expectancy for UV lamps in general range from between 6000 to 10,000 hours (Cheremisinoff and Young, 1975; Scheible and Bassell, 1979; Wolfe, 1990) as a result of ageing and due to deposits of burnt off electrodes in the inner material which decreases the UV emission (Ellis, 1991). Thampi (1990) suggested that the lamp life is longer in continuous operation. Ellis (1991) reported that LPMLs, unlike HPMLs or MPMLs, can be stopped and started immediately or in few seconds which is an important feature.

4.2 Factors affecting UVR disinfection efficiency

Disinfection efficiency can be affected by factors classified into primary and secondary factors. Primary factors are those that affect UV dose via attenuation of I which are related directly to the effluent quality.

Primary factors

The primary factors, having the biggest influence include organic matter content, suspended solids concentration, colour, and turbidity (Czarnecka-Nieminska, 1985; Harris, 1986; Cardenas *et al.*, 1986-87; Savolainen, 1991; Darby *et al.*, 1993; Dizer *et al.*, 1993).

Sewage contains substances such as proteins, phenolic compounds, and urea which absorb UV light at 253.7 nm (Czarnecka-Nieminska, 1985; Harris, 1986; Wolfe, 1990; Jackson, 1994; Cairns *et al.*, 1995). These parameters reduce UVR transmittance through the disinfected effluent (Czarnecka-Nieminska, 1985; Cairns *et al.*, 1995). For example, Qualls *et al.* (1985) showed that survival of indicator organisms were inversely related to number of particles/ml of sample of irradiated secondary effluent. In an earlier study, in 1983 they noted that the differences in disinfection efficiencies of raw wastewater and secondary effluents resulted from differences in particle size of both qualities.

Turbidity and SS scatter or absorb UVR, hence, have an attenuation effects on I. Their concentration in UV disinfected effluents correlated negatively with UV transmittance (Jackson, 1994). SS can also shield bacteria from UVR thus aggregation of bacteria and viruses in SS provide some degree of protection (Qualls *et al.*, 1983; Harris *et al.*, 1987; Wolfe, 1990). Scheible and Bassell (1979) and White *et al.* (1986) suggested that the SS and BOD of UVR disinfected effluents should be below 30 mg/l in order to meet microbial standards set for restricted irrigation (FC 200/100). Positive statistically significant correlations have been observed between turbidity, absorbance, and survival rates of indicator organisms after UVR of wastewater effluents (Harris *et al.*, 1987; Darby *et al.*, 1993). Absorbance of disinfected effluents is also a function of dissolved organic matter (Thampi, 1990; Savolainen, 1991).

Secondary factors

Secondary factors are those that affect UVR effectiveness, performance or efficiency indirectly and as a result of accumulation in the course of time. These factors include operational factors, such as lamp configuration, lamp age, mixing regime, quantity of flow (Q) and other factors. For example, intensity of UVR is affected by lamp age. It is reported that UV dose efficiency could be reduced by 0.6% a day due to ageing of lamp and by 2.0% a day due to fouling (White *et al.*, 1986; Savolainen, 1991).

Temperature and pH can affect the performance of UVR systems indirectly by reducing I (Czarnecka-Nieminska, 1985). Higher and lower temperatures may change the vapour pressure and will reduce output by 1 - 3% for each degree (C) (Thampi, 1990*)*. Extreme changes in the background working temperature in a UV systems may cause a shift to a different wave length (Bohn, 1991). However, Severin *et al.* (1983a) studied the effect of temperature on the irradiation of *E. coli* at three different temperatures; 5 C, 20 C and 35 C, and concluded that *E. coli* showed a relative insensitivity to radiation at temperature change of that range. Moreover, Abu-Ghararah (1994) observed lower resistant of FC indicator bacteria at higher operation temperatures in the range of 10 to 45 C which is a counter-balance effects of relatively higher effluent temperatures in hot climates. The author concluded that lower temperature (10 °C) required higher UV dose to achieve same efficiency obtained at 45 °C.

In conclusion, the primary factors affecting UVR efficiency can be controlled by the use of sand filtration before UV disinfection to insure a consistent effluent quality reaching the UV system, to account for high seasonal variations. Nieuwstad *et al.* (1991) stressed that higher effluent transmittance will save costs of UV disinfection treatment in the long run. Effects of secondary factors can be mitigated by tertiary filtration as well, but could be controlled by good monitoring, operation, and maintenance programs.

4.3 Mechanisms of UV inactivation

Damages to DNA or its components is believed to be the main inactivation mechanism in UVR disinfection (Scheibel and Bassell, 1979; Severin *et al.*, 1983b; White *et al.*, 1986). When a cell is exposed to UV photons, new bonds (dimers) are formed between two adjacent thymine monomers in the DNA strand. The new bond inhibits further replication and consequently, the cell is “inactivated”. Pyrimidine bases thymine (T) and cytosine (C) are more sensitive to UV than purine bases (almost by factor of 10) (Friedberg, 1985; Mitchell and Karentz, 1993; Holm-Hansen *et al.*, 1993). Therefore, the wave length 240 - 290 nm radiation kills most organisms by altering the DNA or RNA, which are so large molecules that they are easily hit by UVR photons, thus blocking the normal cell division. The reaction is that DNA absorbs light strongly at this region with a peak at 254 - 260 nm. In addition, the induction of ATP damage by UVR is reported to occur in UVR microbial cells (Holm-Hansen *et al.*, 1993).

Most living cells are equipped with special mechanisms and hence naturally capable of instantaneous repair of a certain amount of DNA injury. The repair mechanisms: photoreactivation, dark repair and post replication repair have been presented in details elsewhere (Friedberg, 1985; Harris *et al.*, 1987 Mechsner *et al.*, 1991; Mitchell and Karentz, 1993). The short-term photoreactivation is the most pronounced type (Scheible and Bassell, 1979; Ellis, 1991; Mechsner *et al.*, 1991). However, it is a function of the dose but many micro-organisms do not possess photoreactivation ability (Baron, 1997)

5. Cost analysis of alternative means of disinfection

Capital cost of UV systems are reported to be only 10% higher than chlorine systems of the same capacity (Zukovs *et al.,* 1986; Cardenas *et al.,* 1986-87). The operational costs were found to be similar for both systems. However, it is argued that total costs (capital, operation and maintenance) of UV systems will be much less with the use of high effluent quality. This is attributed to the need for lower dosage and contact time, hence, smaller size UV systems will be needed. The role of sand filtration is obvious to cut running costs (operation and maintenance). Generally, UV costs have been found to be less than that of O_3 and ClO_2 due to lower installation and operational costs (Whitby *et al.*, 1984; Wolfe, 1990, Blatchely *et al.,* 1996) as it appears from Table 1.

Table 1 Relative unit costs comparisons of different disinfectants with respect to UV systems

Cost	UV	Chlorine	Chl. / Dechl.	O_3	ClO_2	Author
Capital	1	1.45	1.80	7.27	-	(Whitby *et al.*,1984)
Operation	1	2.11	2.84	6.73	-	(Whitby *et al* 1984)
Total	1	0.32	1.2	1.26	1.63	(Wolfe, 1990)

Conclusions

UV disinfection systems have proved to provide effective, efficient and practical engineering practice (Scheible and Bassell, 1979; Qualls *et al.,* 1985; Thampi and Sorber, 1987; Qualls *et al.*, 1989; Wolfe, 1990; Ellis, 1991; Jackson, 1994). Even though, the benefits and contributions of Cl_2 to public health can not be denied, total elimination of Cl_2 disinfection practice is considered due to environmental concern. UV emerges as an alternative environmental-friendly disinfectant (Scheible and Bassell, 1979). Three major factors make UV disinfection increasingly popular over the other methods: the adverse environmental effects of chlorine, the high cost of ozone, and the advancement of UVR systems through research and development which overcomes its previous shortcomings (Harris *et al.,* 1987; Ellis, 1991). This is manifested in an increase UV permits in many countries as a water and wastewater disinfection technique (Harris *et al.,* 1987). For example, a single lamp can handle a flow rate up to 220 m^3/d (Cheremisinoff and Young, 1975) (i.e., population equivalent 1500-2000 capita) and by arranging units in parallel or series, higher flow rates can be treated.

A comparison between UVR, Ozone, and Chlorine which are the most widely used for water as well as wastewater disinfectants is presented in following Table 2.

Table 2. Comparison of the alternative means of disinfections.

	UV	OZONE	CHL. / DECHL.
Capital cost	low	Highest	low
Operating cost	lowest	High	low
Efficiency	excellent	Unreliable	good
Virudice	good	Good	poor
Interference, nitrogen	no	Yes	yes
By-products	none	unknown	THM
Effect of TSS	average	high	medium
pH effect	no	moderate	yes
Temperature effect	no	moderate	yes
Toxic chemicals	no	yes	no
Operational problems	low	high	average
Complexity	simple	complex	moderate
Corrosivness	no	yes	yes
Safety on site	minimum	moderate	substantial
Toxicity	none	high	high
Contact time	2-6 sec.	10 min.	30 - 60 min.
Dosage	10 - 30 J/cm^2	10 - 50 mg/l	5 - 20 mg/l
Capacity	large-scale	small-scale	large-scale

(from Czarnecka-Nieminska , 1985; EPA, 1992)

Research is underway to explore disinfectants for water and wastewater other than the conventional previously described practices. Unconventional disinfectants that might be the future practices include the exploitation of solar energy (Davies-Colley *et al.,* 1994; Acher *et al.*, 1997), high energy electrons, (Farooq *et al.,* 1993), gamma radiation (Pandya *et al.,* 1987; Farooq *et al.,* 1993), pulse electric field (Grahl and Märkl, 1996), photocatalytic oxidation (Bekbölet and Araz, 1996), high heat (Fayer, 1994), and electrochemical treatment (Patermarakis and Fountoukidis, 1990). However, the practices and efficiencies of these systems are not proved and their research work is still very preliminary.

تعقيم مياه الصرف الصحي المعالجة لأغراض إعادة الاستخدام

صالح بن محمد المقرن

مستشار بوكالة المياه – وزارة المياه والكهرباء – السعودية

توجد كثير من الممرضات في مياه الصرف الصحي والتي تصل إلى محطات المعالجة بأعداد كبيرة، وعـــادة ما تتخطى مراحل المعالجة المختلفة بما فيها الترشيح. وفي مياه الصرف الصحي الغير معالجة (الخام) يكون متوســـط عدد بكتريا القولون 1×10^{7} لكل 100 ملل، ويمكن لمحطة معالجة صرف صحي تدار وتشغل بطريقة سليمة أن تقضي على 99% من هذه البكتريا والتي تعتبر مؤشراً لنوعية المياه الأحيائية ليصبح متوسط عددها بالتالي في ميـــاه الفائض هائلاً أيضاً 1×10^{5} مما يستوجب عمل تعقيم لتلك لامياه قبل استخدامها لتكـــون مـــصادر مـــستدامة للاستخدامات المختلفة وعلى رأسها الزراعية . وفي هذه الدراسة يتم إبراز المعقمات أو المطهرات التالية : الأشـــعة فوق البنفسجية الكلور والأوزون من خلال القيام بتحليل فاعلية واقتصادية كل منها بهدف وضع خط منطقـــي لاختيار أنسبها من منظور عناصر مختلفة وزوايا عدة تتعرض لها الدراسة.