WASTE STABILIZATION PONDS

AND

CONSTRUCTED WETLANDS

DESIGN MANUAL



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Part One

Chapter One: Introduction

1.1 Importance of Waste Stabilization Ponds and Constructed Wetlands in Wastewater Treatment in Tropical Climates

Waste Stabilization Ponds (WSP) and Constructed Wetlands (CW) have proven to be effective alternatives for treating wastewater, and the construction of low energy-consuming ecosystems that use natural processes, in contrast to complex high-maintenance treatment systems, will hopefully lead to more ecologically-sustainable wastewater treatment in the future. CWs and WSPs also have the capability of meeting the demand for a high percentage removal of pathogenic organisms, compared to conventional technologies. CWs and WSPs combined, and joined with other technologies, may be important for even more improved performance of water cleaning systems. WSP's and CW's are now well-established methods for wastewater treatment in tropical climates. Their many advantages include: simplicity, low cost, low maintenance, low energy consumption, robustness, and sustainability. While WSPs are most commonly used for treating domestic wastewaters, they are also successfully used for treating industrial wastewater, including water that contains agro-industrial wastes. One of the potential advantages of using constructed wetlands is that they do not allow mosquitoes to breed (sub-surface flow wetland). The process of designing WSPs and wetlands, and predicting their performance, is improving rapidly as we gain more experience with these systems.

Many countries in tropical climates use WSPs for wastewater treatment (e.g., Tanzania, Kenya, Malawi, Uganda, Zambia, Botswana, Zimbabwe). Many of these systems have been performing below the required standards, due to lack of proper operation and maintenance (Kayombo *et al*., 1998). Constructed wetlands have not yet received the deserved attention as an alternative method for wastewater treatment.

1.2 Overview

This manual provides comprehensive technical information about the planning, design, construction and operation of WSPs and CWs for a wide range of applications. To understand

and design WSPs and CWs requires the involvement of expertise in a variety of fields, including chemistry, hydrology, soil science, plant biology, natural resources, environmental management, ecology, environmental engineering, surveying, and project management.

1.3 Why Produce a Manual for Waste Stabilization Ponds and Constructed Wetlands?

This manual has been produced to provide necessary information regarding most aspects of the design and implementation of WSPs and CWs. Lack of such information, and an integrated approach, has led to the development of many WSPs and CWs that are inappropriate, under-performing, or poorly designed or maintained. The reasons for these problems include:

- Lack of appreciation by many designers of the complex, physical, biological and chemical processes within WSPs and CWs;
- Lack of consistency in design, construction and operation aimed at optimal performance;
- Lack of appropriate design tools and methodologies suitable for local conditions; and
- Changing nature of rapidly-developed technology

Although WSPs and CWs are often relatively simple in engineering terms, they are extremely complex ecological systems. Performance of WSPs and CWs relies not only on good design, but also good construction and operation. This design manual for WSP and CW has adopted the design approaches detailed by D.D. Mara (WSP design approaches), and the design approach for the constructed wetlands given by Department of Land and Water Conservation, New South Wales (Volume 1 and 2). Part one of this manual provides a description of the design of WSPs, while Part two provides a detailed description on the design of CW's. The management of WSP and CW is discussed in Part three, while Part four focuses on modelling of WSP and CW ecosystems. Specific design examples are given as appendices in the file, "**Design examples.**"

1.4 Design Tools and Rapidly-Developing Technology

WSP and CW designers come from many different backgrounds or disciplines, including civil engineering, environmental engineering, microbiology, chemical engineering, soil science, or natural resources management. Many of the WSP and CW design tools and models have been

adopted from countries with temperate climates. Thus, not all models can be transferred and used in tropical climate countries. For example, the hydrology and climate in tropical African is significantly different to that of most States in the United States or Europe. Nutrient removal in these countries has a high priority, whereas the removal of pathogenic organisms has a high priority in tropical countries. Most empirical model or design tools developed and used on-site or regionally-specific WSPs or CWs data. Different characteristics, in terms of climate and hydrology, can lead to problems when models are transferred without appropriate modification for local conditions. Further, using simple tools or "rule of thumb" methods, in lieu of appropriate design techniques, often results in malfunctions or a reduced efficiency in the effect of WSP's and CW's.

1.5 Objectives

The objectives of this waste stabilization and constructed wetland manual are as follows:

- To provide WSP and CW designers, builders and operators with appropriate information to develop, implement and operate WSP and CW for a range of applications and design objectives;
- To provide standard systems approach that can be adopted universally, and which can accommodate a development technology, with changes in information concepts and ideas with time;
- Provide theoretical background on the biological, chemical and physical processes of WSPs and CWs, the current state of the technology and technical knowledge on how to design, operate and maintain the systems; and
- Provide theoretical knowledge on how the models can be used in the best manner to describe the systems.

Chapter Two: Waste Stabilization Ponds

2.1 Application of Waste Stabilization Pond Systems

Waste Stabilization Ponds (WSPs) are large, shallow basins in which raw sewage is treated entirely by natural processes involving both algae and bacteria. They are used for sewage treatment in temperate and tropical climates, and represent one of the most cost-effective, reliable and easily-operated methods for treating domestic and industrial wastewater. Waste stabilization ponds are very effective in the removal of faecal coliform bacteria. Sunlight energy is the only requirement for its operation. Further, it requires minimum supervision for daily operation, by simply cleaning the outlets and inlet works. The temperature and duration of sunlight in tropical countries offer an excellent opportunity for high efficiency and satisfactory performance for this type of water-cleaning system. They are well-suited for low-income tropical countries where conventional wastewater treatment cannot be achieved due to the lack of a reliable energy source. Further, the advantage of these systems, in terms of removal of pathogens, is one of the most important reasons for its use.

2.2 Types of Waste Stabilization Ponds and Their Specific Uses

WSP systems comprise a single string of anaerobic, facultative and maturation ponds in series, or several such series in parallel. In essence, anaerobic and facultative ponds are designed for removal of Biochemical Oxygen Demand (BOD), and maturation ponds for pathogen removal, although some BOD removal also occurs in maturation ponds and some pathogen removal in anaerobic and facultative ponds (Mara, 1987). In most cases, only anaerobic and facultative ponds will be needed for BOD removal when the effluent is to be used for restricted crop irrigation and fish pond fertilization, as well as when weak sewage is to be treated prior to its discharge to surface waters. Maturation ponds are only required when the effluent is to be used for unrestricted irrigation, thereby having to comply with the WHO guideline of >1000 faecal coliform bacteria/100 ml. The WSP does not require mechanical mixing, needing only sunlight to supply most of its oxygenation. Its performance may be measured in terms of its removal of BOD and faecal coliform bacteria.

2.3 Processes in Waste Stabilization Ponds

2.3.1 Anaerobic ponds

Anaerobic ponds are commonly 2 – 5 m deep and receive wastewater with high organic loads (i.e., usually greater than 100 g BOD/m^3.day, equivalent to more than 3000 kg/ha.day for a depth of 3 m). They normally do not contain dissolved oxygen or algae. In anaerobic ponds, BOD removal is achieved by sedimentation of solids, and subsequent anaerobic digestion in the resulting sludge. The process of anaerobic digestion is more intense at temperatures above 15^o C. The anaerobic bacteria are usually sensitive to pH <6.2. Thus, acidic wastewater must be neutralized prior to its treatment in anaerobic ponds. A properly-designed anaerobic pond will achieve about a 40% removal of BOD at 10^o C, and more than 60% at 20^o C. A shorter retention time of 1.0 - 1.5 days is commonly used.

2.3.2 Facultative ponds

Facultative ponds (1-2 m deep) are of two types: Primary facultative ponds that receive raw wastewater, and secondary facultative ponds that receive particle-free wastewater (usually from anaerobic ponds, septic tanks, primary facultative ponds, and shallow sewerage systems). The process of oxidation of organic matter by aerobic bacteria is usually dominant in primary facultative ponds or secondary facultative ponds.

The processes in anaerobic and secondary facultative ponds occur simultaneously in primary facultative ponds, as shown in Figure 2.1. It is estimated that about 30% of the influent BOD leaves the primary facultative pond in the form of methane (Marais 1970). A high proportion of the BOD that does not leave the pond as methane ends up in algae. This process requires more time, more land area, and possibly 2 -3 weeks water retention time, rather than 2 -3 days in the anaerobic pond. In the secondary facultative pond (and the upper layers of primary facultative ponds), sewage BOD is converted into "Algal BOD," and has implications for effluent quality requirements. About 70 – 90% of the BOD of the final effluent from a series of well-designed WSPs is related to the algae they contain.

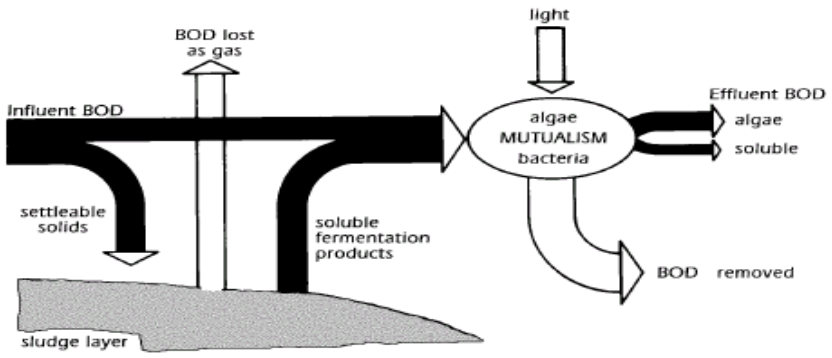


Fig. 2.1 Pathways of BOD removal in primary facultative ponds (After Marais, 1970)

In secondary facultative ponds that receive particle-free sewage (anaerobic effluent), the remaining non-settleable BOD is oxidised by heterotrophic bacteria (*Pseudomonas, Flavobacterium, Archromobacter and Alcaligenes* spp). The oxygen required for oxidation of BOD is obtained from photosynthetic activity of the micro-algae that grow naturally and profusely in facultative ponds

Facultative ponds are designed for BOD removal on the basis of a relatively low surface loading (100 – 400 kg BOD/ha.day), in order to allow for the development of a healthy algal population, since the oxygen for BOD removal by the pond bacteria is generated primarily via algal photosynthesis. The facultative pond relies on naturally-growing algae. The facultative ponds are usually dark-green in colour because of the algae they contain. Motile algae (*Chlamydomonas* and *Euglena*) tend to predominate the turbid water in facultative ponds, compared to none-motile algae (*Chlorella*).

The algal concentration in the pond depends on nutrient loading, temperature and sunlight, but is usually in the range of 500 - 2000 μg chlorophyll-*a*/liter (Mara, 1987). Because of the photosynthetic activities of pond algae, there is a diurnal variation in the dissolved oxygen concentration. The dissolved oxygen concentration in the water gradually rises after sunrise, in response to photosynthetic activity, to a maximum level in the mid-afternoon, after which it falls to a minimum during the night, when photosynthesis ceases and respiratory activities consume oxygen. At peak algal activity, carbonate and bicarbonate ions react to provide more carbon dioxide for the algae, leaving an excess of hydroxyl ions. As a result, the pH of the water can rise to above 9, which can kill faecal coliform. Good water mixing, which is usually facilitated by

wind within the upper water layer, ensures a uniform distribution of BOD, dissolved oxygen, bacteria and algae, thereby leading to a better degree of waste stabilization.

2.3.3 Kinetic processes in facultative ponds

The growth of the mixed culture was studied at concentrations ranging between 200 - 800 mg COD/l, in a series of batch chemostat reactors. From laboratory data, the specific growth rate (μ) was determined, using the modified Gompertz model (Kayombo *et al.* 2003). There are several growth models in the literature used to evaluate growth parameters, examples being the models of Gompertz (1825), Richards (1959), Stannards *et al*., (1985), Schnute (1981), logistic models, etc. Most of these models describe the number of microorganisms, without considering the consumption of substrate. Zwietering *et al,* (1990) derived a modified mathematical relationship based on the Gompertz model for the increase in biomass over time, which relates the population size over time to the specific growth rate, lag time, and asymptotic level of organisms, as shown in equation (2.2):

$$y = \exp\left\{-\exp\left[\frac{\mu_m e}{A}(\lambda - t)\right] + 1\right\} \qquad (2.2)$$

Where μ_m is a specific growth rate (day^{-1}), A is a maximum asymptotic value, and λ is the lag time (days). The re-parameterized Gompertz equation by Zwietering *et al*., (1990) was rewritten by Wijtzes *at al*., (1995), so as to determine the maximum biomass (X_{max}), as shown in equation (2.3):

$$\ln(X_t) = \ln(X_0) + \{\ln(X_{max}) - \ln(X_0)\}\exp\left\{-\exp\left[\frac{\mu_m e}{\{\ln(X_{max}) - \ln(X_0)\}}(\lambda - t) + 1\right]\right\} \qquad (2.3)$$

Where t is time (hours or days), X_t is the biomass at time t (mg/l), X_0 is the asymptotic biomass at time zero, and X_{max} is maximum biomass (mg/l).

The maximum specific growth rate (μ_{max}) and half saturation coefficients (K_s) were calculated using the Monod kinetic equation. The Monod (1949) equation is the most commonly used model to relate microbial growth to substrate utilization. Monod found that, in pure cultures and continuous reactor systems, the relationship between the bacterial growth and substrate availability in the system, with a single growth-limiting substrate, could be expressed empirically, as shown in equation (2.4):

$$\mu = \frac{\mu_{max} S}{K_s + S} \qquad (2.4)$$

Where, μ is a specific growth rate (day^{-1}), μ_{max} is the maximum growth rate (day^{-1}), S is the Substrate concentration (mg/l), and K_S is the half saturation coefficient (mg/l).

To determine the biological parameters in the Gompertz model, the simulation was done with Stella II® software, which integrated the model using the built-in 4th Order Runge Kutta. The specific growth rate and lag time (λ) determined at each concentration of organic carbon, using the Gompertz model, is shown in Table 2.1. The values in the brackets are the correlation coefficients (R^2) between the measured and simulated results for specific growth rates of heterotrophic bacteria and algae biomass.

Table 2.1 Specific biomass growth rates

Substrate (mg COD/l)	μ A (day^{-1})	λ (day)	μ A chl-*a* (day^{-1})	λ (day)	μ HB (day^{-1})	λ (day)
0.00	0.000	0	0.000	0.0	0.000	0.0
200	0.345(R^2: 0.86)	3.5	0.484(R2: 0.97)	3.5	0.360 (R^2:0.93)	5.0
300	0.420 (R^2: 0.98)	4.0	0.555 (R2: 0.97)	3.4	0.460 (R^2:0.98)	4.0
400	0.453 (R^2: 0.98)	4.5	0.568 (R2: 0.97)	4.0	0.517 (R^2: 0.95)	4.5
500	0.473 (R^2: 0.98)	4.8	0.568 (R2:0.96)	4.5	0.563 (R^2:0.94)	4.8
600	0.466 (R^2: 0.84)	3.8	0.500 (R2: 0.84)	4.0	0.560 (R^2: 0.88)	3.8
700	0.474 (R^2: 0 93)	4.3	0.470 (R2: 0.94)	5.3	0.560 (R^2: 0.95)	4.3
800	0.470 (R^2: 0.86)	4.0	0.405 (R2: 0.68)	4.5	0.555 (R^2: 0.89)	3.9

Explanation: μA and μ HB are specific growth rates of algae and heterotrophic bacteria, respectively; μ A chl-a is the specific growth of algae, based on chlorophyll-a.

The specific growth rate of algae and heterotrophic bacteria obtained from the modified Gompertz equation was then fit into the Monod model (i.e., Equation 2.4) to determine the μ_{max} and K_s. The maximum observed growth rate (μ_{max}) for heterotrophic bacteria was 3.8 day^{-1}, with a K_S of 200 mg

COD/l. The μ_{max} for algal biomass, based on volatile suspended solids, was 2.7 day^{-1}, with a K_S of 110 mg COD/l. The μ_{max} of algae, based on chlorophyll-*a,* was 3.5 day^{-1} with a K_S of 50 mg COD/l. Table 2.2 summarizes the kinetic constants for processes in secondary facultative ponds.

Table 2.2 Maximum and half-saturation coefficients

Organisms	μ_{max} (day^{-1})	K_S (mg COD/l)	Regression coefficient (R^2)
Heterotrophic bacteria	3.8	200	0.99
Algae (based on volatile solids)	2.7	110	0.99
Algae (based on chlorophyll-*a*)	3.5	50	0.96

The observed specific substrate removal by heterotrophic bacteria varied between the concentrations of substrate used, with an average value of 0.82 (mg COD/mg biomass). Thus, the determined Monod kinetic parameters are useful for defining the operation of secondary facultative ponds. The specific substrate utilization rate in the bioreactors was directly proportional to the specific growth rate.

Specific substrate removal rate (q) -- The substrate utilization rate was determined at a steady-state growth rate for each batch reactor. The observed substrate utilization rate was between 0.68 - 0.97 (mg COD/mg biomass). The average substrate removal rate was 0.82 (mg COD/mg biomass). The maximum substrate utilization rate of 0.97 was achieved at a substrate concentration of 400 mg/l. Thus, the bioreactor yield was 1.03 - 1.47 mg biomass/mg COD. The obtained values compare well to the trend of the specific growth rate obtained from the bioreactors. The substrate utilization rate was directly related to the specific growth rate of heterotrophic bacteria, as also was shown by Papkov *et al.,* 2000.

Although the Monod Equation was developed for the pure culture of bacteria growing in a single organic substrate, the results from this work indicate the model may be used correctly for a mixed culture of algae and heterotrophic bacteria growing in a heterogeneous-mixed organic carbon. The (K_S) obtained are the characteristic of the wastewaters treated by SFPs, which contains a complex mixture of organic carbon. Further, it appears that the Monod constants depend on the nature of the

operation of the system. The parameters obtained during this experimental work are applicable for the SFPs where algae and bacteria grow symbiotically. The Monod constants for algae depend on the method used in determining the biomass. In this research, the reported values are based on the chlorophyll-*a* and volatile solids values, thus adding more knowledge to our understanding of the processes of facultative ponds.

2.3.4 Effects of pH on the growth rate of heterotrophic bacteria and algae in secondary facultative ponds

Knowledge of the effects of the pH on the growth rate of algae and heterotrophic bacteria, and its subsequent impacts on the degradation of organic matter, is useful for better operation and design of secondary facultative ponds (SFP). The kinetics of microbial growth and product formations are influenced by diurnal variation in the pH in the pond resulting from diurnal variations in the carbon dioxide (Fritz et al., (1979). Algae require large quantities of dissolved carbon dioxide during photosynthesis, causing a depletion of carbon dioxide (CO_2) and leading to a shift in the carbonate – bicarbonate (CO_3^{-2} - HCO_3^-) equilibrium, resulting in an increase in pH due to the formation of hydroxyl (OH^-) ions (Neel,*et al*., 1961; Poon *et al.,* 1986; Van der Ploeg, 1982). The effects of pH on the growth rate of heterotrophic bacteria and algae in Secondary Facultative Ponds (SFP) were investigated, using batch growth at a pH value between 5 and 11. The optimum pH and pH constant for algae and heterotrophic bacteria was determined using the pH model presented by Henze *et al.,* (1995):

$$f(pH) = \mu_{max(XB)}\left(\frac{K_{pH}}{K_{pH} + y}\right) \qquad (2.5)$$

and

$$y = 10^{|optpH - pH|} - 1 \qquad (2.6)$$

Where y is the hydrogen ion activity, K_{pH} is the pH constant, pH is the value of the wastewater pH, and optpH is the optimum pH at which the growth of microorganism is maximum.

The determination of the specific growth rate of heterotrophic bacteria and algae under the influence of the pH, based on the data obtained from the reactors, was done by using the modified Gompertz model presented by Zwietering *et al.*, (1990), as shown in Equation 2.2. Equation 2.2

was programmed with Stella II® software. The integration was done with the built-in 4th Order Runge Kutta. The natural logarithmic values of the measured biomass data were compared with the predicted specific growth rates by the modified Gompertz model. The observed specific growth rates and other coefficients obtained are as shown in Table 2.3 and 2.4 for heterotrophic bacteria and algae biomass. The linear regression coefficient (R^2) between the measured and model-predicted specific growth rates also is shown in Table 2.3 and 2.4.

Table 2.3 Specific growth rates for heterotrophic bacteria and algae

pH	μHB (day^{-1})	λ (day)	A	R^2
5	0.320	1.8	0.36	0.74
6	0.336	1.6	0.10	0.82
7	0.368	2.0	0.10	0.63
8	0.369	1.8	0.13	0.75
9	0.350	3.2	0.12	0.66
10	0.357	5.8	0.10	0.53
11	0.327	1.6	0.10	0.78

Table 2.4 Specific growth rate for algae

pH	μAlgae (day^{-1})	λ (day)	A	R^2
5	0.213	2	0.43	0.76
6	0.220	1.9	0.44	0.80
7	0.235	1.6	0.35	0.58
8	0.261	3.6	0.20	0.73
9	0.255	1.8	0.20	0.61
10	0.242	5.8	0.49	0.75
11	0.246	4.5	0.51	0.59

The results indicate that, for a pH value higher than 8, the chlorophyll-*a* content decreases. However, the specific growth rate of heterotrophic bacteria and algae was high with a pH value between pH 7 to 8. At a pH value above 9, the specific growth rate of both biomasses started to decrease.

The optimum pH and pH constant for microbial activities in SFP was determined with the model presented by Henze *et al.* (1995), as shown in Equations 2.5 and 2.6. The specific growth rate of microorganisms in the SFP was increasing from a pH value of 5, reaching a maximum value at a pH between 6.5 and 8. The observed optimum pH for heterotrophic bacteria was 8.5, and the pH constant was 200. The optimum pH for algae was 6.8, and the pH constant was 179 (Kayombo *et al*., 2001). The parameters were obtained at a maximum growth rate of 3.6 $day^{-1}$and 3.45 day^{-1} for

heterotrophic bacteria and algal biomass, respectively. The region of pH insensitivity for heterotrophic bacteria was between 7 and 9. However, the region of pH insensitivity for algae was between 6.5 and 8. The range between the optimum pH for algae and heterotrophic bacteria was 1.7 pH units. Table 2.5 shows the parameters obtained for the influence of the pH on the growth rate of heterotrophic bacteria and algae.

Table 2.5 The optimum and pH constant for algae and heterotrophic bacteria.

Parameter/biomass	Heterotrophic bacteria	Algae
KpH	200	179
Optimum pH	8.5	6.8
Regression coefficient (R^2)	0.76	0.78

It was concluded that the pH simultaneously affects algae and heterotrophic bacteria processes in facultative ponds with an almost equal magnitude. The model reported by Henze *at al.*, (1995) was found to simulate well the influence of pH on algae and heterotrophic bacteria biomasses.

2.3.5 Diurnal variation of dissolved oxygen in facultative waste stabilization ponds

Oxygen tension in WSP's is an operational parameter with a great deal of daily and hourly variation. The rate of oxygen production is a function of the algal concentration. Because algal growth is both light and temperature - dependent, the rate of oxygen production (photosynthetic) follows the same pattern. Temperature also is a parameter exhibiting marked seasonal and daily variation in WSP's. It influences photosynthesis, the growth of microorganisms, and the bio-decomposition of organic carbon in the system. The fluctuation in pH influences the kinetics of microbial growth, species competition and product formations in the pond (Fritz *et al*., 1979). Microbial species can grow within a specific pH range, which typically extends over 3 to 4 pH units, with an optimum growth rate near the mid-point of the range. Values of pH up to 11 are not uncommon in WSP's, with the highest levels being reached in the late afternoon.

Thus, the combined effects of changes in temperature, pH and light intensity may have a marked effect on microbial activities in the pond, than when only one factor is considered. In reality, changes in these forcing functions and processes occur simultaneously; thus, their influence on the processes should be determined at this level. The objective of the work reported herein was to use the long-term data collected from the ponds to determine the manner in which pH, temperature and light intensity influence the production and utilization of dissolved oxygen in SFWSP. The

influence of these forcing functions on the DO model was determined for their combined effect, rather than their effect at the individual level.

The dissolved oxygen sub-model was developed in order to depict the combined influences of light, pH, temperature and carbon dioxide on the processes of dissolved oxygen production and utilization in SFWSP. The model was formulated on the basis of the work of Chen and Orlob (1975), being modified to include the influence of pH and carbon dioxide (Kayombo *et al.,* 2000), as follows:

$$PHOTO = \mu_{\max} f\left(T, pH, Substrate, Light\right) \tag{2.7}$$

$$\frac{\partial DO}{\partial t} = \left(\mu_{\max} f\left(T, pH, CO, L\right) \right) AlgB + QC_{in} - \left(k_{20}\theta^{(T-20)} \frac{DO}{K_{DO}+DO} \right) COD - \left[RA\max f(T) \right] AlgB - Q_{eff}C_{eff} \tag{2.8}$$

Where PHOTO refers to photosynthesis, μ_{max} is the maximum algal growth (day^{-1}), and f[T, pH, Substrate, Light) is the function of temperature, pH, substrate, and light intensity on the specific growth rate of algae. The model is presented in annex III-B.

The forcing functions in the dissolved oxygen model were light intensity, carbon dioxide, temperature and pH. It was found that temperature; light and pH influence the photosynthesis process, based on the multiplicative formulation of forcing functions. The model was calibrated and validated by using the average daily data from the Secondary Facultative Waste Stabilization Pond 1 and 11 (SFWSP1 & 11), yielding a linear regression coefficient of 0.87 during calibration, and 0.78 during validation. Based on the model results, the rate of DO production in relation to dry algal biomass was 1.599 mg DO/mg dry weight (equivalent to 35.905 mg DO/mg chlorophyll-*a*). This correlation between the observed data and model prediction indicates the assumption inherent in the mathematical model formulation of the processes is valid for describing DO production and usage in the ponds.

The developed and validated oxygen sub-model of SFWSP reveals that all forcing functions simultaneously affect the rate of photosynthesis, based on the multiplicative function. It also

suggests that, for a balanced system, the quantity of DO produced by the photosynthesis process is sufficient to keep the system healthy. Based on the model, the major oxygen utilization was related to total respiration. Although the developed model is valid for SFWSPs, with sufficient data it also may be extended to maturation ponds. The model provides the rate that controls the oxygen production at which all forcing functions fluctuate simultaneously, a study that might be difficult to conduct at the laboratory level. The rates obtained during the model calibration and validation are the average governing rates of the performance of the pond under the influence of all forcing functions (Table 2.6).

Table 2.6 The model parameters used and sources of estimation (Kayombo *et al.*, 2000)

Parameter	Range	Model calibration	Unit	Reference
μ	0.48	0.43	day^{-1}	Rich, 1996
θ	1.036 – 1.104	1.104	day^{-1}	EPA, 1985
μ_{max}	0.1 - 11	7.4	day^{-1}	Asaeda, 1997
k_{20}	0.1 – 1.2	0.6	day^{-1}	EPA, 1985
IK	10 - 300	198	$\mu E/m^2.sec$	Asaeda, 1997
KpH	150 - 250	250		Henz *et al.,* 1995
Rmax	0.03 – 0.92	0.45	day^{-1}	Asaeda, 1997
Topt		28	^oC	model calibration
Tmin		6	^oC	model calibration
optpH	6 - 11	6.9		Casey, 1996
K_{CO2}	0.5 – 0.6	0.13	mg/l	Chen and Orlob, 1975

These results lead to the conclusion that the most useful rate that may be used to govern the processes from the WSP should be those determined from the combined influence of the various forcing functions.

2.4 Maturation Ponds

The maturation ponds, usually 1-1.5 m deep, receive the effluent from the facultative ponds. Their primary function is to remove excreted pathogens. Although maturation ponds achieve only a small degree of BOD removal, their contribution to nutrient removal also can be significant. Maturation ponds usually show less vertical biological and physicochemical stratification, and are well-oxygenated throughout the day. The algal population in maturation ponds is much more diverse than that of the facultative ponds, with non-motile genera tending to

be more common. The algal diversity generally increases from pond to pond along the series (Mara, 1989). Although faecal bacteria are partially removed in the facultative ponds, the size and numbers of the maturation ponds especially determine the numbers of faecal bacteria in the final effluent. There is some removal of solids-associated bacteria in anaerobic ponds, principally by sedimentation. The principal mechanisms for faecal bacterial removal in facultative and maturation ponds are now known to be:

(a) Time and temperature;
(b) High pH (> 9); and
(c) High light intensity, combined with high dissolved oxygen concentration.

Time and temperature are the two principal parameters used in designing maturation ponds. Faecal bacterial die-off in ponds increases with both time and temperature (Feachem *et al.,* 1983). High pH values (above 9) occur in ponds, due to rapid photosynthesis by pond algae, which consumes CO_2 faster than it can be replaced by bacterial respiration. As a result, carbonate and bicarbonate ions dissociate, as follows:

$$2\ HCO_3^- \rightarrow CO_3^{2-} + H_2O + CO_2 \qquad (2.9)$$

$$CO_3^{2-} + H_2O_2 \rightarrow 2\ OH^- + CO_2 \qquad (2.10)$$

The resulting CO_2 is fixed by the algae, and the hydroxyl ions accumulate, often raising the pH to values above 10. Faecal bacteria (with the notable exception of *Vibrio cholerae*) die very quickly at pH values higher than 9 (Pearson *et al.,* 1987*c*). The role of high light intensity and high dissolved oxygen concentration has recently been elucidated (Curtis *et al.,* 1992). Light of wavelengths between 425 – 700 nm can damage faecal bacteria by being absorbed by the humic substances ubiquitous in wastewater. They remain in an excited state sufficiently long to damage the cell. Light-mediated die-off is completely dependent on the presence of oxygen, as well as being enhanced at high pH values. Thus, the sun plays a three-fold role in directly promoting faecal bacterial removal in WSP, and in increasing the pond temperature, and more indirectly by providing the energy for rapid algal photosynthesis. This not only raises the pond pH value

above 9, but also results in high dissolved oxygen concentrations, which are necessary for its third role; namely, promoting photo-oxidative damage.

2.5 Nutrient removal in waste stabilization ponds

In anaerobic ponds, organic nitrogen is hydrolysed to ammonia. Thus, the effluent from anaerobic ponds usually has higher concentrations of ammonia than in raw sewage. In facultative and maturation ponds, ammonia is incorporated into algal biomass. At high pH values, ammonia leaves the pond through volatilization. There is little evidence for nitrification (and hence denitrification, unless the wastewater has a high nitrate content) (Mara, 1997). This is due to the fact that the population of the nitrifying bacteria is low because of the lack of physical attachment sites in the aerobic zone. Total nitrogen and ammonia removal from WSP can reach 80 and 95%, respectively (Mara, 1997).

Phosphorus removal in WSP is associated with its uptake by algal biomass, precipitation and sedimentation. Mara (1997) suggested that the best way to remove much of the phosphorus in the wastewater by WSP is to increase the number of maturation ponds. However, both nitrogen and phosphorus must be removed in order to prevent eutrophication in receiving waterbodies. The common practice in the design of the WSP is not based on nutrient removal; rather, it is based on BOD and faecal coliform removal.

2.6 Design of Waste Stabilization Ponds

2.6.1 Design parameters

There are four important design parameters for WSP, including temperature, net evaporation, flow and BOD. The climate also is important inasmuch as the processes responsible for BOD_5 and fecal bacterial removal are temperature-dependent. Further, algal photosynthesis depends on solar insulation, itself a function of latitude and cloud cover. Cloud cover periods are seldom a problem because the solar insulation during the day in tropical and sub-tropical regions generally greatly exceeds the saturation light intensity of the algae in the ponds. The design temperature usually is the mean air temperature in the coolest month (or quarter). The pond water

is usually 2-3^{o} C warmer than the air temperature in the cool season, with the reverse also being true.

Because the bacteria responsible for treatment are mesophilic, high temperatures are not a problem. However, low temperatures can be since they slow down the treatment process. In the case of the methanogenic bacteria (crucial to anaerobic digestion), methane production virtually ceases below temperatures of 15^{0} C. Thus, in areas where the pond temperature remains below 15^{0} C for more than a couple of months of the year, careful consideration should be given to deciding whether or not anaerobic units are needed. Net evaporation (evaporation minus rainfall) must be taken into account during the design of facultative and maturation ponds, but not for anaerobic ponds (Arthur, 1976). Anaerobic ponds generally have a scum layer, which effectively prevents significant evaporation.

Total nitrogen and free ammonia (NH_3, rather than $NH^{+}_4 + NH_3$) are important in the design of wastewater-fed fishponds. Typical concentrations of total nitrogen in raw domestic wastewater are 20-70 mg N/l, and total ammonia ($NH_4^{+} + NH_3$) concentrations are 15 – 40 mg N/l. Faecal coliform numbers are important if the pond effluent is to be used for unrestricted crop irrigation or for fishpond fertilization. Grab samples of the wastewater may be used to measure the faecal coliform concentration if wastewater exists.

2.6.2 Estimation of water flows and BOD concentrations

The mean water flows should be carefully estimated, since they have direct effects on the size of the ponds and the construction costs. A suitable design is 85% of the in-house water consumption. The BOD may be measured if wastewater exists, based on 24-hour flow weighted data. Alternatively, the BOD may be estimated from the following equation:

$$L_i = 1000B/q \qquad (2.11)$$

Where Li is wastewater BOD (mg/l), B is BOD contribution (g/capita.day), and Q is wastewater flow (L/ capita.day).

Values of B vary between 30 to 70 g/capita.day, with rich communities producing more BOD than poor communities (Campos and von Sperling, 1996). In medium-sized towns, a value of 50 g/capita.day is more suitable (Mara and Pearson, 1987). A typical design figure for an urban area in a developing country would be 40 - 50 grams BOD_5/capita.day (Arthur, 1976). Although it is dangerous to generalize, in view of the wide variations which can be expected with differing social customs, religion, etc., a BOD_5 per capita contribution of 40 grams/day, with a wastewater contribution of about 100 liters/capita.day is probably a reasonable initial estimate where there is a household water supply; however, flows also may be considerably less. The usual range of faecal coliform in domestic wastewater is $10^7 – 10^8$ faecal coliform/100 ml, with a suitable deign value being 5 x 10^7/100 ml.

2.6.3 Design of anaerobic ponds

The anaerobic ponds are designed on the basis of volumetric loading (λv, g/m^3/d), which is given by:

$$\lambda_v = L_i Q / V_a \qquad (2.12)$$

Where L_i is influent BOD (mg/l), Q is flow rate (m^3/day), and V_a is anaerobic pond volume (m^3). Meiring *et al*., 1998 recommended that the loading should be between 100 – 400 g/m^3.day, in order to maintain anaerobic conditions. Once the organic loading is selected, the volume of the pond is then determined with the use of Equation 2.12. The hydraulic retention time is then calculated, using Equation 2.13, as follows:

$$t_{an} = V_a / Q \qquad (2.13)$$

A retention time less than one day should not be used for anaerobic ponds; if it occurs, however, a retention time of one day should be used, and the volume of the pond should be recalculated. Table 2.7 illustrates the permissible loadings to the anaerobic ponds.

Table 2.7 Design value of permissible volumetric BOD loadings on, and percentage BOD removal in, anaerobic ponds at various temperatures (from Mara and Pearson, 986; Mara *et al*. 1997)

Temperature (^{o}C)	Volumetric loading (g/m^{3}.day)	BOD removal (%)
< 10	100	40
10 - 20	20T – 100	2T+20
20-25	10T+100	2T+20
>25	350	70

2.6.4 Design of facultative ponds

2.6.4.1 Kinetic models for the design of facultative ponds

Facultative ponds can be designed on the basis of kinetic or empirical models. The rate at which the organic matter is oxidized by bacteria is a fundamental parameter in the rational design of biological wastewater treatment systems. It has been found that BOD removal often approximates first-order kinetics; that is, the rate of BOD removal (rate of oxidation of organic matter) at any time is proportional to the quantity of BOD (organic matter) present in the system at that time. This is expressed mathematically in Equation 2.14 as:

$$dL/dt = -k_1 L \qquad (2.14)$$

Where L is the quantity of BOD remaining (= organic matter to be oxidized) at time "t", and k_1 is first-order rate constant for BOD removal (day^{-1}).

The simple approach to the rational design of facultative ponds assumes they are completely mixed reactors in which BOD_5 removal follows first-order kinetics (Marais and Shaw, 1961). The rational equation for the design is illustrated in Equation 2.15:

$$L_e/L_i = 1/(1+k_1 t) \qquad (2.15)$$

Rearranging Equation 2.15:

$$t = (L_i/L_e - 1)(1/k_1) \qquad (2.16)$$

Where t is the retention time (days).

The mid-depth area of the pond is calculated using Equation 2.17:

$$A = Qt/D \tag{2.17}$$

Where Q is the volumetric flow rate (m^3/day), D is the pond depth (m), and A is the mid-depth area (m^2). Substituting "t" from Equation 2.16 into Equation 2.17, the mid-depth area of the pond will be:

$$A = \left(Q/Dk_1\right)\left(L_i/L_e - 1\right) \tag{2.18}$$

The value for k_1 at 20° C was found to be 0.3 day^{-1} (Mara, 1986), while the value of k_T is calculated using Equation 2.12. Note that the rate (k_1) is a gross measure of bacterial activity and, consistent with almost all parameters that describe a biological growth process, its value is strongly temperature-dependent. Its variation with temperature is usually described by an Arrhenius equation (Equation 2.19):

$$k_T = k_{20}\theta^{(T-20)} \tag{2.19}$$

Where θ is the Arrhenius constant, whose value is usually between 1.01 - 1.09. However, the typical values of θ for the design of waste stabilization ponds range between 1.05 - 1.09. Note that the temperature should be taken as the mean temperature of the coldest month. The design example shown in the folder **"Design examples" (Annex 1)** illustrates how to apply the above-noted equations in designing the facultative ponds.

2.6.4.2 Empirical models for the design of facultative ponds

Although several methods are available for designing facultative ponds, Mara (1976) recommended that facultative ponds should be designed on the basis of surface loading (with the reasons stated in the sections above, λ_s, kg/ha.day), which is give by:

$$\lambda_s = 10L_iQ/A_f \tag{2.20}$$

Where L_i is the concentration of influent sewage (mg/l), and A_f is the facultative pond area (m^2). The selection of the permissible design value of λ_s is usually based on the temperature. The

temperature dependence indicates that the design values of λ_s increases with temperature. The earliest relationship between λ_s and temperature was given by McGarry and Pescode (1970), and later by Mara (1976). The Mara (1976) equation is as shown in Equation 2.21:

$$\lambda_s = 20T - 120 \tag{2.21}$$

However, an appropriate λ_s and temperature relationship is presented by Mara (1987) as:

$$\lambda_s = 350\left[1.107 - 0.002T\right]^{(T-20)} \tag{2.22}$$

Once the surface loading has been selected, the area of the facultative pond can be calculated from Equation 2.20, and its retention time (θ_f, day) is calculated from Equation 2.23:

$$\theta_f = A_f D / Q_m \tag{2.23}$$

Where D is the pond depth (usually 1.5m), and Q_m is the mean flow (m^3/day).

The mean flow is the mean of the influent and effluent flows (Q_i and Q_e), the latter being the former less net evaporation and seepage. Thus Equation 2.23 becomes:

$$\theta_f = \frac{A_f D}{\left[\frac{1}{2}\left(Q_i + Q_e\right)\right]} \tag{2.24}$$

If seepage is negligible, Q_e is given by:

$$Q_e = Q_i - 0.001 A_f e \tag{2.25}$$

Where e is net evaporation rate (mm/day). Thus, Equation 2.24 becomes:

$$\theta_f = \frac{2A_f D}{\left[2Q_i - 0.001 A_f e\right]} \tag{2.26}$$

A minimum retention time value of 5 days should be adopted for temperatures below 20^{o} C, and 4 days for temperature above 20^{o} C. This is to minimize hydraulic short-circuiting, and to give algae sufficient time to multiply (i.e., to prevent algal washout).

2.6.4.3 Design of maturation ponds for faecal coliform removal

The method of Marais (1974) is generally used to design a pond series for faecal coliform removal. This assumes that faecal coliform removal can be reasonably well represented by a first-order kinetic model in a completely-mixed reactor. The resulting equation for a single pond is given by:

$$N_e = \frac{N_i}{(1 + k_T \theta)} \tag{2.27}$$

Where N_e and N_i are the number of faecal coliform/100 ml in the effluent and influent, k_T is the first-order rate constant for faecal coliform removal (d^{-1}), and Θ is a retention time (day).

For a series of anaerobic, facultative and maturation ponds, Equation 2.27 becomes:

$$N_e = \frac{N_i}{\left[(1 + k_T \theta_a)(1 + k_T \theta_f)(1 + k_T \theta_M)^n\right]} \tag{2.28}$$

Where the sub-scripts, a, f and m, refer to the anaerobic, facultative and maturation ponds, and *n* is the number of maturation ponds. Equation 2.28 assumes that all the maturation ponds are equally-sized, which is the most efficient configuration (Marais, 1974), but which may not be topographically possible (in which case, the last term of the denominator in Equation 2.28 is replaced by: $[(1+k_T\Theta_{m1})\ (1+k_T\Theta_{m2}).\ldots\ldots(1+k_T\Theta_{mn})]$).

The value of k_T is highly temperature-dependent. Marais (1974) found that:

$$k_T = 2.6\ (1.19)^{T-20} \tag{2.29}$$

Thus, k_T changes by 19% for every change in temperature of 1^{o} C. Equation 1.20 contains the implicit assumption that the value of k_T determined from Equation 2.29 is equally valid in anaerobic, facultative ponds. Although this is not true, Pearson *et al*. (1996*a*) nevertheless found

that Equation 2.28 represents the removal of faecal coliform in a series of ponds as a whole reasonably well. Maturation ponds require careful design to ensure that their faecal coliform removal follows that given by Equations 2.28 and 2.29.

Examination of Equation 2.28 shows that it contains two unknowns (Θ_m and *n*), since by this stage of the design process, Θ_a and Θ_f will have been calculated, N_i measured or estimated, N_e set (e.g., 1000/100 ml for unrestricted irrigation), and k_T calculated from Equation 2.29. The best approach to solving Equation 2.28 is to calculate the values of Θ_m corresponding to n = 1, 2, 3, etc., and then adopt the following rules to select the most appropriate combination of Θ_m and *n,* namely:

(a) $\Theta_m \nsucc \Theta_f$ and

(b) $\Theta m \prec \theta_m^{min}$

Where θ_m^{min} is the minimum acceptable retention time in a maturation pond. This term is introduced to minimize hydraulic short-circuiting and prevent algal washout. Marais (1974) recommends a value of 3 days, although at temperatures below 20^o C, values of 4-5 days are preferable. The remaining pairs of Θ_m and n, together with the pair θ_m^{min} and ñ, where ñ is the first value of n for which Θ_m is less than θ_m^{min}, are then compared, and the one with the least product selected, since this will identify the least land area requirements. The BOD loading on the first maturation pond must be checked, and must not be higher than that on the preceding facultative pond; in fact, it is preferable that it be significantly lower. The maximum BOD loading in the first maturation pond should be 75% of that on the preceding facultative pond. It is not necessary to check the BOD loadings on subsequent maturation ponds, as the non-algal BOD contribution to the load on them is very low.

The loading on the first maturation pond is calculated on the assumption that 70% of unfiltered BOD has been removed in the preceding anaerobic and facultative ponds (or 80% for temperatures above 20^o C). Mara and Pearson, (1987) also suggested that 90% cumulative removal in anaerobic and facultative ponds, and then 25% in each maturation pond, for

temperatures above 25° C (80% and 20%, respectively, for temperatures below 20° C), when the BOD is based on filtered BOD values. Thus:

$$\lambda_{S(m1)} = \frac{10(0.3L_i)Q}{A_{m1}} \tag{2.30}$$

or, since $Q\theta_{m1}=A_{m1}D$:

$$\lambda_{S(m1)} = 10(0.3L_i)D/\theta_{m1} \tag{2.31}$$

The maturation pond area is calculated from a rearrangement of Equation 2.26, as follows:

$$A_m = 2Q_i\theta_m/(2D + 0.001e\theta_m) \tag{2.32}$$

The design example in the folder **"Design examples" (Annex 1)** indicates how the maturation pond is designed according to the above-noted procedures.

2.6.5 Helminth egg removal

Helminth eggs are normally removed by sedimentation, with the process occurring in the anaerobic or primary facultative ponds. If the final effluent is to be used for restricted irrigation, it is necessary to ensure that it contains no more than one egg per liter. Analysis of egg removal in the pond has yielded the following relation reported by Ayres *et al.* (1992):

$$R = 100[1 - 0.14\exp(-0.38\theta)] \tag{2.33}$$

Where R is percent egg removal, and θ is a retention time (day). The equation corresponding to the lower 95% confidence limit of Equation 2.33 is:

$$R = 100[1 - 0.41\exp(-0.49\theta + 0.0085\theta^2)] \tag{2.34}$$

Table 2.8 Design values of percent helminth egg removal (R) (in individual anaerobic, facultative or maturation ponds for hydraulic retention times (θ) in the range of 1 – 20 days, as calculated from Equation 2.34)

θ	R	θ	R	θ	R
1.0	74.67	4.0	93.38	10	99.29
1.2	76.95	4.2	93.66	10.5	99.39
1.4	79.01	4.4	93.40	11	99.48
1.6	80.87	4.6	94.85	12	99.61
1.8	82.55	4.8	95.25	13	99.70
2.0	84.08	5.0	95.62	14	99.77
2.2	85.46	5.5	96.42	15	99.82
2.4	87.72	6.0	97.06	16	99.86
2.6	87.85	6.5	97.57	17	99.88
2.8	88.89	7.0	97.99	18	99.90
3.0	89.82	7.5	98.32	19	99.92
3.2	90.68	8.0	98.60	20	99.93
3.4	91.45	8.5	98.82		
3.6	92.16	9.0	99.01		
3.8	92.80	9.5	99.16		

Equation 2.34 or Table 2.8 is recommended for use in the design. Annex A provides guidance on how to design a waste stabilization pond for the removal of Helminth eggs.

2.6.6 Design of WSP for nutrient removal

The design equation for nutrient removal in WSP's is based on equations developed in North America; thus, the designer should use these equations with the caution that they might not accurately predict the expected performance. The equation for ammonia-nitrogen (NH_3 + NH^+_4) removal in individual facultative ponds was presented by Pono and Middlebrooks (1982). The equation for temperatures below 20^o C is as follows:

$$C_e = C_i / \{1 + [(A/Q)(0.0038 + 0.000134T)\exp((1.041 + 0.044T)(pH - 6.6))]\} \quad (2.35)$$

And for temperatures above 20^o C:

$$C_e = C_i / \{1 + [5.035x10^{-3}(A/Q)][\exp(1.540x(pH - 6.6))]\} \quad (2.36)$$

Where C_e is ammonia-nitrogen concentration in pond effluent (mg N/L), C_i is ammonia-nitrogen concentration in pond influent (mg N/L), A is pond area (m^2), and Q is influent flow rate (m^3/day).

Total nitrogen removal in the individual facultative and maturation ponds was presented by Reed (1995), as follows:

$$C_e = C_i \exp\left\{-\left[0.0064(1.039)^{T-20}\right]\left[\theta + 60.6(pH - 6.6)\right]\right\} \quad (2.37)$$

Where Ce and Ci is the total nitrogen concentration in the pond effluent and influent, respectively (mg N/L), T is temperature (^{o} C; range: 1-28^o C), and θ is retention time (days; range: 5-231 days). The pH values used in the above equations may be estimated as follows:

$$pH = 7.3\exp(0.0005A) \quad (2.38)$$

Where A is influent alkalinity (mg $CaCO_3$/L).

The design example in the folder **"Design examples" (Annex 1)** shows how to use these equations to design WSP's for nutrient removal.

Part Two

Chapter Three: Constructed Wetlands

3.1 General

Constructed wetlands (CWs) are planned systems designed and constructed to employ wetland vegetation to assist in treating wastewater in a more controlled environment than occurs in natural wetlands. Hammer (1990) defines constructed wetlands as a designed, manmade complex of saturated substrate, emergent and submerged vegetation, animal life, and water that simulate wetlands for human uses and benefits. Constructed wetlands are an "eco-friendly" alternative for secondary and tertiary municipal and industrial wastewater treatment. The pollutants removed by CW's include organic materials, suspended solids, nutrients, pathogens, heavy metals and other toxic or hazardous pollutants. In municipal applications, they can follow traditional sewage treatment processes. Different types of constructed wetlands can effectively treat primary, secondary or tertiary treated sewage. However wetlands should not be used to treat raw sewage and, in industrial situations, the wastes may need to be pre-treated so that the biological elements of the wetlands can function effectively with the effluent. CW's are practical alternatives to conventional treatment of domestic sewage, industrial and agricultural wastes, storm water runoff, and acid mining drainage.

3.2 Types of Constructed Wetlands

Constructed wetlands for wastewater treatment can be categorized as either Free Water Surface (FWS) or Subsurface Flow (SSF) systems. In FWS systems, the flow of water is above the ground, and plants are rooted in the sediment layer at the base of water column (Figure 3.1). In SSF systems, water flows though a porous media such as gravels or aggregates, in which the plants are rooted (Figure 3.2). Table 3.1 illustrates the type of wetlands, vegetation types and water column contacts in constructed wetlands.

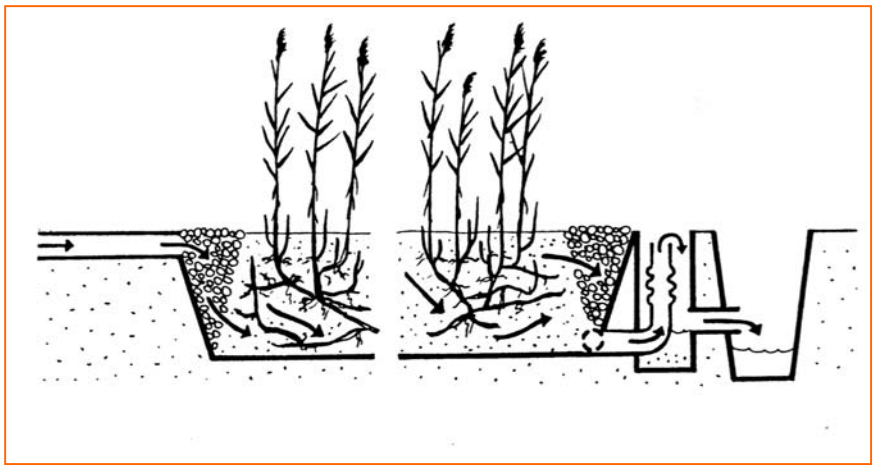


Fig. 3.1 Emergent macrophyte treatment system with horizontal sub-surface flow (Brix 1993).

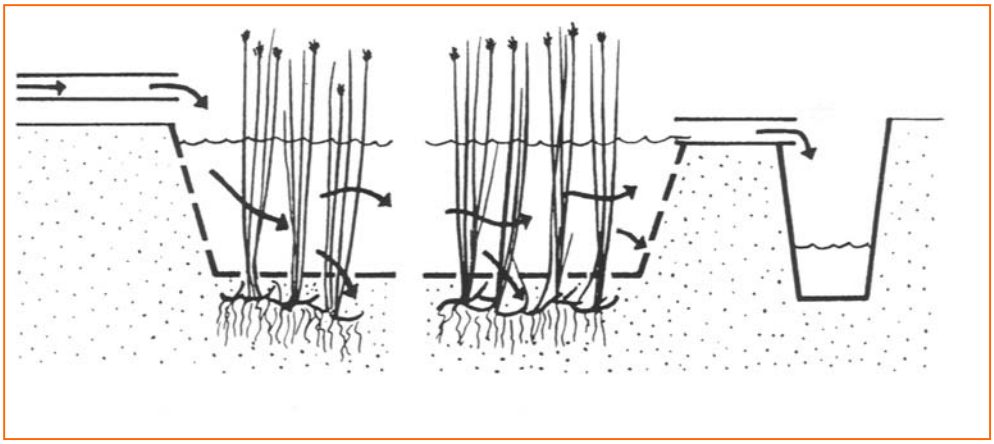


Fig. 3.2 Emergent macrophytes treatment system with surface flow (Brix, 1993)

Table 3.1 Vegetation type and water column contact in constructed wetlands

Constructed wetland type	Type of vegetation	Section in contact with water column
Free water surface (FWS)	Emergent	Stem, limited leaf contact
	Floating	Root zone, some stem / tubers
	Submerged	Photosynthetic part, possibly root zone
Sub-surface flow (SSF)	Emergent	Rhizome and root zone

SSF systems are most appropriate for treating primary wastewater, because there is no direct contact between the water column and the atmosphere. There is no opportunity for vermin to breed, and the system is safer from a public health perspective. The system is particularly useful for treating septic tank effluent or grey water, landfill leachate and other wastes that require removal of high

concentrations organic materials, suspended solids, nitrate, pathogens and other pollutants. The environment within the SSF bed is mostly either anoxic or anaerobic. Oxygen is supplied by the roots of the emergent plants and is used up in the Biofilm growing directly on the roots and rhizomes, being unlikely to penetrate very far into the water column itself. SSF systems are good for nitrate removal (denitrification), but not for ammonia oxidation (nitrification), since oxygen availability is the limiting step in nitrification.

There are two types of SSF systems: horizontal flow SSF (hSSF) and vertical flow SSF (vSSF). The most common problem with hSSF is blockage, particularly around the inlet zone, leading either to short circuiting, surface flow or both. This occurs because of poor hydraulic design, insufficient flow distribution at the inlet, and inappropriate choice of porous media for the inlet zone. Properly-designed SSF systems are very reliable.

FWS systems are very appropriate for polishing secondary and tertiary effluents, and for providing habitat. The environment in the FWS systems is generally aerobic at, and near, the surface, tending toward anoxic conditions near the bottom sediment. The microbial film grows on all available plant surfaces, and is the main mechanism of pollutant removal. FWS usually exhibits more biodiversity than does SSF systems.

The objective of using CWs is to remove organic matter, suspended solids, pathogenic organisms, and nutrients such as ammonia and other forms of nitrogen and phosphorus. The growing interest in wetland system is due in part to recognition that natural systems offer advantages over conventional activated sludge and trickling filter systems. When the same biochemical and physical processes occur in a more natural environment, instead of reactor tanks and basins, the resulting system often consumes less energy, is more reliable, requires less operation and maintenance and, as a result costs less. They also are used for removing heavy metals and toxic compounds. This manual is concerned with the design, operation and maintenance of sub-surface flow constructed wetlands.

3.3 Advantages and disadvantages of constructed wetlands

Constructed wetlands are (1) relatively inexpensive to construct and operate, (2) easy to maintain, (3) provide effective and reliable wastewater treatment, (4) relatively tolerant of fluctuating hydrologic and contaminant loading rates (optimal size for anticipated waste load), and (5) provide indirect benefits such as green space, wildlife habitats and recreational and educational areas. The disadvantages are (1) the land requirements (cost and availability of suitable land), (2) current imprecise design and operation criteria, (3) biological and hydrological complexity and our lack of understanding of important process dynamics, (4) the costs of gravel or other fills, and site grading during the construction period, and (5) possible problems with pests. Mosquitoes and other pests could be a problem for an improperly designed and managed SSF. The system may be used for small communities and, therefore, may be located close to the users. The dependence of wetland community on hydrologic patterns is most obvious in the change in species composition resulting from alterations in water depths and flows.

3.4 Configuration, Zones and components of constructed wetlands

Influents to constructed wetland can range from raw wastewater to secondary effluents. Most constructed wetlands have the following zones: inlet zone, macrophyte zone, littoral zone and outlet zones. The components associated in each zones are as shown in Table 3. 2, and can include substrates with various rates of hydraulic conductivity, plants, a water column, invertebrate and vertebrates, and an aerobic and anaerobic microbial population. The water flow is maintained approximately 15 – 30 cm below the bed surface. Plants in wastewater systems have been viewed as nutrient storage compartments where nutrient uptake is related to plant growth and production. Harvesting before senescence may permanently remove nutrients from the systems. Within the water column, the stems and roots of wetland plants significantly provide the surface area for the attachment of microbial population. Wetland plants have the ability to transport atmospheric oxygen and other gases down into the root to the water column. Most media used include crushed stones, gravels, and different soils, either alone or in combination. Most beds are underlain by impermeable materials to prevent water seepage and assure water level control. Wastewater flows laterally, being purified during contact with media surface and vegetation roots. The sub-surface

Table 3.2 Wetland zones and their associated components

Zones	Components	Functions
Inlet zone	Inlet structure, splitter box	Flow distribution across the full width at a minimum of 3 – 5 m interval
Macrophyte zone	Porous bed/substrate, open water, vegetation, island, mixing baffles, flow diversion	To provide the substrate with high hydraulic conductivity; to provide surface for the growth of Biofilm; to aid in the removal of fine particles by sedimentation or filtration; to provide suitable support for the development of extensive root and rhizome system for emergent plants.
Deep water zone	Usually deeper, non-vegetated	Reduce short circuiting by re-orienting flow path; reduce stagnant areas by allowing for mixing by wind; enable UV disinfections of bacteria and other pathogens; provide habitat for waterfowl.
Littoral zone	Littoral area	Littoral vegetation protects embankment from erosion; littoral vegetation serves to break up wave action.
Outlet zone	Collection devices, spillway, weir, outlet structures	Control the depth of the water in the wetland; collect the effluent water without creating of dead zones in the wetlands; provide access for sampling and flow monitoring.

zone is saturated and generally anaerobic, although excess DO conveyed through the plant root system supports aerobic microsites adjacent to the root and rhizomes.

3.5 Processes in Sub-surface Flow Constructed Wetlands (SSFCW)

Wetland can effectively remove or convert large quantities of pollutants from point sources (municipal, industrial and agricultural wastewater) and non-point sources (mines, agriculture and urban runoff), including organic matter, suspended solids, metals and nutrients. The focus on wastewater treatment by constructed wetlands is to optimise the contact of microbial species with substrate, the final objective being the bioconversion to carbon dioxide, biomass and water. Wetlands are characterized by a range of properties that make them attractive for managing pollutants in water (Bavor and Adcock, 1994). These properties include high plant productivity, large adsorptive capacity of the sediments, high rates of oxidation by microflora associated with plant biomass, and a large buffering capacity for nutrients and pollutants. Table 3.3 provides an overview of pollutant removal mechanisms in constructed wetlands (Mitchell, 1996).

3.5.1 Biological processes

There are six major biological reactions involved in the performance of constructed wetlands, including photosynthesis, respiration, fermentation, nitrification, denitrification and

Table 3.3 Overview of pollutant removal mechanisms

Pollutant	Removal Processes
Organic material (measured as BOD)	Biological degradation, sedimentation, microbial uptake
Organic contaminants (e.g., pesticides)	Adsorption, volatilization, photolysis, and biotic/abiotic degradation
Suspended solids	Sedimentation, filtration
Nitrogen	Sedimentation, nitrification/denitrification, microbial uptake, volatilization
Phosphorous	Sedimentation, filtration, adsorption, plant and microbial uptake
Pathogens	Natural die-off, sedimentation, filtration, predation, UV degradation, adsorption
Heavy metals	Sedimentation, adsorption, plant uptake

microbial phosphorus removal (Mitchell, 1996b). Photosynthesis is performed by wetland plants and algae, with the process adding carbon and oxygen to the wetland. Both carbon and oxygen drive the nitrification process. Plants transfer oxygen to their roots, where it passes to the root zones (rhizosphere). Respiration is the oxidation of organic carbon, and is performed by all living organisms, leading to the formation of carbon dioxide and water. The common microorganisms in the CW are bacteria, fungi, algae and protozoa. The maintenance of optimal conditions in the system is required for the proper functioning of wetland organisms. Fermentation is the decomposition of organic carbon in the absence of oxygen, producing energy-rich compounds (e.g., methane, alcohol, volatile fatty acids). This process is often undertaken by microbial activity. Nitrogen removal by nitrification/denitrification is the process mediated by microorganisms. The physical process of volatilization also is important in nitrogen removal. Plants take up the dissolved nutrients and other pollutants from the water, using them to produce additional plant biomass. The nutrients and pollutants then move through the plant body to underground storage organs when the plants senesce, being deposited in the bottom sediments through litter and peat accretion when the plants die.

Wetland microorganisms, including bacteria and fungi, remove soluble organic matter, coagulate colloidal material, stabilize organic matter, and convert organic matter into various gases and new cell tissue (Mitchell, 1996a). Many of the microorganisms are the same as those occurring in conventional wastewater treatment systems. Different types of organisms, however, have specific tolerances and requirements for dissolved oxygen, temperature ranges and nutrients.

3.5.2 Chemical processes

Metals can precipitate from the water column as insoluble compounds. Exposure to light and atmospheric gases can break down organic pesticides, or kill disease-producing organisms (EPA, 1995). The pH of water and soils in wetlands exerts a strong influence on the direction of many reactions and processes, including biological transformation, partitioning of ionized and un-ionised forms of acids and bases, cation exchange, solid and gases solubility.

3.5.3 Physical processes

Sedimentation and filtration are the main physical processes leading to the removal of wastewater pollutants. The effectiveness of all processes (biological, chemical, physical) varies with the water residence time (i.e., the length of time the water stays in the wetland). Longer retention times accelerate the remove of more contaminants, although too-long retention times can have detrimental effects.

3.5.4 Limitations of wetland processes

3.5.4.1 Process rates

The chemical and biological processes occur at a rate dependent on environmental factors, including temperature, oxygen and pH. Metabolic activities are decreased by low temperature, reducing the effectiveness of pollutant uptake processes relying on biological activity. Low oxygen concentrations limit the processes involving aerobic respiration within the water column, and may enhance anaerobic processes, which can cause further degradation of water quality. Many metabolic activities are pH-dependent, being less effective if the pH is too high or low.

3.5.4.2 Hydrological limitations

The capacity of wetlands to treat wastewater is limited, both in terms of the quantity of water, and the total quantity of the pollutants. Hydraulic overloading occurs when the water flow exceeds the design capacity, causing a reduction in water retention time that affects the rate of pollutant removal. Pollutant overloading occurs when the pollutant input exceeds the process removal rates within the wetland (White *et al.,* 1996). Hydraulic overloading may be compensated for by using surcharge mechanisms, or the design may be based on a flush principle, whereby large water flows bypass the wetland when used for storm water treatment (White *et al.,* 1996). Inflow

variations are typically less extreme for wetlands treating municipal wastewaters, with incoming pollutant loads also being more defined and uniform.

3.5.5 Wetland nitrogen processes

The most important nitrogen species in wetlands are dissolved ammonia (NH^+_4), nitrite (NO^-_2), and nitrate (NO^-_3). Other forms include nitrous oxide gas (N_2O), nitrogen gas (N_2), urea (organic), amino acids and amine (Kadlec & Knight, 1996). Total nitrogen in any system is referred to as the sum of organic nitrogen, ammonia, nitrate and nitrous gas (Organic-N + NH^+_4 + NO^-_3 + N_2O). The various nitrogen forms are continually involved in transformations from inorganic to organic compounds, and vice-versa. Many of these transformations are biotic, being carried out by nitrobacter and nitrosomonas (Kadlec & Knight, 1996). As it undergoes its various transformations, nitrogen is taken up by wetland plants and microflora (preferentially as NH^+_4, and NO^-_3), some is leached to the subsoil, some is liberated as gas to the atmosphere, and some flows out of the wetland, normally in a dissolved form. Organic nitrogen comprises a significant fraction of wetland biota, detritus, soils, sediments and dissolved solids (Kadlec and Knight, 1996). It is not readily assimilated by aquatic plants, and must be converted to NH^+_4, or NO^-_3 through multiple conversions requiring long reaction time (Kadlec & Knight, 1996). The process of biological nitrogen removal follows several sequences. Nitrification first takes place, generally in the rhizosphere and in biofilms (aerobic process). Denitrification may then follow, occurring in soils and below the oxidized microzone at the soil/water interface, as it is an anaerobic process (Broderick et al., 1989).

Nitrification is a two-step process catalysed by Nitrosomonas and nitrobacter bacteria. In the first step, ammonia is oxidized to nitrite in an aerobic reaction catalyzed by Nitrosomonas bacteria, as shown in Equation 3.1:

$$NH^+_4 + O_2 \xrightarrow{+\text{Nitrosomonas}} H^+ + NO^-_2 + H_2O \qquad (3.1)$$

The nitrite produced is oxidized aerobically by nitrobacter bacteria, forming nitrate (Equation 3.2), as follows:

$$NO^-_2 + O_2 \xrightarrow{\text{Nitrobacter}} NO^-_3 \qquad (3.2)$$

The first reaction produces hydroxonium ions (acid pH), which react with natural carbonate to decrease the alkalinity (Mitchell, 1996a). In order to perform nitrification, the nitrosomonas must compete with heterotrophic bacteria for oxygen. The BOD of the water must be less than 20 mg/l before significant nitrification can occur (Reed et al., 1995). Temperatures and water retention times also may affect the rate of nitrification in the wetland. Denitrification is the process in which nitrate is reduced in anaerobic conditions by the benthos to a gaseous form. The reaction is catalyzed by the denitrifying bacteria Pseudomonas spp. and other bacteria, as follows:

$$NO^-_3 + \text{Organic-C} \xrightarrow{\text{Denitrifying Bacteria}} N_2\,(NO\ \&N_2O)_{(G)} + CO_{2(G)} + H_2O \qquad (3.4)$$

Denitrification requires nitrate, anoxic conditions and carbon sources (readily biodegradable). Nitrification must precede denitrification, since nitrate is one of the prerequisites. The process of denitrification is slower under acidic condition. At a pH between 5-6, N_20 is produced. For a pH below 5, N_2 is the main nitrogenous product (Nuttall et al., 1995). NH^+_4 is the dominant form of ammonia-nitrogen at a pH of 7, while NH_3 (present as a dissolved gas) predominates at a pH of 12. Nitrogen cycling within, and removal from, the wetlands generally involves both the translocation and transformation of nitrogen in the wetlands, including sedimentation (resuspension), diffusion of the dissolved form, litter fall, adsorption/desorption of soluble nitrogen to soil particles, organism migration, assimilation by wetland biota, seed release, ammonification (mineralisation) (Orga-N – NH^+_4), ammonia volatilization (NH^+_4 – NH_3 (gas)), bacterially-mediated nitrification/denitrification reactions, nitrogen fixation (N_2, N_2O (gases – organic-N)), and nitrogen assimilation by wetland biota (NH^+_4, Nox organic – N, with NO_x usually as NO^-_3). Precipitation is not a significant process due to the high solubility of nitrogen, even in inorganic form. Organic nitrogen comprises a significant fraction of wetland biota, detritus, soils, sediments and dissolved solids (Kadlec and Knight, 1996).

3.5.6 Phosphorus removal

Phosphorus is an essential requirement for biological growth. An excess of phosphorus can have secondary effects by triggering eutrophication within a wetland, and leading to algal blooms and other water quality problems. Phosphorus may enter a wetland in dissolved and particulate forms. It exits wetlands in outflows, by leaching into the sub-soil, and by removal by plant and animals. Phosphorus removal in wetlands is based on the phosphorous cycle, and can

involve a number of processes. Primary phosphorus removal mechanisms include adsorption, filtration and sedimentation. Other processes include complexation/precipitation and assimilation/uptake. Particulate phosphorus is removed by sedimentation, along with suspended solids. The configuration of constructed wetlands should provide extensive uptake by Biofilm and plant growth, as well as by sedimentation and filtration of suspended materials. Phosphorus is stored in the sediments, biota, (plants, Biofilm and fauna), detritus and in the water. The interactions between compartments depend on environmental conditions such as redox chemistry, pH and temperature. The redox status of the sediments (related to oxygen content) and litter/peat compartment is a major factor in determining which phosphorus cycling processes will occur. Under low oxygen conditions (low redox potential), phosphorus is liberated from the sediments and soils back into the water column, and can leave the wetland if the anaerobic condition is not reversed (Moss et al., 1986).

3.5.7 Suspended solids

Solids may be derived from outside a wetland (e.g., inflows and atmospheric inputs), and from within a wetland from plankton (zooplankton and phytoplankton), and plant and animal detritus. With low wetland water velocities and appropriate composition of influent solids, suspended solids will settle from the water column within the wetland. Sediment resuspension not only releases pollutants from the sediments, it increases the turbidity and reduces light penetration. The physical processes responsible for removing suspended solids include sedimentation, filtration, adsorption onto Biofilm and flocculation/precipitation. Wetland plants increase the area of substrate available for development of the Biofilm. The surface area of the plant stems also traps fine materials within its rough structure.

3.5.8 Pathogen removal

Pathogens are disease-causing organisms (e.g., bacteria, viruses, fungi, protozoa, helminthes). Wetlands are very effective at removing pathogens, typically reducing pathogen number by up to five orders of magnitude from wetland inflows (Reed at al., 1995). The processes that may remove pathogens in wetlands include natural die-off, sedimentation, filtration, ultra-violet light ionization, unfavorable water chemistry, temperature effects, predation by other organisms and pH (Kadlec & knight 1996). Kadlec and Knight (1996) showed that vegetated wetlands seem more

effective in pathogen removal, since they allow a variety of microorganisms to grow which may be predators to pathogens.

3.5.9 Heavy metal removal

Heavy metals is a collective name given to all metals above calcium in the Periodic Table of Elements, which can be highly toxic, and which have densities greater that 5g/cm^3 (Skidmore and Firth, 1983). The main heavy metals of concern in freshwater include lead, copper, zinc, chromium, mercury, cadmium and arsenic. There are three main wetland processes that remove heavy metals; namely, binding to soils, sedimentation and particulate matter, precipitation as insoluble salts, and uptake by bacteria, algae and plants (Kadlec & Knight, 1996). These processes are very effective, with removal rates reported up to 99% (Reed et al., 1995). A range of heavy metals, pathogens, inorganic and organic compounds present in wetlands can be toxic to biota. The response of biota depends on the toxin concentration and the tolerance of organisms to a particular toxin. Wetlands have a buffering capacity for toxins, and various processes dilute and break down the toxins to some degree.

3.6 Abiotic Factors and their Influence on Wetlands.

3.6.1 Oxygen

Oxygen in wetland systems is important for heterotrophic bacterial oxidation and growth. It is an essential component for many wetland pollutant removal processes, especially nitrification, decomposition of organic matter, and other biological mediated processes. It enters wetlands via water inflows or by diffusion on the water surface when the surface is turbulent. Oxygen also is produced photosynthetically by algae. Plants also release oxygen into the water by root exudation into the root zone of the sediments. Many emergent plants have hollow stems to allow for the passage of oxygen to their root tissues. The oxygen-demand processes in wetlands include sediment-litter oxygen demand (decomposition of detritus), respiration (plants/animals), dissolved carbonaceous BOD, and dissolved nitrogen that utilizes oxygen through nitrification processes (Kadlec & Knight, 1996). The oxygen concentration decreases with depth and distance from the water inflow into the wetland. It is typically high at the surface, grading to very low in the sediment –water interface.

3.6.2 pH

The pH of wetlands is correlated with the calcium content of water (pH 7 = 20 mg Ca/L). Wetland waters usually have a pH of around 6-8 (Kadlec and Knight, 1996). The biota of wetlands especially can be impaired by sudden changes in pH.

3.6.3 Temperature

Temperature is a widely-fluctuating abiotic factor that can vary both diurnally and seasonally. Temperature exerts a strong influence on the rate of chemical and biological processes in wetlands, including BOD decomposition, nitrification and denitrification.

3.7 Process Design for Constructed Wetlands

3.7.1 Hydrology

It is not safe to ignore water exchanges with the atmosphere, mainly because they can significantly contribute to water flows. Rain causes two opposing effects, including (1) dilution of waters, thereby reducing material concentrations, and (2) increased water velocity, increasing the water retention time within a wetland. The presence of vegetation may retard the evapotranspiration, although wetland evapotranspiration is usually 0.8 times the Class A pan set at an adjacent open site. Preparation of an accurate hydrological budget is needed to properly design a constructed wetland. The water balance to a wetland can be calculated as follows:

$$\frac{dV}{dt} = Q_i + Q_e + P - ET \tag{3.5}$$

Where Q_i is the influent wastewater flow (volume/time), Q_e is the effluent wastewater flow (volume/time), P is the precipitation (volume/time), ET is the evapotranspiration (volume/time), V is the volume, and t is time. The equation does not consider the inflow from, and to, the groundwater, since the SSF wetlands should be lined.

3.8 Design Approaches for Constructed Wetlands

The characteristic wastewater parameters to be treated by the CW include BOD, COD, suspended solids (SS), nitrogen compounds, phosphorus compounds, heavy metals, and pathogenic

organisms. Three approaches have been used to design constructed wetlands. An empirical approach is based on two different "rule of thumb" approaches (Reed *et al*., 1995; Kadlec and Knight 1996). In the United Kingdom, constructed wetlands have been designed using a rule of thumb method. In contrast, both Reed et al. (1995) and Kadlec & Knight (1996) consider wetlands as attached-growth biological reactors, therefore using a first-order plug flow kinetics model as the basis for their performance equations. The removal of soluble BOD in the SSFCW is due to microbial growth attached to the plant roots, stems, leaf litter and substrates. Both Reed *et al*. (1995) and Kadlec and Knight (1996) admit that BOD_5 removal in SSFCW can be described with first-order plug flow kinetics. First-order kinetics simply means that the rate of removal of a particular pollutant is direct proportional to the remaining concentration at any point within the wetland cell. Two idealized mixing theories may be applied:

- Completely mixed reactor -- The concentration is the same as the effluent concentration at any point in the reactor;
- Plug flow -- The concentration of the reactant decreases along the length of the flow path through the reactor.

Plug flow obviously provides a more appropriate description of the flow pattern in constructed wetlands.

The main difference between Reed's approach and that of Kadlec and Knight is the basis for their choice of rate constants. Reed's equations use a volumetric, temperature-dependent basis, with the calculations being based on the available volume of the wetland and the average water temperature. Kadlec and Knight's equations assume an areal basis; thus, the rate constant is related only to the wetland surface area, and temperature changes are considered significant only for nitrogen removal. One of the limitations of the Kadlec and Knight approach is that, as the required effluent concentration approaches the minimum pollutant concentration, their predicted area for the treatment increases exponentially, leading to severe over-estimates of the required area in cases near the pollutant concentration limit.

Kadlec and Knight's rate constants are the result of analyzing the performance of all the FWS and SSF constructed wetlands listed in the North American Data Base (NADB), which were not

designed using the first-order plug flow approximation. It should be noted that Kadlec and Knight's statistical analysis of the available data is comprehensive, and the authors point out that caution should be used in applying their so-called "global" parameter values to new systems. Reed included temperature-dependence term for all pollutant of general interest in the design of constructed wetland.

3.8.1 Reed's method for the design of constructed wetlands

The equations of Reed *et al*. (1995) are based on the first-order plug flow assumption for those pollutants that are removed primarily by biological processes, including biochemical oxygen demand (BOD), ammonia (NH_4) and nitrate (NO_3). Reed suggests separate equations for total suspended solids (TSS) and total phosphorus (TP), based on regression analyses of an early version of the NADB (Knight *et al*., 1993) for constructed wetlands. For the removal of pathogenic organisms in constructed wetlands, he suggests the same approach as that used for waste stabilization ponds. The design equations based on Reed *et al*. (1995) are as presented below:

For removal of BOD, NH_4 and NO_3 in constructed wetlands:

$$\ln\left(\frac{C_i}{C_e}\right) = K_T t \tag{3.6}$$

and

$$t = \frac{V_f}{Q} = \frac{LWy}{Q} = \frac{nLWy}{Q} = \frac{A_s yn}{Q} \tag{3.7}$$

The actual retention time in constructed wetlands is a function of the porosity of the substrate used, as defined by the third and fourth expression in Equation 3.7. The substrate porosity may be obtained as $n = V_v/V$. The value n is defined as the remaining cross-sectional area available for flow:

$$K_T = K_R \theta_R^{(T_W - T_R)} \tag{3.8}$$

and

$$A_s = LW = \frac{Qt}{yn} = \frac{Q\ln\left(\frac{C_i}{C_e}\right)}{K_T yn} = \frac{\left[Q(\ln C_o - \ln C_e)\right]}{K_T yn} \quad (3.9)$$

Alternatively:

$$C_e = C_i \exp\left(\frac{-A_s K_T yn}{Q}\right) \quad (3.10)$$

and

$$HLR = \frac{100Q}{A_s} \quad (3.11)$$

Where A_s = treatment area of the wetland (m^2), C_e = outlet effluent pollutant concentration (mg/l), C_i = influent pollutant concentration (mg/l), HLR = hydraulic loading rate (cm/day), K_R = rate constant at reference temperature (day^{-1}), K_T = Rate constant at temperature TW (day^{-1}), L = length of the wetland (m), n = porosity (percent, expressed as decimal fraction), Q = average flow rate through the wetland (m^3/day), t = hydraulic residence time (day-1), T_W = water temperature (^oC), T_R = reference temperature (^oC), V_f= volume of wetland available for water flow (m^3), W = width of the wetland (m), y = depth of the wetland (m), θ_R = temperature coefficient for rate constant, and V_v and V are the volume of the voids and total volume, respectively. Table 3.4. shows the parameters for the design of the two types of constructed wetlands based on the Reed *et al.* (1995) equation.

Table 3.4 Temperature coefficient for rate constant for Reed *et al.* (1995[a]) design equations

Parameter	BOD removal	Nitrification [b] (NH_4 – removal)	Denitrification [b] (NO_3 – removal)	Pathogen (removal)
T_R (^{o}C)	20	20	20	20
Residual (mg/l)	6	0.2	0.2	-
For free water surface wetlands:				
K_R (day^{-1})	0.678	0.2187	1.00	2.6
θ_R	1.06	1.048	1.15	1.15
For sub-surface flow wetlands:				
K_R (day^{-1})	1.104	K_{NH} [c]	1.00	2.6
θ_R	1.06	1.048	1.15	1.19

[a] *All rate coefficients are for temperature greater than 1^o C;*
[b] *nitrification/denitrification are not possible at temperatures below 0^o C;*
[c] $K_{NH} = 0.01854 + 0.3922\ (rz)^{2.6077}$ **(3.12)**
Where K_{NH} = SSF nitrification rate constant (day^{-1}), and rz = depth of bed occupied by root zone (percent, expressed as decimal fraction).

For TSS removal:

In SSF wetlands:

$$C_e = C_i(0.1058 + 0.0011HLR) \quad (3.13)$$

In FWS wetlands:

$$C_e = C_e(0.1139 + 0.00213HLR) \quad (3.14)$$

For Pathogen Removal:

Reed argues that the mechanisms for pathogen removal are essentially the same in both waste stabilization ponds and constructed wetlands. Where pathogen removal has been investigated in constructed wetlands, Equation 10 has proven to be conservative; hence, it is useful as a predictor:

$$C_e = \frac{C_i}{(1 + tK_T)^n} \quad (3.15)$$

Where C_e = effluent faecal coliform concentration (number/100 mL), C_o = influent faecal coliform concentration (number/100 mL), K_T = temperature dependent rate constant (day^{-1}), n = number of cells in series, and t = hydraulic residence time (days).

The reliance of this formula on the number of cells in series tends to suggest that, for optimal pathogen removal, the number of cells should be maximised.

For TP removal:

In both SSF and FWS wetlands:

$$C_e = C_i.e^{\left(\frac{-K_p}{HLR}\right)} \quad (3.16)$$

Where Kp is the first order phosphorous reaction rate (= 2.73 cm/day).

Note that the cross-sectional area of the flow is then calculated as:

$$A_c = Q/k_s S \quad (3.17)$$

Where A_c is the cross-sectional area of wetland bed (d*W) perpendicular to the direction of flow (m^2), d is the depth (m), k_s is the hydraulic conductivity of the medium (m^3/m^2.day), and S is the slope of the bed or hydraulic gradient (as a fraction or decimal). The bed width is then calculated as follows:

$$W = A_c / d \tag{3.18}$$

The cross sectional area and bed width are determined on the basis of Darcy's law, as follows:

$$Q = k_s A_s S \tag{3.19}$$

The bed cross-sectional area and bed width is independent of temperature and organic loading, since they are controlled by the hydraulic characteristics of the media.

3.8.2 Kadlec and Knight design method

Kadlec and Knight (1996) presume a first-order decay, plug flow model for all pollutants, including BOD, TSS, total phosphorous (TP), total nitrogen (TN), organic nitrogen (OrgN), ammonia nitrogen (NH4-N), oxidised nitrogen (NOx-N), and faecal coliform (FC). Their model is based on areal rate constants, which are independent of temperature. The Kadlec and Knight model may be less sensitive to different climatic conditions:

$$\ln\left(\frac{C_e - C^*}{C_i - C^*}\right) = \frac{-k}{q} \tag{3.20}$$

$$q = \frac{365Q}{A_s} \tag{3.21}$$

$$A_s = \frac{365Q}{k} \ln\left(\frac{C_i - C^*}{C_e - C^*}\right) \tag{3.22}$$

Alternatively:

$$C_e = C^* + (C_i - C^*)\exp\left(\frac{-A_s k}{365Q}\right) \quad (3.23)$$

Where A_s = treatment area of the wetland (m^2), C_e = target effluent concentration (mg/l), C_i = target influent concentration (mg/l), C* = background pollutant concentration (mg/l), k = first order aerial rate constant (m/yr), q = hydraulic loading rate (m/yr), and Q = average flow rate through the wetland (m^3/day).

Kadlec and Knight (1996) advocate the use of the "global" parameters they determined from plug flow analysis of performance data available to date on the North American Data Base (NADB) (Knight *et al*., 1993) in other systems. They suggest that specific parameters should be locally determined prior to investment in a full-scale system, in order to ensure suitability of design.
The global parameters to be used for this design are shown in Table 3.5. Where values for q other than 1.00 are given, it indicates that the parameter value varies with temperature.

Systems layout

Components of the CW include preliminary/primary treatment units and the CWs cell (or cells). The CW includes substrate, vegetation and biological organisms contained within the physical configuration that can be described as an attached growth biological filter.

Configuration

Configuration should enhance wastewater distribution to maximize contact between wastewater, substrate and vegetation by minimizing short-circuiting. The configuration should consider the degree of pre-treatment, required treatment area, available land shape and slope, length-to-width ratio, desired bed slope, amount of excavation and grading to obtain cell depth and slope, substrate type, collection pipes, and operation and maintenance flexibility. There are three possible configurations for wetland cells used for wastewater treatment: In parallel, in series, or a combination of the two. The main advantage for cells in parallel is the flexibility and redundancy in operation, so that individual cells can easily be taken off-line for maintenance and repair. The water flow can be re-directed to other cells, allowing for ongoing operation of the wastewater treatment

Table 3.5 Preliminary model parameter values for Kadlec and Knight's design equations (Kadlec and Knight, 1996)

Parameter	BOD	TSS	OrgN	NH_4-N	No_x-N	TN	TP[a]	FC
For free water surface wetlands:								
k_{20} (m/yr)	34	1000[b]	17	18	35	22	12	75
Q	1.00	1.00	1.05	1.04	1.09	1.05	1.00	1.00
C* (mg/l)	$3.5+0.053C_i$	$5.1+0.16C_i$	1.50	0.00	0.00	1.50	0.02	300[c]
q	1.00	1.065						
For sub-surface flow wetlands:								
k_{20} (m/yr)	180	1000b	35	34	50	27	12	95
q	1.00	1.00	1.05	1.04	1.09	1.05	1.00	1.00
C* (mg/l)	$3.5+0.053C_i$	$7.8+0.063C_i$	1.50	0.00	0.00	1.50	0.02	10[c]
q	1.00	1.065						

[a] *for non forested wetlands;*
[b] *rough unsubstantiated estimate, settling rate determination preferred (Kadlec and Knight, 1996);*
[c] *central tendency of widely variable values.*

plant and the wetland. When the wetland cells operate in series, the water moves sequentially from one cell to next, forming a chain of wetland cells. The main operational advantage of cells in series is minimization of short-circuiting, leading to better overall treatment in the system. From the reactor theory, an ideal plug provides the best pollutant removal. The selected configuration must be based on a clear understanding of the objectives, influent water quality, desired effluent water quality, hydraulic regime and site specific constraints and opportunities (Kadlec and Knight, 1996). An L/W value as low as 1 is recommended for SSF (Hammer, 1990), while an L/W ratio of at least 10 is required for the SF. The bed slope for SSF should be 2% or less, while that for SF should be 0.5% or less. The substrate depth should be 0.6 m.

Preliminary/primary treatment

CWs should be preceded with screens, or comminution as preliminary treatment facilities. Imhoff tanks, septic tanks and stabilization pond(s) should be used as primary treatment facilities. Small communities with water flow rates less than 380 m^3/day may use septic tanks as primary treatment facility to reduce organic solids.

Part Three

Chapter Four: Operation and maintenance of waste stabilization ponds and constructed wetland

4.1 Operation and Maintenance of Waste Stabilization Ponds

The operation of the ponds includes the commissioning (starting up), maintaining and monitoring of its performance.

4.1.1 Start-up phase for waste stabilization ponds

Anaerobic ponds must be filled with raw sewage and seeded with sludge from conventional sewage treatment plants or septic tanks. After filling and seeding, the pond should gradually be loaded up to the design-loading rate. The pond contents should have a pH above 7, to allow the development of methanogenic bacteria. Lime or soda ash is added, if necessary, to rise the pH in the pond. If the sewerage system is new, and the flow rate, as well as the loading rate, to the anaerobic pond is low, the sewage may be bypassed till the flow rate and the loading rate from the sewerage systems satisfies the condition to be discharged in the pond. It is important to have a bypass from the anaerobic pond that will be used during de-sludging. It is also recommended that the ponds be commissioned during the beginning of the hot season, in order to allow the quick establishment of microorganisms of importance for the waste stabilization ponds.

The facultative pond should be commissioned before the anaerobic pond, in order to avoid odors during the release of anaerobic pond effluent to an empty facultative pond. During the start-up phase of the facultative and maturation ponds, the ponds should be filled with freshwater (from tap, river or wells), thereby allowing the gradual development of algae and heterotrophic bacteria population. However, the primary facultative pond must be seeded as an anaerobic pond. If freshwater is not available, the secondary facultative and maturation ponds should be filled with raw sewage and left for 3-4 weeks, to allow the development of the microbial population.

4.1.2 Maintenance

Maintenance of the ponds should be carried out regularly to avoid odors, flies and mosquito nuisances. The routine maintenance includes:

- Removing screenings and grit from the inlet and outlet works;
- Cutting grasses on the embankment, and removing it so that it does not fall in the ponds;
- Removing floating scum and floating macrophytes from the surface of the maturation and facultative ponds;
- Spraying scum on the surface of the anaerobic ponds and not removing it, since this will help the treatment processes;
- Removing any accumulated solids in the inlet and outlet works;
- Repairing any damaged embankment as soon as possible; and
- Repairing any damage of the fences or gates.

The operator must be given precise information on what must be done at the pond site. A clear form should be prepared, clearly showing the tasks to be performed by the operator, and which should be counter-checked weekly by the foreman/supervisor. Mara (1987) recommends that, for proper pond operation, there should be a manager, assistant manager, engineers, work foreman, laboratory chemist, assistant laboratory chemist, technicians, artisans and clerks. The number of staff members will depend on how extensive the project is, as well as the population to be served.

Anaerobic ponds require desludging when they are one third full of the sludge by volume.

$$n = V/3Ps$$

Where V is the volume of the anaerobic pond (m^3), P is the population served, and s is the sludge accumulation rate (m^3/person year).

The value of "s" is usually 0.04 m^3/person.year. The precise time to de-sludge the anaerobic pond is sometimes determined by the use of a "white towel." Mara (1987) recommends an annual de-sludging frequency. The sludge from the pond may be disposed of in sludge lagoons or

tankers that transport it to a landfill site, agricultural land or other suitable disposal area. The sludge disposal should be done in accordance with local regulations.

4.1.3 Monitoring and evaluation of performance

Frequent monitoring of the final effluent quality of a pond system is required to address the following needs:

- Regular assessment regarding whether or not the effluent complies with the local discharge or reuse standards; and
- Detection of any sudden failure, or determining if the pond effluent has started to deteriorate; it also may help identify the causes of the problem(s) and the remedial actions to be taken.

Two levels of monitoring are recommended. Level 1 involves taking samples of the final effluent at least monthly or weekly, for analysis of the parameters for which effluent discharge or reuse requirements exist. Level 2 is implemented when the Level 1 monitoring indicates that the effluent is failing to meet necessary standards, thereby requiring a more detailed study. Twenty-four hour flow-weighted composite samples are preferable for most of the parameters to be analyzed, although grab samples are necessary for some of them (pH, temperature, faecal coliform). The methods for collecting composite samples are as follows:

- Automatic sampler -- which takes samples every 1-2 hours, with subsequent manual flow weighting if this is not done automatically;
- Grab samples -- every 1-3 hours, with subsequent manual flow weighting; and
- Column samples – near the final pond outlet.

A full evaluation of the performance of the pond is a time-consuming, expensive process, requiring experienced personnel to interpret the results. Details on how to collect samples for evaluating the performance of the pond is given by Mara (1987). Evaluation of the pond performance has the following significance:

- Provides information on how the pond is either under-loaded or overloaded;
- Indicates how much loading may safely be added to the system as the community increases; and
- Indicates how the future design can be improved in the region.

4.2 Operation and maintenance of constructed wetlands

4.2.1 Commissioning

Sometimes commissioning of the wetland is referred as the time from planting to the date where the wetland is considered operational. Operation during this period should ensure an adequate cover of the wetland vegetation. The water level within the wetland during this time needs to be controlled carefully, to prevent seedling from being desiccated or drowned. Once the plants are established, the water level may be raised to operational level. Plant loss may occur during the commissioning, therefore requiring transplanting.

4.2.2 Operation

The operation of a constructed wetland depends on the type of wetland, and the number of preliminary treatment units used for wastewater treatment. Constructed wetlands are designed to be passive and low maintenance, thereby not requiring continual upkeep. Constructed wetland, however, are dynamic ecosystems, with many variables that require managing. If not, problems may occur when the operator does not understand the needed operation and maintenance, the wetland is either hydraulically or organically overloaded, unavoidable disasters (e.g., flooding, drought) occur, the wetland is plagued by weed problems and/or if excessive quantities of sediments, litter and pollutants accumulate and are not removed from the wetland. The management of the constructed wetlands consists of four tasks, as outlined below in Table 4.1

Not all constructed wetlands are the same, given that they can be designed for a range of objectives. These objectives will determine the kind of operation and management activities needed to be undertaken. Thus, the operation and management of a constructed wetland must be tailored to a particular constructed wetland, reflecting desired objectives and site-specific constraints, including local hydrology, salinity, climate and relevant aspects of public safety. The essential elements of the operation and maintenance of a constructed wetland must include:

- Description of the wetland and its objectives;
- List of tasks or management; and
- Monitoring activities, including inspection checklists.

Table 4.1 Management of constructed wetlands

Tasks	Example
Operational control	Varying water level
Monitoring	Water quality, habitat, flora and fauna
Inspection	Structures and embankments
Maintenance	Repair damage to the structures and control weeds

The operation of a constructed wetland after commissioning must include:

- Maintaining the embankments;
- Removing litter and debris;
- Checking the water flow rate to a constructed wetland to determine if it is in accordance with the design;
- Removing any blockages in the inlet and outlet works;
- Replacing plants as required;
- Removing any unwanted weed species from the constructed wetland;
- Checking the plants for any sign of diseases;
- Protecting the deep open water;
- Correcting erosion and slumping; and
- Checking for any signs of over-flooding (for sub-surface flow constructed wetlands).

These tasks may be addressed in the form of a checklist to direct the required maintenance, and to identify who should be immediately contacted in the event of problems.

4.2.1 Monitoring

Monitoring selected performance parameters should provide sufficient information to measure performance in meeting wetland objectives. If water quality improvement is the primary objective, the performance indicator should either be presented as a concentration or load at the outlet, or a comparison of inflow and outflows, also in terms of concentrations or loads. If monitoring results indicate that the system is not working according to the objectives, corrective measures must be applied. Improvement of water quality may be assessed by monitoring a range of inflow and outflow water-quality parameters. Useful parameters for monitoring wetland performance include dissolved oxygen (DO), BOD, COD, total phosphorus, orthophosphorus,

total nitrogen, total Kjeldhal nitrogen, ammonia nitrogen, oxidized nitrogen, faecal coliform, pH, suspended solids, electrical conductivity, and heavy metal concentrations. Water flow rates to and from the constructed wetland also must be measured. The sampling may be done using either an automatic or manual sampler. Samples within the wetland must sometimes be taken for the purpose of comparison.

4.2.3 Decommissioning and refitting

Decommissioning and refitting of a constructed wetland may take place if its design lifetime is over. At the end of its design life, a wetland will be either be refitted, or decommissioned if no longer required. Refitting may be required when the accumulation of wetland sediments is adversely affecting wetland performance, or when changing catchment conditions require modifications of the wetland. Major refits may include the removal of accumulated peat, including aquatic plants, and replacements of substrates. Decommissioning of a wetland may be required if the land supporting it is utilized for other purposes, or if the wetland functioning is unable to achieve the original design objectives.

Part Four

Chapter Five: Modelling of Waste Stabilization Ponds and Constructed Wetlands

5.0 Biogeochemical Dynamic Models for Waste Stabilization Ponds and Constructed Wetlands

5.1 Background

A model is an idealization of a real situation, in which the most important components are identified and their interactions described and used as a tool to solve problems (Jorgensen, 1986; Daigger, 1982). An ecological model is regarded as a mathematical presentation of the biochemical activities relevant to the ecology of a secondary facultative pond. Energy flows in ecosystems are uni-directional and non-cyclic, whereas nutrient movement is cyclic. An ecosystem is a functional unit, in that its components are connected by flows of energy and matter. The type of ecological model to be formulated is determined by the nature of the problem to be solved, the ecosystem being addressed, and the data available (Jorgensen, 1992). In its mathematical formulation, an ecological model consists of 5 components, including: (i) Forcing functions or external variables; (ii) State variables; (iii) Mathematical equations; (iv) Parameter coefficients; and (v) Universal constants. The level of organisation of the model depends on the nature of the problem to be solved. There are many classes of ecological models, including stochastic, deterministic, reductionist, holistic, steady-state, static, distributed, and lumped-parameter models. The level of organisation of the ecosystem, and the external factors, help decide what type of model is suitable for describing a specific ecological system. A combination of approaches, such as empirical studies, comparative studies, and experimental modelling, should be used to provide a whole picture of an ecosystem. Ecosystems are typically so complex that it is sometimes too difficult to try to represent all the system's features accurately using only one approach.

Understanding the processes and reactions of an ecosystem is a basis for evaluating the ability of nature to self-design its forcing functions, thereby providing a guide for choosing the appropriate biotechnological method. It is not only important to know how humans can influence the ecosystem processes, and how the ecosystem components are linked, but also how changes in one ecosystem

can produce changes in a neighboring ecosystem. An example in wastewater treatment is that it often is necessary to use primary treatment units, followed by secondary or biological treatment processes. Any disturbances that may occur in the primary treatment units also will affect the secondary biological treatment processes. Waste stabilization ponds and constructed wetlands are ecosystems with defined physical, chemical and biological processes. They are considered an ecosystem because of the interactions between their organisms and their surrounding environment.

5.1.1 Elements of ecological modelling

During any formulation of physical, biological of chemical processes of an ecosystem, model, it is necessary to know the elements of each ecosystem, in order to have a clear understanding of what should be included in the model.

5.1.2 State variables

State variables refer to the variables that describe the state of an ecosystem, being those variables that constitute its main properties. It is the change in state variables that characterize the dynamic system. There are four ways in which a state variable can change over time (Hardisty *et al.,* 1994): *Damped* -- the state variables stay at or fluctuate around a fixed level; *Explosive condition* -- the state variables move progressively away from an initial value; *Periodic condition* – the state variables move between extremes through time; *unsystematic condition* -- a state variable has no perceptible pattern of changes over time (i.e., the change is chaotic).

5.1.3 Forcing functions

In this case, the forcing functions refer to variables of an external nature that influence the state variables of an ecosystem. The ecological model is used to predict what will change in an ecosystem when the forcing functions are varied with time. The forcing functions to the system that may be influenced by humans are the control functions, such as the sewage flow rate to the pond, which may be controlled to a maximum or minimum extend. In contrast, the sun's light intensity cannot be controlled since it is a natural factor not influenced by human beings. The relationship between the forcing functions and state variables forms the core of an ecosystem model (Mitsch and Jorgensen, 1989).

5.1.4 Mathematical equations

Mathematical modelling provides a quantitative description of the ecosystem that must be modeled. With the use of mathematical models, the rates of the processes occurring in the ecosystem, and their stoichiometric interactions with the compounds, are formulated mathematically. The mathematical equations are used to represent the biological, chemical and physical processes occurring in the system of interest. Mathematical formulations should be incorporated in a solution procedure that considers the physical constraints and characteristics imposed by the system in which the processes are taking place (e.g., temperature, mixing). The relation between the state variables and forcing functions can readily be depicted by using mathematical equations.

5.1.5 Parameters

These are coefficients that can be found in mathematical representations of the ecosystem processes. These are considered constants for a specific ecosystem, or part of an ecosystem.

5.2 Modelling Procedures

Ecological modelling, like any other modelling procedure, must undergo several critical steps prior to being accepted, as described in the following section.

5.2.1 Definition of the problem

The problem to be addressed and solved should be clearly defined. For the goals of this report, the task is to formulate a model that will include the interactions of algae and bacteria in the overall removal of organic carbon in secondary facultative ponds. All the selected forcing functions are used to evaluate the extent to which they influence the state variables, and the organic carbon removal process.

5.2.2 Model selection

This is a stage at which the conceptualization of a model, in the form of a diagram, is undertaken. It provides information on which state variables, forcing functions and process equations are required in the model. There are two steps involved. A model may be theoretically developed that specifies the required variables and parameters, along with the associated equations.

The second step is called numerical specification and validation, which requires the theoretically-developed model to be run in a computer. At this stage, one may include, or reduce the bulkiness of the model, using scientific arguments.

5.2.3 Preliminary simulations

During this step, preliminary simulations of the system being modelled are performed. The procedure is useful for identifying data deficiencies, theoretical gaps and the most important parameters.

5.2.4 Mathematical equations

The process of the oxidation of organic matter, the growth of biomass, and the influence of the forcing functions on the state variables, are linked together with the use of equations. Mathematical models describe the relationship between the model components, using specific mathematical expressions. The main concern in this step usually is the selection of the equation that correctly describes the processes. Figure 5.1 shows the necessary steps used for ecological modelling.

The mathematical models proposed to describe the growth of microorganisms fall into two categories; namely structured and unstructured. The structured approach considers some aspects of the internal structure, and variations of the internal composition of microorganisms, and is important for the treatment of transient growth. The complexity associated with the structured models (i.e., large number of parameters) has limited their use to a few well-defined growth conditions. The unstructured models are based on a simplification of the complexity of living organisms, through the assumption that the amount of biomass present in the culture provides sufficient information about microbial activities (Tan *et al*., 1994). The Monod (1949) equation is the most widely-used unstructured mathematical model for a single substrate-limited microbial growth.

5.2.5 Model validation

A model must be validated before it can be used with confidence for its intended purpose. This is done by running the calibrated model with a new set of data, with physical parameters and forcing functions to reflect new conditions. The model is revised each time, until

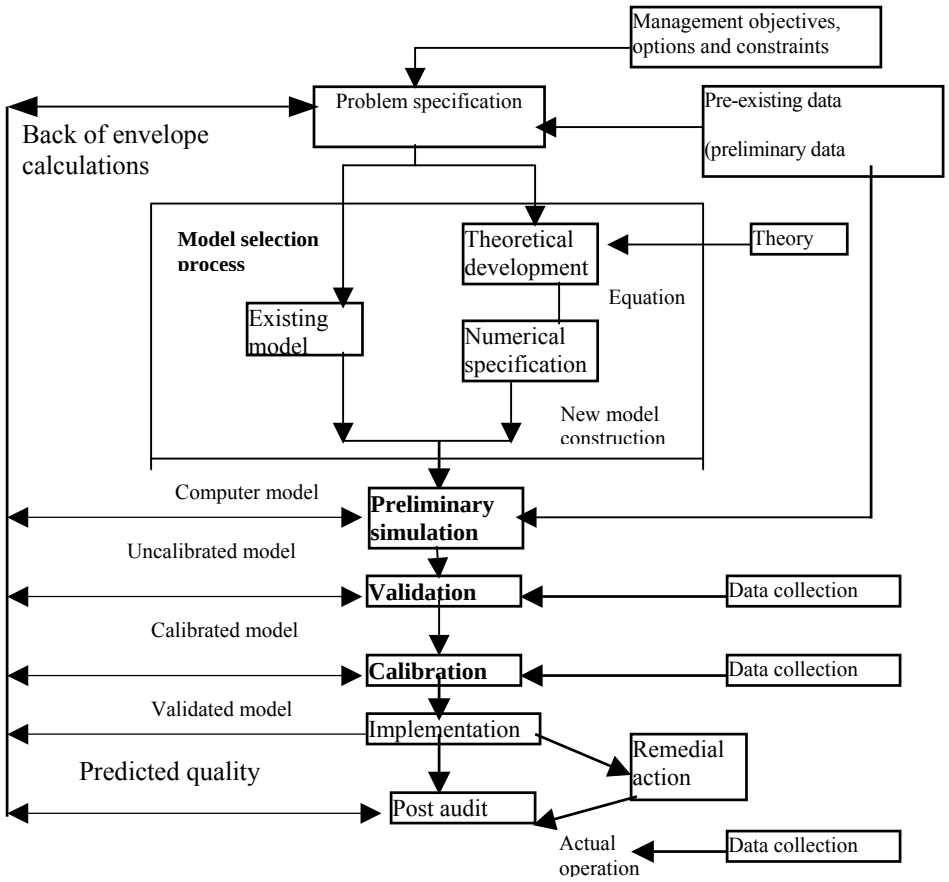


Figure 5.1 A schematic diagram of the ecological modelling process, highlighting the needed information

it is verifiably and consistently accurate. In contrast, the kinetic coefficients must now be kept fixed at the values derived during the original calibration exercise. When the new model is validated, it becomes an effective predictive tool for the range of conditions defined by the original calibration and validation data set. If there is no match between predicted and real data, the model often can be analysed to determine the possible reasons for the discrepancy.

5.3 Advantages of Modelling

The process of modelling reduces the real world to a smaller world that may be more easily analyzed. If a model is made systematically, one may go back to the first step of model formulation, in order to increase the number of state variables or forcing functions. It is possible in a modelling

exercise to refine the conceptual model to fit the requirements of the formulated model. During the model calibration phase, it is possible to reduce the number of forcing functions and state variables, provided this gives a more accurate representation of the system being modeled. It is also possible to view the behavior of the model outcome, making it possible to change the values of coefficients in order to approximate the real outcome. Once a model is formulated, it can give a clear indication of the type of data to be collected from field studies for model calibration and validation. If one starts with data collection, and finishes with the model formulation, there is no doubt that many data will not be useful for the model, often making it necessary to collect even more field data, thereby also making the first collected data set obsolete. Thus, a systematic understanding of the modelling procedure is a fundamental requirement.

5.4 Models for Waste Stabilization Ponds and Constructed Wetlands

The models for waste stabilization ponds and constructed wetlands are presented in Annexes 3 and 4. The model of organic carbon transformation in waste stabilization ponds is based on the growth processes of heterotrophic bacteria and algae in the pond system. The forcing functions (temperature, light, etc.) are well used to predict the dominant processes in the transformation of organic matter. The model on the transformation of nitrogen also was formulated, calibrated and validated.

The coupled model of organic carbon in the primary facultative ponds and sub-surface wetlands also is presented in Annex 4, followed by a model of nitrogen transformation in the horizontal sub-surface flow constructed wetland. The conceptual model and processes equations also are presented.