Chapter 2

Literature review

2.1 Introduction

Mara (1976, 2004) described waste stabilization ponds as large shallow basins enclosed by earthen embankments in which wastewater is biologically treated by natural process involving pond algae and bacteria. Anaerobic, facultative and maturation ponds are the three principal types of waste stabilization ponds. These pond systems are normally arranged in a single or parallel series depending on the requirements of the final effluent quality. A long hydraulic retention time is employed to treat wastewater due to the slow oxidation rate of organic wastes. Typical hydraulic retention times range from 4 days to 100 days depending on the temperature of a particular region.

Waste stabilization ponds are the most effective, appropriate wastewater treatment facilities in areas with warm climates where sufficient land is available and where the temperature is most favourable for their operation (Mara, 1976). Waste stabilization ponds have also been used widely in cold climate regions (Abis, 2002). Mara (2004) describes in detail the overwhelming advantages of waste stabilization ponds over conventional sewage treatment plants in terms of capital cost, operation and maintenance costs, pollutants removal and the heavy metal removal. However, waste stabilization ponds release odour when they are overloaded.

2.1.1 Anaerobic ponds

Anaerobic ponds are the first ponds in a series of waste stabilization ponds that receive raw sewage. Geometrically, anaerobic ponds are constructed to have a ratio of length to width of about 1- 2:1 with a depth ranging from 2 to 5 m. They receive a high organic BOD loading of more than 100 g BOD_5/m^3/day. The suspended solids settle by gravity to the bottom of pond where they are oxidised anaerobically due to the absence of dissolved oxygen.

Mara and Pearson (1998) recommended that anaerobic ponds should be used in warm climates arguing that well designed and properly maintained anaerobic ponds achieve 60 % BOD removal at 20^{o}C and over 70 % at 25^{o}C. Research findings by Silva (1982) reported that anaerobic ponds operated extremely well at short retention times and it was not good practice to design anaerobic ponds at long retention times as this did not offer any advantage.

Many wastewater engineers are afraid to include anaerobic ponds in waste stabilization ponds series due to odour production caused by hydrogen sulphide gas (H_2S). Mara (2001, 2004) strongly advised that well-designed anaerobic ponds with pH of 7.5 do not produce odour. In addition, limiting sulphate compounds (SO_4) concentration in raw wastewater to less than 300 mg/l is one way of eliminating odour release in anaerobic ponds. This could be achieved if the source of potable drinking water complies with the WHO (2003) guideline, which limits the sulphate concentration in raw water to 250 mg/l.

2.1.1.1 Classic design approach of anaerobic ponds

Anaerobic ponds are designed using the volumetric organic BOD loading rate. This process design approach has been developed using experimental data of full-scale operational anaerobic ponds. Meiring *et al.* (1968) proposed that the permissible volumetric organic BOD loading rate should be within a range of 100 - 400 g/m^3/day to ensure the satisfactory treatment performance of anaerobic ponds. The upper limit of the volumetric organic BOD loading rate (400 g/m^3/day) was established to avoid the risk of odour release due to the hydrogen sulphide gas (H_2S).

Mara and Pearson (1986) observed that the sulphate (SO_4) concentration of 500 mg/l in the domestic wastewater was still capable of producing odour if the volumetric organic BOD loading rate of 400 g/m^3/day was used in the design. They suggested that the organic BOD loading rate should be reduced to 300 g/m^3/day to provide an adequate margin of safety against odour. However, Mara (2004) revised the design BOD loading rate from 300 g/m^3/day to 350 g/m^3/day following the WHO (2003) guideline for the drinking water, which limits the sulphate concentration in raw water to 250 mg/l. It was suggested that the design BOD loading rate (350 g/m^3/day) could not produce odour at the recommended sulphate concentration (250 mg/l). In addition, the design of anaerobic ponds would use less land compared with the earlier BOD loading rate (300 g/m^3/day).

However, it should be noted that this process design approach was not based on the actual hydraulic flow patterns that existed in anaerobic ponds. The design approach assumed that the completely mixed hydraulic flow pattern was realised in anaerobic ponds due to the use of the low ratio of length to width (1- 2:1) of the pond dimensions. Research findings by Peña *et al.* (2000, 2002); Tchobanoglous *et al.* (2003) have shown that the completely mixed hydraulic flow pattern is not achieved in operational anaerobic ponds due to the changes of wind velocity, temperature, influent momentum and the density variation of the wastewater despite the use of square shape dimensions in the anaerobic pond geometry.

Peña *et al.* (2000) observed that the mean hydraulic retention time is not attained in anaerobic ponds due to the existence of the hydraulic short-circuiting and stagnation regions that are inherent in many ponds. These hydraulic deficiencies are significant when the sludge volume occupied more than 50% of the anaerobic pond volume. It can be inferred that the mean hydraulic retention time, which is designed using the volumetric organic BOD loading rate, is not achieved in practice despite the use of the completely-mixed hydraulic flow model.

2.1.2 Facultative Ponds

Facultative ponds normally follow anaerobic ponds in waste stabilization pond series. They are usually 1- 2 m deep and are geometrically designed to have a high ratio of length to width (up to 10:1) to simulate the hydraulic plug flow in these ponds (Mara, 2004; Mara *et al.* 2001). There are two types of facultative ponds: primary facultative ponds that receive raw wastewater and secondary facultative ponds that receive settled wastewater effluent from anaerobic ponds. The term facultative is used because a mixture of aerobic and anaerobic conditions is found in the pond (Mara, 2004). Aerobic conditions are maintained in upper layer while anaerobic conditions exist towards the bottom of the pond.

Mara (2004) reported that facultative ponds are coloured dark green due to large number of micro-algae. Facultative ponds may also occasionally appear red or pink when slightly overloaded due to the presence of anaerobic purple sulphide-oxidizing photosynthetic bacteria. It has been observed by Mara (2004) that the concentration of algae in a facultative pond depends on BOD loading and temperature. Chlorophyll *a* concentration in the range of 300 - 2000 µg per litre is normally found in a well operating facultative pond. Facultative ponds are designed for BOD removal on the basis of the relatively low surface BOD loading in the range of 100 - 350 kg/ha/day (˜ one tenth of the BOD loading rate in the anaerobic pond) to permit the development of a healthy algal population. This sustains the supply of oxygen by photosynthetic algae to the pond bacteria for BOD oxidation.

2.1.3 Design principles of facultative ponds

According to Mara (1976) and Marecos do Monte and Mara (1987), the process design of facultative ponds is based on rational and empirical approaches. The empirical design approach has been developed using performance data of existing waste stabilization ponds. Rational methods model pond performance by employing kinetic theory of biochemical reactions in association with the hydraulic flow regime.

2.1.3.1 Surface BOD Loading

McGarry and Pescod (1970) correlated performance data of primary facultative ponds from 143 different climatic conditions and reported that BOD removal was between 70 - 90%. The statistical model found that pond performance was related to the surface BOD loading with high correlation coefficient of 0.995. It was observed that the maximum surface BOD loading rate (λ_s), at which a primary facultative pond could become anaerobic (pond failure), was related to the ambient air temperature by equation 2.1:

$$\lambda_s = 60(1.099)^T \qquad 2.1$$

where:

λ_s = surface BOD loading (kg/ha/day)

$T =$ temperature (^{o}C)

Mara (1987) adopted the McGarry and Pescod's failure model by incorporating a factor of safety to ensure the safe design of facultative ponds. This model ensures that healthy concentration of algae is maintained in facultative ponds to avoid development of anaerobic condition. The tentative global equation that was developed by Mara (1987) is presented in equation 2.2:

$$\lambda_s = 350(1.107 - 0.002T)^{T-25} \qquad 2.2$$

where:

λ_s = surface BOD loading (kg/ha/day)

$T =$ temperature (^{o}C)

Determination of the facultative pond area (A_f) is derived using equation 2.3:

$$A_f = \frac{10 L_i Q}{\lambda_s} \qquad 2.3$$

where:

L_i = influent BOD_5 concentration (mg/l)

$Q =$ mean flow (m^3/day)

The mean hydraulic retention time (θ_f) is calculated using equation 2.4, which is based on the pond volume ($A_f D_f$) and the mean influent and effluent flow ($0.5(2Q_i$- $0.0001eA_f$)):

$$\theta_f = \frac{2 A_f D_f}{(2Q_i - 0.001 e A_f)} \qquad 2.4$$

where:

D_f = depth of facultative pond (m)

$e =$ net evaporation (mm/day)

θ_f = retention time (days)

2.1.3.2 von Sperling's design approach to facultative ponds

von Sperling (1996) noticed that input design parameters used in deriving equations (2.1- 2.4) are not known with high certainty in developing countries due to the limitation of research resources. He proposed an 'uncertainty principle' to design facultative ponds based on random design values selected in a range of each design parameter depending on the degree of its certainty. In this method, Monte Carlo design simulations are employed to design facultative pond area, mean hydraulic retention time and the effluent BOD concentration. Banda *et al.* (2005); Sleigh and Mara (2003) compared the design of waste stabilization ponds using modern and classic methods. It was found that modern design methods are appropriate when upgrading existing waste stabilization ponds and new waste stabilization ponds should be designed using classic methods.

Although Monte Carlo design simulations give confidence to the resultant designs so produced (area and effluent quality), the design approach assumes that the theoretical retention time is achieved during the operational period of the facultative pond. However, many researchers have observed that the theoretical retention time is not achieved in waste stabilization ponds due to the existence of the hydraulic short-circuiting and formation of stagnations (Mangelson and Watters, 1972; Lloyd *et al.* 2003). These hydraulic factors are inherent in many operational waste stabilization ponds. Effects of thermo-stratification and wind velocity are thought to cause the hydraulic short-circuiting that diminishes the treatment efficiency of facultative ponds.

Another weakness of both Monte Carlo design simulations and the surface BOD loading approach is the assumption that the complete-mix hydraulic flow pattern is initiated by effects of wind and thermo-stratification (Mara, 2004; Marais, 1974). Tracer experiments show that complete mix hydraulic regime is never achieved in facultative ponds by the mixing process induced by effects of wind and the thermo-stratification (Shilton, 2001; Marecos do Monte and Mara 1987; Mangelson and Watters, 1972).

It is worth noting that the classic design approach and Monte Carlo design simulations do not account for the effects of baffles or various types of inlet and outlet structures on the treatment efficiency of facultative ponds at the design or operational stages. As a result, there is no optimisation of the resultant design. There is a risk that the design of the facultative pond can require substantially more land than is actually necessary.

Mangelson and Watters (1972) present a detailed study of the hydraulic flow patterns of waste stabilization ponds that were fitted with various configurations of baffles and inlet structures using tracer experiments. They found that the 70% pond-width baffles when fitted across the longitudinal axis of the pond at a uniform separation improved the pond hydraulics significantly. In addition, there was a significant reduction in the volume of stagnation regions in baffled waste stabilization ponds. These benefits are not exploited in both classic and modern design approach when sizing facultative ponds.

Mara (2004) proposed a design model of predicting the effluent BOD concentration in facultative ponds based on the first-order kinetic reaction and this was combined with the complete mix hydraulic flow regime. This design model is presented in equation 2.5 as follows:

$$L_e = \frac{L_i}{(1 + K_1 \theta_f)} \qquad 2.5$$

$$K_{1(T)} = K_{1(20)} (1.05)^{T-20}$$

where:

$K_{1(T)}$ = first-order rate constant for BOD removal (day^{-1}) at temperature T ^oC

L_e = effluent BOD_5 (mg/l)

L_i = influent BOD_5 (mg/l)

$K_{1(20)}$ = 0.3 day^{-1} for primary facultative pond, 0.1 day^{-1} for secondary facultative pond at 20^oC

Equation 2.5 assumes that the hydraulic flow pattern in a facultative pond is a completely mixed. However, it is recognised that this hydraulic flow pattern is never achieved in operational facultative ponds (Thirumurthi, 1969). As a result, the design engineer does not have confidence of the predicted BOD effluent quality. Nevertheless, the first-order rate constant for BOD removal proposed by Mara (2004) can be confidently used in CFD to simulate the BOD removal in facultative ponds (Chapter 3).

2.1.3.3 CFD-based design of facultative ponds

CFD is generic flow equations that can simulate precisely the hydraulic flow patterns in reactors such as facultative ponds. The simulation of the hydraulic flow pattern is achieved precisely because CFD equations are based on momentum, mass and energy conservation principles. CFD can also simulate precisely the pollutant removal in facultative ponds by including the source term function that represents the decay of *E. coli* or BOD_5 removal in the default scalar transport equation. Using CFD to design waste stabilization ponds, limitations of the classic design approach, Monte Carlo design simulations and the ideal flow patterns can be overcome (Chapter 5).

CFD models can assess more precisely the improved treatment performance of facultative ponds at both the design and operational stage compared with classic and modern design procedures when baffles of various configurations are fitted to modify the pond geometry (Shilton and Harrison, 2003a, 2003b).

Researchers have realised that thermo-stratification and wind effects initiate the hydraulic short-circuiting that diminish the treatment efficiency of facultative ponds. These effects are not included in the classic design approach and the complete mix hydraulic flow model when designing and assessing the treatment performance of facultative ponds. Incorporation of these effects into a process design method could be a significant way of assessing realistically the performance of facultative ponds under these effects.

Thermo-stratification effects can be simulated in the CFD model by including the wastewater density function that depends on the temperature of wastewater layers along the pond depth. The temperature profile in the pond can be obtained by installing *i -buttons* (temperature loggers) supplied by Maxim Integrated Products (UK), Ltd at the required intervals (Abis and Mara, 2006). The monitored temperature could be used to calculate the density of wastewater layers defined for thermo-stratification in the model.

Effects of wind velocity are included in the CFD model by applying a corresponding shear stress on the horizontal top surface of the pond. This becomes the frictional boundary condition of the model. The prevailing wind direction is taken into account by applying the wind speed in reference to the direction of the wastewater flow in the pond. This can be parallel or perpendicular to the direction of the wastewater flow in the pond. Design engineers could benefit tremendously using CFD in assessing the treatment efficiency of facultative ponds when effects of thermo-stratification and wind velocity are significant at the design and operational stages of facultative ponds.

2.1.4 Maturation Ponds

Maturation ponds are classically designed for excreted pathogen removal if the practice of unrestricted crop irrigation is required. Although models of pathogen removal are based on *E. coli*, which is not itself a pathogen, it has been established that *E. coli* is a suitable faecal indicator of bacterial and viral pathogens in wastewater (Feachem *et al.* 1983). The mechanisms of *E. coli* removal in waste stabilization ponds are very close to that of faecal viral and bacterial pathogens (Curtis, 1990). Marais' (1974) equation is usually used to model the *E. coli* removal in a series of anaerobic, facultative and maturation ponds. This equation is the recommended for designing maturation ponds (Mara, 2004). However, the Marais' equation has received strong criticisms from various researchers as being unrealistic and unsafe because the equation is based on the ideal complete mix hydraulic flow regime that is not realised in operational maturation ponds (Thirumurthi, 1974, 1969; Arceivala, 1983; Polprasert and Bhattaria, 1983; Nameche and Vasel, 1998; von Sperling, 1999). These researchers suggest that the dispersed hydraulic flow pattern is the most practical flow pattern that can be achieved in operational maturation ponds. It is proposed that the first-order rate constant of the *E. coli* removal should be combined with the dispersed hydraulic flow pattern model.

Pearson *et al.* (1995, 1996) and Mara (2004) argued that Marais' (1974) model could be safely used to design and evaluate the treatment performance of waste stabilization ponds that are optimally loaded. However, the treatment efficiency of waste stabilization ponds diminishes over time due to the BOD overloading (Banda *et al*. 2005). Consequently, the Marais' (1974) model is not precise in predicting the *E. coli* removal. When this happens, Monte Carlo design simulations and von Sperling's (1999) empirical equations should be used to assess the treatment performance of the overloaded waste stabilization ponds. Banda *et al.* (2005) advise that modern design approach is appropriate when upgrading the capacity of overloaded waste stabilization ponds.

2.2 Hydraulic flow patterns in waste stabilization ponds

The flow of wastewater in waste stabilization ponds is classified as (i) completely mixed, (ii) plug flow and (iii) dispersed (arbitrary) flow depending on the assumptions proposed by the designer. These classic hydraulic flow patterns are currently used for designing the hydraulic retention time in waste stabilization ponds. In addition, these hydraulic flow patterns are integrated with biochemical process of the first-order kinetic reaction to simulate the decay of *E. coli* and BOD_5 in waste stabilization ponds. It is interesting to note that the completely mixed flow pattern is mostly used to design and assess the treatment performance of waste stabilization ponds (Mara, 2004).

2.2.1 Completely mixed flow pattern

Levenspiel (1972) describes complete mixing as a reactor in which the contents are well stirred and uniform throughout. The completely mixed reactor produces effluent quality that has similar composition as fluid elements within the reactor.

Researchers have used the completely mixed flow pattern in simulating hydraulic flow patterns in waste stabilization ponds (Marais and Shaw, 1961; Marais, 1974; Mara, 1976). Marais (1974) used the completely mixed hydraulic flow pattern when deriving the model of *E. coli* removal in waste stabilization ponds and this model is currently used to design maturation ponds. The completely mixed model proposed by Marais and Shaw (1961) is given by the basic relationship shown in equation 2.6:

$$\frac{L_e}{L_i} = \left[\frac{1}{1 + K_{BOD_C}\theta_f}\right]^n \qquad 2.6$$

$$K_{BOD_C} = 0.3(1.05)^{(T-20)} \qquad 2.7$$

where:

L_e = effluent BOD_5 (mg/l)

L_i = influent BOD_5 (mg/l)

K_{BOD_C} = completely-mixed first-order rate constant for BOD removal (day^{-1})

θ_f = mean hydraulic retention time in facultative pond (days)

n = number of ponds in series

T = mean temperature of the coldest month (^oC)

Utilization of equation 2.6 requires the assignment of the effluent BOD concentration. Mara (1976) recommends that the effluent BOD concentration should be in a range of 50 - 100 mg/l (usually 70 mg/l). Equations 2.6 and 2.7 enable the mean hydraulic retention time to be calculated. The area of waste stabilization pond is determined using equation 2.8:

$$A_f = \frac{Q\theta_f}{D_f} \qquad 2.8$$

where:

Q = design flow (m^3/day)

D_f = facultative pond depth (1.5 - 2 m)

A_f = area of the facultative pond (m^2)

Mara (1976) suggested that the assumptions made by Marais and Shaw (1961), that the first-order rate of BOD removal remains constant over the mean hydraulic retention time was not realistic. They argued that the BOD first-order rate constant removal retarded exponentially with the hydraulic retention time. Levenspiel (1972) believed that completely mixed reactors could be achieved when the influent wastewater flow was ideal steady state. In reality, waste stabilization ponds receive quasi-steady state influent flow due to the daily variation of water usage and ground water infiltration and these factors prevent the development of steady state flow in waste stabilization ponds. This convinced Thirumurthi (1974) to propose the dispersed hydraulic flow pattern in simulating the hydraulic flow pattern in facultative ponds for the realistic determination of the mean hydraulic retention time.

Marais and Shaw (1961) suggested that wind mixing and temperature difference were principal factors that initiated the completely mixed flow in waste stabilization ponds. In contrast, Shilton and Harrison (2003a) and Tchobanoglous *et al.* (2003) argued that wind mixing and temperature difference without mechanical mixers could not develop the completely mixed flow pattern in waste stabilization ponds. However, Mara (2004) argued that the first-order kinetic removal of pollutants (*E. coli* and BOD) in waste stabilization ponds was well represented by the completely mixed model. This argument was supported by research findings of Pearson *et al.* (1995, 1996) who compared the accuracy of Marais' (1974) equation with the observed first-order rate constant removal of *E. coli* in facultative and maturation ponds. They observed that Marais' (1974) equation was sufficiently accurate to predict the observed effluent *E. coli* numbers in a series of waste stabilization ponds that were optimally loaded.

2.2.2 Plug flow pattern

Levenspiel (1972) defined the plug flow pattern as the ideal steady state flow reactor that was characterised by orderly flow of fluid elements with no elements of fluid overtaking or mixing with other elements ahead or behind. Plug flow pattern has internal mixing of fluid but no mixing or diffusion along the flow path takes place. It was suggested by Levenspiel (1972) that the necessary condition for plug flow development was for the residence time in the reactor to be same for all fluid elements.

Plug flow hydraulic regime was considered as the rational basis of designing efficient waste stabilization ponds (Thirumurthi, 1969). This design approach is the most efficient as it ensures that wastewater pollutants attain the theoretical hydraulic retention time. Plug flow model tries to eliminate the hydraulic short-circuiting and stagnation regions formation that are inherent in many waste stabilization ponds. Thirumurthi (1969, 1974) strongly recommended that waste stabilization ponds be designed based on a plug flow model in order to maximise their hydraulic performance and treatment efficiency. Plug flow pattern can be achieved by fitting a large number of conventional baffles in a waste stabilization pond such that the width of the baffle openings and baffle compartments are the same. However, there is a performance risk that BOD overloading may be initiated in the first baffle compartment because the installation of baffles could concentrate the influent BOD_5 in the baffle compartments.

Reed *et al.* (1988) used the plug hydraulic flow model to design primary facultative ponds and their model is presented in equation 2.9:

$$\frac{L_e}{L_i} = e^{-K_{BODP}\theta_f} \qquad 2.9$$

where:

L_e = effluent BOD (mg/l)

L_i = influent BOD (mg/l)

K_{BOD_P} = plug flow first-order rate constant for BOD removal (day^{-1})

K_{BOD_P} is related to any temperature as shown in equation 2.10 as follows:

$$K_{BOD_P} = K_{BOD_{P20}}(1.06)^{T-20} \qquad 2.10$$

$K_{BOD_{P20}}$ = first-order rate constant of BOD_5 removal at 20 ^oC (day^{-1})

T = mean temperature of the coldest month (^oC)

Reed *et al.* (1988) suggested that the $K_{BOD_{P20}}$ value depended on the surface BOD loading rate and advised that, if the value of $K_{BOD_{P20}}$ was not known, a value of 0.1day^{-1} could be confidently adopted. The plug flow model calculates the mean hydraulic retention time required for the specified BOD removal requirements. If the average design flow is known, the required pond volume may be calculated by multiplying the average design flow and the mean hydraulic retention time. The limitation of this plug flow model in warm climate areas is the limited organic surface BOD loading rate of 112 kg/ha/day proposed. This model could be inappropriate in warm climate regions where higher organic BOD surface loading rate has been reported to be appropriate (Mara, 2002 and 2004).

The plug hydraulic flow regime is considered as unrealistic because zero longitudinal mixing of wastewater flow in waste stabilization ponds is difficult to achieve (Thirumurthi, 1969; Thirumurthi, 1974; Marecos do Monte and Mara, 1987). Wehner and Wilhelm (1956) argued that plug flow conditions could only be achieved if the length of the wastewater travel in the pond was infinity. However, all facultative ponds have finite limited lengths. Infinite length of liquid travel cannot be attained in practice and the proposed plug flow model was indeed difficult to achieve in practice.

Research has been carried out with the use of baffles to increase the distance of the wastewater flow in facultative and maturation ponds (Pearson *et al.* 1995, 1996; Reed *et al.* 1988; Mangelson and Watters, 1972). Although these researchers suggested that baffles could initiate plug flow in waste stabilization ponds, surprisingly there is no design procedure that recommends the number, position and length of baffles that could form plug flow in waste stabilization ponds.

Although the approximate plug flow model could be achieved by fitting a large number of conventional baffles in facultative ponds, some researchers (Shilton and Harrison, 2003a; Banda *et al.* 2006a) have expressed concern of the possibility of BOD overloading in the first baffle compartment. This might reduce the treatment efficiency of the plug flow pond as the algae concentration could decrease with increased loading of ammonia and sulphur concentration (Pearson *et al.* 1987). This potential risk of BOD overloading in the first baffle compartment has not been researched in the operational facultative ponds. It is not surprising to note that researchers recommend the use of conventional baffles in maturation ponds where BOD removal is not the primary objective.

2.2.3 Dispersed hydraulic flow regime

Mara (2004) argues that the ideal steady state of complete mix and plug flow pattern are difficult to achieve in practice. Levenspiel (1972) observed that real reactors never fully follow an ideal steady state flow regime. It was noted that deviations from the ideal flow regime are quite considerable. These deviations from the ideal steady-state are caused by channelling of wastewater from the inlet to outlet, recycling of wastewater and creation of stagnation regions that are inherent in all reactors including waste stabilization ponds.

Thirumurthi (1969) recommended that waste stabilization ponds be designed as dispersed flow reactors because waste stabilization ponds are neither plug flow nor completely mix reactors. He proposed the use of the first-order equation of Wehner and Wilhelm (1956) when designing facultative ponds. The proposed dispersed hydraulic flow model is presented in equations 2.11 – 2.14 as follows:

$$\frac{L_e}{L_i} = \frac{4ae^{\frac{1}{2d_f}}}{(1+a)^2 e^{\frac{a}{2d_f}} - (1-a)^2 e^{\frac{-a}{2d_f}}} \qquad 2.11$$

$$a = \sqrt{(1 + 4K_{BOD_D}\theta_f d_f)} \qquad 2.12$$

$$K_{BOD_D} = K_{BOD_{D20}}(1.09)^{T-20} \qquad 2.13$$

The degree of inter-packet mixing that takes place is expressed in term of a dimensionless ‘dispersion number’ defined as:

$$d_f = \frac{D}{vl} \qquad 2.14$$

where:

L_e = effluent BOD (mg/l)

L_i = influent BOD (mg/l)

K_{BOD_D} = dispersed flow first-order rate constant for BOD removal at T temperature (day^{-1})

$K_{BOD_D 20}$ = dispersed flow first-order rate constant for BOD removal at 20 °C temperature (day^{-1})

θ_f = mean hydraulic retention time in facultative pond (days)

d_f = dispersion number

D = coefficient of longitudinal dispersion (m^2/h)

v = mean velocity (m/h)

l = mean path length of a typical particle in the pond (m)

T = minimum pond temperature in the coldest month (°C)

Thirumurthi developed a chart to facilitate the use of the complicated equation 2.11 where ($K_{BOD_D} \times \theta_f$) is plotted against the percentage of the BOD remaining in the effluent for various dispersion numbers, d_f varying from zero for a plug flow to infinity for a complete mixed reactor.

The observed dispersion numbers in waste stabilization ponds range from 0.1 to 4 with most values not exceeding 1.0. The difficulty which is encountered in designing facultative ponds using the dispersed hydraulic flow model lies in the fact that at design stage, the value of dispersion number (d_f) and the first-order rate constant for BOD removal (K_{BOD_D}) are not known. Thirumurthi proposed that values of K_{BOD_D} should be developed based on various environmental conditions that are known to be toxic to the pond ecology. This is cumbersome and expensive laboratory work such that the resulting value of K_{BOD_D} could not be determined with high level of accuracy. Dispersion number has been suggested to be obtained from tracer experiments in existing waste stabilization ponds. Interestingly, at the design stage of new waste stabilization ponds, dispersion number may not be determined since there could be no existing waste stabilization ponds with similar BOD loading conditions and geographical location.

Recently, von Sperling (2002) compared four dispersion number models proposed by Polprasert and Bhattarai (1985); Agunwamba *et al.* (1992); Yanez (1993) and von Sperling (2002). von Sperling used Monte Carlo design simulations to predict the variation of dispersion number of four models. It was found that the dispersion number models proposed by Polprasert and Bhattarai (1985) and Agunwamba *et al.* (1992) were not accurate in predicting the dispersion number. Dispersion number models developed by Yanez (1993) and von Sperling (2002) predicted accurately the dispersion number since they produced a narrow variation of dispersion numbers.

Dispersion number models developed by Polprasert and Bhattarai (1985) and Agunwamba *et al.* (1992) not only show a weakness by predicting a wide range of dispersion numbers, but they also require the designer to assume some design variables such as kinematic viscosity, shear velocity and flow velocity when using the equation. It is also considered that measurements of these design variables in existing waste stabilization ponds cannot be determined accurately. In addition, these design variables depend on other factors such as temperature and the influent momentum that vary significantly on a daily basis and this increases the inaccuracy of the prediction.

von Sperling's (2002) dispersion number model that was developed for facultative and maturation ponds is given in equation 2.15 as follows:

$$d_f = \frac{1}{\left(\frac{L}{W}\right)} \qquad 2.15$$

where:

d_f = dispersion number

L = pond length (m)

W = pond width (m)

It can be seen that equation 2.15 is much simpler to use in the hydraulic design of waste stabilization ponds because it requires dimensions of the pond geometry. von

Sperling’s (2002) model can be solved simultaneously with the Welhner and Wilhelm’s (1956) dispersed flow model to determine the retention time to design facultative pond area. One of the drawbacks with the use of the dispersed hydraulic flow regime is that the first-order rate constant of BOD removal changes exponentially over the hydraulic retention time and this could result into either over-design or under-design of facultative ponds (Mara, 1976). Another drawback of designing facultative ponds based on the dispersed hydraulic flow regime is that the dispersion number changes continuously due to the unsteady state of the influent momentum and effects of wind and thermo-stratification. These factors are not incorporated in the dispersion number model. It is not surprising to note that the predicted treatment performance of waste stabilization ponds that is based on the dispersed hydraulic flow pattern is not accurate (Buchauer, 2006).

Tchobanoglous *et al.* (2003) noted that effects of thermo-stratification initiate the hydraulic short-circuiting that diminishes the treatment performance of waste stabilization ponds. This often occurs when the influent wastewater flow entering the waste stabilization pond is either colder or warmer than the wastewater in the pond. Thermo-stratification effects cause the influent wastewater to flow directly to the pond outlet in a fraction of the theoretical hydraulic retention time without mixing into the full pond volume. It is surprising to note that the effects of thermal hydraulic short-circuiting are not included in the dispersed hydraulic flow regime.

Wind effect is another physical factor that is not taken into account in the dispersed hydraulic flow regime. It is known that wind speed and its prevailing direction induces shear stress at the top surface of the pond and this affects the hydraulic flow pattern in waste stabilization ponds (Wood, 1997). The extent of the wastewater mixing, which is initiated by the wind velocity can significantly change dispersion numbers and this could diminish the treatment efficiency of waste stabilization ponds. Wind velocity has also been noticed to cause hydraulic short-circuiting in waste stabilization ponds with a large surface area and a large inlet pipe (Shilton and Harrison, 2003a). Wind effects are not included in the dispersed hydraulic flow model when assessing the treatment performance of waste stabilization ponds. As a result, the prediction of the effluent quality from waste stabilization ponds cannot be guaranteed to meet the consent requirements set by the Environmental Protection Agency.

2.2.4 Tracer Experiments

2.2.4.1 Residence time distribution curves

Chemical engineering relies on the use of tracer experiments to predict the overall hydraulic flow pattern of the fluid in a reactor. Levenspiel (1972) recognised the need for precisely knowing the complete velocity distribution map for the fluid and argued that such knowledge could be excellent in predicting the hydraulic flow pattern of a reactor. Levenspiel (1972) admitted that there was no available design tool that could predict velocity distribution in a reactor. This was true at that time because CFD codes were not widely used due to the low performance of computer hardware. This led chemical engineers to use tracer experiments to study the hydraulic behaviour of fluid in a reactor.

Tracer experiments can be employed to obtain the mean hydraulic retention time and dispersion number in waste stabilization ponds. The experiment involves the addition of the tracer chemical at the pond inlet and its concentration is measured over time at the pond outlet. The principle of the tracer experiment is based on the fact that different fluid elements take different lengths of time in passing a reactor. This technique has been widely used by researchers to study the hydraulic performance of waste stabilization ponds (Mangelson and Watters, 1972; Lloyd *et al.* 2003; Shilton, 2001). Levenspiel (1972) advised that tracer experiments are suitable in ideal flow that has steady state without reaction and density change of a single fluid that passes through a reactor.

2.2.4.2 The mean hydraulic retention time

The mean hydraulic retention time ($\bar{t}$) in a waste stabilization pond is calculated using the normalised residence time distributions curve that is obtained from the tracer experiment (Levenspiel, 1956). The theoretical equation that is used to determine the mean hydraulic retention time in a waste stabilization pond is given by equation 2.16:

$$\bar{t} = \frac{\int_0^\infty tc\,dt}{\int_0^\infty c\,dt} \qquad 2.16$$

where:

t = time (seconds)

c = tracer concentration (mg/l)

$\bar{t}$ = mean hydraulic retention time (days)

For practical purposes, experimental data of the tracer concentration and residence time are taken at a number of discrete times and equation 2.16 is modified into equation 2.17:

$$\bar{t} \cong \frac{\sum t_i c_i \Delta t_i}{\sum c_i \Delta t_i} \qquad 2.17$$

where:

$\bar{t}$ = mean residence time (seconds)

c = tracer concentration (mg/l)

t_i = discrete time (seconds)

t_Δ = discrete time difference (seconds)

c_i = tracer concentration at time, t_i (mg/l)

The next most important hydraulic parameter that is obtained from the normalised residence time distribution curve is the variance (σ^2) of the mean residence time. This is obtained from equation 2.18:

$$\sigma^2 = \frac{\int_0^{\infty}(t-\bar{t})^2 cdt}{\int_0^{\infty} cdt} \qquad 2.18$$

Equation 2.18 is normally expressed in discrete form (equation 2.19) for practical purposes as:

$$\sigma^2 = \frac{\sum (t-\bar{t})^2 c_i \Delta t_i}{\sum c_i \Delta t_i} \qquad 2.19$$

Equation 2.19 is very useful because it is matched with the variance equation of theoretical curves of the residence time distribution curve given by equation 2.20 to obtain the dispersion number $\left(\frac{D}{ul}\right)$ in the pond.

$$\frac{\sigma^2}{\bar{t}} = 2\frac{D}{ul} - 2\left(\frac{D}{ul}\right)^2\left(1 - e^{-\frac{ul}{D}}\right) \qquad 2.20$$

where:

σ^2 = variance (seconds)

$\frac{D}{ul}$ = dispersion number

u = velocity (m/s)

l = length of the liquid travel (m)

Although a single value of the dispersion number is used to assess the hydraulic performance of the waste stabilization pond, researchers are aware that the dispersion number varies significantly over the residence time period of the pond. Its value depends significantly on the extent of the climatic condition and the influent momentum that exists on the day the tracer experiment was carried out. This weakness casts doubt on the accuracy of the tracer experiment in assessing realistically the hydraulic performance of waste stabilization ponds. As a matter of fact, the mean retention time that is obtained from the normalised residence time distribution curve is always less than the theoretical retention time $\left(\bar{t} < \frac{V}{Q}\right)$ due to formation of dead zones. This is thought to be caused by the effects of wind velocity, thermo-stratification and channelling flow pattern in the pond. These factors initiate hydraulic short-circuiting in waste stabilization ponds and so reducing the theoretical retention time (Tchobanoglous *et al.* 2003).

Tracer experiments have contributed significantly to the understanding of hydraulic flow patterns that exist in waste stabilization ponds (Shilton, 2001; Marecos do Monte and Mara, 1987; Mangelson and Watters, 1972). Tracer experiments explain qualitatively the effects of wind and thermo-stratification on the hydraulic and treatment efficiency of waste stabilization ponds. The experiments cannot provide reliable experimental data of the effluent quality as the daily climatic conditions play significant role in controlling the hydraulic flow pattern in the pond.

When the mean hydraulic retention time and dispersion number are significantly different from theoretical values, the decision is often made to improve the hydraulic performance of the pond. In most cases, baffle installation is proposed to reduce the hydraulic short-circuiting as this increases the effective retention time in the pond (Shilton and Harrison, 2003a).

Using tracer experiment as a hydraulic design tool, one should realise that this approach cannot be used at the design stage of new waste stabilization ponds because the experiment is carried out in existing pond systems. Interestingly, CFD can be used at the design stage of waste stabilization ponds to simulate tracer experiments using the time-dependent equation of the scalar transport equation or the species transport equation. Baléo *et al*. (1991); Shilton (2001); Salter (1999); Wood (1997); Sweeney (2004) all used CFD to simulate the residence time distribution curves in waste stabilization pond models. This design approach allows the designer to make informed decisions regarding ways of improving the hydraulic performance of waste stabilizations ponds rather than relying on tracer experiments that are difficult to achieve successfully in a full-scale waste stabilization pond.

It is worth noting that tracer experiments cannot assess quantitatively the treatment efficiency of waste stabilization ponds when effects of wind and thermo-stratification are significant. Interestingly, CFD models can include simulated effects of wind and thermo-stratification when assessing the hydraulic performance and treatment efficiency of waste stabilization ponds under these effects.

Although tracer experiments enable calculation of the mean hydraulic retention time in the pond, the technique cannot predict the retention time of wastewater in stagnation regions. As a result, it is difficult to assess the extent of the effective volume that is useful in the treatment of wastewater in the waste stabilization pond. CFD can calculate the residence time at all points in the pond and this can help designers to investigate physical design interventions that can minimise the extent of the hydraulic short-circuiting and stagnation regions in waste stabilization ponds (Baléo *et al*. 1991).

2.3 Effects of baffles on the performance of waste stabilization ponds

The current design procedures for waste stabilization ponds are not modified to include the improvement in the treatment efficiency and hydraulic performance that is initiated when baffles of various configurations are fitted in the pond. Mangelson and Watters (1972) investigated the hydraulic and treatment performance of waste stabilization ponds when various baffle configurations were fitted in a proto-type pond with dimensions of 12.19 m long, 6.10 m wide and 1.07 m deep. Eight different baffle configurations were investigated to determine the optimal baffle configuration that provided the maximum hydraulic and treatment efficiency in a proto-type pond. Figures 2.1 and 2.2 show the general arrangement of baffles in the two-baffle pond that employed the 60% pond-width baffles and the 90% pond-length baffles respectively. The baffle lengths, L_b and L_l were expressed as the percentage of the pond-width (6.10 m) and pond-length (12.19 m) respectively.

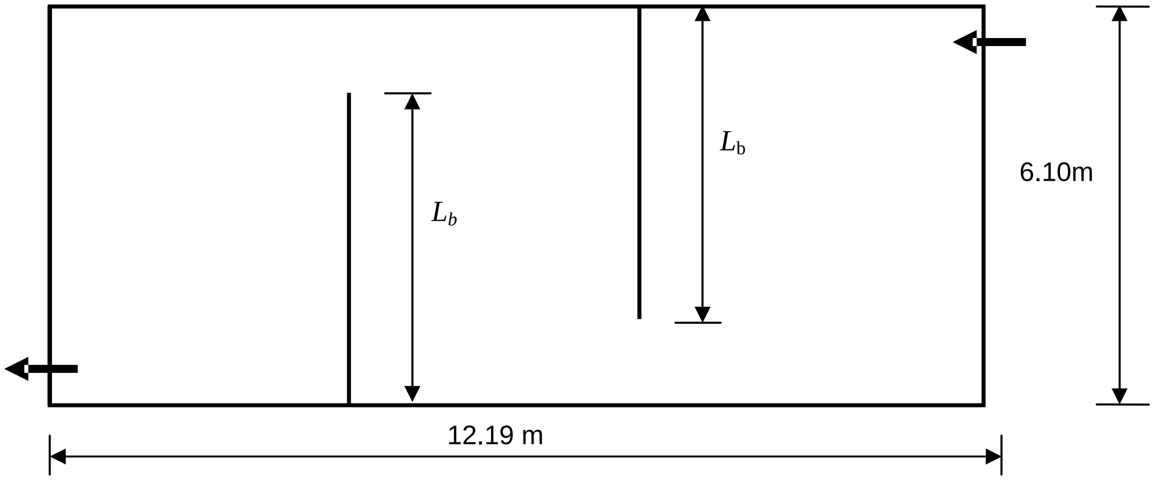

Figure 2.1 The general arrangement of the 60% pond-width baffles (L_b) in the two-baffle proto-type pond.

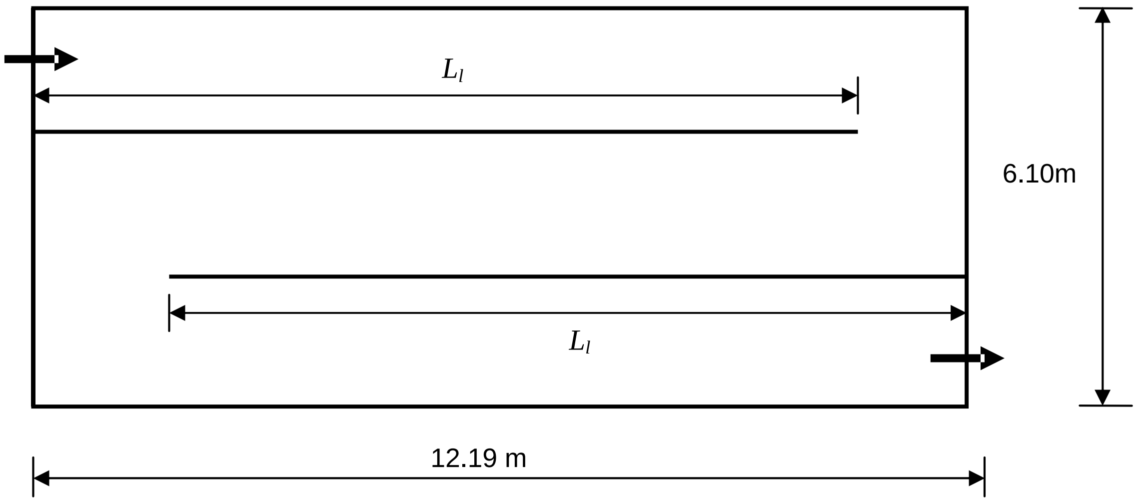

Figure 2.2 The general arrangement of the 90% pond-length baffles (L_l) in the two-baffle proto-type pond.

The first baffle configuration (Figure 2.1) was carried out using a two-baffle pond and was fitted with the 60% pond-width baffles (L_b = 3.66 m). The baffles were spread at a uniform separation along the longitudinal axis of the pond. The second configuration (Figure 2.2) employed the 90% pond-length baffles (L_l = 10.36 m) and the baffles were spread again at a uniform separation along the width of the pond. The treatment efficiency of the proto-type pond was defined as the ratio of the BOD removed to the influent BOD concentration while the hydraulic efficiency was defined as the ratio of the average retention time to the theoretical retention time of the pond. Tracer experiments were used to obtain the normalised residence time distribution curves for the computation of the average retention time.

The treatment and hydraulic efficiency of the unbaffled proto-type pond were 65% and 34% respectively and these were used as the baseline of the investigation. The treatment efficiency of the two-baffle pond with the 60% pond-length baffles and 90% pond-width baffles were 75% and 84% respectively while the hydraulic efficiency were 58% and 75% respectively. The results suggested that the longitudinal baffles (90% pond-length baffles) should be the first option when installing two baffles in waste stabilization ponds. However, the costs of procuring longitudinal baffles should be considered and evaluated with other costs such as the availability of the land before choosing this option.

The second investigation was carried out using a four-baffle pond. The pond was tested with the 60% pond-width baffles (L_b = 3.66 m) and the 90% pond-width baffles (L_b = 5.49 m) that were spread at a uniform separation along the longitudinal axis of the pond. The treatment efficiency of the pond with the 60% pond-width baffles and the 90% pond-width baffles were 83% and 81% respectively while the hydraulic efficiency were 72% and 70% respectively. It was concluded that increasing the length of baffles from 60% to 90% pond-width did not offer significant improvement in the treatment performance when baffles were placed along the longitudinal axis of the pond.

Six-baffle pond and eight-baffle pond were also tested with various baffle length. The six-baffle pond was tested with the 50% pond-width baffles (L_b = 3.05 m) and the 70% pond-width baffles (L_b = 4.27 m) while the eight-baffle pond was tested with the 70% pond-width baffles and the 90% pond-width baffles (L_b = 5.49 m). The baffles were spread at a uniform separation along the longitudinal axis of the pond.

The treatment efficiency of the six-baffle pond using the 50% pond-width baffles and 70% pond-width baffles were 87% and 89% respectively while the hydraulic efficiency were 70% and 79% respectively. The treatment efficiency of the eight baffle pond that used the 70% pond-width baffles and the 90% pond-width baffles were 88% and 84% respectively while the hydraulic efficiency were 86% and 81% respectively. Mangelson and Watters (1972) concluded that the 70% pond-width baffles were the optimal baffle length that should be adopted in order to improve the hydraulic and the treatment performance of waste stabilization ponds.

However, one should note that Mangelson and Watters (1972) did not investigate the effects of wind and thermo-stratification on the treatment and hydraulic performance of baffled waste stabilization ponds. In addition, performance assessments of *E. coli* removal were not carried out to evaluate the optimal baffle configuration that gives the maximum treatment efficiency in the baffled waste stabilization ponds that were investigated. Furthermore, assessment of BOD overloading condition in the baffle compartment was not carried out and this is likely to occur in a primary facultative pond that is fitted with a large number of baffles.

Although the performance assessments of the baffled laboratory-scale pond studied by Mangelson and Watters (1972) provides useful information regarding the hydraulic and treatment performance of full scale-baffled ponds, the results do not represent the realistic performance of full-scale baffled waste stabilization ponds. The recommended approach of evaluating the realistic assessments of the hydraulic and treatment efficiency of full-scale baffled waste stabilization ponds is to use pilot-scale ponds that mimic the conditions of full-scale waste stabilization ponds that are under the continuous influence of wind and thermo-stratification effects.

Kilani and Ogunrombi (1984) used laboratory-scale ponds to investigate the effects of baffles on the treatment and hydraulic efficiency of waste stabilization ponds. Baffles were fitted along the longitudinal axis of the pond. The dimensions of the laboratory-scale pond were 100 cm long, 50 cm wide and 10 cm deep. The unbaffled laboratory-scale pond was used as a control of the experiment. Three-baffle pond, six-baffle pond and nine-baffle pond were investigated. The publication does not indicate the length of baffles that were used in the pond. The treatment performance of the pond was assessed by observing the dispersion number, the effluent BOD and COD concentration in the pond effluent. It was noted that BOD removal increased with the increasing number of baffles. However, there was no significant improvement in the COD removal when baffles were installed in the pond. The BOD removals in the three-baffle pond, six-baffle pond and nine-baffle pond were 81%, 86% and 89% respectively while that of COD was 84%, 84.2% and 84.2% respectively.

The results of the dispersion number in the three-baffle pond, six-baffle pond and nine-baffle pond were 0.126, 0.112 and 0.096 respectively indicating the initiation of plug flow pattern with increasing number of baffles. The baffled-laboratory ponds developed isothermal condition due to the shallow depth (10 cm) that was used and this eliminated the effects of thermo-stratifications on the performance of the baffled ponds. The results of the baffled-laboratory ponds cannot be generalised to describe the treatment performance of full-scale baffled waste stabilization ponds that develop more frequently thermal stratification. Full-scale ponds are considerably deeper (1.5 – 2 m) than the laboratory ponds (0.1 m) that were studied. However, the results do indicate that facultative ponds can be fitted with baffles to enhance the hydraulic performance and BOD removals although the improvement in COD removal is not significant. In addition, there is an assurance that BOD overloading cannot be initiated in the first baffle compartment.

Pedahzur *et al.* (1993) investigated the treatment performance of the four-baffle secondary facultative pond (Kibbutz Sha'alabim pond) located in the Judea plains. The dimensions of the facultative pond were 50 m long, 30 m wide and 1 m deep. The average retention time of the secondary facultative pond was 5 days. Four baffles were fitted in the pond along the longitudinal axis of the pond. However, the publication does not indicate the baffle configuration (i.e., baffle spacing and length) that was employed. The treatment efficiency of the four-baffle secondary facultative pond was reported in terms of the microbiological and chemical quality of the influent and effluent. In addition, tracer experiments were used to assess the hydraulic performance of the baffled secondary facultative pond. The applied BOD loading was 340 – 950 kg per ha per day and this is more than 200% when compared with the recommended surface BOD loading suggested by (Mara, 1987).

It was observed that the installation of four baffles did not improve the hydraulic performance and treatment efficiency of the facultative pond. There was unsatisfactory removal of faecal coliform and BOD in the baffled secondary facultative pond. It was suggested that thermo-stratification initiated the hydraulic short-circuiting that reduced the hydraulic retention time in the pond. Although it was recognised that the facultative pond was overloaded by more than 200%, conclusions cannot be drawn to suggest that the hydraulic short-circuiting due to thermo-stratification effect was responsible for the poor performance of the baffled secondary facultative pond.

Results of CFD flow patterns (Chapter 5) show that hydraulic short-circuiting can be initiated in facultative ponds with poor design of baffle configurations. This may occur when the width of flow channel in baffled compartments is greater than that at baffle openings. However, the results of Pedahzur *et al.* (1993) showed that baffled secondary facultative ponds do not always perform satisfactory if they are overloaded and thermally stratified.

Pearson *et al.* (1995, 1996) investigated the treatment performance of complex arrangement of five-series waste stabilization ponds that employed different geometry, depth and the hydraulic retention times. Two anaerobic ponds were operated at volumetric loading of 187 g BOD per m^3 per day while the secondary facultative pond was operated at the surface organic loading of 217 kg BOD per ha per day. One of the tertiary maturation ponds was fitted with baffles such that the ratio of the effective length: breadth was greater than 100:1. It was observed that this baffled maturation pond was more efficient at faecal coliform removal than other tertiary maturation ponds.

Although the results of the baffled maturation pond were encouraging, conclusions could not be drawn to suggest that the treatment performance of baffled primary facultative ponds could be similar to that of baffled maturation ponds. It is known that the hostile environmental conditions that remove *E. coli* in maturation ponds are different from those found in facultative ponds. As a matter of fact, maturation ponds are designed based on pathogen removal while facultative ponds are designed for BOD removal. Nevertheless, the experimental data from baffled pilot-scale primary facultative ponds presented in Chapter 6 show that baffles can improve significantly the treatment efficiency and hydraulic performance of facultative ponds. This can be one area of optimizing classic design methods in reducing the land area requirements for the construction of waste stabilization ponds.

Muttamara and Puetpailboon (1996, 1997) evaluated the treatment performance of baffled laboratory ponds and baffled pilot-scale ponds. The laboratory-scale pond had dimensions of 150 cm long, 50 cm wide and 15 cm deep and this neglected thermo-stratification effects due to its shallow depth. The dimensions for the baffled pilot-scale pond were 30 m long, 3.3 m wide and 1.0 m deep. The laboratory-scale pond was investigated with two, four, and six baffle configurations while the pilot-scale pond was tested with six baffle configurations. The hydraulic efficiency and physicochemical parameters were used to assess the treatment performance of the baffled ponds. It was observed that the hydraulic efficiency in the pond increased with increasing number of baffles. The six-baffled pilot-scale pond indicated more plug flow conditions with a decrease in the dispersion number. The results revealed that more than 65% of total nitrogen and 90% of ammonia were removed in the six-baffle pilot-scale pond. It was concluded that the treatment performance of waste stabilization ponds can be increased significantly by installing baffles.

Other researchers (Zanotelli *et al.* 2002; Von Sperling *et al.* 2002, 2003) have also observed that baffles improve the hydraulic and treatment efficiency of waste stabilization ponds. However, these researchers did not investigate effects of wind velocity and thermo-stratification on the performance of baffled waste stabilization ponds.

Shilton and Harrison (2003a, 2003b) and Shilton and Mara (2005) adopted the findings of Mangelson and Watters (1972) in respect of the use of 70% pond width-baffles. A 2D-CFD model was used to assess the treatment performance of the baffled facultative pond. It was found that *E. coli* removal in the model increased with the number of baffles installed along the longitudinal axis of the pond. *E. coli* removals of 4.22 - 5.92 log units were obtained in a primary facultative pond with 2 - 4 baffles. For these cases, the width of flow channel in baffle compartments was greater than the width of flow channel at the baffle openings. Simulation of ponds with a large number (ten or more) of conventional baffles was not undertaken because the anticipated high effluent quality cannot be economically justified.

However, research into baffled facultative ponds with a large number of 70% pond-width baffles is very significant because performance assessments of the initiated plug flow pond can be evaluated against possible pond failure due to BOD overloading in the first baffle compartment. In addition, this configuration of baffled pond may form a width of flow channel in baffle compartments that is less than the width of flow channel at the baffle openings. In this situation stagnations can develop in the baffled pond due to a reduction of velocity magnitude at the baffle opening. This may reduce the effective pond volume and may reduce the expected pond performance.

It should also be noted that Shilton and Harrison (2003a, 2003b) and Shilton and Mara (2005) did not include the effects of thermo-stratification and wind velocity when assessing the treatment performance of baffled facultative ponds. These shortfalls have been investigated in this research.

2.3.1 Short-baffles in facultative ponds

Persson (2000) believed that short baffles when placed near the inlet could improve significantly the treatment and hydraulic performance of waste stabilization ponds. Using a 2D-hydraulic model of a hypothetical waste stabilization pond, it was found that a small island of 2% pond area (a circular baffle) located near the inlet achieved very little short-circuiting, with effective pond volume being 96% of the actual pond volume. However, experimental work was not carried to validate the results of the hydraulic model. One should also note that 2D-models are not sufficiently precise to predict the pond performance due to their inability to model the actual pond depth and the geometry of the inlet and outlet structures (Wood, 1997). 2D models use 1 m as the default dimension of pond depth, size of the inlet and outlet pipes. However, the depth and size of the inlet and outlet pipes in waste stabilization ponds could be different to the 1m dimension used in 2D models.

Shilton and Harrison (2003a, 2003b) found that short baffles near the inlet and outlet were as effective as two long baffles of 70% pond-width in reducing *E. coli* levels. The research focussed only on the 15% pond-width baffles. In addition, the optimum positions of short baffles from the inlet and outlet structures were not investigated. It is necessary to investigate a wide range of short baffles (15 - 30% pond-width) and the optimum baffle positions that could improve the treatment and hydraulic performance in waste stabilization ponds. These research limitations have been accounted for in Chapter 5. Although the use of short-baffles in waste stabilization pond seems promising, experimental work is required to establish whether BOD overloading can be initiated in the pond due to the installation of the short baffle near the inlet structure.

2.4 Effects of wind velocity on the performance of waste stabilization ponds

Brissaud *et al*. (2000, 2003); Frederick and Lloyd (1996); Lloyd *et al*. (2003); Vorkas and Lloyd (2000) noted that wind speed diminishes the treatment performance of waste stabilization ponds due to the initiation of the hydraulic short-circuiting. They observed that even at low wind speeds of 0.5 – 2.6 m/s, hydraulic short-circuiting can develop in waste stabilization ponds that are isothermal. Surprisingly, the classic and modern design methods do not account for the effects of wind speed and its prevailing direction when designing and evaluating the treatment performance of waste stabilization ponds (Banda *et al*. 2005).

Shilton and Harrison (2003a, 2003b) suggested that the momentum of the influent supplied by the inlet pipe can overcome wind effects thus obviating the concerns of the hydraulic short-circuiting caused by the wind effects. It was argued that the inlet momentum can sustain the flow pattern in waste stabilization ponds during the residence time period. However, validation of this theory on existing waste stabilization ponds under windy conditions was not carried out.

Research into wind effects on the hydraulic flow pattern has shown that wind produces a shear stress on top surface of pond and this alters the general flow pattern in the waste stabilization pond (Sweeney, 2004; Shilton, 2001). With advances and availability of computational technology, it is now recommended to use CFD to include wind speed and its direction when designing and evaluating the hydraulic flow pattern in waste stabilization ponds (Shilton, 2001; Sweeney, 2004; Wood, 1997). The superiority of CFD-based design of waste stabilization ponds over the classic and modern design methods is that wind shear stress can be included at the design or operational stage of waste stabilization ponds and detailed knowledge of the flow and the treatment performance in the ponds can be obtained.

Banda *et al.* (2006a) used a CFD model with the incorporation of wind effects to assess the treatment performance of a facultative pond in terms of *E. coli* removal. The wind effects were applied in the model as a shear stress across the top surface of the pond. The facultative pond was assumed to have isothermal conditions, so there was no short-circuiting associated with thermal-stratification. Shilton and Harrison's (2003a, 2003b) power theory was used to assess the significance of the inlet momentum and the wind effects with particular respect to the hydraulic flow patterns. Shilton and Harrison used a 30-day retention time to show that the inlet momentum was significant over the wind effects when the wind had a velocity of 2.8 m/s in a similar model.

However, Banda *et al.* (2006a) used a 4-day retention time to increase substantially the inlet momentum. Using Shilton and Harrison's power theory (2003a), the wind speed of 4 m/s provided power of 0.82 kW over the pond surface area of 640 m × 320 m. The power supplied by the influent momentum was 22 kW, so the contribution of the wind effects was 4%. It was argued that the effect of wind on the flow pattern of the wastewater flow was so small that the resulting flow pattern could be deemed to be sustained by the inlet momentum. With this significant inlet momentum, the wind effect was considered to be negligible in influencing the treatment performance of a facultative pond. Interestingly, the work demonstrated that with a wind speed of 4 m/s in the opposite direction to the wastewater flow, the *E. coli* removal was reduced by 31% compared to a facultative pond with no wind. Banda *et al.* (2006a) argued that if Shilton and Harrison's (2003a, 2003b) theory was satisfactory, the effects of wind on the *E. coli* removal could have been insignificant as the inlet momentum was 96% greater than the wind effects. The work of Banda *et al.* (2006a) demonstrated that wind speed could affect the treatment performance of waste stabilization ponds despite the high influent momentum that could control the flow pattern in the pond.

The design of waste stabilization ponds may be suboptimal if the effects of wind speed and direction are not taken into account in the geometric design of pond systems. It should be noted that even at a low wind speed of 0.5 m/s, Fredrick and Lloyd (1996) observed short-circuiting in ponds that had a 12-day retention time with isothermal conditions. The Shilton and Harrison hydraulic guidelines should be revised to include the effects of wind speed and direction on hydraulic flow patterns in waste stabilization ponds as the guidelines emphasize the significance of the influent momentum neglecting the wind effects. It is necessary to carry out further research of wind effects on the treatment performance of waste stabilization ponds. This could be a realistic way of designing waste stabilization ponds.

2.5 Effects of thermal stratification on the performance of waste stabilization ponds

Thermo-stratification is one of the main causes of the hydraulic short-circuiting that diminishes the treatment performance of waste stabilization ponds. Thermo-stratification is characterized with high vertical thermal gradients. It is argued that the high turbidity of the wastewater (mainly by algae) provides favourable conditions for the occurrence of thermo-stratification (Kellner and Pires, 2002). During the summer season, layers of wastewater near the pond surface receive a larger amount of thermal energy compared to the deeper layers and this results in a temperature difference between the surface and the bottom of the pond. Consequently, a density profile appears, with the less dense layers located at the surface of the pond and the densest ones close to the bottom. It is argued by Kellner and Pires (2002) that thermo-stratification in the water column induces alterations in the hydraulic flow pattern and a decrease of the useful volume of the pond. This often happens when thermally stratified wastewater layers remain stable for a considerable period of time. The influent may only mix with stratified wastewater layers of similar density and temperature properties. A significant pond volume may not be utilised in mixing the influent with the entire wastewater in the pond.

Bokil and Agrawal (1977) observed thermo-stratification in shallow waste stabilization ponds with depth of about 35.5 cm. The findings from this research contradicted the results of Kilani and Ogunrombi (1984) and Muttamara and Puetpailboon (1996, 1997) who found that the effects of thermo-stratification on shallow laboratory-scale ponds were not significant enough to influence the hydraulic and the treatment performance of waste stabilization ponds.

Fritz *et al.* (1980) developed a complicated model to predict temperature in waste stabilization ponds that develop thermo-stratification. One should note that the developed model cannot design accurately the temperature in waste stabilization ponds because it requires various input design parameters such as longitude, latitude, daily values of cloud cover fraction and relative humidity of a particular site that are difficult to obtain locally especially in developing countries where research resources are severely limited.

Gu and Stefan (1995) presented a detailed investigation of the types of thermo-stratification that occur in waste stabilization ponds. They used a simulation model that predicted the vertical temperature profile in the waste stabilization pond. Validation of the thermo-stratification model was carried out with the observed vertical temperature profile that was obtained in the pond. The correlation of the model with the experimental data was so satisfactory that the work demonstrated an innovative way of simulating effects of thermo-stratification in waste stabilization ponds.

Gu and Stefan (1995) observed that waste stabilization ponds can be completely mixed during consecutive day and night, or they can be stratified during the day and mixed during the night or they can be continuously stratified during several days and nights. The type of thermo-stratification that occurs continuously for several days and nights could adversely diminish the treatment performance of the pond due to the absence of the mixing of wastewater layers. The author believes that any model that simulates continuous thermo-stratification over the retention time period could provide realistic assessment of risks initiated by thermo-stratification on the treatment performance of waste stabilization ponds. CFD simulation of thermo-stratification effects in the pilot-scale primary facultative pond has been achieved by developing the wastewater density function that depends on the temperature of the wastewater layers in the pond (Chapter 6).

Pedahzur *et al.* (1993) observed that the effects of thermo-stratification reduced significantly the treatment performance of a secondary facultative pond that was fitted with four baffles. It was found that the hydraulic retention time in the baffled facultative pond was reduced substantially due to the short-circuiting caused by thermo-stratification. It was also found that the removal of BOD, *E. coli* and nutrients were not satisfactory.

Salter (1999) observed improved hydraulic performance of a two-baffle CFD model that simulated the effects of thermo-stratification. It was noted that the unbaffled CFD model indicated significant hydraulic short-circuiting caused by thermo-stratification effects. On the contrary, Abis and Mara (2006) found that effects of thermo-stratification did not diminish the treatment performance of the three unbaffled pilot-scale primary facultative ponds, which were operated for three years at Esholt sewage treatment works in Bradford. It is worth noting that these pilot-scale primary facultative ponds were operated at long retention times (60 – 90 days) and this would probably reduce the hydraulic short-circuiting associated with thermo-stratification effects as the velocities of wastewater in the pond were very small (0.026 - 0.038 m/s).

Recently, Banda *et al.* (2006b) used a 3D-CFD model to simulate thermo-stratification effects on the treatment performance of a pilot-scale primary facultative pond that was fitted with two and four baffle configurations. The temperature profile in the pilot-scale pond primary facultative pond was monitored using "*i-buttons*" that were supplied by Maxim Integrated Products (UK), Ltd. *I-buttons* were used to measure and record temperature at hourly interval. The observed temperature profile during the winter season showed minimal variation and was assumed to be constant at 5^{o}C. The wastewater density at this temperature was 1000 kg/m^3 (Perry and Green, 1984). However, thermo-stratification developed in the pond during the summer season. The temperature varied from 17^{o}C at the pond surface to 12^{o}C at the bottom level. A density-temperature model was developed based on data of Perry and Green (1984) for the range of temperature of 0^{o}C – 40^{o}C. This resulted in five different *E. coli* decay rates that were used in the model.

It was concluded that although the predicted CFD-log removals of *E. coli* in the model and the pilot-scale pond were not identical, the predicted effluent *E. coli* counts were in the same order of magnitude. The physicochemical parameters indicated that there was no significant difference in treatment efficiency of the pilot-scale pond with effects of thermo-stratification and isothermal conditions. It appeared that the hydraulic short-circuiting was not significant because of the long retention time (30 days) that was used. The results showed that CFD has the capacity of simulating precisely the treatment performance of waste stabilization ponds under thermo-stratification effects.

2.6 The basic CFD equations

CFD is a generic flow model that calculates velocity, temperature, pressure and scalar variables (tracer, *E. coli* numbers, BOD concentration etc) at all points in a reactor. The CFD equations are developed using conservation equations of mass, momentum and energy (Patankar, 1980; Versteeg and Malalasekera, 1995). Derivation of the CFD equations is based on partial derivatives techniques (Fluent Manual, 2003). The finite volume method is used to integrate the CFD differential equations to facilitate the computation of non-linear equations. Patankar (1980); Versteeg and Malalasekera (1995) provide detailed derivations of CFD equations in three dimensions for the time dependent flow with compressible fluids. Equation 2.21 presents the mass conservation equation for a fluid that exhibits unsteady state flow characteristics.

$$\frac{\partial \rho}{\partial t} + \frac{\partial(\rho u)}{\partial x} + \frac{\partial(\rho v)}{\partial y} + \frac{\partial(\rho w)}{\partial z} = 0 \qquad 2.21$$

where:

$\rho =$ density of fluid (kg/m^3)

$u =$ velocity in x direction (m/s)

$v =$ velocity in y direction (m/s)

$w =$ velocity in z direction (m/s)

$\partial x, \partial y, \partial z =$ differential change in distance (m)

$\partial t =$ differential change in time (s)

For an incompressible fluid in steady state conditions, the density ρ in equation 2.21 is constant and the equation becomes:

$$\frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z} = 0 \qquad 2.22$$

The second conservation equation that is used in CFD is the momentum equation. The momentum equation is developed based on the *Newton's Second law* of motion. Simplification of the momentum equation involves the use of the *Navier-Stokes* equation and is very useful for the application of the finite volume method. Three momentum equations are used to calculate the fluid velocity in the CFD model. In the X-direction, the momentum equation is written as:

$$\frac{\partial(\rho u)}{\partial t} + div(\rho u U) = \frac{-\partial p}{\partial x} + div(\mu grad u) + S_{mx} \qquad 2.23$$

where:

$$div(\rho u U) = \frac{\partial(\rho uu)}{\partial x} + \frac{\partial(\rho uv)}{\partial y} + \frac{\partial(\rho uw)}{\partial z}$$

$$div(\mu grad u) = \frac{\partial}{\partial x}\left(\mu \frac{\partial u}{\partial x}\right) + \frac{\partial}{\partial y}\left(\mu \frac{\partial u}{\partial y}\right) + \frac{\partial}{\partial z}\left(\mu \frac{\partial u}{\partial z}\right)$$

S_{mx} = momentum source term (N/m^3)

μ = dynamic viscosity (kg/m-s)

ρ = density (kg/m^3)

p = pressure (N/m^2)

U = vector velocity (m/s) = $[u, v, w]$

Additional momentum equations that are similar to equation 2.23 are used to calculate the fluid velocity in the Y and Z - directions in the CFD model.

The third equation that is used in the CFD is the energy conservation equation, which is based on the first law of thermodynamics. This equation is presented in equation 2.24 as:

$$\frac{\partial(\rho i)}{\partial t} + div(\rho i U) = -p\,div U + div(k grad T) + \Phi + S_i \qquad 2.24$$

where:

$$div(\rho i U) = \frac{\partial(\rho iu)}{\partial x} + \frac{\partial(\rho iv)}{\partial y} + \frac{\partial(\rho iw)}{\partial z}$$

$$pdivU = p\left(\frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z}\right)$$

$$div(kgradT) = \frac{\partial}{\partial x}\left(k\frac{\partial T}{\partial x}\right) + \frac{\partial}{\partial y}\left(k\frac{\partial T}{\partial y}\right) + \frac{\partial}{\partial z}\left(k\frac{\partial T}{\partial z}\right)$$

i = internal energy (J)

S_i = energy source term (J/s/m^3)

k = thermal conductivity (W/m/K)

ρ = density (kg/m^3)

p = pressure (N/m^2)

U = vector velocity (m/s) = $[u, v, w]$

T = temperature (K)

Φ = energy dissipation (J/s/m^3)

However, the internal energy variable (i) in equation 2.24 is related to temperature (T) and the specific heat capacity (C_v) as:

$$i = C_V T \qquad 2.25$$

where:

C_V = specific heat capacity of fluid (J/kg/K)

The solution of equations (2.21 - 2.25) are appropriate for a CFD model with laminar flow characteristics (i.e., Reynolds number less than 4000). However, when the Reynolds number at the pond inlet and outlet is more than 4000, the turbulent flow could be assumed in the model. Two additional turbulent equations such as the standard $k-\varepsilon$ are included in the CFD model (Patankar, 1980).

The standard $k-\varepsilon$ turbulent model has been extensively validated in a number of flow problems (Versteeg and Malalasekera, 1995). This equation is commonly used in the CFD-based design of waste stabilization ponds (Shilton, 2001; Salter, 1999; Sweeney, 2004). Launder and Spalding (1974) developed the standard $k-\varepsilon$ model to simulate turbulent flow characteristics in CFD models and these equations are presented as:

$$\frac{\partial(\rho k)}{\partial t} + div(\rho kU) = div\left[\frac{\mu_t}{\sigma_k} gradk\right] + 2\mu_t E_{ij}.E_{ij} - \rho\varepsilon \qquad 2.26$$

$$\frac{\partial(\rho\varepsilon)}{\partial t} + div(\rho\varepsilon U) = div\left[\frac{\mu_t}{\sigma_\varepsilon} grad\varepsilon\right] + C_{1\varepsilon}\frac{\varepsilon}{k} 2\mu_t E_{ij}.E_{ij} - C_{2\varepsilon}\rho\frac{\varepsilon^2}{k} \qquad 2.27$$

$$\mu_t = \rho C_\mu \frac{k^2}{\varepsilon} \qquad 2.28$$

where:

$$div(\rho kU) = \frac{\partial(\rho ku)}{\partial x} + \frac{\partial(\rho kv)}{\partial y} + \frac{\partial(\rho kw)}{\partial z}$$

$$div(\rho\varepsilon U) = \frac{\partial(\rho\varepsilon u)}{\partial x} + \frac{\partial(\rho\varepsilon v)}{\partial y} + \frac{\partial(\rho\varepsilon w)}{\partial z}$$

$$gradk = \frac{\partial k}{\partial x} + \frac{\partial k}{\partial y} + \frac{\partial k}{\partial z}$$

$$grad\varepsilon = \frac{\partial\varepsilon}{\partial x} + \frac{\partial\varepsilon}{\partial y} + \frac{\partial\varepsilon}{\partial z}$$

k = kinetic energy (J/s/m^3)

ε = dissipation of kinetic energy (J/s/m^3)

$C_\mu = 0.09; \sigma_k = 1.00; \sigma_\varepsilon = 1.30; C_{1\varepsilon} = 1.44; C_{2\varepsilon} = 1.92$

U = vector velocity (m/s) = $[u, v, w]$

The FLUENT manual (2003) and Patankar (1980) provide detailed derivations of the general scalar transport equation that is useful for the simulation of pollutant removal in waste stabilization ponds and the equation is presented as:

$$\frac{\partial(\rho\phi)}{\partial t} + div(\rho\phi U) = div(\Gamma grad\phi) + S_\phi \qquad 2.29$$

where:

$$div(\rho\phi U) = \frac{\partial(\rho\phi u)}{\partial x} + \frac{\partial(\rho\phi v)}{\partial y} + \frac{\partial(\rho\phi w)}{\partial z}$$

$$grad\phi = \frac{\partial\phi}{\partial x} + \frac{\partial\phi}{\partial y} + \frac{\partial\phi}{\partial z}$$

$$div(\Gamma grad\phi) = \frac{\partial}{\partial x}\left(\frac{\Gamma\partial\phi}{\partial x}\right) + \frac{\partial}{\partial y}\left(\frac{\Gamma\partial\phi}{\partial y}\right) + \frac{\partial}{\partial z}\left(\frac{\Gamma\partial\phi}{\partial z}\right)$$

$S_\phi = A + B\phi$ = source term of ϕ (kg/m^3/s)

$\phi =$ pollutant concentration (*E . coli* numbers per 100 ml or BOD_5)

$\Gamma =$ coefficient of diffusivity (kg/m/s)

$\rho =$ density (kg/m^3)

$U =$ velocity vector (m/s) = $[u, v, w]$

Integration of equation 2.29 over the control mesh volume (*cv*) yields equation 2.30:

$$\frac{\partial}{\partial t}\left(\int_{CV}\rho\phi dv\right) + \int_{CV} div(\rho\phi U)dv = \int_{CV} div(\Gamma grad\phi)dv + \int_{CV} S_\phi dv \qquad 2.30$$

where:

cv = mesh volume (m^3)

Versteeg and Malalasekera (1995) applied Gauss's divergence theorem to simplify equation 2.30 such that the face area of a cell was used in CFD computations. In waste stabilization ponds, steady state flow could be assumed because the concentration profile of wastewater pollutants in the pond does not change with time due to the long period of the pond operation. The simplified discrete form of the general scalar transport equation 2.30 is expressed as:

$$\frac{\partial}{\partial t}\left(\int_{CV}\rho\phi dv\right) = 0$$

$$\sum_{f}^{N_{faces}} \rho_f v_f \phi_f . A_f = \sum_{f}^{N_{faces}} \Gamma_\phi (\nabla\phi)_n . A_f + S_\phi V \qquad 2.31$$

where:

$N_{faces} =$ number of faces enclosing mesh

$\phi_f =$ value of ϕ convected through face f

$\rho_f v_f . A_f$ = mass flux through the mesh face (kg/s)

$(\nabla\phi)_n$ = magnitude of $\nabla\phi$ normal to face f

A_f = area of face in a mesh f (m^2)

V = volume of mesh (m^3)

Equations 2.30 and 2.31 are used in commercial CFD software to simulate the transport of pollutants in flow models. Understanding the mathematical derivations of the scalar transport equation is very significant because one can use CFD confidently to simulate precisely the decay of scalar variables such as *E. coli* and BOD in waste stabilization ponds. In order to simulate the decay of *E. coli* and BOD in waste stabilization ponds using CFD, the general scalar transport equation 2.29 should be modified by adding a source term function that represents the decay of the particular pollutant. However, for the simulation of the tracer experiment, it is not necessary to modify the default scalar transport because there is no addition and destruction of tracer as it passes through the pond.

The scalar variable that has been commonly simulated in the CFD-based design of waste stabilization ponds is the tracer experiment. This is achieved by solving the time-dependent scalar transport equation 2.29. The technique involves solving first the steady state flow equations without the scalar transport equation, and then solving the time dependent scalar equation without the state flow equations. This technique is not difficult to implement because the default scalar transport equation 2.29 is solved without any modification.

Shilton and Harrison (2003a, 2003b); Vega *et al.* (2003) used CFD to model the decay of *E. coli* numbers and BOD concentration in a primary facultative and anaerobic ponds respectively. The default scalar transport equation was modified by including the *inbuilt* source term function that is available in PHOENICS and MIKE-21 software. The first-order rate constants of *E. coli* and BOD removal were the only parameters that were defined in the model. The results of the CFD model were satisfactory because the *inbuilt* source term function in this commercial software has been derived correctly with proper units.

Sweeney (2004) used CFD to simulate the decay of *E. coli* in waste stabilization ponds using FLUENT software. The source term function that represents the decay of *E. coli* numbers was developed and was included in the scalar transport equation (2.29). Surprisingly, the CFD model did not remove *E. coli* at a temperature of $25^{o}C$ in a waste stabilization pond that had 4-days hydraulic retention time. It is interesting to note that the source term function was not developed correctly because it had improper dimensions and units.

CFD users should be aware that development of additional functions that modify the default CFD equations should be based on satisfactory dimensional analysis and fluid mechanics principles. It should also be noted that the *inbuilt* source term function used by Shilton and Harrison (2003a, 2003b); Vega *et al.* (2003) are satisfactory in predicting the pollutant removal when waste stabilization ponds have isothermal conditions. The *inbuilt* source term function is not reliable to model the pollutant decay in waste stabilization ponds with the effects of thermo-stratification because a constant value of the first-order rate constant removal for the pollutant decay is used for all cells in the model. The realistic estimation of the pollutant decay in waste stabilization ponds that are thermally stratified would require different first-order rate constants for the individual wastewater layers with different temperature.

Using dimensional analysis, Chapter 3 shows that the source term function that represents the pollutant removal in CFD depends on the density of wastewater and the first-order rate constant removal of the particular pollutant. The challenge that could be faced is the determination of the precise value of the first-order rate constant removal that could be used in the source term function. When waste stabilization ponds are thermally stratified, the density of the wastewater and temperature vary along the vertical profile of the water column and this changes the first-order rate constant. Hence, the source term function that represents the decay of pollutants in the pond should be developed to reflect the temperature variation and the wastewater density along the pond depth.

2.6.1 Review of CFD PhD theses on waste stabilization ponds

2.6.1.1Wood (1997) PhD thesis

Wood *et al.* (1995, 1998) demonstrated the potential benefits that CFD could provide in improving the hydraulic design of waste stabilization ponds. It was argued that CFD-based design of waste stabilization pond could assess precisely the improvement in the hydraulic and treatment performance of waste stabilization ponds that are fitted with baffles of various configurations.

Wood (1997) pioneered the technique of simulating tracer experiment in the CFD model of waste stabilization ponds. The time dependent scalar transport equation was used to calculate the tracer concentration at the pond outlet. A 2D model was used to replicate the tracer experiment that was carried out in the laboratory pond by (Mangelson and Watters, 1972). FIDAP software was used in the CFD model. It was noted that the simulated residence time distribution curves did not replicate satisfactorily the residence time distribution curves observed by Mangelson and Watters (1972).

Wood decided to use a 3D model to improve the accuracy of the time dependent scalar transport equation. This was achieved by including the depth of the pond, inlet and outlet structure in the model. It was found that the 3D model replicated more satisfactorily the Mangelson and Watters's residence time distribution curves than the 2D model. Wood (1997) advised that a 2D model should never be used to model the hydraulic flow patterns in waste stabilization ponds because it failed to represent precisely the pond depth, the inlet and outlet pipes in the CFD model. In addition, a 2D model cannot simulate precisely thermo-stratification effects that indicate temperature variation along the pond depth (Benelmouffok and Yu, 1989).

Wood simulated the hydraulic flow patterns of a full-scale waste stabilization pond in Australia using a 3D model. Validation of the model was carried out using tracer experiments. It was found that the CFD model was not successful in predicting the flow patterns in the full-scale waste stabilization pond because the experimental data of the simulated tracer experiment did not agree with that of the observed tracer experiment. Effects of wind were blamed for the unsuccessful validation of the model because the CFD model did not include wind effects.

Wood included wind effects in the model by applying a shear stress due to wind velocity on the top surface of the pond. The magnitude of the wind shear stress was expressed as a percentage (3%) of the measured wind velocity. This provided the boundary condition at the top surface of the CFD model. However, Sweeney (2004) argued that the complex interaction that exists between the wind velocity and the surface water velocity could not be simulated by such simple boundary condition. Sweeney (2004) used the empirical equation developed by van Dorn (1953) for computing the wind shear stress on the top surface of the pond. Although Wood (1997) included wind effects in the 3D model, the experimental data from the simulated tracer experiment did not agree satisfactorily with that from the observed tracer experiment.

Wood (1997) proposed design techniques that could improve the treatment performance of waste stabilization ponds. Three baffles of various configurations were used to modify the geometry of the full-scale waste stabilization pond following the recommendations of (Thackston *et al.* 1987; Kilani and Ogunrombi 1987; Mangelson and Watters, 1972) that baffles improved the hydraulic performance of waste stabilization ponds.

Wood observed that hydraulic short-circuiting in a baffled waste stabilization pond model was significantly reduced compared with waste stabilization pond model that had no baffles. It was concluded that CFD was the innovative design tool that could predict precisely the treatment performance of baffled waste stabilization ponds.

Wood (1997) noted that the use of the time-dependent scalar transport equation with a source term was the innovative way of modelling the pollutant removal in waste stabilization ponds. However, this was not carried out. It was suggested that the development of a sub-model of the source term function that represented the pollutant removal was more complex and difficult to validate. However, the use of the scalar transport equation with a source term function that represented the pollutant removal was a significant step in assessing realistically the performance of waste stabilization ponds. This could have been the best approach for comparing the treatment performance of baffled waste stabilization pond models rather than relying on residence time distribution curves.

Thermo-stratification effects were not simulated in the CFD model of waste stabilization ponds studied by Wood (1997). However, it was recognised by Wood that incorporation of thermo-stratification effects in the CFD model could provide detailed information regarding the flow patterns and treatment performance of waste stabilization ponds that were thermally stratified. Thermo-stratification effects initiate the hydraulic short-circuiting that diminishes the treatment performance of full-scale waste stabilization ponds. Any CFD model that does not include thermo-stratification effects cannot provide realistic performance assessments of waste stabilization ponds that are thermally stratified.

2.6.1.2 Salter (1999) PhD thesis

Using FLUENT software, Salter (1999) used a CFD model to investigate the hydraulic performance of five series of lagoons operated by Thames Water Company in Thailand that treated industrial wastewater. Steady state flow with isothermal conditions was assumed in the CFD model. A three-dimensional model was employed to represent precisely the waste stabilization pond geometry following the recommendations of Wood (1997). The standard turbulence $k-\varepsilon$ model was employed because the inlet Reynolds number at the pond inlet was more than 4000. Wind effects were not included in the model and no explanation was given to its omission. The CFD model used a 18 × 48 × 8 structured grid as the main computational mesh.

The CFD model indicated that significant hydraulic short-circuiting was responsible for the poor performance of the operational lagoons. The model predicted a reduced hydraulic retention time due to the early arrival of the simulated tracer at the pond outlet. Two baffles were included in the model to reduce the hydraulic short-circuiting. The two-baffle pond model showed that the installation of the baffles reduced significantly the hydraulic short-circuiting. Surprisingly, Salter did not validate the CFD model with the experimental data from the operational lagoons.

Salter advised that the CFD model had been validated by results elsewhere, a weak reason indeed to justify the results of the model.

Although Salter (1999) attributed thermo-stratification effects as the cause of the hydraulic short-circuiting, it was also necessary to investigate wind effects on the hydraulic performance of such large lagoons (surface area of the pond ˜ 6 ha). Many researchers have observed that wind effects initiate hydraulic short-circuiting in waste stabilization ponds that have large surface area (Banda *et al.* 2006a; Brissaud *et al.* 2000, 2003; Frederick and Lloyd, 1996; Lloyd *et al.* 2003; and Vorkas and Lloyd 2000).

Salter (1999) suggested that a combination of mechanistic models with CFD is the innovative way of designing rational waste stabilization ponds. It was argued that designers could predict precisely the effluent quality during the design and operational stages of waste stabilization ponds. Salter argued that this approach would ensure that the effluent quality meets the stringent consent requirements placed on wastewater treatment works. Surprisingly, Salter did not combine the CFD with mechanistic models to improve the design of waste stabilization ponds.

Salter (1999) is the first researcher who has used CFD to simulate thermo-stratification effects in waste stabilization ponds. The inlet boundary temperature was varied in the model depending on the time of the day. The temperature boundary was set to 31^{o}C for the 8-hours duration and was 28^{o}C for the 16-hours duration. The same technique was used for the top surface boundary.

Salter assessed the hydraulic flow pattern in the CFD model with thermo-stratification effects by simulating tracer experiment using species transport equation that is available in FLUENT software. The simulated residence time distribution curves showed the existence of the hydraulic short-circuiting in the pond. Thermo-stratification was thought to diminish the hydraulic performance of the unbaffled lagoon. Salter argues that the warm influent rapidly reached the pond outlet in a fraction of the mean residence time of the pond.

Two baffles were fitted in the model with simulated effects of thermal stratification along the longitudinal axis of the pond. The two-baffle pond model showed that there was a reduction in the degree of the hydraulic short-circuiting compared with the unbaffled pond model. This is the first time that effects of thermo-stratification were simulated in the CFD model in assessing the hydraulic performance of waste stabilization ponds. Although Salter (1999) was successful in simulating effects of thermo-stratification in the CFD model, it should be noted that experimental data from tracer experiments provide qualitative information regarding the hydraulic flow patterns that exists in waste stabilization ponds. The drawback of relying on tracer experiments to assess the hydraulic performance of waste stabilization ponds is that the model cannot predict the residence time of wastewater in stagnation regions. As a result, it is difficult to assess the effective volume of the pond that is useful in the treatment of the wastewater.

Another weakness of the thermal-CFD model used by Salter (1999) is the assumption that the boundary pond temperatures of 31^{o}C and 28^{o}C remain constant for the entire duration of 8-hours and 16-hours of each day respectively. It is a fact that temperatures in a particular area vary significantly on a daily basis. It is recommended that empirical temperature model for boundary regions should have been developed based on the continuous monitoring of temperature at the inlet, outlet and the top surface of the pond. Sweeney (2004) used this approach when developing empirical temperature models that were assigned to the boundary regions of the thermal-CFD model.

It is also interesting to note that Salter (1999) did not use the CFD as a reactor model to predict the treatment efficiency of the two-baffle pond model and the unbaffled pond model. Other researchers (Shilton and Mara, 2005; Shilton and Harrison, 2003a; Banda *et al.* 2006a) have used this rational approach to assess the improvement in the treatment performance of baffled waste stabilization ponds instead of relying on tracer experiments that give quantitative estimates of the hydraulic performance of waste stabilization ponds.

2.6.1.3 Shilton (2001) PhD thesis

Shilton (2001) carried out a detailed CFD modelling of a laboratory and a full-scale waste stabilization pond in New Zealand. The CFD models investigated the hydraulic and the treatment performance that occurred in waste stabilization ponds when baffles or inlet structures of various configurations were fitted in the pond to modify the pond geometry. Shilton constructed a laboratory pond based on the principles of geometric similarity. The dimensions of the laboratory pond (2.715 m × 1.750 m × 0.125 m) were scaled from a typical waste stabilization pond that was designed following standard procedures (Mara, 2004).

Shilton did not use geometric similarity principles in fixing the depth of the laboratory pond. It was found that the depth of the laboratory pond was too small to be used for practical purposes. Instead, a convenient laboratory pond depth of 125 mm was chosen. PHOENICS software was used in the CFD to assess the hydraulic flow patterns in the laboratory pond.

In order to validate the CFD model, Shilton carried out extensive tracer experiments that were compared with simulated tracer experiments in terms of the normalised residence time distribution curves. In addition, measurements of velocity field in the laboratory pond were carried out and these were compared with those from the CFD model. Shilton found that the simulated tracer experiment predicted satisfactorily the tracer experiment due to the agreement of the residence time distribution curves.

With the confidence gained in simulating the tracer experiment in the laboratory pond, Shilton (2001) simulated tracer experiments in the full-scale waste stabilization pond. It was observed that the simulated residence time distribution curves from the CFD model did not agree satisfactorily with the observed residence time distribution curves.

Wind effects were included in the model to improve the accuracy of the simulated residence time distribution curves. It was found that the inclusion of wind effects in the model improved significantly the residence time distribution curves in predicting the hydraulic flow patterns of the full-scale waste stabilization pond.

However, Shilton (2001) did not simulate effects of thermo-stratification in the CFD model to realistically assess the treatment performance of the full-scale waste stabilization ponds. Shilton gave no explanation to its omission. Various researchers (Kellner and Pires, 2002; Gu and Stefan, 1995; Fritz *et al.* 1980; Salter, 1999) have blamed thermo-stratification as the main cause of the poor hydraulic and treatment performance in waste stabilization ponds. Simulation of thermo-stratification effects in the CFD model of waste stabilization ponds could enable the precise prediction of *E. coli* numbers and BOD concentration in the pond effluent (Salter, 1999).

Shilton (2001) integrated CFD with the *inbuilt* source term function that represents the *E. coli* decay to predict the *E. coli* numbers in the pond effluent. Shilton (2001) reports that PHOENICS software has been arranged to define a source term (S) in terms of a user defined coefficient, (C) and a value (V). The source term function as defined by Shilton and Harrison (2001) is expressed in equation 2.32 as:

$$S_\phi = C_\phi\left(V_\phi - \phi_n\right) \qquad 2.32$$

where:

S_ϕ = source term of ϕ

C_ϕ = a user defined coefficient

V_ϕ = Value

ϕ_n = scalar variable at node (n)

Shilton noted that the rate of decay of *E. coli* in waste stabilization ponds was expressed as equation 2.33:

$$\frac{dX}{dt} = -kX \qquad 2.33$$

where:

X = *E. coli* numbers remaining per 100 ml

k = first-order rate constant of *E. coli* removal (day^{-1})

Equation 2.32 is equivalent to equation 2.33 when $V_\phi = 0$ and $C_\phi = k$. This technique was used to simulate the *E. coli* decay in the hypothetical standard facultative pond that was fitted with baffles of various configurations. This CFD modelling procedure works satisfactorily in PHOENICS and MIKE-21 software because these codes have *inbuilt* source terms that are correctly developed. Other CFD codes (FLUENT and FIDAP) do not have *inbuilt* source term functions that represent *E. coli* and BOD removal. The CFD modeller is required to develop source term functions from first principles that should be dimensionally and mathematically consistent with other terms of the scalar transport equation 2.29.

It should be noted that the *inbuilt* source term function predicts precisely the *E. coli* removal when isothermal conditions are assumed in waste stabilization ponds (Shilton and Harrison, 2003a, 2003b). The source term function cannot simulate precisely the pollutant removal in waste stabilization ponds that are thermally stratified. Chapter 3 provides detailed procedure of developing versatile source term functions that could be used to simulate the removal of *E. coli* and BOD removal in waste stabilization ponds that develop both isothermal and thermo-stratification conditions.

It is surprising to note that Shilton did not include wind effects in the CFD model that simulated the *E. coli* decay in the hypothetical standard facultative pond with baffles of various configurations. Effects of wind are significant on the treatment performance of waste stabilization ponds that have large surface area. Interestingly, the surface area of the facultative pond that was simulated in the CFD was approximately 21 hectares and the effects of wind in this pond size could be significant (Banda *et al.* 2006b). Mara (2004) suggested that wind velocity could be beneficial or detrimental to the treatment efficiency of waste stabilization ponds depending on the prevailing wind direction in relation to the wastewater flow in the pond. This research area needs further investigation in order to exploit the benefit of wind towards the treatment performance of waste stabilization ponds.

2.6.1.4 Sweeney (2004) PhD thesis

Sweeney (2004) used a 3D CFD model (FLUENT software) to model the hydraulic performance of a hypothetical and the full-scale waste stabilization pond that was located in Australia. Effects of wind and thermo-stratification were included in the CFD model. A CFD model without wind effects and under isothermal condition was modelled first to establish the baseline investigation.

Tracer experiments were carried out to validate the CFD model. Effects of wind were incorporated in the model by applying the wind shear stress on the horizontal top surface of the pond. Numerical data (simulated residence time distribution curves) from the CFD showed that wind velocity initiated the hydraulic short-circuiting in the waste stabilization pond model.

A second CFD model with simulated thermo-stratification effects assessed the treatment performance of waste stabilization ponds that are thermally stratified. The results of the model were assessed in terms of the effluent *E. coli* numbers. Sweeney developed a source term function that used the first-order rate constants removal developed by Kayombo *et al.* (2000) and Curtis *et al.* (1992) in the default scalar transport equation 2.29 to predict the *E. coli* removal in the waste stabilization pond. The boundary conditions of the CFD model with thermo-stratification effects (inlet, outlet and the top pond surface) were determined by developing empirical temperature equations using the temperature data that was monitored hourly.

It is worth reviewing in detail the CFD methodology that was used by Sweeney to simulate the *E. coli* decay in the hypothetical waste stabilization pond. A hypothetical pond with dimensions of 20 m × 10 m × 1.4 m was simulated in the CFD model. The diameters of the pond inlet and outlet were 600 mm and this gave the cross sectional area of 0.28 m^2. The inlet and outlet pipes were located at the opposite diagonal corners of the pond. The pond volume was meshed with 4000 cells. The mean hydraulic retention time of the pond was 4 days. Isothermal conditions were assumed in the pond at a temperature of 25^{o}C. The source term function that represents the *E. coli* decay in the model was based on Curtis *et al.*'s (1992) first-order rate constant removal and this was included in the FLUENT solver.

The scalar transport equation of *E. coli* predicted a 0.025 log removal. It was realised that the CFD simulation of *E. coli* removal was far from satisfactory. Effort was made to improve the accuracy of the log removal of *E. coli* by increasing the oxygen concentration in Curtis *et al.*’s (1992) model. The CFD model predicted a marginal increase of 0.0012 log removal of *E. coli*. The CFD-predicted log removal of *E. coli* is a negligible value indeed. Sweeney compared the predicted CFD-log removal of *E. coli* with the Marais’ (1974) model at a similar retention time (4 days) and temperature (25^{o}C). It was found that the Marais’ (1974) model predicted 1.56 log removals of *E. coli* giving a difference of 99.84% compared with the results of the CFD model.

Analysis of the source term function developed by Sweeney (2004) show that the function is not dimensionally and mathematically consistent with other terms of the scalar transport equation 2.29. Chapter 3 shows that the units of the source term function in the scalar transport equation are $kgm^{-3}s^{-1}$ not s^{-1} as noted in the Sweeney’s source term function (See Chapter 3 for development of the source term function). The CFD literature review shows that it is very important to understand the basic CFD equations when developing sub-models that simulate the treatment efficiency and the hydraulic performance of waste stabilization ponds.

2.7 Summary of the literature review

In conclusion, the chapter has demonstrated that the classic and modern design procedures for waste stabilization ponds are based on the completely mixed flow and plug flow that are not realised in practice. However, the dispersed flow pattern may be realised in operational waste stabilization ponds and this could form a rational basis for designing the pond hydraulics. It has been shown that these design procedures cannot assess the improvement in the hydraulic and treatment performance of waste stabilization ponds that are fitted with baffles of various configurations.

It has also been shown that tracer experiments provide quantitative estimates of the hydraulic performance of waste stabilization ponds. However, tracer experiments cannot assess precisely the hydraulic performance of waste stabilization ponds when effects of wind and thermo-stratification are significant. In addition, tracer experiment can only be used in operational waste stabilization ponds as the experiment cannot be carried out at the design stage of new waste stabilization ponds.

CFD overcomes the limitations of current design procedures for waste stabilization ponds. CFD can be used as a reactor model to simulate precisely the pollutant removal and the spatial residence time distribution in waste stabilization ponds. This can be carried out by developing source term functions that modify the default scalar transport equation. This could enable a designer to identify physical design interventions that can improve the treatment performance of waste stabilization ponds.

The chapter has shown that effects of wind are not included in the current design procedures for waste stabilization ponds. Incorporation of wind effects into the CFD model can provide detailed information regarding flow patterns and the treatment performance of waste stabilization ponds under windy conditions.

Baffles can improve significantly the hydraulic and the treatment performance of waste stabilization ponds that are optimally loaded. Results of full-scale baffled waste stabilization ponds show that the design and construction of waste stabilization can be optimized to use the minimal land. However, experiments are required to investigate effects of baffles in reducing the hydraulic short-circuiting associated with thermo-stratification and wind effects. Another area worth researching is the investigation of the optimal baffle configuration that initiates the plug flow pattern in baffled waste stabilization ponds. The available literature does not give guidance on the number and length of baffles that provide economic baffle configuration. Numerical experiments using CFD models can be used to investigate the suggested problems.