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A System Dynamics Approach to Modelling the Degradation of Biochemical Oxygen Demand in a Constructed Wetland Receiving Stormwater Runoff

Leslie A. Mudgett

A SYSTEM DYNAMICS APPROACH TO MODELLING THE DEGRADATION OF BIOCHEMICAL OXYGEN DEMAND IN A CONSTRUCTED WETLAND RECEIVING STORMWATER RUNOFF

THESIS

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Presented to the faculty of the School of Engineering

of the Air Force Institute of Technology

In Partial Fullfillment of the

Requirements for the Degree of

Master of Science in Engineering and Environmental Management

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Captain, USAF

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Leslie A. Mudgett

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Abstract

The objective of this research was to develop a tool to aid the Air Force Environmental Manager in the identification of the design parameters of a constructed wetland system that may be optimized to provide a desired biochemical oxygen demand (BOD) removal efficiency during the treatment of Air Force stormwater runoff. The objective is achieved through the development and use of a system dynamics model which simulates the hydrological functions of a constructed wetland as well as the processes within the wetland responsible for degradation of BOD. Based on literature review, the primary mechanism responsible for the degradation of BOD within a constructed wetland system is degradation due to microbial populations in the form of both suspended biomass and biofilm found on the surface of vegetation and the wetland floor. The model was run for constructed wetlands of various surface areas, each subjected to a range of stormwater influent rates and influent concentrations. The hydraulic retention times, organic loading rates and BOD removal efficiencies were determined for each case. Scatter plots of both hydraulic retention times and organic loading rates vs. removal efficiency indicated a clear relationship between both hydraulic retention time and removal efficiency as well as organic loading rate and removal efficiency. Several runs of the model also indicated that larger surface areas, greater length to width ratios and greater depths contributed to lower BOD concentrations in the water column.

A SYSTEM DYNAMICS APPROACH TO MODELLING THE DEGRADATION OF BIOCHEMICAL OXYGEN DEMAND IN A CONSTRUCTED WELTAND RECEIVING STORMWATER RUNOFF

I. Introduction

General Issue

The Environmental Protection Agency (EPA) declared its goal to "restore and maintain the chemical, physical and biological integrity of the Nation's waters" (Clean Water Act Title I, sec 101) in 1977 with the passage of the Federal Water Pollution Control Act (later amended and renamed the Clean Water Act (CWA)). The initial focus of regulation resulting from the CWA and subsequent amendments has historically been on the discharges of industrial process waters and publicly owned treatment works (POTWs). As these problems gradually came under control, concern shifted to the quality of stormwater discharges.

The EPA conducted several studies to determine the content of stormwater originating from residential, commercial and light industrial areas. The results of these studies showed that stormwater contains many of the contaminants that discharges from industrial processes and POTW's contain, and often in high quantities (Feeney, 1992:1). This prompted the promulgation of stormwater regulation, albeit limited. The regulation of stormwater is dynamic. As the most serious sources are grappled with and brought under control, regulation will likely extend to all sources. Those currently subject to recently promulgated stormwater regulation include "public and private facilities that discharge stormwater via one or more point sources or into the waters of the United States, either directly or through a separate storm sewer system" (Feeney, 1992: tab 100, 3). The discharging facility must also meet one of the five categories of stormwater dischargers identified by the National Storm Water Program, one of which includes facilities that engage in industrial activity. The Air Force counts itself a member of the regulated community largely due to the significant amount of industrial activity occurring on Air Force installations.

Current regulation requires stormwater dischargers to evaluate potential Best Management Practices (BMP) to control stormwater discharges and implement them where appropriate. Capt. Pete Ridilla and Lt. Brad Hoagland addressed several BMP's including the use of constructed wetlands, in their thesis entitled "Analysis of Best Management Practices for Storm Water Compliance at Air Force Airfields." Their thesis included a decision support framework to aid the Environmental Manager in determining the most appropriate BMP for that base. Ridilla and Hoagland considered the following factors: cost, manpower and maintenance requirements, non-point source pollution removal effect and suitable site conditions. In comparison with other proposed BMP's, constructed wetlands fared well. Capt. Mark Smekrud subsequently studied the use of constructed wetlands as a stormwater run-off BMP in his thesis entitled "A Preliminary System Dynamics Model of a Constructed Wetland for the Mitigation of Metals in USAF Stormwater." He concluded that "properly sized CW systems can offer an effective technology approach to controlling metal concentrations in AF stormwater." (Smekrud, 1994:87). The results of both theses indicate the viability of a constructed wetland as a BMP for the treatment of stormwater run-off. However, due to the variables associated with stormwater run-off, such as the quantity and quality of runoff associated with separate storm events, further research is certainly warranted.

Although limited information is available regarding the use of constructed wetlands for the treatment of stormwater runoff, extensive documentation regarding the use of constructed wetlands for the treatment of wastewater is available. In fact, many constructed wetlands exist for that purpose. "At least 300 constructed wetlands in North America and over 500 wetlands in Great Britain and Europe currently are used to treat municipal, industrial, and agricultural wastewater." (Knight, 1994:30). Many small communities are incorporating constructed wetland systems into their waste water treatment systems to aid them in meeting water quality standards. As the cost of constructing and operating conventional treatment systems increases, the constructed wetland becomes an appealing alternative. Among some of Hammer's reasons for offering constructed wetlands as a "promising alternative to conventional treatment plants" are: "Constructed wetland systems (1) are relatively inexpensive to construct and operate; (2) are easy to maintain; (3) provide effective and reliable wastewater treatment..." (Hammer, 1990:16). Constructed wetlands have proven to be effective in removing pollutants such as biochemical oxygen demand (BOD), suspended solids (SS), nitrogen and others from municipal wastewater.

The functions of natural wetlands as nutrient sinks and buffering zones lend themselves to the application of wastewater treatment (US EPA, 1988:1). "Wetlands can effectively remove or convert large quantities of pollutants from point sources and nonpoint sources" through processes such as natural filtration, sedimentation, biological decomposition, and absorption to name a few (Hammer, 1990:12). In the interest of conserving the natural wetland as a valuable ecological resource, we turn to the constructed wetland to perform the same function of water quality improvement. We have seen the effectiveness of constructed wetlands in the treatment of municipal and industrial wastewater, but again, research regarding the use of constructed wetlands for the treatment of stormwater is limited.

Several variables associated with the characteristics of stormwater make it difficult to predict the efficiency of a wetland stormwater system. Unlike municipal wastewater systems where the influent to the constructed wetland is often well characterized and consistent, the concentrations of pollutants in stormwater can vary a great deal. Urquhart researched AF stormwater characteristics and documented results in her thesis titled "Status of Stormwater Pollution in the United States" and found that "pollutants range from the conventional, such as BOD and pH, to a wide variety of unconventional pollutants, including pesticides, volatiles, metals, organic and inorganic compounds." (Urquhart, 1994:42). Types and concentrations of pollutants will not only vary by facility, but will vary at the same facility in a short period of time. For example, the concentration of pollutants during a rain storm will be greatest during the early stages of the storm, and will diminish over the course of the storm. The flowrate of stormwater runoff during rain storms is also irregular and maximum flow rates may exceed the capacity of the wetland, whereas municipal wastewater flow rates can be controlled. The flow rate of influent into the wetland affects detention time and in turn the ability of the wetland to effectively remove pollutants. Variables such as those mentioned prompt some skeptics such as Livingston to state that "Due to variations in stormwater characteristics and poor understanding of wetland processes that remove pollutants, treatment efficiency predictions of a wetland stormwater system are not possible" (Hammer, 1994:255). However, better understanding of wetland processes and characterization of stormwater pollutants can provide information essential to the design of an effective constructed wetland.

The limited studies available regarding the pollutant removal effectiveness of constructed wetlands have revealed several parameters and variables influencing the efficiencies of the constructed wetland. Some of these parameters are variations in local hydrology, detention times, rates of runoff, water level fluctuations, and seasonality. The design parameters of a constructed wetland to include surface area, volume, depth of water, length to width ratio, inlet and outlet structure and vegetation also determine a wetlands effectiveness in removal of specific pollutants. These design parameters must be optimized in order to achieve the desired effect. Constructed wetland design for the purpose of wastewater treatment has historically been based on the desired removal of pollutants such as BOD and SS. An understanding of the fate of such pollutants as they are transported through a constructed wetland system can aid in the determination of critical parameters and the development of an effective constructed wetland design.

Problem Statement

Stormwater dischargers are required by Federal Regulation to evaluate BMP's and implement them where appropriate. Previous research by the Air Force and others has identified several BMP's for the management and treatment of stormwater runoff. One such proposed BMP is the use of constructed wetlands. Limited data on the effectiveness of constructed wetlands in the removal of pollutants warrants further research. Research has been conducted focusing on "controlling and mitigating the effects of trace metals in stormwater runoff from a typical AF installation....to assist Base Civil Engineers and Environmental Managers in deciding whether a constructed wetland is applicable to their specific situation" (Smekrud, 1994:6). This information should be supplemented with further research regarding the effectiveness of constructed wetland removal of other pollutants. A tool is needed to aid in the determination of those parameters that affect constructed wetland effectiveness and pollutant removal efficiency.

Research Objectives

The purpose of this research is to develop a tool to aid in the identification of the design parameters of a constructed wetland system that may be optimized to provide a desired BOD removal efficiency during the treatment of Air Force stormwater runoff . This objective will be achieved through the development of a system dynamics model which will attempt to simulate the processes within a constructed wetland system responsible for the degradation of BOD, a significant pollutant found in AF stormwater. This model will provide the AF with a tool to determine those critical parameters and their optimal values that will achieve the desired BOD removal efficiency.

Investigative Questions

The following investigative questions will be used to guide the research and aid in accomplishing the research objective:

(1) What are the sources and characteristics of BOD in AF stormwater? What is the typical BOD content in AF stormwater?

(2) What processes within a CW occur that affect the degradation of BOD?

(3) What are the critical parameters associated with a CW that affect the degradation of BOD?

Scope

This study reviews recently promulgated regulations which impose the requirement for stormwater management on the AF. It describes the characteristics of AF stormwater with emphasis on the source and expected concentrations of BOD. The functions of natural and constructed wetlands are described as well as current applications in wastewater treatment. The processes within a constructed wetland responsible for the transport or transformation of BOD are determined and used as the basis of a system dynamics model which models the accumulation of BOD over time in a constructed wetland system. Several runs of the model are performed to aid in the determination of optimal values of those design parameters which effect the BOD removal efficiency of the wetland. This model considers only the removal of BOD in optimizing constructed wetland design parameters. Other significant pollutants are not considered. The model is verified during the course of development, however, validation using field data is not accomplished at this time.

II. Literature Review

Introduction

This chapter begins with a description of stormwater runoff, its sources, characteristics and resulting effects on the surrounding environment. A brief look at the history of stormwater legislation is made as well as reference to AF requirements regarding management of stormwater runoff. The pollutant of interest in this research is biochemical oxygen demand (BOD), therefore a detailed discussion of BOD and its degradation characteristics is provided. Several best management practices are discussed before focusing on the wetland alternative. Wetlands are discussed in detail, to include a description of wetland purifying functions and those inherent characteristics of wetlands responsible for those functions. Two types of constructed wetlands for the purpose of wastewater treatment are described as well as examples of existing wetland systems. Finally, the importance of the degradation capabilities of microbial populations within a wetland system and the factors that affect the population are discussed in detail.

Stormwater Runoff

Stormwater, as defined by the Code of Federal Regulations (CFR), consists of "stormwater run off, snow melt run off and surface run off and drainage." (40CFR122.26(b)(13)). This includes all major flows that result from precipitation events. In undeveloped areas, stormwater run off is handled naturally by the hydrologic cycle. Urbanization and its resulting increase in paved and impervious surfaces affects the flows and pollutant load in stormwater. Stormwater flowing over surfaces such as roofs, roads, parking lots, industrial areas, lawns and agricultural areas carries pesticides, oil and grease, heavy metals and other wastes to receiving waters. Adverse impacts on receiving waters include increased oxygen demand, turbidity, bacterial loading and toxicity.

In comparison with other forms of wastewater, "separate storm wastewaters are significant sources of pollution, typically characterized as having solid concentrations equal or greater than those of untreated sanitary wastewater and BOD concentrations approximately equal to those of secondary effluent" (Field, 1993:4). A study performed in Durham, North Carolina confirms the claims that stormwater can be nearly as much of a problem as sanitary wastes. They found that "when compared to the raw municipal waste generated within the study area the annual urban runoff of COD was equal to 91% of the raw sewage yield; the BOD yield was equal to 67%; and the SS yield was 20 times that contained in the raw municipal wastes" (Field, 1994:5). Research in Florida revealed that stormwater-associated pollution is responsible for:

1. 80 to 95 percent of the heavy metals loading to Florida surface waters;
2. Virtually all of the sediment deposit in State waters;
3. 450 times the suspended solids going to receiving waters and 9 times the load of BOD_5 substances when compared to loads from secondarily treated sewage effluent; and
4. Nutrient loads comparable to those in secondarily treated sewage effluent discharges (Livingston, undated:289).

A comparison of typical values for storm flow discharges with background levels, combined sewer overflow and sanitary sewage is given in Table 2.1.

Table 2.1

Comparison of Typical Values for Storm Flow Discharges[a]

	TSS	VSS	BOD	COD	KN	Total N	PO_4-P	OPO_4 P	Lead
Back-ground Levels	5-100	____	0.5-3	20	____	0.05-0.5[b]	0.01-0.2[c]	____	<0.1
Storm-water Runoff	415	90	20	115	1.4	3-10	0.6	0.4	0.35
Combined Sewer Overflow	370	140	115	375	3.8	9-10	1.9	1.0	0.37
Sanitary Sewage	200	150	375	500	40	40	10	7	____

TSS = Total Suspended Solids, VSS = Volatile Suspended Solids, BOD = Biochemical Oxygen Demand, COD = Chemical Oxygen Demand, KN = Total Kjeldahl Nitrogen, Total N = Total Nitrogen, PO_4 = Phosphate, P = Phosphorous, OPO_4 = Organic Phosphate

[a] All values mg/L, [b] NO_3 as N, [c] Total phosphorus as P (Field, 1993:6)

Concern for the effects of contaminated stormwater run off dates back to 1964 with the initiation of the Storm and Combined Sewer Pollution Control Research, Development, and Demonstration Program (SCSP). The mission of the SCSP was to develop methods for controlling pollution from urban stormwater discharges, combined sewer overflows (CSO), and excessive inflow and infiltration (Field, 1993:3). The SCSP examined three types of discharges to include CSO's, storm drainage from separate storm systems (sewered or unsewered) and another form of CSO, overflow from sanitary lines infiltrated with stormwater. The focus of this paper will be on storm drainage from separate storm systems.

Continued concern into the 1980's prompted the US EPA to establish the Nationwide Urban Runoff Program (NURP) for the purpose of "characterizing pollutant types, loads, and effects on receiving water quality" (Praner and Sprewell, 1992:20).

Data collected during 1981 and 1982 from 81 sites during more than 2300 separate storm events was summarized in a report published by the EPA in 1983. The NURP identified the following pollutants as those that characterize urban run off:

Total Suspended Solids (TSS)
Biochemical Oxygen Demand (BOD)
Chemical Oxygen Demand (COD)
Total Phosphorous (TP)
Soluble Phosphorous (SP)
Total Kjeldahl Nitrogen (TKN)
Nitrite and Nitrate ($NO_{2\&3}$)
Total Copper (Cu)
Total Lead (Pb)
Total Zinc (Zn)

Analysis of the NURP data indicated that stormwater pollutant characteristics could not be inferred from the land use category or geographic location, however, pollutant loads per unit area are much higher for commercial areas due to the higher degree of imperviousness of those areas (Praner and Sprewell, 1992:24). The NURP report also found that "no correlation was found between event mean concentrations (EMC's) and run-off volumes, indicating that EMC's and run-off volumes are, for the most part, independent of each other" (Stahre and Urbonas, 1990:278). The EMC refers to the flow weighted average concentration for each pollutant. The data from all sites surveyed was consolidated to provide an overall description of the general characteristics of urban run off as shown in Table 2.2.

Table 2.2

Median EMC for All NURP Sites by Land Use Category

CONSTITUTE	RESIDENTIAL		MIXED		COMMERCIAL	
(mg/L)	Median	CV	Median	CV	Median	CV
BOD	10.000	0.41	7.800	0.52	9.300	0.31
COD	73.000	0.55	65.000	0.58	57.000	0.39
TSS	101.000	0.96	67.000	1.10	69.000	0.85
Pb	0.144	0.75	0.114	1.40	0.104	0.68
Cu	0.033	0.99	0.027	1.30	0.029	0.81
Zn	0.135	0.84	0.154	0.78	0.226	1.10
TKN	1.900	0.73	1.290	0.50	1.180	0.43
$NO_{2\&3}$	0.736	0.83	0.558	0.67	0.572	0.48
TP	0.383	0.69	0.263	0.75	0.201	0.67
SP	0.143	0.46	0.056	0.75	0.080	0.71

(Stahre and Urbonas, 1990:279)

Some studies show that the most significant water quality effects are a result of first flush when the early stages of a storm flush the accumulated pollutants from urban areas. One example of this first flush effect is in Florida where first flush is considered to be the first 2.5 cm of stormwater which carries 90% of the pollution load. (Hammer, 1994:254). Stahre notes that other studies do not support this point of view (Stahre and Urbonas, 1990:280). Due to these conflicting findings, Starhe does not recommend that collection and treatment of first flush volume only, unless it can be shown that 20% of the runoff contains 80% of the pollutants.

Air Force Stormwater Characteristics. In an effort to characterize typical non point source pollutants generated on an Air Force Base, Praner and Sprewell determined that the majority of Air Force activities and land uses were comparable with those in the urban category. For this reason, they focused their research on the ten pollutants identified in the NURP study. They looked at two models that predict total non point source pollutant loading in stormwater runoff and compared the results with those from a sampling and analysis program they conducted at the Air Force Academy during three rainstorm events. The first model is a manual model called the Unit Quantity Model. This model requires a site specific runoff coefficient (Rv) value which is a function of topography, soil type, vegetative cover, and the degree of imperviousness. Information regarding the land area and annual rainfall is also required. The second model is a computer model called ProStorm which requires data regarding rainfall, topography, soil type, degree of imperviousness, vegetative cover and land area. Unfortunately, the BOD results from the sampling and analysis program were considered invalid due to laboratory error and insufficient resources. The thesis authors however contend that "the models presented provide an adequate means of characterizing non point source (NPS) pollution on an Air Force base" (Pràner and Sprewell, 1992:87). They suggest that either method "will provide the necessary information required to effectively implement Best Management Practices targeting those pollutants of concern" (Praner and Sprewell, 1992:88).

Urquhart collected data regarding concentrations of pollutants in stormwater effluent from twenty four Air Force bases. Urquhart consolidated the data and determined the percent data exceeding the bench mark values set by the EPA for those pollutants (Urquhart, 1994:45-46). Table 2.3 shows the results from limited data for the ten pollutants identified by the NURP study. Not every base is required to test for the ten pollutants identified under the NURP study. (Note that total phosphorus and soluble phosphorus are consolidated as total phosphorus here).

Table 2.3

Percent Data from AF Bases Exceeding EPA Benchmark Values

	TSS	BOD	COD	Total P	TKN	$NO_{2\&3}$	Cu	Pb	Zn
%	12	40	29	33	0	68	96	26	50

(Urquhart, 1994:45-46)

In 1992, in an effort to meet requirements of the EPA's Group Application for a National Pollutant Discharge Elimination System (NPDES) permit, eleven AF bases provided the following stormwater data as reported by Smekrud. Benchmark values are as reported by Urqhart.

Table 2.4

AF Stormwater Sample Data (1992)

Constituent (unit)	EPA Benchmark Values	Composite Range (Mean)	Grab Sample Range (Mean)
TSS (mg/L)	100	4-312 (71.07)	4-650 (181.49)
BOD_5 (mg/L)	9	2-42 (12.61)	1.0-18.45 (7.52)
COD (mg/L)	65	5-60 (24.84)	5-225 (48.25)
TKN (mg/L)	105	0.19-3.0 (1.19)	0.21-3.30 (1.23)
$NO_{2\&3}$ (mg/L)	0.68	0.12-15.0 (1.39)	0.11-2.46 (.55)
Total P (mg/L)	0.33	0.04-0.57 (0.25)	0.01-0.80 (0.29)
Oil and Grease (mg/L)		0.8-2.90 (1.83)	0.2-5.8 (1.83)
pH*	6.5-9	6.8-9.4 (8.0)	5.0-9.4 (7.03)
Metals (micrograms/L)			
Cu	0.009	<1.0-50.0	<5.0-15.6**
Pb	0.0337	<14-20	<2.0-52.0
Zn	0.065	<14-94	<20-348

*pH is not necessarily a pollutant but included as an important characteristic of the runoff

**Most copper samples were measured at <50 micrograms/L

(Smekrud, 1994:13; Urqhardt, 1994:26)

History of Stormwater Legislation and Regulation. In 1972, the Clean Water Act (CWA) required EPA to issue National Pollutant Discharge Elimination System (NPDES) permits for every point source discharge of pollutants to waters of the United States. In response to this requirement, less than 70,000 industrial wastewater facilities and Publicly Owned Treatment Works (POTW's) were permitted. If every stormwater point source were permitted, it is estimated that greater than 7 million additional facilities would require permits (Feeney, 1992: Tab 700, Appendix A, 501). The 1987 CWA amendments relaxed the requirements regarding those facilities subject to regulation and requiring NPDES permits.

Those subject to the stormwater regulations are any one of the following five categories of stormwater dischargers:

- facilities already covered by an NPDES permit for stormwater;
- facilities that engage in industrial activity;
- large (>250,000 population) municipal separate storm sewer systems; and
- medium (>100,000 and <250,000 population) municipal separate storm sewer systems; and
- facilities that the EPA administrator (or an NPDES state administrator) determines to have stormwater discharges contributing to a violation of water quality, or that are "significant contributors" of pollutants to waters of the United States.

The largest category consists of those engaged in industrial activity, which includes many federal facilities engaging in various industrial activities. Industrial facilities were required to apply for NPDES permits for stormwater discharges by 1 October 1992. (Feeney, 1992: Vol. 1, Tab 100, 3). The Water Quality Act of 1987 states that permits for discharges associated with industrial activity must meet best available technology/best control technology based requirements. The purpose of the

NPDES permits is to ensure that discharges do not violate the water quality standards in receiving bodies of water since stormwater discharges are subject to water quality based standards. In order to meet the requirements of the NPDES General Stormwater permit for industrial activities, Air Force bases must develop Storm Water Pollution Prevention Plans (SWPPP) and implement best management practices (BMP's) to eliminate or reduce pollutant discharges from stormwater runoff.

Biochemical Oxygen Demand. Biochemical oxygen demand (BOD) was identified by the NURP as one of the primary pollutants that characterize urban stormwater runoff, and is also a primary pollutant of concern in industrial and municipal wastewaters. Table 2.3 and 2.4 also call attention to the fact that BOD is a pollutant of concern for the Air Force. Table 2.3 notes that 40% of the data collected indicate exceedance of EPA benchmark values, and Table 2.4 shows a mean value of BOD concentrations exceeding EPA benchmark values.

BOD can be defined as the "measure of oxygen consumption required by the microbial oxidation of readily degradable organics and ammonia" (Atlas and Bartha, 1993:360). The measure of BOD is commonly used as a measure of the efficiency of wastewater treatment processes. It is also used to determine the concentration of organic matter in industrial and urban wastewater and to predict its subsequent oxygen demand on receiving bodies of water. Wastewater with a high BOD content discharged to natural waterways will exhaust the dissolved oxygen (DO) supply in the receiving water and slow its self-purification processes. The water becomes anaerobic, killing fish and other organisms that depend on oxygen to live. The decomposition of these dead organisms will in turn give rise to greater oxygen demand, further lowering the amount of dissolved oxygen available. The turbidity associated with a high BOD content can interfere with photosynthetic oxygen regeneration, continuing to worsen the problem. (Atlas and Bartha, 1993:361). Since the process of returning oxygen to the water by rearation or photosynthetic oxygen regeneration is much slower than the microbial utilization of oxygen in the presence of an abundance of organic material, the BOD content of discharged waters must be closely monitored to prevent the above effects from occurring. An understanding of the characteristics of BOD, degradation process of BOD and the external factors that affect the degradation rate can aid in the design of treatment systems for the removal of BOD.

Organic Content of Wastewater. Organic compounds found in wastewaters are composed of a combination of carbon, hydrogen, oxygen, nitrogen plus other elements such as sulfur, phosphorus and iron (Metcalf and Eddy, 1991:65). The principle groups of organic substances are proteins, carbohydrates, fats and oils and synthetic organic molecules ranging from simple to very complex in structure. Some synthetic organic molecules include surfactants, organic priority pollutants, volatile organic compounds and agricultural pesticides. Priority pollutants are those designated by the EPA and subject to control under the Clean Water Act.

Several tests have been developed to determine the organic content of wastewaters. Lab methods used to measure gross amounts (>1 mg/L) of organic matter include the 5-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total organic carbon (TOC) and theoretical oxygen demand (ThOD) tests. The COD is a measure of the oxygen required to chemically oxidize organic matter in waste. COD is typically higher than BOD because more compounds can be chemically oxidized than biologically oxidized. The TOC test is "especially applicable to small concentrations of organic matter" (Metcalf and Eddy, 1991:82) and often produces values slightly less than the actual amount present. The ThOD is a stoichiometric determination of organic content which can be calculated from a balanced reaction equation if the chemical formula of the organic matter is known. The BOD_5 is "an index of the biodegradable organics present" used to "assess concentration and composition of organic matter in raw water supplies, wastewaters, treated effluents, and receiving waters and to determine the efficiency of treatment processes" (Eaton and others, 1994:27)). The most widely used parameter of organic pollution in wastewater and surface water is BOD_5. It is used to:

> (1) to determine the approximate quantity of oxygen that will be required to biologically stabilize the organic matter present, (2) to determine the size of waste treatment facilities, (3) to measure the efficiency of some treatment processes, and (4) to determine compliance with wastewater discharge permits (Metcalf and Eddy, 1991:71).

Natural Sources of BOD. Other sources of BOD entering a natural treatment system such as a wetland are macrophyte generation of dissolved organic matter and water column primary producers such as phytoplankton, periphyton and submerged macrophytes. Kadlec notes that "since a natural wetland function is the production of carbonaceous material, a non-zero background BOD is found in all wetlands, in most cases about 5 mg/l" (Mitsch, 1994:341). Moshiri recognizes this fact also and states that " a primary function of macrophytes in natural wetlands is to generate photosynthetically large amounts of organic carbon" (Moshiri, 1993:5). Organic carbon is released by the macrophytes in both particulate form and as dissolved organic matter (DOM). According to Moshiri, the portion released as DOM may add up to 30 to 40 % of the total net productivity of the macrophytes and can be released within hours of senescence. Moshiri also notes that most macrophytes exprerience "more or less continuous senescence and sloughing of a portion of their foliar and rooting tissues"

(Moshiri, 1993:5). Cronk and Mitsch found that in four newly constructed freshwater wetlands, water column primary producers contributed an estimated 17 to 67% of the net above ground carbon production of each wetland, with the remaining attributed to macrophyte productivity (Cronk, 1994:449). Although rates of algal dissolved organic matter release have been reported to range from less than 1% of net primary productivity all the way up to the rate equal to net primary productivity, on average, most values are less than 20% (Wetzel, 1975:244). Wetzel notes that "bacterial utilization of excreted organic compounds is extremetly rapid" and further states that:

> Organic substrates released by the macrophytes and algae in part are actively utilized by epiphytic bacteria. Dissolved organic compunds not utilized by this association, or adsorbed within or to monocarbonate surfaces, enter the pool of littoral dissoved organic matter for further bacterial processing (Wetzel, 1975:400).

The BOD source term associated with macrophyte secretion can be quantified by using estimates of net primary productivity (NPP) values specific to the type of macrophyte of interest. Richardson published NPP values for several fresh water wetland ecosystems, to include cattail and reed marshes (Greeson and others, 1978:135). His data calls out both above ground and below ground net productivity values for several species. Table 2.5 contains NPP values for some of the species reported by Richardson.

Table 2.5

Marsh Net Primary Productivity

Wetland Type and *Dominant Species*	Net Productivity (m.t. ha^{-1} yr^{-1})		
	Above Ground	Below Ground	Total
Cattail			
Typha latifolia	5.3	15.1	20.4
Typha latifolia	7.2	14.9	22.1
Typha (hybrid)	16.8	14.8	31.6
Typha sp.	18.5	16.0	34.5
Typha sp.	15.7	10.5	26.2
Avg. Cattail NPP	12.7	14.26	26.96
Reed			
Typha angustifolia & Phragmites communis	8.1	18.0	26.1
Phragmites communis	7.8	6.2	14.0
Scirpus lacustiris	13.3	12.2	25.5
Juncus effusus	16.7	1.9	18.6
Avg. Reed NPP	11.48	9.58	21.06

(Greeson and others, 1978:135)

The BOD Oxidation Reaction. The aerobic decay of organic matter under biological conditions occurs in multiple steps during which a portion of the organic matter is oxidized to carbon dioxide and water and part is used to produce new microbial organic matter. The weight of cells produced per weight of substrate utilized is called the biomass yield. The amount of oxygen used in the repetitive process up to any time is a measure of the biochemical oxygen demand. The following quantitative relationship represents the theoretical amount of oxygen required to convert a given amount of organic matter to carbon dioxide, water and ammonia:

$$C_nH_aO_bN_c + (n + \frac{a}{4} - \frac{b}{2} - 3\frac{c}{4})O_2 \rightarrow nCO_2 + (\frac{a}{2} - \frac{3}{2}c)H_2O + cNH_3$$

(Sawyer and others, 1994:528)

The rate of the above reaction depends a great deal on the environmental and other external conditions such as microbial population numbers and their ability to acclimate to the substrate; temperature; nature and concentration of the organic matter; available dissolved oxygen; availability of nutrients; and pH. However, Streeter and Phelps presented a basic model to predict the distributions of BOD and DO concentrations in streams by making the following generalization: "The rate of the biochemical oxidation of organic matter is proportional to the remaining concentration of unoxidized substance, measured in terms of oxidizability" (Phelps and others, 1948:309).

Streeter and Phelps determined that the following first order rate reaction was representative of the BOD reaction: $-\frac{dL}{dt} = KL$, and when integrated becomes:

$L_t = L_o\, e^{-kt}$ or $L_t = L_o\, 10^{-Kt}$, where k = 0.434K and L_t is the BOD remaining at any time t, L_0 is the initial BOD value, and k is the reaction rate constant base e and K is the reaction constant base 10. It has been found experimentally for sewage treatment plant effluent, that at 20 degrees C, the constant K has a value of 0.1 day^{-1}. Phelps provides the rate of completion of the normal BOD reaction (i.e. K = 0.1 day^{-1}) at 20 degrees C in terms of the percent of ultimate BOD oxidized. Those values are as shown in Table 2.6.

It is commonly accepted that temperature effects reaction rates. Streeter and Phelps adopted the value of 1.047 as a temperature coefficient to describe the relative increase in the reaction rate for an increase of one degree C. That is, the velocity of the reaction is increased by 4.7% for a rise of one degree C (Phelps and others, 1948:310). Phelps provides a table of values for the BOD specific to sewage which relate the time and temperature of a BOD reaction to the 5-day BOD value (BOD_5) at 20 degrees C.

These values can be used to determine the oxygen demand on a stream at a given time and temperature based on the standard BOD_5 value given for the waste.

Table 2.6

Rate of Completion of the Normal BOD Reaction for the Effluent from a Sewage Treatment Plant

Time (days)	Oxidized (%)	Time (days)	Oxidized (%)
1	21	9	87
2	37	10	90
3	50	11	92
4	60	12	94
5	68	14	96
6	75	16	97
7	80	18	98
8	84	20	99

(Phelps and others, 1948:311)

Further studies on domestic waste have shown that k values vary considerably from day to day and are typically much higher than the originally presumed constant 0.1. In general it was found that the k value varies a great deal depending on the nature of the waste. "Typical values were given as 0.39, 0.66, and 0.17 (day^{-1}) for raw, primary and secondary treatment effluents, respectively" (Kawashima and others, 1989:1003). The reaction rate has a significant effect on the course of the BOD reaction as is illustrated by Table 2.7 and Figure 2.1 which compare percent of total BOD exerted over time between different reaction rate constants:

Table 2.7

Significance of Reaction Rate Constant k upon BOD
(Percent of Total BOD exerted)

Time (days)	*k = 0.05*	*k = 0.10*	*k = 0.15*	*k = 0.20*	*k = 0.25*
1	11	21	29	37	44
2	21	37	50	60	68
3	29	50	64	75	82
4	37	60	75	84	90
5	44	68	82	90	94
6	50	75	87	94	97
7	55	80	91	96	98
10	68	90	97	99	99+
20	90	99	99+	99+	99+

(Sawyer and others, 1994:538)

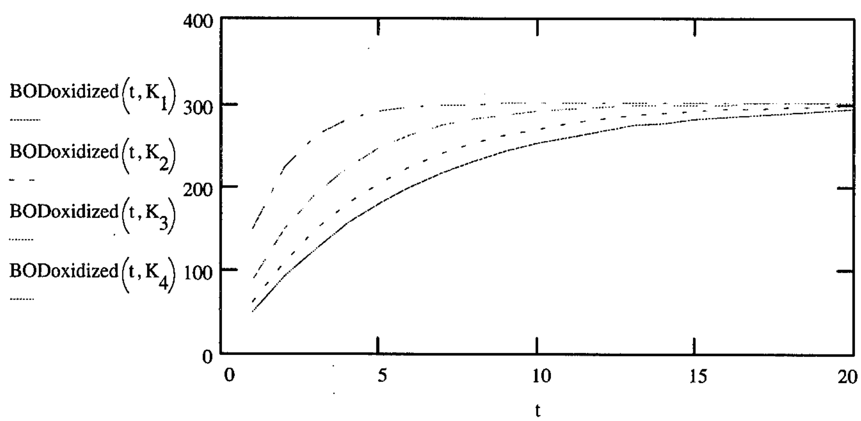


Figure 2.1

The Effect of K on BOD for a Given L Value

where $K_1 = 0.08$; $K_2 = 0.10$; $K_3 = 0.15$; $K_4 = 0.30$, L is the total or ultimate oxygen demand and t is in days.

The variation in the value of k is likely a function of the nature of the organic matter and the ability of the organisms to utilize the organic matter. Substances that are soluble and therefore readily available give high k values and therefore faster reaction rates. Those that occur in "colloidal and course suspension must await hydrolytic action before it can diffuse into the bacterial cells where oxidation can occur" (Sawyer and others, 1994:539). These substances are characterized by much lower k values and slower reaction rates. Many industrial wastes contain synthetic organic chemicals which may delay the reaction rate. These wastes may not have a sufficient population of organisms which can oxidize them immediately and an acclimation period may be required. It is important to determine a fairly representative value of k for a particular organic waste stream so that the BOD at any time t can accurately be predicted from the first order reaction rate equation previously discussed. These k values can be determined by making several BOD observations on a particular sample at different time periods to establish the shape of the curve.

Independent of the rate constant k, the demand for oxygen can be affected by the presence of nitrifying organisms which oxidize noncarbonaceous matter, specifically nitrogen in the form of ammonia. Since these nitrifying organisms are typically lower in numbers, the effect of their demand for oxygen is not seen in the first 8 to 10 days (Sawyer and others, 1994:531). This effect is represented as two stages, carbonaceous and nitrogenous. The standard BOD_5 test to predict ultimate oxygen demand (by extrapolation) was chosen to prevent the process of nitrification from affecting measurements of carbonaceous BOD. Wastes such as those from biological treatment units however often contain sufficient nitrifying populations to affect even the 5 day incubation period. Higher temperatures can also expedite the onset of any existing nitrifying organisms. Nitrification inhibiting agents are often used during the BOD_5 test to prevent interference from these organisms.

Swamee and Shekhar address the phenomenon of a two phased carbonaceous stage. They present a "generalized BOD exertion equation valid over all the phases and stages of the phenomenon" (Swamee and Shekhar, 1991:901). This includes the two phases of the carbonaceous stage and the nitrogenous stage. They begin with the basic first order rate equation and include terms to account for inclusion of a lag phase in the carbonaceous stage due to an unacclimated seed. They then add terms to account for the lag time and rate of the nitrogenous stage. Finally, they consider a plateau within the carbonaceous stage. The result is a generalized equation which models BOD exertion including all or some of the phases and stages. Several parameters including slopes of the curve during each phase/stage, apparent times for reaching the end of each phase/stage, apparent values of ultimate carbonaceous and nitrogenous BOD and lag times between phases/stages. These parameters "can be evaluated readily by plotting the BOD curve on double logarithmic paper" (Swamee and Shekhar, 1991:902). No other research that discusses any inaccuracies associated with modeling BOD without considering two distinct phases of carbonaceous BOD was found. For the purposes of many modeling efforts, such as the design of treatment systems, this two phased modeling approach appears unnecessary.

Best Management Practices

Urquhart assumed that the minimum current requirements for AF bases are those outlined in the "NPDES General Permit for Storm Water Dischargers from Industrial Activities" (Urquhart, 1994:28). This general permit requires that facilities develop a Storm Water Pollution Prevention Plan (SWPPP) which includes identification and description of potential sources of pollution and implementation of Best

Management Practices (BMP's) to reduce pollution. Praner and Sprewell provide a "compendium of US EPA recognized Best Management Practices to effectively manage NPS pollution" (Praner and Sprewell, 1992:7). They categorized urban BMP's based on Federal Regulation 40CFR 130.2(m) as either structural, requiring physical construction activities to implement, nonstructural associated with regulatory and educational programs, or operations and maintenance control measures such as housekeeping activities. Ridilla and Hoagland reviewed several nonstructural, low structural, and structural BMP's. Non structural practices include efforts such as pollution prevention and land use planning. Low structural practices "are applied at the source or upland areas of a watershed and control runoff in new developments or mitigate existing problems in developed areas" (Ridilla and Hoagland, 1993:22). Structural measures include infiltration systems, filter strips, porous paving, detention systems and wetlands among several others.

Field discusses several structural management alternatives which include land management practices to reduce stormwater runoff and pollutants before they enter the downstream drainage system; collection system controls for wastewater interception and transport; storage facilities and finally treatment. According to Field, "The most promising and common approach to urban storm flow management involves the integration of control and treatment" (Field, 1993:31). Collection systems and storage facilities used in storm drainage systems extend system capacities, provide flow equalization and water quality enhancement, prior to final treatment. These structures must be sized based on data regarding the intensity, duration and frequency of rainfall. The collected stormwater is then released from the structure to a treatment system by a flow regulator.

Field provides a review of best management practices for urban stormwater runoff control to include the following structural control measures:

(1) *Detention Ponds* have the dual purpose of reducing flood damages downstream and reducing nonpoint pollution from storm runoff. Pollutant removal mechanisms include particle settling and decay (Field, 1993:192-193).

(2) *Infiltration Facilities* such as infiltration trenches and basins or porous pavement both "allow stormwater runoff to filter through the soil column where pollutant removal by physical, chemical and biological processes take place" (Field, 1993:197).

(3) *Vegetative Filter Strips* can be used as a treatment stage preceding another practice. The filter strip "serves to slow down overland flow, allowing sediments and pollutants to settle out or infiltrate" (Field, 1993:199).

(4) *Wetlands* which are often used for final treatment of municipal wastewater, are now considered a viable BMP for stormwater runoff treatment. Studies have shown integrated detention pond/wetland systems are effective in removing metals and suspended solids (Field, 1993:201).

According to Novotny and Chesters, as reported by Ridilla and Hoagland, the following should be considered when selecting the most appropriate BMP to implement:

- Type of land activity.
- Physical conditions in the watershed.
- Pollutants to be controlled.
- Site-specific conditions. (Ridilla & Hoagland, 1993:18)

Wetlands

Wetlands are broadly defined as transitional areas between open water and dry land. (US EPA, 1993a.). According to EPA Wetlands Fact Sheet #9, the US Army Corps of Engineers and the US EPA have used the following definition for regulatory purposes since the 1970's:

> Wetlands are areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas (US EPA, 1993b).

Wetlands are typically characterized by emergent vegetation such as cattails, rushes, and reeds. They may also support floating plant species such as water hyacinths and duck weed as well as submerged plants such as pondweed. Natural wetlands are a highly productive ecosystem that benefit several trophic levels from bacteria up through plants, animals and humans. Wetlands also provide the valuable function of flood control by buffering downstream areas through slowing the flow of flood water. They may also recharge groundwater aquifers or serve as discharge areas for surfacing groundwater (Hammer, 1990:11). Of primary importance is the function of water quality improvement that wetlands provide by removing pollutants from both point and nonpoint sources through various intrinsic mechanisms. Recognition of this valuable capability has lead to the exploitation of this natural system for the purpose of wastewater treatment.

The primary components of a wetland that influence the ability to treat wastewater include the plants, soils, bacteria and animals (Reed, Middlebrooks, Crites, 1988:166). Each of these components perform complex functions which serve to aid in the physical, biological or chemical transformation of pollutants entering the system.

Vegetation. "Natural wetlands are populated by different plant types adapted for growth in water or saturated soil" (Hammer, 1990:73). These aquatic macrophytes are divided into free floating and rooted forms. The rooted forms are further subdivided into emergent, floating and submerged classes. Certain types of macrophytes in wetlands have several properties that lend themselves as suitable components of constructed wetlands for the purpose of wastewater treatment. The functions that these plants perform include stabilization of surface beds, physical filtration, facilitation of sedimentation, insulation against frost, attenuation of sunlight to prevent growth of algae, temperature moderation, transfer of oxygen to the rhizosphere, and surface area for attached microbial growth (US EPA, 1988:3). The presence of emergent vegetation also influences water movement through the wetland by reducing flow, depending on density of the vegetation. Emergents can also have a substantial effect on the water level due to transpiration rates.

Constructed wetlands of the free water surface type are typically populated by the emergent class of vegetation. The most frequently used for freshwater marshes include cattails, reeds, rushes, bulrushes and sedges (US EPA, 1988:25; Mitsch, 1993:605). The EPA wetland data base reports the cattail and bulrush as the most common vegetation type in wetlands surveyed. These types of emergents are widespread, able to tolerate a range of environmental conditions, and presumed to have positive impacts on the transfer of oxygen to the rhizosphere. Mitsch, however, notes that cattail is a rapid colonizer of limited wildlife value and is therefore viewed by some as undesirable (Mitsch, 1993:608). The Maryland Created Wetland Design Standards refer to both cattail and the common reed as aggressive volunteers that are unacceptable species in constructed wetlands (Hammer, 1990:257). The EPA, however notes that cattail are "hardy, capable of thriving under diverse environmental conditions, and easy to propagate and thus represent an ideal plant species for constructed wetlands" (EPA, 1988:25). Daukus et al. also describe a constructed wetland for the purpose of stormwater management that was planted with cattail comprising 30% of the total number of tubers (Hammer, 1990 :689). Reed et al. note that attempts to control diversity of vegetation in a constructed wetland may be fruitless because cattails and reeds or bulrushes often tend to dominate due to high nutrient levels (Reed and others, 1988:169). The dominant type of vegetation as well as its density will also be determined by the depth of water in the wetland.

Soil. Wetland soils are a mixture of mineral sediment and organics, water, and pore space (Hammer, 1990:43). The composition of mineral soil and organics determines the physical and chemical properties of the soil, and in turn affects the suitability of the soil in a wetland constructed for wastewater treatment. The influent to a wetland typically contains mineral solids which settle out and deposit on the soil surface, reducing turbidity downstream. This process of mineral sediment deposition is relatively irreversible, incorporating the minerals into the soil storage compartment indefinitely (Moshiri, 1993:293). The accumulation of organic soil or peat is a product of an internal wetland process. It is formed when biomass production within the wetland exceeds the decomposition rate (Moshiri, 1993:294). Accumulation of organic matter in a wetland is a slow process as compared with the accumulation of mineral deposits. Accretion rates for organic material in wetlands have been found to be one sixth the rate of mineral sedimentation and annual mass accumulations an order of magnitude less for organics as compared with mineral deposits (Moshiri, 1993:294). Organic soil accumulation may even be considered negligible if the plant litter produced every year is decomposed

(Moshiri, 1993:294). The deposition of both organic material and mineral sediments take with them both nutrients and contaminants.

Wetland soils, or substrates, provide physical support for plants, a reactive surface area for complexing ions, anions, and compounds as well as attachment surfaces for microbial populations. According to Faulkner et al., "the effectiveness and capacity of a soil to remove/retain contaminants is a function of soil-wastewater contact" (Hammer, 1990:42). This soil wastewater contact is a function of the hydraulic conductivity, a physical property of the soil composition, which determines water movement through the soil. "Fine-textured silty or loamy soils permit more soil-water contact" because of their higher total porosity and lower hydraulic conductivity. In contrast, sandy or gravelly soils, characteristic of the mineral soil type, have a higher hydraulic conductivity and lower porosity than organic soil types and allow water to move rapidly through the soil allowing less soil-water contact.

The saturated conditions of wetland soils induce an anaerobic environment. The amount of dissolved oxygen available to microbes, plus the extent of soil-water contact, are important factors in the process of microbial degradation of organic matter at the soil surface of the wetland. Therefore, the physical properties such as total porosity and hydraulic conductivity of the soil affect the capacity for microbial degradation of BOD in the sediment.

The chemical properties of the soil determine the wetland's ability to remove wastewater constituents by the following mechanisms: (1) ion exchange/nonspecific adsorption; (2) specific adsorption/precipitation; and (3) complexation. Ion exchange is determined by the reactivity of the soil, a function of surface area and surface charge of soil particles. The process involves ionically bonded cations on the solid surface exchanging with other cations in the soil solution. "The cation exchange capacity measures the soil's capacity to hold cations on exchange sites and varies widely among different soils" (Hammer, 1990:43). For example, organic soils have higher CECs than mineral soils. Specific adsorption reactions occur when a ligand occupies a position within the coordination sphere of the cation. Complexation primarily refers to metals binding with soil organic matter.

The bacterial component of the wetland may be the most important, though it is the least understood (State of MD Sediment and Stormwater Division, 1991:35). The bacteria within the water column, attached to vegetation, and on the wetland soil surfaces are the primary mechanisms responsible for the biological transformation of organic matter. These processes create a demand on the existing dissolved oxygen supply which affects the health of the wetland system. The processes of microbial degradation of organic matter and consequential oxygen demand are discussed in subsequent sections.

Natural Wetlands. Since natural wetlands are considered by law to be "waters of the United States" and therefore require a permit for any discharge into them, the decision to utilize a natural wetland for treatment purposes is not economically favorable. Natural wetlands are also characterized by extreme variability in functional components making it very difficult to predict responses to wastewater application. It is Moshiri's opinion that use of wetlands for treatment of wastewater could lead to "disastrous results in ecosystems where recovery from long term damage could take many decades" and that "natural wetlands should not be used deliberately as wastewater treatment systems, but should be preserved for environmental conservation" (Moshiri, 1993:10). Constructed wetlands however perform all of the same functions and provide the same advantages as natural wetlands regarding wastewater treatment, without the difficulties associated with regulatory requirements or the dangers of damaging a natural resource. As compared with natural wetlands, constructed wetlands allow a greater degree of control of the system. Site selection, sizing, composition of substrate, type of vegetation, retention time, flow pattern, etc. can all be determined by the design of the system. Constructed wetland systems also offer several advantages over conventional treatment systems such as :

1. low cost of construction and maintenance
2. low energy requirements
3. low training requirements for personnel
4. more flexible and less susceptible to variations in loading rate.

Due to the rising costs of construction, operation and maintenance of conventional wastewater treatment facilities, interest has turned to the use of natural systems such as wetlands. The EPA's inventory of constructed wetlands in the US shows that the primary purpose for the construction of wetlands has been for municipal wastewater treatment. Since the quality of stormwater runoff has become an increasing environmental concern, and recent regulation requires the implementation of best management practices for stormwater management, wetlands are considered a natural alternative. Natural wetlands perform several stormwater management functions to include: "conveyance and storage of stormwater, which dampens effects of flooding; reduction of flood flows and velocity of stormwater, which reduces erosion and increases sedimentation; and modification of pollutants typically carried in stormwater" (Hammer, 1990:259). Other ancillary benefits noted are that the construction of these systems "provide a source of fill to the developer, premium lakefront property, open space, recreational area, and aesthetic enhancement" (Hammer, 1990:259). Although the nature of stormwater differs a great deal from that of typical municipal wastewater as previously discussed, the functions of the wetland are the same, as are the typical designs. The following section presents the predominant types of wetlands constructed for the purpose of wastewater treatment. These systems are suitable for the treatment of stormwater as well as municipal or industrial wastewater treatment.

Constructed Wetlands. Two principal types of wetlands have been used for the purpose of wastewater treatment: the free water surface (FWS) and subsurface flow (SSF) systems also referred to as vegetated submerged bed (VSB) systems. The US EPA initiated a project in 1990 to collect and catalog data on existing wetland treatment systems, specifically those that treat municipal and industrial wastewaters as well as stormwaters. Agricultural and mining wastewater systems were not included. The database is not complete as the project was ended in 1993 due to funding limitations. The data base does however contain data on 178 sites which include 203 separate systems consisting of 323 treatment cells. Municipal sites number 154, 9 are industrial and 6 stormwater the remaining 8 are not categorized. Of the 178 sites, 24% are natural wetlands, the remaining 76% are constructed. Of the constructed wetland systems, 68% are FWS and the rest are VSB. The data base shows that the VSB systems are most common in the southern US with the FWS more widely distributed throughout the US.

Although both types of systems are considered "attached growth biological reactors and operate similarly to microbial activity occurring in trickling filters, RBC units..." (Reed and Brown, 1992:776), several factors such as the physical construction, primary mechanism for reaeration, the critical surface area available for microbial growth, and the prevalent hydraulic mixing regime differ between the two types of systems. The following section describes the physical characteristics of each system and provides a comparison of the benefits and drawbacks of each.

Free Water Surface Wetland Systems. The FWS system typically consists of a shallow basin or channel which may or may not be lined with an impermeable barrier, depending on soil permeability and whether it is necessary to prevent seepage to and infiltration from ground water. Whether or not a barrier is used, a suitable medium is required to support emergent vegetation. Typical substrates used range from gravel or mine soils to clay or peat (Moshiri, 1993:61). Water flows through the wetland at a shallow depth, ranging from 2 to 36 inches with a typical depth of 12-18 inches (Reed and Brown, 1992:778), and with a low flow velocity regulated by emergent vegetation. The bottom surface of the wetland is graded to an average slope of about 0.23% to ensure uniform flow (Moshiri, 1993:41). The wetland may consist of open water areas as well as highly vegetated areas. These open water areas allow for rearation at the surface contact with the atmosphere. The surface area of vegetation stems and leaves in the water column serve as the primary site for microbial growth and consequential degradation of pollutants that come into contact with them. An illustration of the cross section of a FWS wetland is shown in Figure 2.2.

Moshiri also describes submerged macrophyte based FWS systems. Because submerged macrophytes thrive only in oxygenated water, these types of systems are not effective in wastewater with a high content of readily biodegradable organic matter because the microbial decomposition of the organic matter will create an anaerobic environment. The submerged macrophytes have their photosynthetic tissue entirely submerged and are able to increase the content of dissolved oxygen during periods of high photosynthetic activity. These periods provide for favorable conditions for degradation of organic matter in the water. These systems are still in the experimental

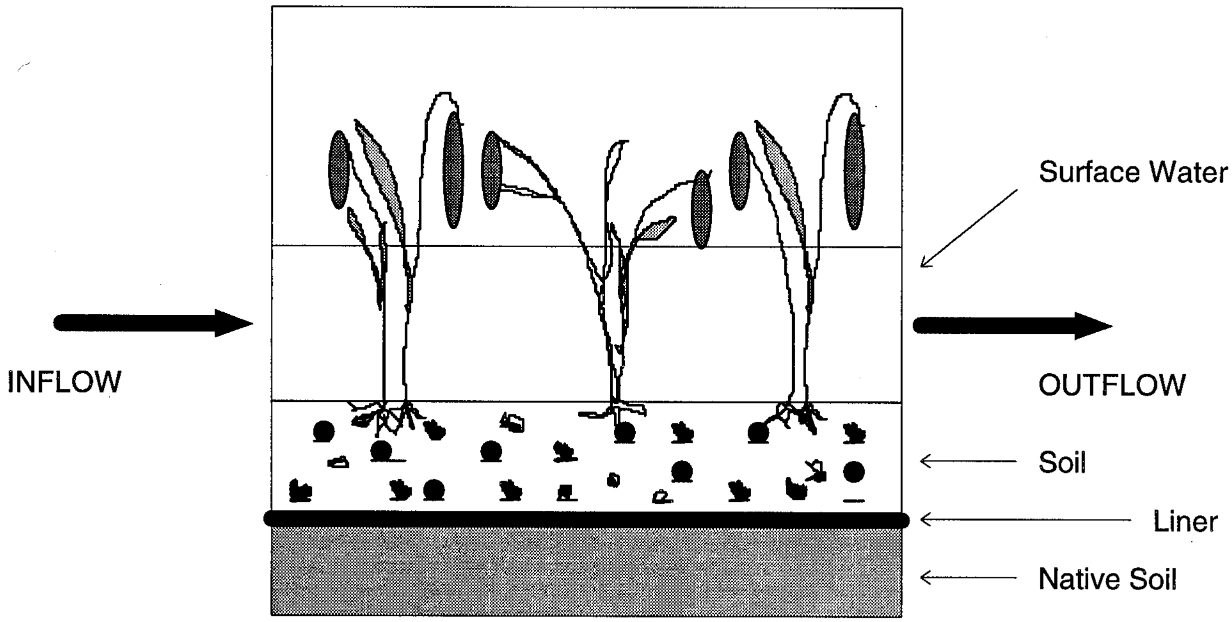


Figure 2.2

Cross-section of a Free Water Surface Wetland

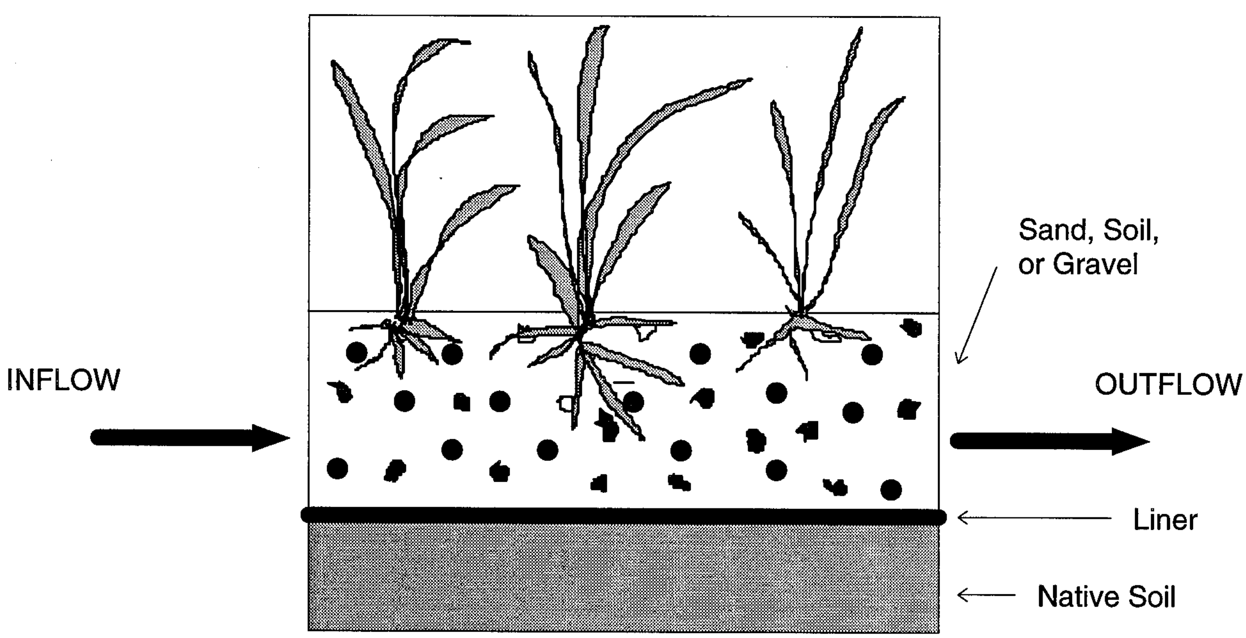


Figure 2.3

Cross-section of a Sub-Surface Flow Wetland stage and it is suggested that systems utilizing submerged vegetation are best suited for "polishing" secondarily treated wastewaters or as a final step in multistage systems (Moshiri, 1993:15).

FWS are similar to natural marshes and provide the ancillary benefit of providing wildlife habitat (Moshiri, 1993:61).

Subsurface Flow Systems. The vegetated submerged bed systems are more recently developed than the FWS systems, though they are widely used in Europe to treat primary effluent and screened raw sewage. Most of the SSF systems in the United States were constructed between 1988 and 1993 (Reed and Brown, 1995:244). The SSF system consists of a basin or channel filled with a media to a depth of 12-30 inches (Reed and Brown, 1992:778) supporting wetland vegetation such that the water flows horizontally through the media with no open surface flow. The media and its porosity are critical factors in determining the surface area available to microbes for degradation of pollutants. Because the system is designed to contain flow below the media surface, there is very little direct reaeration at the surface, therefore the major source of oxygen in SSF systems is through the macrophyte root system. The selection of vegetation with extensive root systems for this purpose is essential. The common reed *Phragmites australis* is typically used (Moshiri, 1993:14). Reed and Brown however found during visits to several sites that roots seldom penetrated to even 12 inches regardless of plant type. This results in anaerobic conditions in the bottom half of the bed which reduces optimal BOD removal (Reed and Brown, 1992:779). Reed and Brown also found that a number of SSF systems they visited experienced surface flow thereby negating any benefits of the submerged media. Investigation of this problem revealed that primarily inorganic trapped solids approached at most 6 percent of the available void spaces, and their presence could be attributed to construction activities. Further investigation of the surface flow problem by Reed and Brown showed that inadequate hydraulic design, not clogging, is the cause. Accurate knowledge of the hydraulic conductivity and porosity of the gravel media is essential since Darcy's law describes the flow regime in this type of wetland (Reed and Brown, 1995:246). See Figure 2.3 for an illustration of the cross-section of a SSF system.

Comparison of the Systems. Reed and Brown analyzed the available data from the EPA sponsored constructed wetland inventory, and conducted site visits to over 20 of the systems inventoried. Construction cost data for 19 FWS and 18 VSB provided by the inventory revealed average costs of $22,000/acre and $215,000/acre respectively. When the costs are calculated on a per unit flow basis however, the VSB systems show an advantage due to the smaller size required to treat the same loading. The costs per unit flow were $0.62/gal for VSB systems and $0.78/gal for FWS systems (Reed and Brown, 1992:778). Freeman however contradicts these figures by listing lower installed cost/gal for FWS systems (Moshiri, 1993:69). The difference in cost/gal provided by Reed and Brown may not even be significant.

Regarding treatment effectiveness, Reed and Brown concluded that both VSB and FWS systems are "capable of effective removal of both BOD and TSS" (Reed & Brown, 1992:779). They suggest that both systems "can be reliably designed for organic loadings up to at least 100 kg/ha/d" but that "a FWS wetland will be larger than an SF wetland for comparable flow and BOD removal goals" (Reed and Brown, 1995:244). They however emphasize the problem of reduced oxygen availability in VSB systems due to the inability of plant root systems to completely penetrate the depth of the submerged bed leading to reduced BOD degradation. Reed and Brown suggest methods such as "water level management in the bed using an adjustable outlet mechanism or alternating operation of parallel cells" in order to induce and maintain root zone development and the related oxygen source" (Reed and Brown, 1995:248). Surfacing of wastewater in VSB systems is also a significant problem that must be addressed through appropriate hydraulic design. Freeman noted that a reduction in loading to 10% of design was required to bring the water level below the surface of a VSB system located in Benton, Kentucky, making the system economically prohibitive (Moshiri, 1993:70). Again, an adequate hydraulic design may have prevented this problem.

Freeman provides a brief comparison of the two types of systems, excluding the treatment effectiveness mentioned above:

Table 2.8

Comparison of FWS and VSB Systems

FWS Systems	VSB Systems
Lower installed cost/gal	Greater assimilation rate, less land required
Simpler hydraulics	No visible flow, less nuisance, vector problems, odors
More natural wetland values can be incorporated into the system wildlife habitat, etc.	More cold tolerant

Constructed Wetland Design Considerations

According to Witthar, the following seven parameters are important in the design of an operational wetland: (1) area requirements, (2) water depth, (3) number of cells, (4) cell shape, (5) flow velocity, (6) wastewater retention time, and (7) substrate (Moshiri, 1993:148).

Area Requirements. The area required for a constructed wetland is often determined by the expected hydraulic loading rate. The EPA concluded from results of one constructed wetland that rates of 200 m^3/ha-d (0.02 m^3/m^2-d) provided maximum treatment efficiencies (US EPA, 1988:24). Witthar reports an average of 24.6 gallons per day per square meter (0.09 m^3/m^2-d) from the results of a survey of existing wetlands treating mine drainage (Moshiri, 1993:148). A report by Reed & Brown showed FWS systems designed for BOD, TSS and NH_3 treatment received hydraulic loading ranging from less than 20 L/m^2/d (0.02 m^3/m^2-d) to nearly 120 L/m^2/d (0.12 m^3/m^2-d) with most receiving between 60 and 80 L/m^2/d (0.06-0.08 m^3/m^2-d) (Reed and Brown, 1992:778).

Organic loading rate is also an important consideration as opposed to a design criteria, in order to maintain aerobic conditions in the wetland. Some wetlands evaluated by Reed operated at organic loading rates ranging from 18 to 116 kg/ha-d. Reed recommends a maximum limit of 110 kg/ha-d to ensure aerobic conditions, and the EPA recommends a maximum mass loading rate of about 112 kg BOD/ha-d (US EPA, 1988:24; Reed, 1988:180). Knight reports typical BOD mass removal efficiencies near 70% or more where mass loading rates are up to 280 kg/ha-day, and much lower removal efficiencies at mass loadings less than 50 kg/ha-day (Moshiri, 1988:49).

Urbonas et al. suggest that when a wetland follows a detention basin or pond, that the surface area of the wetland be sized to be at least 1.5% to 3.0% of the impervious area from which the runoff originates (Urbonas and Stahre, 1993:390).

Water Depth. Several studies show different water depths for FWS systems. Existing systems depths range from 0 - 2 ft (0 - .61 m) with an average water depth of 18 inches (.46 m) (Wetland Database). Reed suggests that maximum depth not exceed

0.6 m and should preferably be in the range of 0.3 to 0.45 m (Reed, 1988:181). Witthar suggests that FWS systems function best when water depth is less than 18 inches (0.46 m) (Moshiri, 1993:148).

The depth of water is also limited by depth requirements of wetland emergent and submerged vegetation. The type of species and its density may be correlated with water depth. Shallower ponds with depths of .3 m tend to be more densely vegetated than deep ponds with depths of .61 m. Studies of emergent vegetation in shallow (.3 m) and deep (.61 m) artificial ponds showed mean densities of 182 plants/m^2 and 31 plants/m^2 respectively (Sediment and Stormwater Division, State of Maryland, 1991:109).

Number of Cells. Multiple cells can be used to raise low levels of dissolved oxygen when high BOD and COD levels are expected. This is more likely the case with municipal wastewater rather than stormwater. Much lower levels of BOD in stormwater runoff would not necessarily create a need for more than one cell.

Cell Shape. The length to width ratio determines the distance the influent travels before exiting the wetland and also affects the velocity of the flow through the wetland. Larger length to width ratios result in greater flow velocity. Reed refers to studies performed in Canada and California which recommend aspect ratios of at least 10:1.

Flow Velocity. The flow velocity is a function of the influent and effluent rates and the cross-sectional area of the surface water. A lower flow velocity allows a longer time for contact with surface areas responsible for degradation process. Witthar recommends flow velocities ranging from 0.1 to 1.0 ft/s (109.7 m/hr) (Moshiri, 1993:149). Flows are also measured in cubic meters per day, and the Wetland

Database shows that most constructed FWS wetlands receive flows ranging from 10 to 10,000 cubic meters/day.

Hydraulic Loading Rate. The hydraulic loading rate is the volume of water received by the system per area per time. The EPA notes that hydraulic loading rates of .015 - .05 m^3/m^2-day have been reported and that for a specific wastewater treatment system at Listowel, Ontario, the hydraulic loading rate of .02 m^3/m^3-day provided maximum treatment efficiencies (US EPA, 1988:24). Reed and Brown also show hydraulic loading rates on FWS wetlands with effluent goals of reducing BOD ranging from .015 - .12 m^3/m^2-day. The Wetlands for Wastewater Treatment Performance Database reports that the bulk of the constructed FWS systems receive hydraulic loading rates ranging from 0 - .05 m^3/m^2-day, the remaining systems surveyed ranged from .05 - .2 m^3/m^2-day.

Retention Time. Retention time is a primary factor in the treatment efficiency of a constructed wetland. The retention time is a function of the hydraulic characteristics of the wetland. Witthar states that retention times range from 0.25 to 75 days with an average of about 5 days (Moshiri, 1993:149). The EPA recommends a detention time of 6-7 days for the treatment of primary and secondary wastewater (EPA, 1988:25).

Substrate and Liners. Daukus et al. suggest that wetland basins contain .46-.60 m of organic soil. Hammer suggests that an impermeable liner be placed .4 to .6 m below the surface of the soil, suggesting similar bed depth. According to Hammer, "Most natural wetlands are perched above an impervious layer that reduces or prevents water loss to underlying strata" (Hammer, 1992:165). The Maryland Created Wetland Design Standards include guidelines that recommend a clay or synthetic liner "if the basin is above the water table and the infiltration rate is high" (Hammer, 1990:256). In order to maintain its hydrology, wetlands must be able to minimize water losses during periods of low or no influent. Without the existence of an impermeable layer, water losses to ground water would exceed water gains from precipitation, runoff or wastewater influent. Both subsurface and surface flow wetlands require an impermeable barrier in order to ensure containment of wastewater and to prevent contamination of groundwater.

In order to determine the most appropriate and cost effective means of providing a relatively impermeable layer, the composition of the soil must be evaluated and hydraulic conductivity determined. "If the conductivity is greater than 10^{-6} or 10^{-7} cm/s, sealing or lining must be considered" (Hammer, 1992:165). If the existing soil consists of "a wide range of silt, sands, and other small particle sizes" and a clay content greater than 10%, then compaction may be an adequate method to ensure a nearly impermeable state (Hammer 1992:165). According to the EPA Design Manual, "sandy clays and silty clay loams can be suitable when compacted" (US EPA, 1988:15). The barrier must be placed approximately 40 to 60 cm below the surface of the soil in order to support vegetation and ensure that root development does not penetrate the barrier and cause leakage (Hammer, 1992:165). The substrate placed above the liner must contain adequate nutrients and porosity in order to support the emergent vegetation. Sandy loam soils are described as soft and friable and along with clay loam soils are appropriate since they both "normally have adequate nutrients, provide good water and gas circulation, and have moderate texture to support the new plants and to permit root or rhizome penetration" (Hammer 1992:203).

Preliminary Treatment

Haan, Barfield and Hayes advise against the use of constructed wetlands as a primary settling system for sediment. Pretreatment to remove sediment may be necessary, depending on the quality of the influent, in order to prevent sediment build up, clogging, and to keep the organic loading at a level that will avoid localized anaerobic conditions. Some type of detention pond is typically used upstream of the receiving constructed wetland. FWS systems are typically loaded less than VSB systems and may require a higher degree of pretreatment than VSB systems, though VSB systems also require pretreatment in order to lessen the possibility of clogging.

According to Breen et al., "If major particulate loads were allowed to enter the wetland component of the system their accumulation over time would eventually alter system morphology and hydrology" (Breen, 1994:106). He continues to state that "the removal of larger particles protects one of the key roles of emergent aquatic macrophytes in the wetlands which is the provision of surface area for the filtering and adhesion of smaller particles" (Breen, 1994:106). The removal of total suspended solids from stormwater also contributes to the removal of some proportion of other pollutants present in the stormwater since "most pollutants appear to have a strong affinity to suspended solids" (Stahre and Urbonas, 1990:279). Wet detention ponds are one means of capturing the runoff and providing for sedimentation of suspended solids. Wet detention ponds are essentially a basin with a permanent pool of water, as opposed to dry basins which are designed to drain completely following a storm event. The detention ponds can also be used to control the peak rate of flow during a storm. The detention pond must therefore be sized to have sufficient volume to handle peak flows.

System Hydrology

System hydrology is an important factor in the performance of a constructed wetland system. An optimum water level is required to support various types of vegetation and other aquatic life. In order to maintain a wetland ecosystem, water losses must be balanced by water gains. The hydraulic residence time is also an important factor in the treatment of pollutants. Therefore, site hydrology is important to the livelihood of the wetland ecosystem as well as the effectiveness of stormwater runoff treatment. System hydrology is determined by influent sources, effluent rate, precipitation, evapotranspiration, infiltration, seepage, hydraulic loading rate and water depth. All of these factors must be quantified and included in a water balance in order to determine an appropriate system design. Infiltration and seepage can be ignored however if the system is designed with an impermeable layer or liner.

According to Hammer, the most important source of water for natural, restored and constructed wetlands is surface runoff, although he found that some natural wetlands do depend on groundwater supplies (Hammer, 1992:24-26). He continues to state however that "attempts to construct wetlands that intercept ground waters have not been very successful due to limited understanding of locations and hydraulic gradients of underground waters" (Hammer, 1992:26). Hammer also warns against dependence on groundwater for water supplies since "little control over the basic management mechanism (hydrology) will be possible and falling groundwater levels could jeopardize the continuance of the wetlands" (Hammer, 1992:134).

In order to maintain a pool of standing water or a low flow rate, physical water control structures are essential. Construction of dikes or berms will retain water to form the needed hydrologic environment and a water control structure will maintain near constant water levels throughout the year. A secondary purpose for the water control structure is to "drain the pool for dike repair or other needed maintenance, or for deep flooding to retard or reverse successional changes" (Hammer,1992:170). Hammer lists the following characteristics of the ideal water control structure:

> 1. provide for fairly precise regulation of water elevations
> 2. have the capacity to raise water levels to the maximum permissible level with a safe margin of dike freeboard, essentially the elevation of the emergency spillway outlet
> 3. have the capacity to completely dewater the pool
> 4. allow changes to be easily made
> 5. not require changes because of increases or decreases in inflows or from precipitation
> 6. consist of simple structures requiring little or no maintenance
> 7. not be susceptible to vandalism
> 8. not be susceptible to blockage from debris or plant growth
> 9. inhibit blockage by beaver or muskrat
>
> (Hammer, 1992:170)

Various types of outlet or water control structures can be used and the design should be based on its primary desired function. The outlet structure must be able to "control the depth of water in the wetlands especially for winter ice conditions where deeper wetland conditions are required to maintain treatment levels" (US EPA, 1988:26).

If the primary input to the wetland is runoff, it can be estimated by various methods. Storm runoff is a function of:

> (1) rainfall amounts and expected frequency
> (2) infiltration rates of the watershed soils
> (3) land use and vegetative cover conditions
> (4) slope of the land in the watershed
>
> (Hammer, 1992:179)

One method for estimating volume and peak rate of runoff will be discussed in the methodology section. In the event that runoff and precipitation are inadequate to maintain necessary water levels, alternate supplies may need to be tapped.

Due to the large surface area typical of FWS systems, precipitation and evapotranspiration can greatly affect the volume of water in the wetland at any given time. Precipitation serves to dilute wastewater thereby reducing pollutant concentrations and increase the velocity of flow, reducing retention times within the wetland. Evapotranspiration increases concentrations and slows water velocity, allowing increased contact time. The combination of water loss from surface water and from the emergent portions of plants is collectively termed evapotranspiration. The rate of evapotranspiration varies as a function of the relative humidity, air and water temperature, wind velocity and duration, and type and density of vegetation (Mitsch 1988:28). Several empirical procedures have been developed which estimate the loss of water due to evapotranspiration (Hammer, 1990:26-29). Some propositions are: (1) Wetland evapotranspiration, over the growing season, is represented by 0.8 times Class A pan evaporation from an adjacent open site; (2) Wetland evapotranspiration and lake evaporation are roughly equal; (3) About half the net incoming solar radiation is converted to water loss on an annual basis; (4) Type of vegetation is not a strong factor in water loss determination (Hammer, 1990:26-29). Class A pan data can be found in *Climatological Data*, published by the US National Oceanic and Atmospheric Administration, Asheville, North Carolina (Hammer, 1990:26); estimates of potential and actual evapotranspiration values can be obtained from the US Soil Conservation Service (Reed, Middlebrooks, and Crites, 1988:210). Metcalf & Eddy suggest using the average monthly evapotranspiration rate of the selected crop as the design evapotranspiration rate (Metcalf and Eddy, 1991:958). Data regarding local precipitation rates can also be obtained from local agencies. Metcalf and Eddy suggest using the wettest year in a 10-year period as a reasonable design precipitation rate (Metcalf and Eddy, 1991:958).

Existing Constructed Wetland Systems

Constructed wetland systems for the purpose of stormwater management often consist of a wet detention pond followed by one or more constructed wetland basins. As previously discussed, both the FWS and VSB require some level of pretreatment, dependent on the quality of the runoff, in order to remove the bulk of larger suspended solids prior to entering the wetland system. Removal of suspended solids keeps the organic loading entering the wetland within reasonable limits and may help avoid localized anaerobic conditions in the early stages of the wetland (Hammer, 1990:319). Wet detention basins with a permanent pool provide the following important functions:

1. dissipation of kinetic energy associated with runoff
2. improved hydraulic control and even distribution of flows over wetland surfaces
3. removal of coarse particulates which reduces the sediment load reaching the wetlands
(Hammer 1990:691)

The performance of these detention ponds is dependent on hydraulic characteristics such as pond area to drainage area ratio, mean depth, storm intensity and duration and antecedent dry-weather period.

A wetland/detention system constructed in Massachusetts for the purpose of treating stormwater runoff from a shopping mall included a wet detention pond sized to represent 1% of their respective catchment area. The pond was designed to attenuate peak flood flows for storms up to the 100-year, 24-hr event, with overflow capability onto an adjacent field in the event of a large storm. Effective residence time in the detention pond for an average storm was determined to be 9-20 days (Hammer 1990:691). This particular detention/wetland system consisted of the detention pond followed by three created wetland basins totaling 0.3 ha. These wetlands are described as shallow marsh communities and therefore are in the category of FWS wetlands. The wetland basins were configured to "promote a long flow path and dispersion through the wetlands to maximize exposure to organic soils and vegetation" (Hammer 1990:689). The wetland basins contain 46-60 cm of organic soil (>12% organic carbon) and were planted with a mixture of emergent plant species which included 30% cattail, 25% arrowhead, 25% bulrush, and 20% sweet flag. Estimations were made for the removal efficiencies expected from the treatment system based on hydraulic loading characteristics of the detention basin and expected wetland treatment efficiency. BOD_5 removal efficiency is expected to range between 50% and 80%. No data on actual removal efficiency was provided.

The state of Maryland developed several guidelines for the construction of shallow wetland stormwater systems as a result of stormwater management regulations. Some of these guidelines are as follows:

> 1. Water inflow from storms, base flow, and groundwater must be greater than water outflow via infiltration and discharge in order to maintain a permanent pool.
> 2. A detention time of 24 hours for the one-year storm enhances pollutant removal and provides the storage volume recovery between storms.
> 3. Five or more wetland species (two primary, three secondary) are needed to match plant requirements to variations in soil type, depth, and water circulation and to promote some active growth for nutrient uptake throughout the growing season.
>
> (Hammer, 1990:258)

Constructed wetland systems for the purpose of stormwater management are also widely used in the state of Florida. These systems typically consist of a wet detention system (permanent water pool) and a littoral zone planted with native aquatic plants (Hammer, 1990:259). The Florida Department of Environmental Regulation developed design and performance standards for wetland stormwater management systems which include monitoring efforts to evaluate their effectiveness. Some of these standards are (paraphrased) as follows:

> 1. The wetland treatment facility is part of a comprehensive stormwater management system that uses wetlands in combination with other BMPs to treat runoff from the first 2.5 cm of rainfall, unless the drainage area is less than 40 ha, in which case the first 1.2 cm of runoff must be treated.
> 2. Pretreatment swales or lakes are used to reduce sediments, oils, greases, and heavy metals, and to attenuate stormwater volumes an peak discharges so that the wetland's hydroperiod is not adversely altered.
> 3. Use shall not adversely affect the wetland by disrupting the normal range of water level fluctuation as it existed prior to construction of the wetland stormwater system.
> 4. Design features of the system shall maximize stormwater residence time, enhancing contact with wetland sediment, vegetation, and microorganisms. (Hammer, 1990:259).

<u>Microbial Population</u>

Microorganisms play a vital role in the self-purification process of natural waters. It has been previously noted that microbial degradation is the primary mechanism for BOD removal in a wetland system. For this reason, an in-depth look at the behavior of microbial populations and the environmental factors that effect their growth and ability to degrade BOD is warranted. If some of these factors can be controlled or manipulated within a wetland system, optimal degradation capabilities may be achieved.

Microbes utilize and mineralize organic nutrients, are responsible for the nitrification of ammonia, and even play a role in the control of allochtonous (foreign) populations and other pathogens in aquatic environments. The existence of such autochtonous (indigenous) microbial populations provides natural waters with the ability to accept low amounts of some pollutants without detrimental effects on other life forms.

Constructed wetlands for the purpose of wastewater or stormwater treatment depend on microorganisms as "the primary agent for removal of organic matter" (Moshiri, 1994:541). In order to effectively design a constructed wetland system in which these removal processes do occur and will continue to do so, we must understand the physical, chemical and nutritional conditions that effect microbial populations. These physical, chemical and nutritional characteristics of the water determine the selection of predominant species as well as the growth rates of the organisms. The primary characteristics include temperature, pH, available dissolved oxygen (DO), available nutrients, and the presence of toxins. Before elaborating on each of these factors, a brief discussion of the classification of microorganisms follows.

Microorganisms are classified as one of three groups: eukaryotes, eubacteria and archaebacteria. Eubacteria and archaebacteria are often collectively referred to as bacteria or prokaryotes. The eukaryote group consists of plants, animals and protists (algae, fungi and protozoa). The difference between eukaryotes and prokaryotes is based on cell structure and function. The eukaryotic cells contain a membrane bound nucleus and structures called organelles that perform functions similar to organs. Important organelles within eukaryotes are the mitochondria which are sites of electron transport and oxidative phosphorylation, and the chloroplasts which are sites of photosynthesis. The prokaryotes are single celled organisms that lack a nucleus and organelles, thereby limiting their ability to perform some functions performed by organelles within eukaryotes. The primary microorganisms important in the treatment of wastewaters are the protists from the eukaryote group, and bacteria, or prokaryotes. Bacteria are particularly important in the treatment of wastewaters due to their role in the decomposition and stabilization of organic matter.

Microorganisms can be further classified by the sources of carbon and energy that they utilize as substrates, as well as the type of electron acceptor the organism uses. Heterotrophs use organic carbon as their source of carbon for growth and reproduction. Autotrophs derive their carbon requirements from the reduction of carbon dioxide, a process which requires a net input of energy. Phototrophs use light as their energy source where chemotrophs derive energy from chemical reactions. Organotrophs use an organic compund as a source of electrons while lithotrophs use an inorganic electron source. The terms chemoheterotroph and chemoorganotroph can typically be used interchangeably as those organisms that use organic energy sources also use organic carbon sources (Gaudy and Gaudy, 1980:360). Chemoheterotrophs are the primary organism responsible for the reduction of organic content in wastewater as they derive their energy from the oxidation of organic compounds. Chemoheterotrophs important in biological treatment consist largely of protozoa, fungi and most bacteria.

As previously mentioned, several factors determine the selection of microbial populations in a habitat, as well as their growth rate and metabolic processes. These factors, typically a function of the environmental conditions, can change continually due to external forces or the organisms themselves. The primary factors are discussed below.

Temperature. Temperature is one of the most influential factors affecting selection of species and microbial growth. Microorganisms have no means for controlling temperature within the cell, therefore it is determined by the external environment. Each microorganism has a characteristic minimum, maximum and optimum temperature. The optimum temperature is that at which growth and reproduction rates are maximized. Growth rates tend to increase with temperature up to the optimal temperature, past which growth rate may drop to zero. For every 10 degree C rise in temperature from minimum to optimum, the growth rate for most organisms increases two to threefold (Gaudy and Gaudy, 1980:178). Increased temperature also increases enzyme activity which suggests an increased rate of substrate removal.

Microorganisms are categorized as either *psychrophiles, mesophiles,* or *thermophiles* based on their optimum growth temperature. *Psychrophiles* have low optimum temperatures (<0 C - <20 C), *mesophiles* are characterized by moderate optimum temperatures (20 C - <40 C) and *thermophiles* by high temperatures (45 C - 90 C). Most microorganisms fall into the *mesophilic* category. Whereas the temperatures above the upper limit for an organism can be lethal to the organism, temperatures below the minimum may only affect growth. Many organisms can remain dormant at lower temperatures. The optimum temperature or that temperature at which growth is most rapid, tends to fall much closer to the organisms maximum tolerable temperature (Gaudy and Gaudy, 1980:177). Table 2.9 gives approximate upper temperature limits for different microorganisms. Note that those capable of growing at very high temperatures (above 60 degrees C) are the prokaryotes, or bacteria.

Table 2.9

Approximate Upper Temperature Limits for Different Microorganisms

Organism	Temperature, degrees C
Protozoa	45-50
Eucaryotic algae	56
Fungi	60
Photosynthetic Bacteria	70-73
Bacteria	>99

(Gaudy and Gaudy, 1980:179)

Temperature also affects the metabolic activity of microorganisms. As with growth rate, activity such as respiration increases with temperature. Enzymes are given temperature quotients, or Q_{10} values based on the increase in activity with 10 degrees C. For example, a Q_{10} value of 2 indicates that a temperature increase of 10 degrees C within the tolerance range of the enzyme will double the activity level (Atlas and Bartha, 1993:219). These temperature quotients however may not be constant over the entire temperature range.

Hydrogen Ion Concentration (pH). Microorganisms are also characterized by a pH range for growth. "The minimum and maximum values that limit growth usually differ by only 3-4 pH units" (Gaudy and Gaudy, 1980:183). Most microorganisms cannot tolerate extreme pH values, however some acidophilic and alkalophilic bacteria exist. The optimum pH value for maximum growth rate is usually midway between the minimum and maximum tolerable values. These values for some organisms are shown in Table 2.10.

Table 2.10

Minimum, Optimum and Maximum pH
in Multiplication of Various Bacteri

Organism	Minimum	Optimum	Maximum
Escherichia coli	4.4	6.0-7.0	9.0
Pseudomonas aeruginosa	5.6	6.6-7.0	8.6
Nitrobacter spp.	6.6	7.6-8.6	1.0
Nitrosomonas spp.	7.0-7.6	8.0-8.8	9.4

(abstracted from Atlas and Bartha, 1993:233)

Gaudy and Gaudy make some general statements concerning the pH preferences of different types of microorganisms:

1. Most bacteria have pH optima near neutrality and minimum and maximum pH values for growth near 5 and 9, respectively.
2. Most fungi prefer an acid environment and have minimum pH values between 1 and 3 with an optimum pH near 5.
3. Most blue-green bacteria have pH optima higher than 7.
4. Most protozoa are able to grow in the pH range 5 to 8, with an optimum pH near 7 (Gaudy and Gaudy, 1980:183).

The pH of an environment affects microorganisms both directly and indirectly. "The pH determines in part the solubility of CO_2, influencing the rates of photosynthesis; the availablility of required nutrients, such as ammonium and phosphate, which limit microbial growth rates in many ecosystems and the mobility of heavy metals, such as copper, which are toxic to microorganisms" (Atlas and Bartha, 1993:233). According to Gaudy and Gaudy:

> utilization of the carbon and energy source, efficiency of substrate utilization synthesis of protein, synthesis of storage materials of different types and release of metabolic products from the cell, as well as other aspects of cellular metabolism were drastically affected by changes in pH over the range within which the organism can grow, 4.6 to 9.6 (Gaudy and Gaudy, 1980:187-188).

The optimum pH for cell growth may not correspond to the enzymatic optima. It should also be noted however, that the changes in the pH of the environment are typically caused by the microorganisms themselves, and unlike the temperature, the pH internal to the cell is not determined by the external pH (Gaudy and Gaudy, 1980:184).

In order to ensure successful biological treatment, the pH of the incoming waste stream may have to be adjusted initially to the neutral range. This may be necessary to prevent any adverse effects on the microbial population in the receiving end.

Oxygen. Another classification for organisms is based on their requirement for molecular oxygen. Organisms that require molecular oxygen are classified as obligate aerobes, and those that cannot grow in the presence of molecular oxygen are obligate anaerobes. Those organisms that can grow in the presence of oxygen but do not require it are falcultative anaerobes. Aerobes use molecular oxygen as an electron acceptor in respiratory metabolism, or aerobic respiration. Some organisms are able to utilize oxidized inorganic compounds such at nitrate or nitrite as an electron acceptor. These organisms are referred to as anoxic.

In the field of biological treatment of wastewater, we are primarily interested in the effects of oxygen concentration on aerobic organisms. The effects of interest are typically those on "respiration rate, biomass yield, cell composition, viability, utilization of soluble and colloidal carbon and energy sources, autodigestion, production of specific enzymes and formation of metabolic products" (Gaudy and Gaudy, 1980:190). Generally, for facultative anaerobes, respiratory activity increases with increased dissolved oxygen (DO) concentrations, up to a critical DO concentration beyond which increased levels of DO do not add to respiration rates. This critical value has been found to be the "DO concentration at which the oxygen uptake rate of the cells is half of the maximum rate recorded for the system when DO is present in abundance" (Gaudy and Gaudy, 1980:191). Although critical DO concentrations seem to range between 0.1

mg/L - 0.5 mg/L, maintenance of 2.0 mg/L is generally recommended. The EPA suggests that available oxygen in wetland systems exceed required oxygen by a factor of two (US EPA, 1988:24). The change in DO concentration can be determined by considering consumption due to the BOD reaction, surface reaeration, photosynthesis and respiration due to algae.

Nutrients. Most living cells are composed of similar types of compounds and therefore require the same elements in relatively the same amounts for maintenance and growth (Gaudy and Gaudy, 1980:195). Four elements comprise the bulk of dry weight of a cell: carbon, oxygen, nitrogen and hydrogen with carbon comprising about 50% of that dry weight. Since the ability of a microorganism to obtain required nutrients from a particular source varies due to the presence or lack of specific enzymes, this determines which organisms wil survive in a given environment. The form in which a nutrient is available is the controlling factor.

The principal nutrients required by cells are nitrogen and phosphorous. "Concentrations of available nitrogen and phosphorus often limit both productivity and decomposition in aquatic habitats" (Atlas and Bartha, 1993:239). Nitrogen is essential for the synthesis of protein. Although nitrogen is available in great abundance as gaseous nitrogen in the atmosphere, few organisms can utilize it in this form and the process referred to as nitrogen fixation, requires a large input of energy. The first detectable product of nitrogen fixation is ammonia. Other common forms of nitrogen are the highly water soluble inorganic nitrogen salts, ammonia, nitrite and nitrate. Ammonia is the most readily utilized form by microorganisms through a process called nitrification. During this process, organisms oxidize ammonia or ammonium ions to nitrite and then to nitrate ions. Both processes produce energy which chemolithotrophs utilize to assimilate CO_2. Both steps of the nitrification are aerobic and obtain required oxygen from the atmosphere and water respectively (Atlas and Bartha, 1993:320).

During denitrification, a type of dissimilatory nitrate reduction, nitrate is converted through nitrite to nitric oxide and nitrous oxide to molecular nitrogen. Organic matter is oxidized throughout this process, which occurs under anaerobic or reduced oxygen tension conditions (Atlas and Bartha, 1993:323). The process of assimilatory nitrate reduction involves the reduction of nitrate and nitrite to ammonia which is then incorporated into amino acids. This can be performed by many bacterial, fungal, and algal species.

Another process which produces ammonia is ammonification. This process takes place when plants, animals and microorganisms convert organic nitrogen to ammonia.

Phosphorous is essential to the growth of algae. Typical forms of phosphorous in aqueous solutions are orthophosphates, polyphosphates and organic phosphates. The orthophosphates are available for biological metabolism without further breakdown.

Kinetics of Microbial Growth. The kinetics of the degradation of BOD are "related to properties of the microbial population and the environmental conditions in the system" (Gaudy and Gaudy, 1980:230). Such environmental conditions were previously discussed and all of these factors can be controlled thereby ensuring the environment is optimal for microbial growth and consequential substrate utilization. The following discussion provides insight into the kinetics of microbial growth and the relationship between growth and substrate utilization.

The general growth pattern of bacteria in terms of variation of mass of the microorganisms with time is characterized by 4 phases: the lag phase during which acclimation to the substrate and environment occurs; the log-growth phase during which there is an excess of substrate and the rate of metabolism and growth is a function of the microorganisms ability to process the substrate; a declining growth phase where the rate of increase in mass decreases due to the limiting supply of substrate, and finally the log-death or endogenous phase where microorganisms are forced to autodigest because of a lack of external carbon source or substrate (Metcalf and Eddy, 1991:368).

The exponential and declining phases of microbial growth are the phases during which substrate removal occurs. The rate of change of concentration of microbial population increases during the exponential phase, then begins to decline until a maximum concentration is reached. This maximum concentration is determined by factors such as nutrient availability, oxygen availability, population density, growth induced changes in the environment and production of toxic substances (Gaudy and Gaudy, 1980:232). In wastewater treatment, it is important to ensure that microbial growth is substrate (carbon) limited rather than limited due to other factors, since the primary mechanism for BOD removal is microbial degradation.

The hyperbolic rate law based on Monod kinetics, is "the most widely used expression for describing the rate of microbial growth as a function of nutrient (substrate) concentration" (Characklis, 1990:238). This rate equation can be used to describe the disappearance of the substrate supporting growth and is a reasonable first approximation for biodegradation in aquatic systems and in soil (Lyman and others, 1982:9-49). The following Monod kinetics rate equation applies to a single-species population of microorganisms on a single carbon substrate, but may also be applicable to mixed-species populations (Lyman and others, 1990:9-49).

$\mu = \dfrac{\mu_{max} S}{K_s + S}$, where μ = specific growth rate, μ_{max} = maximum specific growth rate,

K_s = saturation coefficient, or the concentration of substrate in water supporting a half-maximum growth rate and S = substrate concentration. K_s values range brom 0.1 to 10 mg/L (Lyman, 1990:9-50).

At low substrate concentrations, the growth rate is first order with respect to substrate concentration and may be expressed as follows:

$$\mu = \frac{\mu_{\max}}{K_S} S \quad for \quad S << K_S$$ (Characklis, 1990:239)

This indicates that in systems where low substrate concentrations are typical, an increase in substrate supply will increase its growth rate. In the situation of a constructed wetland receiving stormwater runoff, substrate (BOD) concentrations are typically low enough such that the substrate removal rate is dependant on the substrate concentration, and therefore substrate-limited. An interesting point, according to Characklis, "under substrate-limited conditions, the substrate removal rate is not so sensitive to temperature" (Characklis, 1990:382).

As substrate concentration increases to the point of saturation, the organism cannot assimilate all of the substrate that is available, and the growth rate reaches its maximum. Therefore, at higher concentrations, the growth rate is independant of concentration and expressed as zero order with respect to substrate concentration:

$\mu = \mu_{\max} \quad for \quad S >> K_S$ (Characklis, 1990:239).

The rate of increase in concentration of microorganisms is proportional to the concentration at a given time. This gives the following rate equation to express microbial growth: $\frac{dX}{dt} = \mu X$, where X = microbial population concentration and μ = specific growth rate.

Substituting Monod's expression for growth rate into the rate equation for microbial growth, we get: $\frac{dX}{dt} = \frac{\mu_{max} S}{K_s + S} X$. At low substrate concentrations, we can write: $\frac{dX}{dt} = \frac{\mu_{max}}{K_s} SX$.

The change in microbial population is also affected by a decay rate which decreases the rate of change in the amount of biomass in a system. Decay is due to endogenous metabolism in response to a lack of required energy source for maintenance and is generally considered to be first order with respect to cell concentration (Characklis, 1990:260). The net rate of microbial population accumulation may now be expressed as follows: $\frac{dX}{dt} = \frac{\mu_{max} S}{K_s + S} X - k_d X$ or for low substrate concentration: $\frac{dX}{dt} = \frac{\mu_{max}}{K_s} SX - k_d X$, where k_d = specific decay coefficient.

Substrate utilization can be related to microbial growth by the use of yield coefficient which describes the conversion efficiency of substrate to microorganism mass. Not all substrate metabolized by cells is used to generate more microbial mass. A fraction of the substrate is used for cell maintenance. The value of the cell yield coefficient depends on the nature of the substrate, the microorganism, and environmental conditions. Gaudy and Gaudy show the results of an experiment in which a mixed microbial population exhibited a constant cell yield throughout the substrate removal phase. This allows us to write:

$Y = \frac{\frac{dX}{dt}}{\frac{dS}{dt}}$ where: Y = cell yield, $\frac{dX}{dt}$ = rate of microbial population increase, $\frac{dS}{dt}$ = rate of substrate utilization (Gaudy and Gaudy, 1980:237).

Substrate utilization can now be written in terms of the rate of change of concentration of microbial population and the expected cell yield:

$$\frac{dS}{dt} = \frac{\mu_{max} S}{K_S + S} \frac{X}{Y} \text{ and at low concentrations: } \frac{dS}{dt} = \frac{\mu_{max}}{K_S} \frac{X}{Y} S.$$

The previous expressions for microbial growth and substrate utilization require that the characteristics of both the substrate and microbial population be well defined. The characteristics such as the saturation coefficient, cell yield and maximum growth rate are very specific not only to the microbial population and substrate alone, but specific to the combination of the two. That is, a microbial population may have a specific saturation coefficient when exposed to a particular substrate and a very different saturation coefficient in the presence of yet another substrate. The aquatic environment must also be well characterized in order to determine if the environment is in fact substrate limited. The population may be nutrient limited rather than substrate limited. Or perhaps the natural environment generates sufficient organic matter that maximum growth rates are naturally achieved and maintained such that added amounts of organic matter (BOD) do not significantly affect the growth rate and subsequently the degradation rate. These uncertainties must be investigated further and defined for the wetland environment and substrate source (stormwater runoff).

Biofilms

Microbial populations can exist as suspended biomass within an aquatic system, or they may attach to a surface submerged in an aquatic environment on which they form a biofilm. A biofilm consists of microorganisms immobilized at a support surface or substratum. A system such as a constructed wetland has both a suspended microbial population, and a population existing as biofilm attached to the surfaces of emergent macrpophytes and the sediment surface of the wetland.

As a first approximate, Kawashima and Suzuki in their attempt to develop a model for prediction of BOD removal in streams, assume that biomass grows only on the surface of the river bed and stems of the aqueous plants. They do not consider the affects of suspended biomass (Kawashima, 1989:1004). Brix states that "many of the treatment processes (of aquatic macrophyte-based wastewater treatment systems) are attributed to microorganisms living on and around the macrophyte" and attributes most of the degradation of soluble organic compounds to aerobic degradation by bacteria attached to plant and sediment surfaces (Moshiri, 1994:11). The US EPA also attributes the removal of soluble BOD in FWS weltands to "microbial growth attached to plant roots, stems, and leaf litter that has fallen into the water" (US EPA, 1988:18).

The biofilm system can be thought of as consisting of 5 compartments: (1) the substratum; (2) the base film; (3) the surface film: (4) the bulk liquid; (5) gas. The substratum is the media to which the biofilm attaches. The base film and surface film are commonly referred to as the biofilm. This biofilm compartment contains at least two phases: a continuous liquid phase which contains dissolved and suspended particulate materials, and solids consisting of microorganisms, extracellular material or inorganic particles etc. (Characklis, 1990:6-7). The bulk liquid is typically the water environment that surrounds the biofilm. The gas is the dissolved oxygen that exists in the bulk liquid and the biofilm.

The surface film provides a transition between the bulk liquid compartment and the base film. Some studies also consider the existence of a stagnant liquid film layer that covers the surface of the biofilm (Kawashima and Suzuki, 1989:1004; Polprasert and Agarwalla, 1994:726). This liquid film layer acts as a link between the bulk liquid and the biofilm by transporting substrate to the biofilm through molecular diffusion. Mass transport from the bulk liquid compartment to the biofilm may also control the rate of growth of the biofilm organisms if the transport rate, or diffusion, of substrate into the biofilm is slower than the uptake rate by the microorganisms.

In a constructed wetland system, a biofilm forms on the sediment surface and surfaces of emergent macrophytes. The substrate, BOD, is transported from the bulk liquid, across a stagnant liquid film layer, and into the biofilm by diffusion. In order to determine the rate at which substrate (BOD) is removed from the bulk liquid or surface water compartment, mass transport processes from the surface water to the biofilm must be understood. The rate of mass transport is determined by the concentration gradient between the surface water and the biofilm, the substrate diffusion coefficient in the biofilm, and a first order decay rate constant within the biofilm as well as the thicknesses of the stagnant liquid film and biofilm. The expressions used to determine the rate of mass transport and associated assumptions is presented in greater detail in the following chapter.

Zhang and Bishop studied the structure, activity and composition of biofilms and found that these properties change with depth of the biofilm. They studied the spatial distribution of biofilm properties such as bacterial population, density, living biomass, activity, and porosity. They found that "densities of biofilms in the bottom layers were 4 to 7 times higher than those in the top layers" (Zhang and Bishop, 1994:335). Their conclusions were that due to these spatial distributions, the ratio of effective diffusivity in the biofilm to diffusivity in the bulk liquid also changed with depth. Table 2.10 consists of some of the data obtained by Zhang and Bishop. Based on Zhang and Bishop's findings that "the porosities of biofilms changed from 83-92% in the top layers to 57-64% in the bottom layers for the thicker biofilms, while for thin biofilms, it changed from 70-75% in the top layers to 35-44% in the bottom layers" (Zhang and Bishop, 1994:341), it can be inferred that biofilm 1 in the below table is representative of a thick biofilm, and biofilm 2 a thin biofilm. Zhang and Bishop define a thick biofilm as greater than 500 micrometers and a thin biofilm less than 500 micrometers in thickness.

Table 2.11

Spatial Distributions of the Properties of Biofilms

	Layer	Normalized Distance	Living Biomass (mg/cm^3)	Porosity (%)	D_e/D_b (%)
Biofilm 1	1	0.39	11.8	87.8	71.5
	2	0.56	27.0	72.2	50.8
	3	0.74	37.7	60.5	39.0
	4	1.00	37.1	57.7	36.5
Biofilm 2	1	0.31	27.5	72.0	50.6
	2	0.54	47.0	51.1	31.0
	3	0.77	58.5	37.4	20.8
	4	1.00	56.7	37.5	23.7

D_e/D_b = The ratio of effective diffusivity to diffusivity in the bulk liquid

(Adapted from Zhang & Bishop, 1994:341)

Suspended Biomass

Polprasert and Agarwalla developed a model which considers both substrate consumption in biofilm as well as degradation due to suspended biomass in the water column. Results of their studies of facultative ponds show 47% removal rate by suspended biomass alone and a total of 65% removal by the combined effect of suspended and biofilm biomass (Polprasert and Agarwalla, 1994:729). This indicates that the effects of suspended biomass should not be ignored.

Conclusions

Stormwater runoff can contain significant concentrations of several pollutants that may have an adverse effect on the surrounding environment. The characteristics of stormwater generated on AF installations may be similar to that generated in urban environments or industrial areas. Previous studies of AF stormwater characteristics show that concentrations of BOD, a pollutant also identified by the NURP study, often exceed EPA benchmarks.

A review of possible stormwater best management practices identifies wetlands as a viable option, particularly for the degradation of BOD. Constructed wetlands are currently widely used to treat municipal wastewater. Both the FWS and SSF systems have been shown to successfully lower concentrations of BOD, though both have advantages and disadvantages. Free water surface wetlands are more common within the United States, and more data is available regarding the effectiveness of that type of wetland. They are also typically less costly to construct.

A successful wetland system constructed for the purpose of stormwater management may include a form of preliminary treatment, such as a wet detention pond, for the purpose of removing large amounts of suspended solids that may adversely affect the livelihood and efficiency of the wetland. Other physical characteristics such as a liner, water control structures, vegetation, surface area, water depth and length to width ratio are important in determining the wetlands efficiency in lowering levels of BOD.

Microbial populations in the form of suspended biomass and biofilm on the submerged surface of vegetation and soil surface are the primary factor affecting degradation of BOD. Several environmental factors influence the microbial growth rate and substrate consumption rate, temperature being possibly the most influential factor. Dissolved oxygen concentration is also of concern as respiratory activity changes due to available dissolved oxygen. A minimum concentration of 2 mg/liter DO should be maintained in order to ensure aerobic respiration can be accomplished.

Monod kinetics can be used to describe both the growth rate and substrate utilization rate associated with a specific microbial population and substrate, assuming substrate limited conditions. This requires that the population and substrate be well defined so that characteristics such as saturation coefficient, cell yield and maximum growth rate can be accurately determined.

A system dynamics model that provides the capability to vary above mentioned parameters will aid in identifying those that contribute the most to BOD degradation.

III. Methodology

Introduction

This chapter describes the methodology used to develop a system dynamics model representitive of significant processes within a constructed wetland responsible for the degradation of BOD within the system. The software package, STELLA II, from High Performance Systems Inc. is the tool used to implement the model. System dynamics modelling is based on a framework which allows us to focus on the details of processes that are a part of the larger system, as well as the behavior of the system as a whole. The principal building blocks are stocks or accumulations, and flows or rates of movement to and from a stock. The primary stock of interest in this model is the stock of BOD in the constructed wetland system. The flows into and out of the stock are defined by the processes within the system that physically transport the BOD in a single phase, or transform the BOD by generation or decay.

Physical parameters of a hypothetical constructed wetland system are first provided along with rationale for their selection. The hydrologic functions of the constructed wetland are then described since hydrology is an important factor in modeling the functions and performance of the wetland. Next, processes that may be significant to the degradation of BOD within the system are defined, analyzed, and expressed mathematically. Once several processes are included in the model, physical parameters of the system may be varied to examine their affect on the BOD degradation capabilities of the system. The significance of each process may also be determined during the exercise of parameter variation. Results of parameter variation and conclusions drawn from these excercises will be presented in the next chapter.

Assumptions and simplifications made throughout development of the model are included where necessary.

<u>Constructed Wetland System Physical Description</u>

As previously stated in Chapter 2, both FWS and VSB systems require some level of pretreatment for the purpose of removing the bulk of larger suspended solids prior to entering the wetland system. The system modeled here is a FWS system preceded by a wet detention pond. The wet detention pond serves two purposes: the removal of suspended solids and regulation of flow rate into the constructed wetland. A detailed procedure for design of an appropriate detention pond is not within the scope of this research, however a general guideline for estimating size of the pond is provided as follows.

Detention Pond. The detention pond must have sufficient volume to handle peak flows during and following a single storm event or successive storm events. Several methods exist to calculate the required volume of a detention basin (Stahre and Urbonas, 1990:233). A simplified method which utilizes the Rational Formula is used to estimate an appropriate detention pond size. The Rational Formula is based on a block rainstorm, the simplest form of standardized design storm, which assumes a constant intensity during the rain event. The Rational Formula gives the runoff volume due to a single rain event as follows: $V = T \times C \times I \times A$, where V = runoff volume (ft^3), T = storm duration (sec), C = runoff coefficient (nondimensional), I = average storm intensity for storm duration T (in/hr), A = area of the watershed (acres) (Stahre and Urbonas, 1990:234). Modifying the equation to give the volume in cubic meters gives: $V = 4.013 \times T \times C \times I \times A$, where T is in hours, I is in mm/hour, and A in acres.

The runoff coefficient takes into account the permeability of the watershed surface, and is influenced by such variables as infiltration, ground slope, ground cover, evapotranspiration etc. (Praner and Sprewell, 1992:60). Runoff coefficients range from values as low as 0.05 characteristic of densely vegitated areas, to 0.95 characteristic of asphaltic and concrete surfaces (Praner and Sprewell, 1992:63). Average or overall runoff coefficients for various types of areas as well as those for specific surface types are provided in Appendix A. Praner and Sprewell recommend that a composite basewide runoff coefficient be determined by integration of the values for area type and specific surface type. Haan also notes that a composite runoff coefficient based on percentage of different types of surface in the drainage area be determined (Haan and others, 1994:84). For the purpose of this study, the watershed of concern is assumed a heavy industrial area, which may be representative of an Air Force installation with an airfield. The runoff coefficient, C, used throughout this model is therefore 0.8.

One method for determining the required detention pond volume and dimensions involves calculating the runoff volume for a range of storm durations and plotting the runoff volumes for each duration creating a runoff envelope curve. A line representing the desired release rate from the storage basin is superimposed on the curve and the desired volume can then be determined from the maximum difference between the runoff envelope and the release discharge line as shown in Figure 3.1. (Stahre and others, 1990:234-235). Due to the inaccuracies associated with this simplified method, the Denver Regional Council of Governments recommends that "the Rational Method and all procedures related to it be limited to watersheds having an area that is less than 160 acres" (Stahre and others, 1990:235). This is only one of several methods available for determining an appropriate detention pond size. Again, detailed design of a detention pond is not the purpose of this research and it is assumed that an adequate detention pond is designed that will perform to desired specifications regarding capacity and regulation of flow into the wetland.

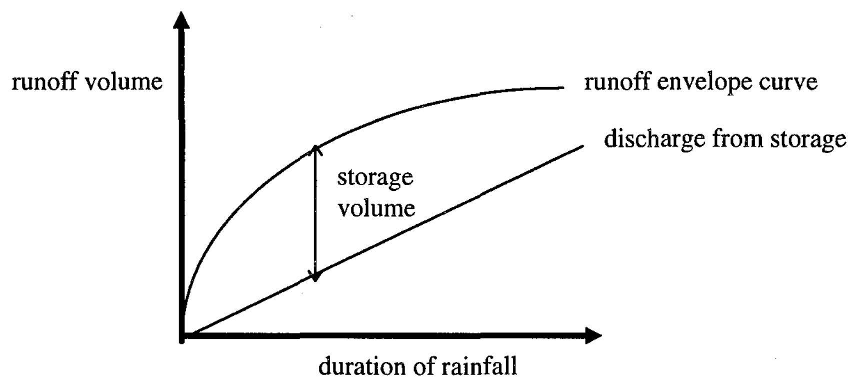


Figure 3.1

Determining Detention Pond Storage Volume
Using the Runoff Envelope Diagram
(Stahre and Urbonas, 1990:235)

Stahre and Urbonas suggest that a surcharge volume (volume above the permanent pool) equal to approximately 13 mm of runoff from the impervious surfaces will capture about 80% to 90% of all runoff events in many cities in the United States (Urbonas and Stahre, 1993:385). An additional 25% will ensure adequate volume is available for sediment accumulation. In order to prevent resuspension of sediment particles, the depth of the permanent pool will be maintained at a minimum of 1.3 meters (Stahre and Urbonas, 1990:296), with a maximum height of 2.4 meters. The length to width ratio will be designed to 3:1 (Stahre and Urbonas, 1990:330).

Using Urbonas' assumption of 13 mm of runoff from an impervious surface area of concern covering 125 acres, the following calculations are made to determine the surcharge volume required. A factor of 1.25 is used to ensure that capacity for sediment accumulation is included.

$V = 4.013 \times 13 \text{ mm} \times 125 \text{ acres} \times 1.25 = 8151.4 \text{ m}^3$

The following calculations can be made to determine the appropriate length, width and area to accomodate the detention pond surcharge volume with a maximum heigth of 1.1 meters, based on a permanent pool heigth of 1.3 meters, and a total maximum detention pond heigth of 2.4 meters:

$$V = 3xWxWx1.1$$

$$W = \sqrt{\frac{V}{3.3}} = 49.7m$$

$$L = 3xW = 149.1m$$

where: V = surcharge volume (m^3), W = detention pond width (m), L = detention pond length (m).

Using these dimensions and assuming that a minimum depth of 1.3 meters is required for the permanent pool, a minimum detention pond volume is calculated:

$$V_{min} = 49.7x149.1x1.3 = 9633.35$$

V_{min} = minimum detention pond volume (permanent pool volume) (m^3)

In the model, the detention pond maximum volume is defined as the permanent pool volume plus the volume of stormwater as previously calculated by the Rational Formula, from which detention pond surface area is calculated. The minimum detention pond volume is used to determine when flow into the wetland will cease. Runoff is released from the detention pond at a predefined, constant rate, until the minimum detention pond volume is reached.

Constructed Wetland. The model is constructed such that the wetland physical parameters such as surface area, surface water depth, length to width ratio, soil porosity, and surface water porosity can be defined in order to determine the effects of varying those parameters on the BOD removal efficiency of the wetland. Appendix B contains the initial or baseline constructed wetland physical parameters utilized in this model, based on the design guidelines provided in Chapter 2, for a watershed area of 125 acres. These values will be varied within the range specified in order to elucidate those physical parameters that have the greatest affect on the ability of the wetland system to degrade BOD.

For the purposes of this model, the bed depth is assumed a constant .5 m. This parameter will not change throughout simulations run using this model, as bed media or depth is not considered a factor affecting the degradation processes. The porosity is also assumed constant and assigned a value of .45 which is expected of appropriate bed media as described in Chapter 2.

Defining System Hydrology

The fundamentals of system hydrology were briefly discussed in Chapter 2. Since most substances in water are measured in terms of concentration, or mass per unit volume of water, we must know the volume of water in the constructed wetland system at any given time. The volume of water is determined by the important hydrologic processes in a given wetland system. The inputs and outputs of wetlands may differ. Some may only be affected by precipitation and evapotranspiration, while others may be fed by ground water or streams etc. The system under consideration here is affected only by inflow due to stormwater runoff, outflow, precipitation and evapotranspiration. The hydrology of the system is defined by the mass balance equation (MBE), which is applied to both the detention pond and the constructed wetland. The mass balance equation is: $\frac{dV}{dt} = Q_i - Q_o + P - ET$, where V = volume of water, t = time, Q_i = flow rate in (volume/time), Q_o = flow rate out (volume/time), P = precipitation rate (volume/time), ET = evapotranspiration rate (volume/time).

Detention Pond. The design volume of the detention pond was previously determined. The dimensions of the DP were determined based on desired L:W ratio and design depth to prevent resuspension of sediment.

The flow rate into the detention pond is determined by the Rational Equation as previously described. The model includes provisions for defining the watershed area of concern, runoff coefficient, and storm intensity. The model allows input of a single storm event, or succesive storm events, at varying intensities over time. The evapotranspiration and precipitation rates are included in the model as a function of DP surface area.

The flow rate of water leaving the DP is determined by a water control structure and may be defined as desired to obtain optimal results.

Constructed Wetland. The constructed wetland initial volume must first be defined, as well as its surface area, L:W ratio, bed depth, surface water depth, soil porosity, and porosity due to vegetation. These parameters determine the volumes of water in each compartment of the system. The constructed wetland hydrologic system can be thought of as divided into two compartments, the subsurface compartment consisting of soil and ground water, and the surface compartment consisting of surface water and vegetation.

The volume of groundwater is assumed constant at maximum capacity since soil porosity is assumed constant and it is also assumed that sufficient water will always be present to fill the subsurface compartment. This volume is calculated as follows: $V_{gw} = A_{cw} \times d_s \times n_s$, where A_{cw} = surface area of the wetland, d_s = depth of the bed, n_s = porosity of the soil, or the ratio of volume of voids to total volume in the subsurface compartment.

The surface water volume changes with time. It's intitial value is determined by the predefined surface area, initial water depth, and porosity due to the presence of vegetation. $V_{sw} = A \times d_{sw} \times n_p$, where V_{sw} = volume of water in the surface water compartment, A = surface area of the constructed wetland, d_{sw} = depth of surface water, n_p = porosity of the system due to existence of vegetation.

The system porosity (vegetative) of a FWS system at Arcata, California and that of a system at Listowel, Ontario were both determined to be 0.75 on the basis of dye study data. Reed suggests that this may be a valid assumption for the general case with a moderately dense stand of emergent vegetation (Reed and others, 1988:180).

The model calculates surface water volume at any time based on the total constructed wetland water volume and ground water volume. The total constructed wetland volume at any time is determined by the hydrologic processes defined in the Hydrology sector of the model.

The flow rate into the constructed wetland is equal to the flow rate out of the detention pond. This flow rate can be controlled to obtain the optimum hydraulic residence time. The model is run at various influent rates to show the effect flow rate has on the efficiency of the constructed wetland. Evapotranspiration and precipitation rates are obtained in the same manner as determined for the detention pond.

The effluent rate from the constructed wetland is determined from the mass balance equation (MBE). The effluent rate is controlled by a water control structure so as to theoretically maintain a constant water volume within the wetland. This does not occur in reality however due to evapotranspiration. Assuming that the system operates at a relatively constant water depth ($dV/dt = 0$) (EPA, 1988:16), the MBE becomes $Q_o = Q_i + P - ET$.

Other quantities or values of interest can be determined from the above hydrologic information. These are calculated by the model and include:

Hydraulic retention time (HRT): the time in days or hours that a volume of water remains in the constructed wetland before it exits with the effluent. This is determined by: $HRT = \frac{2V_{sw}}{Q_i + Q_o}$

Hydraulic loading rate: The volume of water received per area of the constructed wetland per time: $HLR = \frac{Q_i + Q_o}{2A_{cw}}$

Surface water depth: The depth of surface water defines the surface area on the vegetation available for microbial population. Surface water depth is determined by total water volume above the bed divided by the constructed wetland surface area:

$$d_{sw} = \frac{V_{sw} + V_p}{A_{cw}}$$

Vegetation volume and surface area: The vegetation volume is defined by the volume above the bed media and the plant porosity: $V_p = \frac{V_{sw}}{n_p} - V_{sw}$

The total plant surface area is determined by the total plant volume, the surface water depth, and an assumed plant stalk diameter = .14 cm (US EPA, 1988:19). From the surface water depth and assumed diameter of each plant stalk, the surface area per plant and volume per plant is calculated. The approximate number of plants can be determined by dividing the total plant volume previously calculated by volume per plant. The surface area per plant is then multiplied by the number of plants to obtain total plant surface area. All calculations reduce to the following expression for total plant surface area: $A_p = 2\frac{V_p}{r}$, where r = radius of plant stalk.

BOD Transport

Kadlec & Kadlec propose that interpreting data on any water related substance is based on "the concept of mass balance for that substance in some system for some interval of time" (Greeson and others, 1979:437). That is, a mass balance equation can be used to express the accumulation of that substance in terms of the sum of transfers and transformation rates within the system. One way of expressing the conservation of mass equation, or MBE is as follows:

$$\frac{d(Mass\,Storage)}{dt} = \frac{(Mass\,In - Mass\,Out)}{dt} + rate\,of\;generation - rate\,of\;decay$$

This MBE can be applied to the mass of BOD in the constructed wetland system where the change in mass storage with time is equal to the change in mass with time of BOD in all compartments of the CW system. This is equal to the sum of change in mass flux of BOD across the boundaries of each compartment, and any generation or decay terms in each compartment. This same MBE may be applied to each compartment within the system.

The constructed wetland system, for the purpose of this project, is assumed to consist of two compartments containing a mass of BOD at any time. These BOD storage compartments are: (1) the surface water, (2) a biofilm covering the surface area of plant matter and the soil surface. BOD is transported between each compartment, and decay and/or generation terms are associated with each compartment. The mass balance equation will be applied to each compartment. Figure 3.2 gives a visual representation of the BOD transport between compartments, as well as any generation or decay associated with a compartment.

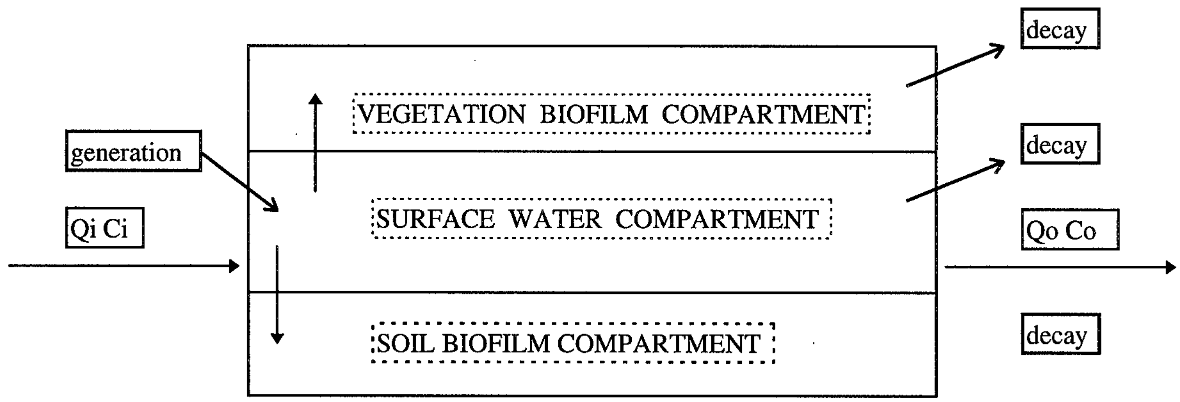


Figure 3.2

BOD Transport Between Surface Water
and Biofilm Compartments

Each compartment must now be analyzed and mechanisms for BOD mass transport in and out of each compartment identified so that the quantity of mass in each compartment can be determined at any time.

Surface Water Compartment. The MBE as applied to the surface water compartment is as follows:

$$\frac{d}{dt}(V_{sw}C_{sw}) = Q_{in}C_{in} - Q_{out}C_{out} - J_b A_b + rate\ of\ generation - rate\ of\ decay$$

where V_{sw} = volume of surface water compartment, C_{sw} = concentration of BOD in the surface water, $Q_{in/out}$ = volumetric flow rate of water entering/leaving the surface water compartment, J_b = mass flux of BOD across the biofilm surface, A_b = the total surface area of biofilm, across which BOD is transported.

The surface water is the only compartment that receives a mass of BOD directly from the constructed wetland influent and loses BOD mass in the constructed wetland effluent. The mass of BOD in the influent is determined based on the assumption that the concentration of BOD in the influent is measured and known. Since the flow rate is regulated by a water control structure, the mass of BOD in the influent can be calculated.

Input of dissolved organic matter into the system due to macrophyte and algae generation is determined by the above ground net primary production values found in Table 2.5. For the purposes of this research, 1/2 the surface area of the wetland is assumed populated by cattail, and 1/2 by bulrush. We can determine the quantity of BOD contributed to the surface water compartment by calculating 35% of the average of the above ground NPP for each the cattail and reed communities applied to1/2 the surface area of the wetland, and converting to appropriate units giving an hourly input to the system.

A rough estimate of water column primary producer release of dissolved organic matter into the water column can be made by assuming that an average of 40% of the wetland net primary productivity is due to water column primary production, the rest by macrophyte, and that 10% of that value is released as dissolved organic matter. This gives the rate of water column primary producer NPP equal to .67 of macrophyte NPP.

BOD is also returned to the surface water when the biofilm becomes so thick that the "adsorbed organic matter is metabolized before it can reach the microorganisms near the media face" (Metcalf and Eddy,1991:404). The microorganisms that do not receive any external organic matter must resort to autodigestion, causing them to lose the ability to cling to the media surface. The shear velocity caused by flow of liquid across the biofilm washes off the biofilm, and a new film begins to grow. The process of losing the biofilm in this manner is termed sloughing, and is a function of the organic and hydraulic loading on the system (Metcalf and Eddy, 1991:404). As with most attached growth biological reactors however, it is assumed that a constant biofilm thickness is maintained in the system. This alleviates the complication of quantifying the change in biofilm thickness in the mass flux equation. At this time, it is assumed that the amount of dissolved organic matter contributed to the surface water concentration by sloughing is negligable.

A mass of BOD leaves the surface water compartment with the effluent. The concentration of BOD within the system is determined at any time by the model, assuming a completely mixed system. The effluent rate is determined assuming that a water control structure can regulate the rate of flow leaving the system to maintain a fairly constant volume. That is, the hydrologic mass balance equation is such that change in volume with time equals zero, and given rate of influent, precipitation, and evapotranspiration, the rate of effluent is determined. The surface water BOD concentration and rate of flow of the effluent give the mass of BOD leaving the system.

BOD is transported between the surface water and both the aerobic biofilm attached to plant matter and that covering the surface of the soil, by diffusion. The pathways for BOD transport to and from the surface water compartment are therefore considered due only to influent, effluent, and diffusion.

Therefore, this model includes degradation due to both suspended and biofilm biomass.

Substrate Consumption Due to Suspended Biomass. Substrate consumption due to suspended biomass is often described by first order kinetics and is given by: $r_s = k_{fs}C_{w,}$ where: r_s = the rate of substrate removal/degradation (mg/liter-hr), k_{fs} = the first order rate constant for suspended biomass (hr^{-1}), C_w = concentration of substrate in the bulk liquid (mg/liter).

In order to determine the suspended biomass first order rate constant for falcultative ponds, Polprasert and Argawalla use Thirumurthi's method which takes an assumed standard environmental condition, and makes corrections to the rate based on given field conditions. In the absence of toxic industrial wastes, the design BOD rate constant is determined as follows:

$$k_{fs} = k_{fss}\{1\text{-}0.083[\log(67.2/L_0)/k_{fss}\} \quad (1)$$

where:

k_{fs} = the first order rate constant at 20 degrees C

k_{fss} = 0.056 day^{-1} ,the standard first-order rate constant based on assumed standard environmental conditions (Thirumurthi, 1974:2095).

L_0 = the organic loading rate in kg/ha-day

A correction must also be made for temperature:

$$(k_{fs})_T = (k_{fs})_{20}\theta^{T-20} \quad (2)$$

where:

$(k_{fs})_T$ = the rate constant at the water temperature (T, degrees C) of interest

$(k_{fs})_{20}$ = the rate constant at 20 degrees C

θ = 1.1

Calculations show that in order for the rate constant to be greater than zero, the organic loading rate must be equal to or greater than .59 kg/ha-hr. The model is then written such that at total organic loading rates less than .59 kg/ha-hr, the first order rate constant for suspended biomass is equal to zero.

BOD Transport into the Biofilm. As previously discussed, constructed wetland systems are compared with attached growth biological reactors such as trickling filters and rotating biological contactors. These types of systems depend on degradation of BOD in biological films, or biofilms, that form on the surface of the system media. The rate of flux of organic material into the biofilm is the rate of removal of BOD from the surface water compartment.

Kawashima and Suzuki developed a model to determine the mass flux of BOD from surrounding water to the biofilm covering the surface of plants in streams (Kawashima, 1989:1003 -1014). This model is based on a double layer of aerobic and anaerobic bacterial films of different thickness, and a stagnant liquid film which surrounds the surface of the biofilm. BOD is transported from the water across the stagnant liquid film to the biofilm surface by molecular diffusion. The flux of BOD across the stagnant liquid film is assumed constant and described according to Fick's Law as follows: $J_{BOD} = \frac{D}{L_1}(C_{sw} - C_p)$, where: J_{bod} = the mass flux of BOD across the stagnant liquid film, D = the molecular diffusivity of the liquid film, L_1 = the thickness of the stagnant liquid film layer, C_{sw} = BOD concentration in the surface water, C_p = BOD concentration at the liquid/biofilm interface.

Polprasert and Agarwalla provide a conceptual illustration of a similar biofilm model in a facultative pond as shown in Figure 3.3.

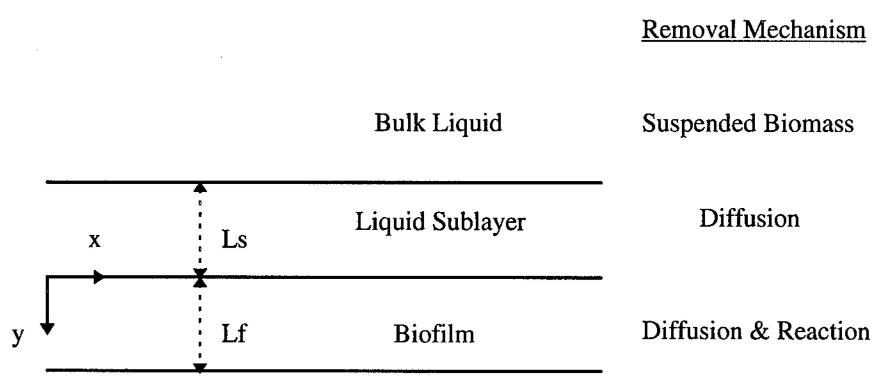


Figure 3.3

Conceptual Illustration of a Biofilm Model

Polprasert and Agarwalla use the following equation given by Lau to determine the flux of BOD at the liquid sublayer/biofilm interface:

$$J\big|_{y=0} = \frac{\tanh(\phi)}{\phi} k_{fa} L_f C_s \tag{3}$$

where:

$J\big|_{y=0}$ = Flux at the liquid sublayer/biofilm surface

$\phi = \sqrt{\dfrac{k_{fa} L_f^{\,2}}{D_f}}$, a characteristic biofilm parameter

k_{fa} = biofilm first order rate constant

L_f = biofilm thickness

$C_s = \dfrac{\alpha}{\alpha + \beta} C_w$, substrate concentration at the liquid sublayer/biofilm interface

$\alpha = \dfrac{D_w}{L_s}$

$\beta = \frac{\tanh(\phi)}{\phi} k_{fa} L_f$

D_f = diffusion coefficient in the biofilm

D_w = diffusion coefficient in water (Polprasert and Agarwalla, 1994:726)

Polprasert and Agarwalla determine the value of k_{fa} in laboratory batch experiments by taking samples of biomass growing on PVC sheets in a facultative pond (Polprasert and Agarwalla, 1994:727). For the purposes of this model, the value of k_{fa} that they determined at 20 degrees C is used, with a correction made for desired temperature.

The thickness of the stagnant liquid film layer is a function of the energy dissipation rate as given by the following equation:

$$L_s = \left(\frac{\varepsilon}{\nu^3}\right)^{-.25} \tag{4}$$

where:

ε = energy dissipation rate

ν = kinematic viscosity of water (Kawashima, 1989:1004)

The energy dissipation rate is related to the velocity of water in the the wetland, and can be determined given the rate of flow of bulk liquid through the wetland. When there is no flow, the energy dissipation rate is determined by wind, and for the purposes of this model, is taken to be a typical value observed in a lake. Kawashima provides a range of values observed in lakes. The value of 10^{-4} m^2/s^3 is used as the no flow condition in this model (Kawashima, 1989:1004).

The thickness of the aerobic biofilm layer is determined using the following equation given by Kawashima and Suzuki:

$$L_f = \frac{D^* L_s}{D} + \sqrt{\frac{D^* L_s}{D}^2 + \frac{2 D^* C_0}{k_{ae}}} \tag{5}$$

where:

L_f = thickness of the aerobic biofilm

L_s = thickness of the stagnant liquid film

D = molecular diffusivity of the BOD in the stagnant liquid film

D* = molecular diffusivity of the BOD in the biofilm

k_{ae} = BOD consumption rate in aerobic biofilm

C_0 = the dissolved oxygen concentration in the liquid

Note: the k_{ae} term is assumed constant in this equation, as is the dissolved oxygen concentration, in order to simplify calculation of the biofilm thickness. The value of k_{ae} chosen is that used by Kawashima (Kawashima, 1989:1007).

The diffusivity of BOD in the liquid film is its diffusivity in water. The effective diffusion coefficient of BOD in water is determined by the following equation:

$$D_w = \frac{13.26 x 10^{-5}}{\mu^{1.14}\ V^{0.589}} \tag{6}$$

where:

D_w = the diffusion coefficient of the organic molecule in water

μ = the solution viscosity in centipoise (10^{-2} cm^{-1} s^{-1}) at the temperature of interest

V = the molar volume of the chemical (cm^3 mol^{-1}) (Schwarzenbach, 1993:199)

Since the concentration of the chemical in the water is relatively small and unlikely to greatly affect the viscosity of the water, the viscosity of water is used in place of solution viscosity. The viscosity of water at 18 degrees Celsius is $1.053x10^{-2}$ g-cm/s (Weast, 1982:F-40).

Kawashima assumes a value of $2x10^{-9}$ m^2/s (temperature not defined) while Polprasert and Agarwalla assume a value of $6.1x10^{-10}$ m^2/s for the diffusivity of a BOD molecule in water at 20 degrees C. For the purposes of this model, the diffusivity of a toluene molecule in water at 18 degrees C is determined using the expression given above. That value is $7.06x10^{-10}$ m^2/s ($2.54x10^{-6}$ m^2/hr), assuming the molar volume of toluene is 131.5 cm^3/mol which is determined by summing the size of the atoms that make up toluene (Schwarzenbach, 1993:198).

The effective molecular diffusivity of BOD in a biofilm is different than diffusivity into a bulk liquid as it accounts for the porosity and tortuosity inherent in biofilm structure. Zhang and Bishop state that the ratio of effective diffusivity in a biofilm to the diffusivity in bulk liquid can be calculated from the porosity of the biofilm (Zhang, 1994:2285). They studied the structure of biofilms that were cultured by laboratory-scale rotating drum biofilm reactors and provide data such as porosities and the associated ratio of bulk to biofilm diffusivity. Preliminary calculations of the thickness of the biofilm in the constructed wetland system defined for this project indicate a thin biofilm as defined by Zhang and Bishop, less than 500 micrometers. The following data taken from Zhang and Bishop's study on the structure of biofilms may be representative of a thin biofilm, and is used in this model to define the biofilm physical properties.

Table 3.1

Spatial Distributions of the Properties of a Thin Biofilm

Layer	Normalized Distance w/in Biofilm	Living Biomass mg/cm^3	Porosity %	D_e/D_b %
1	0.31	27.5	72.0	50.6
2	0.54	47.0	51.1	31.0
3	0.77	58.5	37.4	20.8
4	1.0	56.7	37.5	23.7

Note: For normalized distance, 0 represents biofilm surface, 1.0 represents interface with substratum. D_e/D_b is the ratio of effective diffusivity within the biofilm, to the diffusivity in the bulk liquid.

(Adapted from Zhang and Bishop, 1994:341)

Zhang and Bishop's data show that physical properties of biofilms change with depth and may be considered stratified with respect to density, porosity and even metabolic activity level. To summarize their findings, they showed that densities of biofilms increased with depth, porosity decreased with depth, and that only a fraction of living biomass is metabolically active at bottom layers, whereas almost all bacteria in top layers may be active. This information led them to state that the assumption of a uniform distribution of biofilm properties may be "an over-simplified assumption, valid only in specific cases" (Zhang and Bishop, 1994:335).

For the purposes of this preliminary model, the simplifying assumption of a uniform distribution of physical properties within the biofilm is made. This allows us to model diffusion into only one biofilm compartment. The data given above is used, and diffusivity ratio values are averaged proportional to each layers thickness. A diffusivity ratio of .3305 is used in this model to compute the effective diffusivity.

Now that all terms in the biofilm flux equation are defined, the total mass of BOD removed from the surface water can be determined by multiplying the flux by the total biofilm surface area. This surface area is equal to the sum of the surface area of the submerged portion of vegetation stalks and the surface area of the wetland soil. The surface area of the vegetation is estimated using the defined porosity due to vegetation, current surface water volume, and assumed average diameter of vegetation stalks equal to 14 mm (US EPA, 1988:19).

<u>Summary</u>

The model developed using STELLA II software simulates the relevant processes within a wetland system that affect the degradation of BOD. The hydrology of the wetland is first defined and simulated as the hydrologic processes are the basis of a viable system. The microbial populations in the form of suspended biomass and biofilm on available surfaces are thought to be the primary factor responsible for the degradation of BOD within the system. Estimates of degradation rates associated with the two forms of microbial population are made based on previous research and incorporated in the model.

The model sector entitled "BOD Concentration in Surface Water and Biofilm Compartments" describes the accumulation of BOD in each compartment due to the mechanisms described previously. BOD enters the system in the stormwater influent, and by macrophyte release of DOM. BOD leaves the system in the effluent, through diffusion and degradation in the biofilm and by suspended biomass degradation.

Appendix C contains the STELLA II model which incorporates all of the above processes and associated calculations in order to determine the BOD concentration in the surface water at any given time. The first part of the Appendix contains the equations associated with each sector along with the documentation which provides an explanation of each equation to include units. The second part of the Appendix shows the model construction layer which is organized by sectors such as Hydrology, Constructed Wetland Parameter Calculations, Biofilm Flux Calculations etc..

IV. Results and Analysis

Introduction

This chapter presents some of the data resulting from several runs of the model, as well as a qualitative analysis of the data. Due to the complexity of a wetland system and widely varying conditions that may be imposed on the system, several parameters and combinations of parameters can be investigated, however, only a few scenarios are investigated at this time, and are described below.

The physical parameters of a baseline constructed wetland (see Appendix B) are defined and used for several of the initial runs. The behavior of the system hydrology is first presented to gain an understanding of the water budget and effects of evapotranspiration on the budget during dry periods as well as periods following a single storm event. Next, the effect of varying the dissolved oxygen concentration in the surface water is investigated to determine its effect on the rate of BOD removal by the biofilm. The effects of varying influent rate and influent BOD concentration for four wetlands with different surface areas are investigated, noting the BOD concentration reached during the period influent is introduced. Removal efficiency is plotted throughout the time influent is imposed on the system.

Hydrology

The model was run with initial parameter values as given in Appendix B. No storm event is imposed and the influent rate is zero. Figure 4.1 shows the effects of evapotranspiration on the water budget of the wetland when there is no external water source. After 500 hours (20.83 days), the wetland surface water volume decreased from 5311.5 cubic meters to 3422.97 cubic meters, and the surface water depth decreased from .5 meters to .32 meters. This confirms Kadlec's statement that "the impacts of evaporative processes on a wetland treatment system are not trivial" (Hammer, 1990:38). Kadlec states that "even in northern climates, all applied wastewater can be evaporated during a dry summer season" (Hammer, 1990:38). Decreases in water volume and depth tend to increase the concentration of BOD in the water column due to both reduction in volume of surface water and reduction in the available biofilm area for degradation. Appendix D shows the water budget of the wetland as affected by various influent rates. These graphs should be studied in conjunction with the graphs of BOD concentration as the change in volume of surface water may greatly affect the concentration.

The model was modified to artificially maintain a constant water volume in the wetland by setting the influent rate equal to the evapotranspiration rate. The desired result is observed in Figure 4.2. Constant volume conditions were imposed so that the effects of varying the physical parameters of the system can be investigated without the disturbance of a storm event or the effect of decreasing water volume due to evapotranspiration.

Dissolved Oxygen Concentration

The effects of dissolved oxygen concentration in the water column were investigated under constant volume conditions. Dissolved oxygen concentration in the surface water effects the thickness of the biofilm as determined by Equation (5). The results of increasing the dissolved oxygen concentration in the water column from 0 mg/liter to 9.5 mg/liter, are shown in Figure 4.3. Equation (3) indicates that the flux of BOD at the interface of the liquid sublayer and biofilm interface will increase as biofilm thickness increases. As dissolved oxygen concentration increases, biofilm thickness increases, and flux into the biofilm increases, resulting in a greater rate of removal of BOD from the water column.

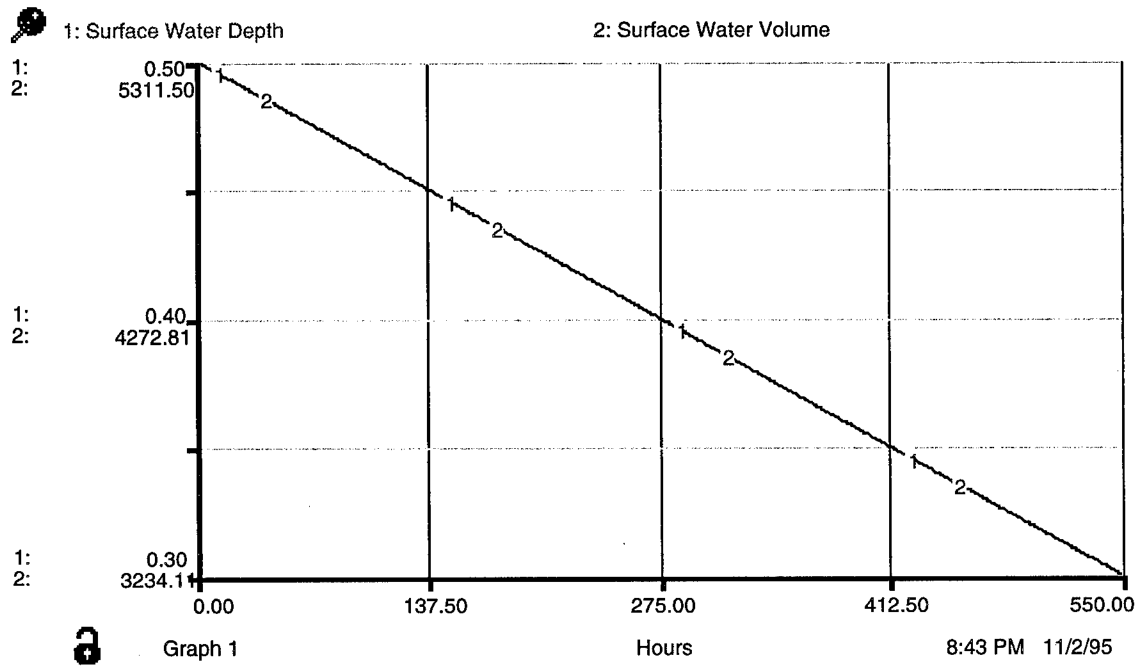


Figure 4.1

Evapotranspiration Effects on Surface Water Volume (m^3) and Depth (m)

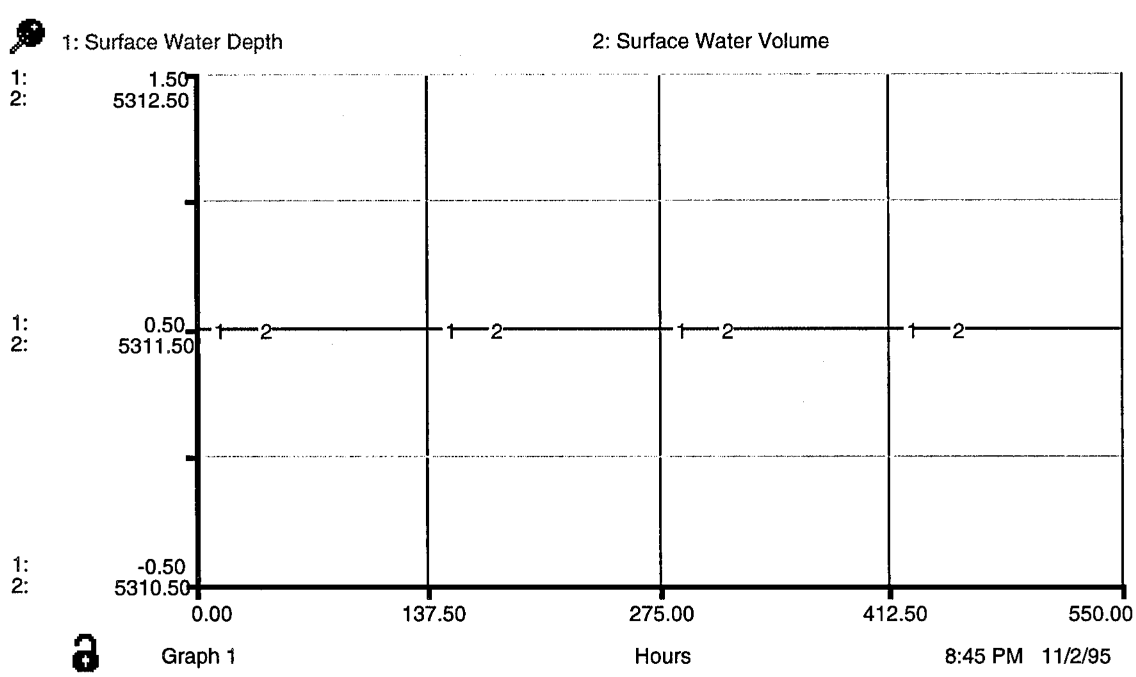


Figure 4.2

Water Budget at Constant Volume Conditions

The remaining runs of the model were made at a constant DO concentration equal to 8.0 mg/liter. This DO concentration is less than the approximate concentration at which water is saturated by DO at the defined water temperature of 18 degrees C (9.5 mg/l) (Sawyer and others, 1994:517). A constant DO value was chosen at this time for simplicity. A value that assumes aerobic conditions are maintained can be supported by Reddy and D'Angelo's statement that:

> The dissolved oxygen concentration of the water column remains relatively high due to:
> (1) a low density of oxygen-consuming organisms; (2) photosynthetic oxygen production by algae; (3) O_2 diffusion from the atmosphere; and (4) advective O_2 transport into the water (e.g. wind-induced mixing) (Mitsch, 1994:310).

It is also mentioned in the NURP study that urban runoff is typically well oxygenated and that no NURP project identified low DO conditions resulting from urban runoff (US EPA, 1983:9-4).

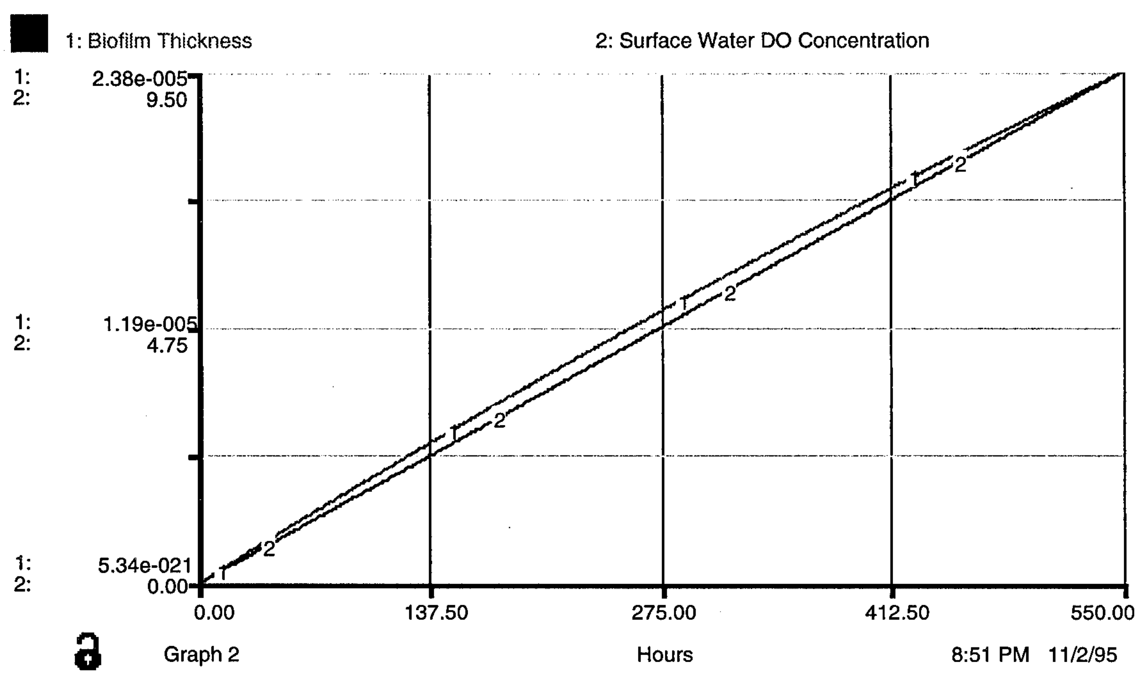


Figure 4.3

Change in Biofilm Thickness (micrometers)
with Dissolved Oxygen Concentration (mg/l)

Equilibrium BOD Concentrations

Baseline constructed wetland parameter values were again used with artificially imposed constant volume conditions to determine pseudo-equilibrium BOD concentrations expected in the wetland due to intrinsic processes. The initial BOD stock was set to zero. BOD accumulates due to macrophyte and algae generation, and eventually reaches an equilibrium concentration when generation equals removal due to diffusion into the biofilm and degradation by suspended biomass. Figure 4.4 shows biofilm flux increasing as surface water concentration increases. Note that the surface water degradation rate equals zero. This is due to Thirmurthis method for determining the first order degradation rate for suspended biomass which requires a minimum

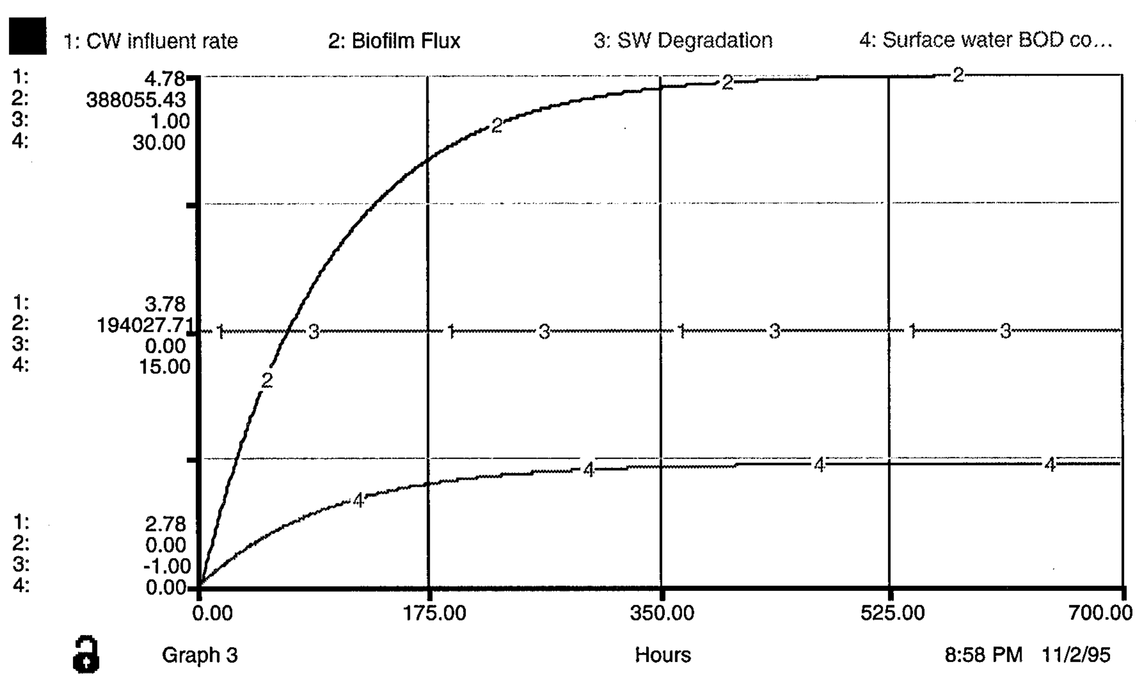


Figure 4.4

Equilibrium Conditions for Baseline Constructed Wetland
(Constant Volume Conditions, No Influent BOD)

organic load of .59 kg/ha-hr in order to obtain a suspended biomass degradation rate greater than zero. The organic loading rate due to macrophyte and algae generation alone is only .27 kg/ha-hr. The apparent equilibrium BOD concentration reached in under 500 hours is 7.08 mg/liter. This may be a reasonable concentration in a wetland with high rates of net primary productivity.

The model was run several more times, again at constant volume conditions and no external BOD source, varying the constructed wetland surface area, surface water initial depth and length to width ratio within the ranges defined in Appendix B. Appendix E lists the parameters and observed equilibrium values for several runs made at constant volume conditions. Table 4.1 contains a summary of the equilibrium values for the baseline constructed wetland with noted parameter changes. Trends resulting from varying parameter values include a decrease in equilibrium BOD levels as constructed wetland surface area, initial depth and length to width ratio are all increased. Change in surface water depth had the greatest affect on final BOD concentration as compared with surface area and length to width ratio. This is may be due to the fact that macrophyte and algae DOM generation are calculated from net primary productivity given in units of mass per area. The surface area remains the same, however, as depth increases, so does the volume of surface water, effectively diluting the concentration of BOD as compared with shallower depths. Area available for biofilm formation however, also increases, thereby increasing the amount of BOD that can be removed from the system.

An increase in constructed wetland surface area reduces the equilibrium BOD value perhaps due only to the increased area for formation of biofilm. The increased area available for biofilm formation may more than compensate for the release of DOM from macrophytes as determined by surface area.

A larger length to width ratio provides for a greater velocity of water through the wetland, which has the effect of reducing the thickness of the liquid film layer as determined by Equation (4). This in turn causes an increase in the biofilm thickness and the resulting flux into the biofilm increases giving a greater removal rate from the surface water.

Table 4.1

Equilibrium Concentrations for the Baseline Constructed Wetland with Parameter Changes (Constant Volume Conditions, No External BOD Source)

Surface Area (m^2)	7578	10000	14164	15175
BOD Conc. (mg/l)	7.54	7.33	7.08	7.03

Surface Water Initial Depth (m)	.25	.5	.75	.9
BOD Conc. (mg/l)	12.06	7.08	5.17	4.49

L:W Ratio	3	5	10	15
BOD Conc. (mg/l)	8.01	7.6	7.08	6.8

Equilibrium BOD concentrations from these runs were used in subsequent runs to determine the initial stock of BOD. For example, subsequent runs examine the effect of a storm event on the BOD removal efficiency of the constructed wetland. If baseline physical parameters are chosen, the initial stock of BOD is based on an equilibrium concentration equal to 7.08 mg/liter and the intitial surface water volume. If the physical parameters are changed, the corresponding equilibrium BOD concentration and appropriate surface water volume are used to determine the initial BOD stock for that run.

Single Storm Event

A single storm event was introduced. The storm event fills the detention pond and water is released to the constructed wetland at a defined rate. The model was run at influent rates of 10, 25, 50, 100 and 150 m^3/hr and influent BOD concentrations at 10, 20, 40, and 60 mg/liter. For each influent rate, a graph of the constructed wetland surface water volume and depth is provided in Appendix D to illustrate how the water budget changes with time. This is important since once the influent has ceased, the volume of water in the wetland begins to decrease due to evapotranspiration, contributing to an increase in concentration. Reduction in surface water also reduces the area of biofilm on vegetation available for diffusion of BOD.

The purpose of these runs is to compare the effects of various influent rates and influent concentrations on surface water BOD concentration and wetland removal efficiency. The physical parameters of the wetland remained constant throughout these runs. Table 4.2 shows the hydraulic retention times and hydraulic loading rates that correspond to each influent rate. These values apply to each set of data in Tables 4.3-4.6. Tables 4.3- 4.6 show the organic loading rates corresponding to each influent rate and concentration for the predefined baseline constructed wetland parameters. These are important values as they may be correlated with removal efficiency. Tables 4.3 - 4.6 also show BOD levels reached during the time influent is imposed on the wetland.

According to the findings in Chapter 2 regarding hydraulic loading rates, the data in Table 4.2 shows that for a wetland of surface area 14164 m^2 , influent rates greater than the 100 m^3/hr would exceed the rate found in most existing FWS systems.

Table 4.2

Hydraulic Retention Times (HRT) and Hydraulic Loading Rates (HLR) for a Constructed Wetland with Surface Area 14164 m^2

Influent Rate (m^3/hr)	*10*	*25*	*50*	*100*	*150*
HRT (days)	27.43	9.68	4.62	2.29	1.53
HLR (m^3/m^2-day)	.017	.042	.084	.168	.254

Table 4.3

BOD Concentrations Reached for the Baseline Constructed Wetland
Influent Concentration 10 mg/l

Influent Rate (m^3/hr)	*10*	*25*	*50*	*100*	*150*
Total Organic Loading Rate (kg/ha-hr)	.34	.45	.63	.98	1.33
BOD Conc. during influx (mg/l)	6.23	5.98	6.32	6.85	7.22

Table 4.4

BOD Concentrations Reached for the Baseline Constructed Wetland
Influent Concentration 20 mg/l

Influent Rate (m^3/hr)	10	25	50	100	150
Total Organic Loading Rate (kg/ha-hr)	.42	.63	.98	1.69	2.39
BOD during influx (mg/l)	7.50	8.28	9.62	11.34	12.37

Table 4.5

BOD Concentrations Reached for the Baseline Constructed Wetland
Influent Concentration 40 mg/l

Influent Rate (m^3/hr)	10	25	50	100	150
Total Organic Loading Rate (kg/ha-hr)	.56	.98	1.69	3.10	4.51
BOD Conc. during influx (mg/l)	10.05	12.59	16.09	20.21	22.55

Table 4.6

BOD Concentrations Reached for the Baseline Constructed Wetland
Influent Concentration 60 mg/l

Influent Rate (m^3/hr)	10	25	50	100	150
Total Organic Loading Rate (kg/ha-hr)	.70	1.33	2.39	4.51	6.63
BOD during influx (mg/l)	12.43	16.80	22.44	28.98	32.65

Watson and others found that most wetland systems have operated at loading rates ranging from 18 - 116 kg BOD_5 /ha-day (.75 - 4.83 kg BOD_5 /ha-hr) and achieve 70 - 95% BOD_5 removal. They suggest that in order to ensure maintenance of aerobic conditions, an upper limit of 110 kg/ha-day (4.58 kg/ha-hr) not be exceeded (Hammer, 1990:341). The organic loading rates in Tables 4.3 - 4.6 range from a low of .34 kg/ha-hr with an influent concentration of 10 mg/l at an influent rate of 10 m^3/hr up to an organic loading rate of 6.63 kg/ha-hr with an influent concentration of 60 mg/l at an influent rate of 150 m^3/hr.

Figure 4.5 illustrates the change in BOD concentration in the baseline constructed wetland receiving stormwater at an influent rate of 25 m^3/hr with a BOD concentration of 20 mg/liter. Both biofilm flux and degradation due to suspended biomass (SW degradation) increase during the period influent is introduced into the wetland. It appears that an equilibrium flux rate and surface water degradation rate are reached during that time, resulting in an equilibrium BOD concentration. As soon as the influent ceases, the biofilm flux rate and surface water degradation rate drop significantly. The cause of the sudden drop in degradation rate due to suspended biomass can be found in the equation for determining the degradation rate constant. The degradation rate constant is dependent on the organic loading rate as given by Equation (1). As previously noted, if the organic loading rate is below .59 kg/ha-hr, the degradation rate is considered insignificant and drops to zero. When the influent which contains a significant concentration of BOD is no longer imposed on the system, the organic loading rate returns to its original level which was previously shown not to be great enough to result in a significant degradation rate due to suspended biomass. The biofilm flux rate drops significantly at the time influent ceases due to the sudden change in flow velocity. Flow velocity effects the liquid film thickness as given by Equation (4), which effects the biofilm thickness as given by Equation (5) and in turn the flux rate as given by Equation (3).

Figure 4.6 illustrates the relationship between influent rate, effluent rate, liquid film layer thickness, biofilm thickness and biofilm flux. During the first 24 hours of the scenario modeled here, no storm event or influent is imposed on the system. The volume of surface water decreases due to evapotranspiration during that time. At 24 hours, a storm event is introduced and the surface water volume increases to its maximum capacity. Once that maximum capacity is reached, the wetland begins to release water and has both an influent and effluent rate, creating a flow velocity through the wetland. The change in flow velocity explains the spikes in both the biofilm thickness and biofilm flux seen in Figure 4.6, as well as the spikes seen in Figures 4.5 and 4.7. The velocity of the flow through the wetland is determined by the influent and effluent rates. During the storm event, the effluent rate equals the influent rate plus the amount of water entering the system due to precipitation, minus the evapotranspiration rate. Once the rain event stops, the effluent rate is equal to the influent rate only minus the evapotranspiration rate. The resulting change in flow rate slightly effects the liquid film thickness as seen in Figure 4.6, which has a much greater effect on the biofilm thickness and in turn the biofilm flux. The flow velocity is high during the storm event and reduces when the rain stops. The increased flow rate of water through the wetland decreases the liquid film layer thickness, increases the biofilm thickness and the flux into the biofilm. This is the spike seen in Figure 4.6. When the effluent rate decreases slightly due to lack of precipitation, the flow rate decreases to a lower rate, slightly increasing the liquid film thickness and reducing the biofilm thickness and flux. When the influent ceases the effluent does as well, reducing the flow velocity to zero and reverting the film thicknesses back to their original thicknesses, and flux rate drops significantly, however, not completely to its previously low level. This is probably due to the increase in surface water concentration, which has the effect of increasing flux rate into the biofilm. The BOD concentration begins to slowly rise once the influent ceases which is likely due to the effects of evapotranspiration reducing the water volume.

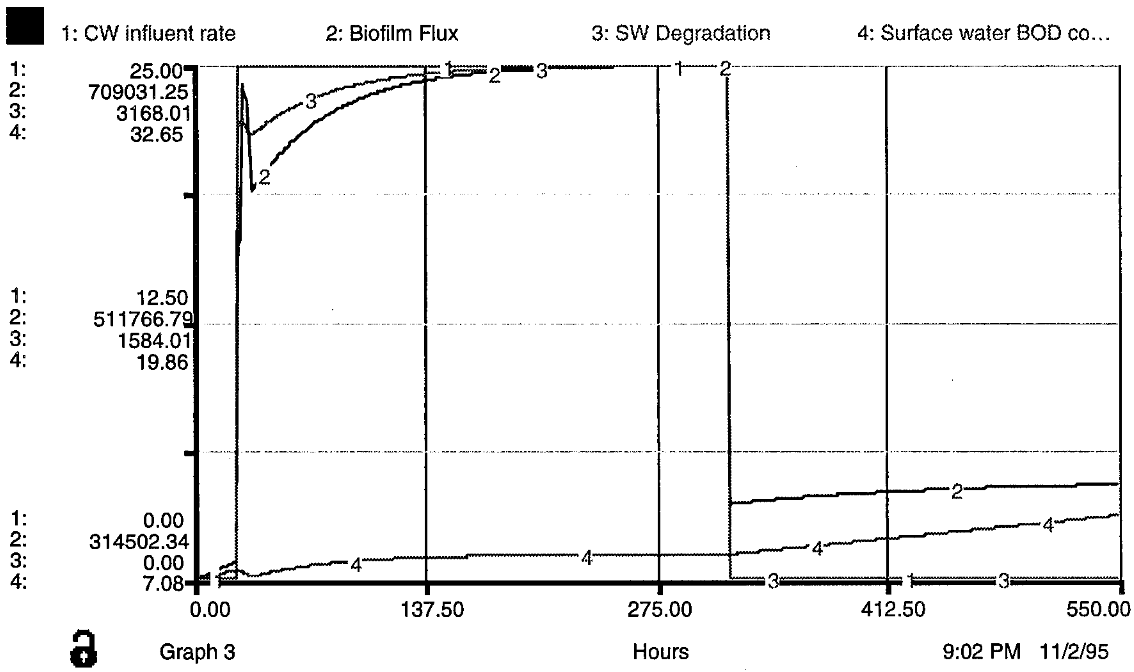


Figure 4.5

Influent Rate 25 m^3/hr, Influent Concentration 20 mg/l

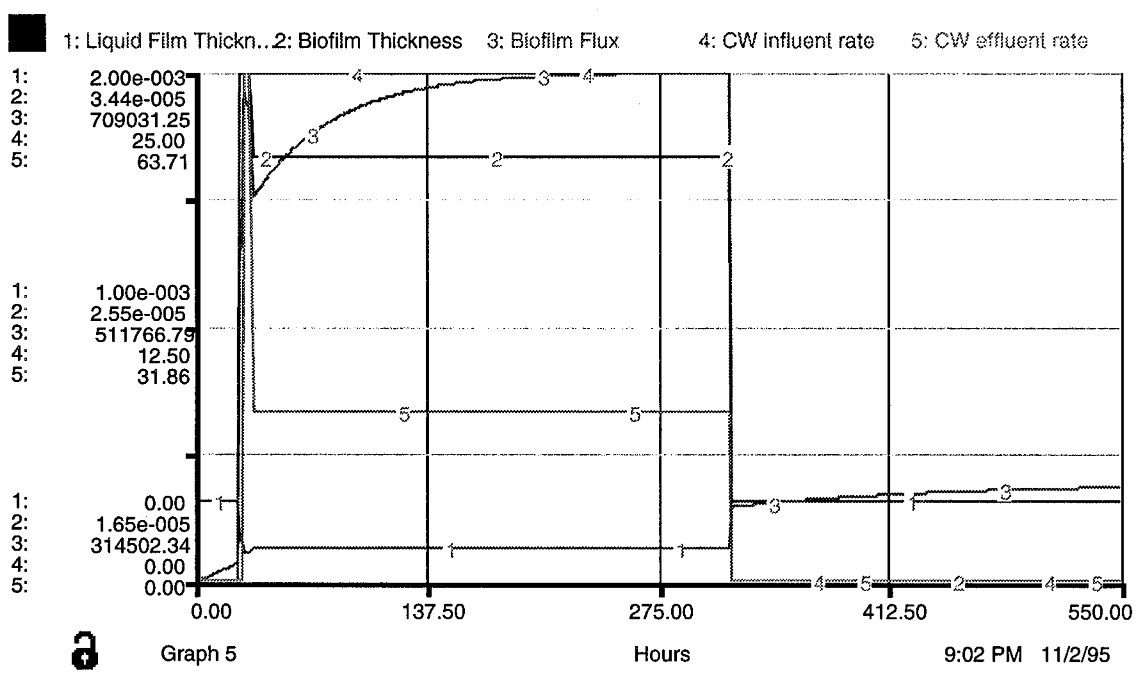


Figure 4.6

Relationship Between Biofilm Flux (mg/hr) and Biofilm Thickness (m)

The BOD removal efficiency is the ratio of total mass of BOD removed from the system to the total mass of BOD entering the system. It is calculated by taking the sum of the rate of BOD removal by the system mechanisms of degradation due to both suspended biomass and flux into the biofilm and dividing that quantity by the rate of BOD entering the system by way of the influent and macrophyte and algae generation.

$$\mathrm{Re}\,moval\ Efficiency = \frac{Biofilm\ Flux\ Rate + Rate\ of\ \ Surface\ Water\ Degradation}{BOD\ Influent\ \ Rate + BOD\ Generation\ Rate}$$

Although this is not the classical definition of removal efficiency that the wastewater engineer would use to determine the efficiency of a treatment system, it does however serve the purposes of this research. The wastewater engineer is interested in a calculation of removal efficiency based only on the influent and effluent concentrations of BOD. The wastewater engineer might use the following expression to calculate a systems' removal efficiency:

$$\mathrm{Re}\,movalEfficiency = \frac{Influent\ \ BOD - Effluent\ \ BOD}{Influent\ \ BOD}$$

Once the removal efficiency of a system is determined, it can then be used to determine the expected BOD concentration in the effluent. The removal efficiency calculated for this research is not used for that purpose. It is used to determine the effects of system parameter changes on system efficiency. Wetland parameter changes not only effect the rate of BOD removal, but also the rate of intrinsic BOD generation. Most conventional treatment systems do not have an intrinsic source of BOD. The removal efficiency calculated for this research accounts for the systems intrinsic generation of BOD and provides a complete picture of change in system efficiency with paramter changes.

The removal efficiencies for each of the runs in Tables 4.3 - 4.6 are shown in Table 4.7. Since the removal efficiency changes with time as seen in Figure 4.7, the efficiencies listed in Table 4.6 are the efficiencies reached by the time the influent ceases. Figure 4.7 shows the removal efficiency reaching an equilibrium value at .80 during influx of stormwater.

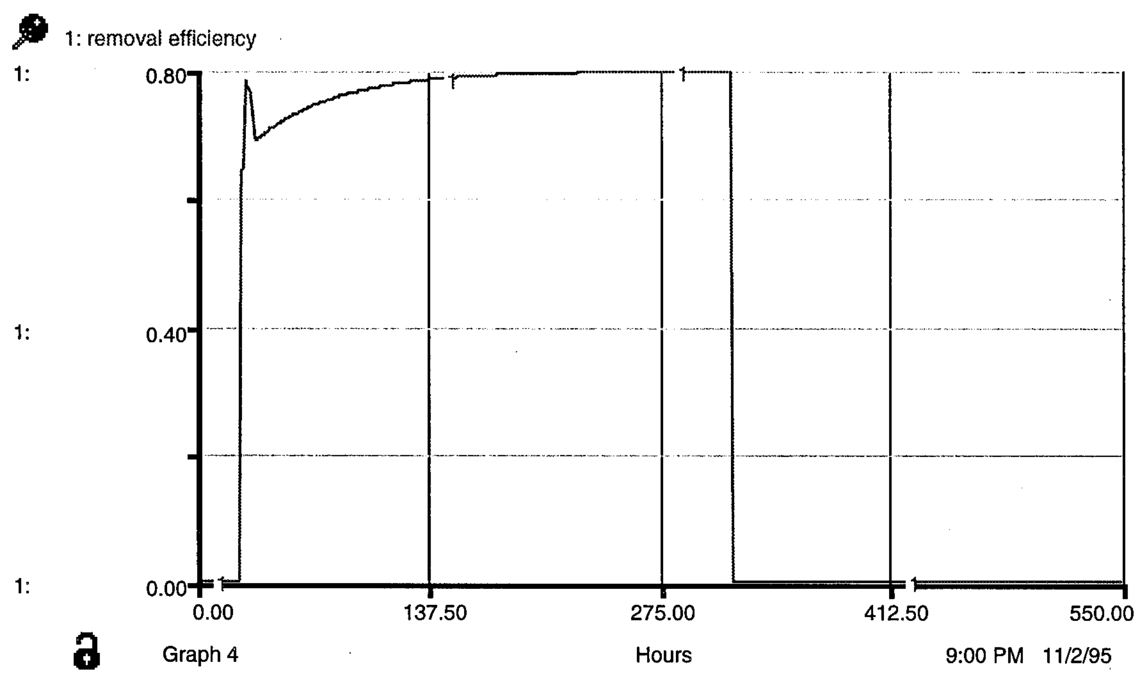


Figure 4.7

Removal Efficiency for Influent Rate 25 m^3/hr and Influent Concentration 20 mg/l.

Table 4.7

Removal Efficiencies for Baseline Parameters

Influent Rate (m^3/hr)	10	25	50	100	150
Hydraulic Retention Time (days)	27.43	9.68	4.62	2.29	1.53

Influent Concentration: 10 mg/l

Organic Loading Rate (kg/ha-hr)	.34	.45	.63	.98	1.33
Removal Efficiency	.92	.81	.67	.53	.44

Influent Concentration: 20 mg/l

Organic Loading Rate (kg/ha-hr)	.42	.63	.98	1.69	2.39
Removal Efficiency	.92	.80	.68	.52	.43

Influent Concentration: 40 mg/l

Organic Loading Rate (kg/ha-hr)	.56	.98	1.69	3.10	4.51
Removal Efficiency	.92	.81	.68	.52	.43

Influent Concentration: 60 mg/l

Organic Loading Rate (kg/ha-hr)	.70	1.33	2.39	4.51	6.63
Removal Efficiency	.92	.81	.68	.53	.43

Table 4.7 shows that removal efficiency for the defined wetland is essentially constant for each influent rate and corresponding hydraulic retention time, even as the organic loading rate increases. However, grouped by influent concentration, as influent rate increases, organic loading rate naturally increases, and removal efficiency decreases. This indicates that hydraulic retention time is the primary factor effecting removal efficiency regardless of organic loading rate.

Three other sets of data were generated by changing the surface area. See Tables 4.8, 4.9, and 4.10. Changing the wetland area affects both the organic loading rate and the hydraulic retention time. A fourth set of data was generated by maintaining the baseline surface area but decreasing the length to width ratio giving the same hydraulic retention time and organic loading rates as for the baseline parameters. See Table 4.11.

Table 4.8

Removal Efficiencies for CW with Surface Area 7578 m^2

Influent Rate (m^3/hr)	10	25	50	100	150
Hydraulic Retention Time (days)	13.28	4.95	2.45	1.23	.83

Influent Concentration: 10 mg/l

Organic Loading Rate (kg/ha-hr)	.41	.6	.93	1.59	2.25
Removal Efficiency	.84	.68	.53	.39	.31

Influent Concentration: 20 mg/l

Organic Loading Rate (kg/ha-hr)	.54	.93	1.59	2.91	4.23
Removal Efficiency	.84	.68	.54	.39	.31

Influent Concentration: 40 mg/l

Organic Loading Rate (kg/ha-hr)	.80	1.59	2.91	5.55	8.19
Removal Efficiency	.84	.68	.54	.39	.32

Influent Concentration: 60 mg/l

Organic Loading Rate (kg/ha-hr)	1.07	2.25	4.23	8.19	12.15
Removal Efficiency	.84	.70	.55	.40	.32

Table 4.9

Removal Efficiencies for CW with Surface Area 10000 m^2

Influent Rate (m^3/hr)	10	25	50	100	150
Hydraulic Retention Time (days)	18.15	6.61	3.23	1.62	1.09

Influent Concentration: 10 mg/l

Organic Loading Rate (kg/ha-hr)	.37	.52	.77	1.27	1.77
Removal Efficiency	.88	.73	.60	.45	.36

Influent Concentration: 20 mg/l

Organic Loading Rate (kg/ha-hr)	.47	.77	1.27	2.27	3.27
Removal Efficiency	.88	.74	.60	.45	.36

Influent Concentration: 40 mg/l

Organic Loading Rate (kg/ha-hr)	.67	1.27	2.27	4.27	6.27
Removal Efficiency	.88	.74	.61	.46	.37

Influent Concentration: 60 mg/l

Organic Loading Rate (kg/ha-hr)	.87	1.77	3.27	6.27	9.27
Removal Efficiency	.88	.75	.61	.46	.37

Table 4.10

Removal Efficiencies for CW with Surface Area 15175 m^2

Influent Rate (m^3/hr)	10	25	50	100	150
Hydraulic Retention Time (days)	29.87	10.43	4.96	2.45	1.64

Influent Concentration: 10 mg/l

Organic Loading Rate (kg/ha-hr)	.34	.44	.60	.93	1.26
Removal Efficiency	.93	.81	.69	.54	.45

Influent Concentration: 20 mg/l

Organic Loading Rate (kg/ha-hr)	.41	.60	.93	1.59	2.25
Removal Efficiency	.93	.81	.69	.54	.44

Influent Concentration: 40 mg/l

Organic Loading Rate (kg/ha-hr)	.54	.93	1.59	2.91	4.23
Removal Efficiency	.93	.82	.69	.54	.44

Influent Concentration: 60 mg/l

Organic Loading Rate (kg/ha-hr)	.67	1.26	2.25	4.23	6.20
Removal Efficiency	.93	.82	.70	.54	.45

Table 4.11

Removal Efficiencies for CW with L:W = 5

Influent Rate (m^3/hr)	10	25	50	100	150
Hydraulic Retention Time (days)	27.43	9.68	4.62	2.29	1.53

Influent Concentration: 10 mg/l

Organic Loading Rate (kg/ha-hr)	.34	.45	.63	.98	1.33
Removal Efficiency	.92	.79	.66	.52	.43

Influent Concentration: 20 mg/l

Organic Loading Rate (kg/ha-hr)	.42	.63	.98	1.69	2.39
Removal Efficiency	.92	.79	.67	.51	.42

Influent Concentration: 40 mg/l

Organic Loading Rate (kg/ha-hr)	.56	.98	1.69	3.10	4.51
Removal Efficiency	.92	.80	.67	.52	.42

Influent Concentration: 60 mg/l

Organic Loading Rate (kg/ha-hr)	.70	1.33	2.39	4.51	6.63
Removal Efficiency	.92	.80	.67	.52	.42

Data from Tables 4.7-4.11 were used in scatter plots to check for a correlation between hydraulic residence time and removal efficiency and a correlation between organic loading rate and removal efficiency. The results are shown in Figures 4.8 and 4.9 respectively.

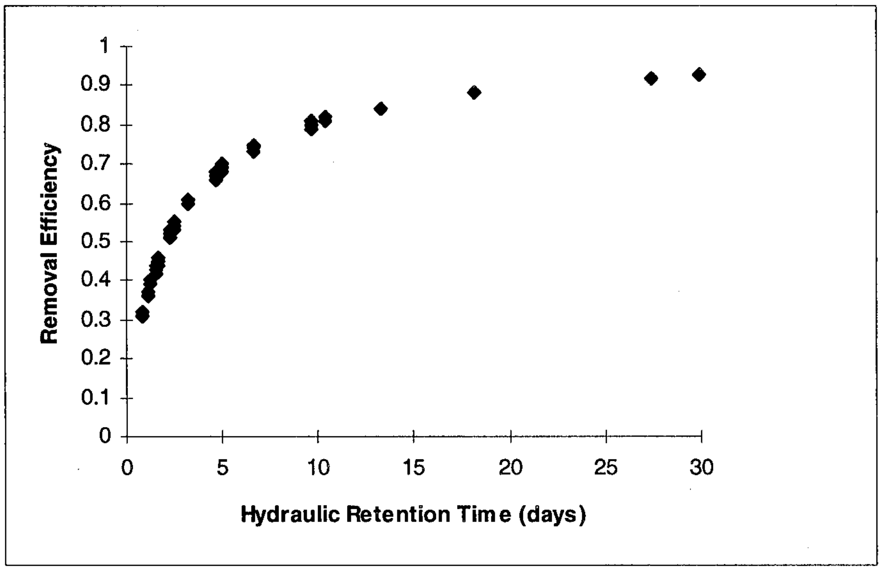


Figure 4.8

Scatter Plot of Hydraulic Residence Time vs. Removal Efficiency

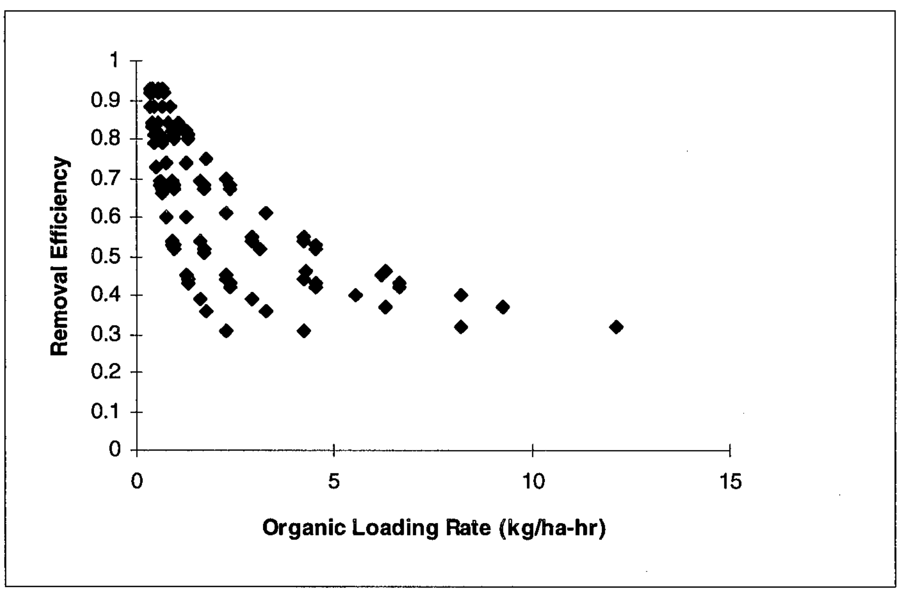


Figure 4.9

Scatter Plot of Organic Loading Rate vs. Removal Efficiency

Figure 4.9 indicates a logarithmic relationship between hydraulic retention time and removal efficiency. As the HRT increases up to 5 days, the increase in removal efficiency is significant. Removal efficiency increases from approximately 32% to approximately 70% at 5 days. Only about 10% efficiency is gained from another 5 days of retention time. Removal efficiency appears to level off at near 90% between 25 and 30 days.

A positive relationship between removal efficiency and organic loading rate is also apparent. Figures 4.10 - 4.13 illustrate this relationship more clearly. The organic loading rates and corresponding removal efficiencies are grouped by influent concentration and plotted seperately. Figures 4.10 - 4.13 show that higher influent concentrations naturally lead to higher organic loading rates for a given wetland, but comparable removal efficiencies to those at lower influent concentrations are achieved. That is, in order to achieve a removal efficiency of approximately 70%, organic loading rates can range from 0.5 kg/ha-hr to approximately 2.5 kg/ha-hr with corresponding influent concentrations of 10 - 60 mg/l. Figures 4.10 - 4.13 indicate that removal efficiency may be a function of both organic loading rate and influent concentration. At organic loading rates below approximately 2 kg/ha-hr, removal efficiencies are highest.

The organic loading rate is not typically used as a design consideration, rather it is monitored as a check for preventing anaerobic conditions from developing in the system. The organic loading expected to result from stormwater runoff is very low as compared with that in municipal wastewater. Most of the data available applies to municipal wastewater treatment systems and the organic loading rates are much higher than most of those in resulting from conditions imposed on this model.

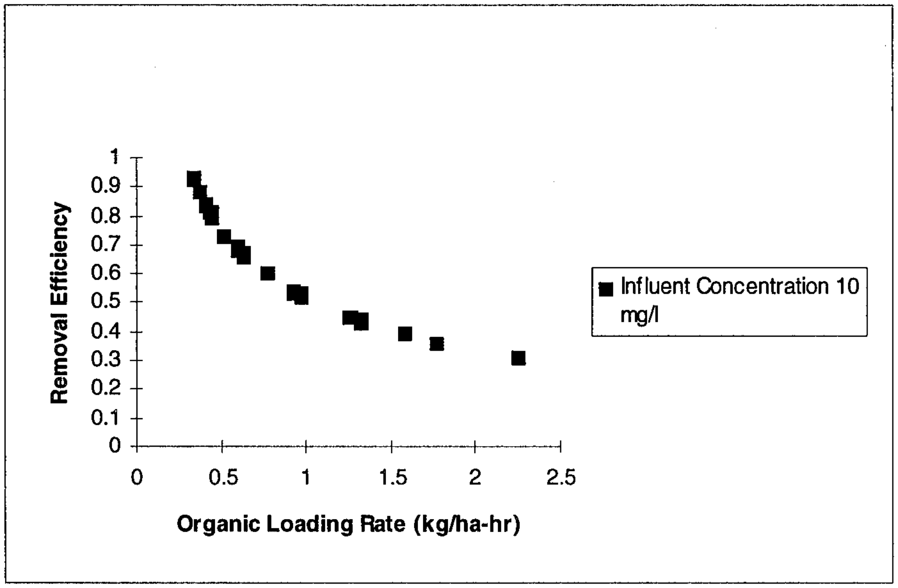


Figure 4.10

Organic Loading Rate vs. Removal Efficiency
(Influent Concentration 10 mg/l)

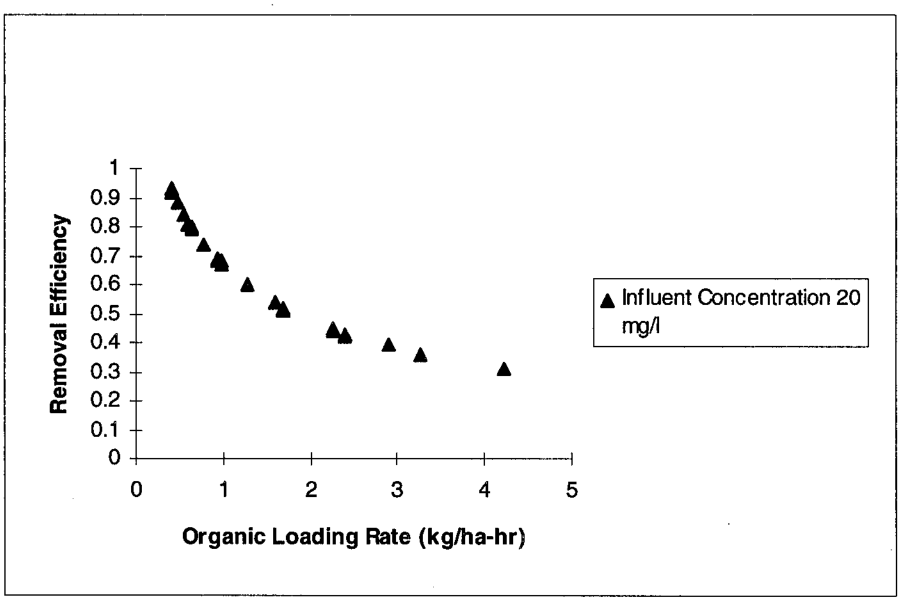


Figure 4.11

Organic Loading Rate vs. Removal Efficiency
(Influent Concentration 20 mg/l)

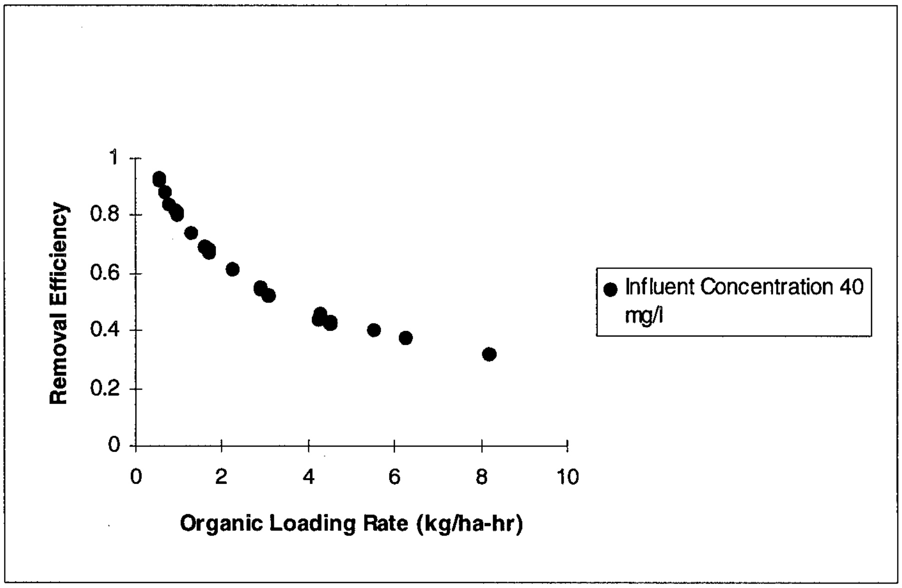


Figure 4.12

Organic Loading Rate vs. Removal Efficiency
(Influent Concentration 40 mg/l)

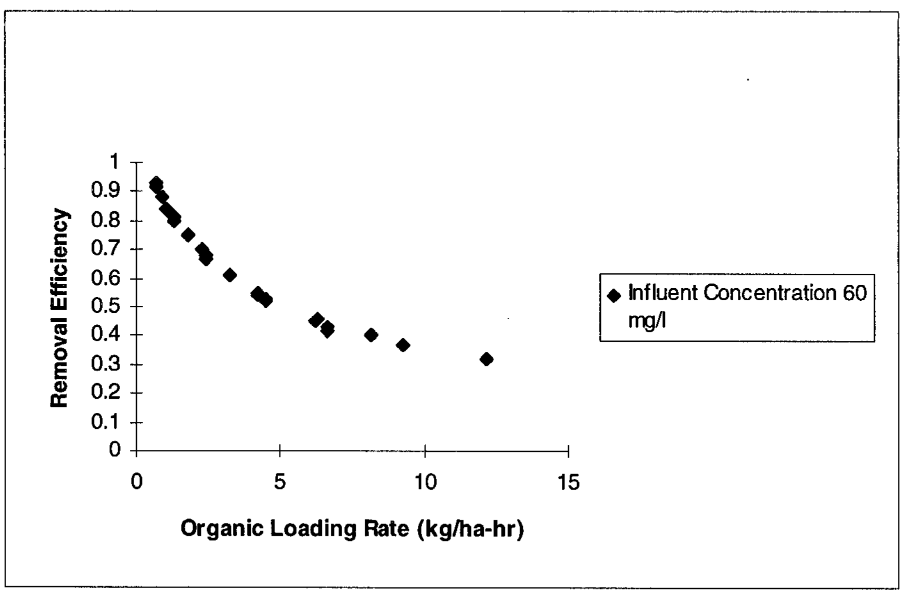


Figure 4.13

Organic Loading Rate vs. Removal Efficiency
(Influent Concentration 60 mg/l)

Figures illustrating change in BOD concentration and removal efficiency identical to Figures 4.5 and 4.7 are included in Appendix F for the remaining influent rates and concentrations applied to the baseline wetland. Some interesting trends with respect to BOD concentrations following the storm event are observed. In all cases where the influent concentration was 10 mg/l during the storm event, the BOD concentration rises following the event. When the influent concentration is increased to 20 mg/l, the BOD increases following the storm event when the influent rate was 10, 25 or 50 m^3/hr, but decreases following influent rates of 100 and 150 m^3/hr. This indicates that higher loading rates may spur higher removal rates. For both influent concentrations 40 and 60 mg/l, the BOD reaches a peak, (as high as 32.65 mg/l in the 60 mg/liter at 150 m^3/hr case) then drops. No explanation for this behavior is offered at this time. It would be interesting to run the model again for each influent rate and concentration, but impose the condition where the system receives influent at the specified rate and concentration for the same period of time in each case.

<u>Summary</u>

The data generated by simulating various physical conditions reveal the areas of the model that require further research and fine tuning in order to simulate wetland processes more accurately. Possible relationships between some of these parameters are, however, noted. First, the effects of evapotranspiration are shown to have a significant effect on the water budget of the wetland, contributing to higher BOD concentrations. Depending on the typical precipitation levels and pan evaporation rates characteristic of the local area, an outside water source other than storm events may be desirable in order to maintain a viable wetland. Field data regarding the effects of evapotranspiration on a wetland should be compared with results from this model. It is important to be sure that the water budget is simulated accurately by the model in order to ensure the most accurate results.

The dissolved oxygen concentration has a significant effect on the development of biofilm thickness, resulting in an effect on the BOD flux rate into the biofilm. A dissolved oxygen mass balance should be included in the model to determine if fluctuations in DO concentration are significant enough to affect biofilm thickness.

The method for determining intrinsic generation of dissolved organic matter may not be representative of true values. This becomes apparent when equilibrium BOD concentrations are determined for various depths. The method for determining macrophyte and algae generation of DOM use in this model is based on the assumption that net primary productivity of the vegetation is only a function of constructed wetland surface area. Chapter 2 references studies that show a possible correlation between density of vegetation and water depth. The method employed by this model assumes constant density of vegetation and mass of DOM generated regardless of depth of water. In actuality, if vegetation is more dense in shallower water, this could result in higher levels of DOM released in shallower wetlands. Field data should be obtained to confirm this.

Based on the calculation of equilibrium conditions alone, the following trends regarding physical parameters are noted: as surface area and length to width ratio increase, the BOD concentration in the water column decreases. An increase in length to width ratio decreases the cross-section in the direction of flow, contributing to an increase in flow velocity, resulting in a decrease in liquid film layer thickness and an increase in both biofilm thickness and flux into the biofilm. An increase in surface area may reduce BOD concentrations due to the increase in available area for biofilm to form.

Also, in every case, BOD removal is primarily due to biofilm degradation as opposed to degradation attributed to suspended biomass.

Finally, removal efficiencies appear to be logarithmically related to hydraulic retention time. A 30% increase in removal efficiency is seen when the HRT increases up to 5 days, and another 20% efficiency is gained up to about 10 days. Gains in removal efficiency after 10 days are not as significant. Another 10% may be gained by extending the hydraulic retention time from 10 to 25 days.

A given removal efficiency is achieved over a range of organic loading rates. For example, in order to achieve a removal efficiency of approximately 80%, the organic loading rate is seen to range from .44 - 1.33 kg/ha-hr. The desired organic loading rate required to achieve that removal efficiency is tied to the influent concentration. At a low influent concentration of 10 mg/l, the organic loading rate required to give a removal efficiency of approximately 80% was 0.44 kg/ha-hr. At the highest influent concentration of 60 mg/l however, the organic loading rate was as high as 1.33 kg/ha-hr. In both cases, the hydraulic retention time is approximately 10 days. The hydraulic retention time seems to be the determining factor regarding removal efficiency, however a range of organic loading rates that will give a specified removal efficiency can be determined.

V. Conclusions and Recommendations

Introduction

This chapter presents the conclusions drawn from the literature review, the results and analysis presented in Chapter Four, as well as answers to the research questions posed in Chapter One. Finally, recommendations are made regarding further research that would improve the accuracy and usefulness of the model developed.

Conclusions

The purpose of this research was to determine the significant processes within a constructed wetland system responsible for the removal of BOD introduced via stormwater influent; develop a systems dynamics model that simulates the hydrologic functions of a wetland and the processes identified above and utilize the model to determine the physical parameters and operational factors that may be controlled to optimize the BOD removal efficiency of the wetland.

Investigative Question One. What are the sources and characteristics of BOD in AF stormwater? What is the typical BOD content in AF stormwater? Stormwater flowing over roofs, roads, parking lots, industrial areas, golf courses etc., carry such pollutants as pesticides, oils and grease, heavy metals and other wastes. Air Force installations consist of industrial areas and airfields as well as urban type areas all of which likely generate stormwater runoff with varying amounts and types of pollutants. Biochemical Oxygen Demand was identified by the NURP study as a pollutant of concern in typical urban stormwaters. Air Force stormwater data also identifies BOD as a pollutant that often exceeds EPA benchmark concentrations. Data from AF stormwater samples show BOD concentrations ranging from 2 to 42 mg/l with a mean of 12.61 mg/l.

Investigative Question Two. What processes within a CW occur that affect the degradation of BOD? The primary mechanism responsible for the removal of BOD from a wetland system is microbial degradation. Microbial populations within a wetland are typically in the form of suspended biomass and biofilm formed on the surfaces of vegetation and the constructed wetland floor. Degradation rates associated with each microbial population are effected by the characteristics of the BOD imposed on the system, the characeristics of the microbial population, surface water temperature and other environmental conditions. The degradation rate constant associated with the suspended biomass is defined by an expression dependent on the organic loading rate and the actual rate of removal is also a function of the current BOD concentration in the water column. The rate of flux of BOD into the biofilm is dependent on the velocity of the flow as this effects biofilm thickness, the surface area of biofilm, and the concentration of BOD in the water column.

Investigative Question Three. What are the critical parameters and operating characteristics associated with a CW that affect the degradation of BOD? Dissolved oxygen concentration in the water column had a significant effect on the rate of BOD flux into the biofilm. This confirms the importance of maintaining aerobic conditions within the system. The effects of evapotranspiration were also noted as significant if precipitation is not sufficient to maintain a minimum volume of water in the system. Loss of water due to evapotranspiration reduces the surface area available for biofilm degradation and serves to concentrate the existing mass of BOD in the system.

Initial runs of the model where constant volume conditions were imposed gave pseudo-equilibrium BOD concentrations, or concentrations believed to be those expected in a wetland due to intrinsic generation of dissolved organic matter. These values ranged from 6.8 mg/l to 12.06 mg/l depending on the physical parameters of the wetland. Increases in surface water depth resulted in the greatest change in equilibrium concentration. Deeper water tended to give lower BOD concentrations. Increased surface area and length to width ratios tended to result in lower BOD concentrations as well.

The results from runs of the model when influent rate and concentration were varied indicate that hydraulic retention time is logarithmically related to the removal efficiency of the wetland. Greatest gains in removal efficiency were realized as hydraulic retention time increase up to approximately 10 days. Gains in removal efficiency were not as great as hydraulic retention time increased from 10 to 30 days. A scatter plot showing organic loading rate vs. removal efficiency also revealed a relationship between the two. Greater influent concentrations did not result in lower removal efficiencies. In fact, greater influent concentrations reached comparable removal efficiencies at higher organic loading rates than lower influent concentrations. However, as organic loading rate decreased in all cases, greater removal efficiencies were achieved.

Design Implications. Hydraulic retention time was the most significant factor in determining removal efficiency. The theoretical hydraulic retention time is a function of both the volume of surface water and the flow rate through the wetland. The volume of surface water is of course a function of the surface area, depth of water and density of vegetation. The depth of water may be limited based on the type of vegetation desired, as some vegetation cannot survive in waters either too deep or too shallow. If the depth can be determined by type of vegetation, a rough estimate of surface area can be calculated. The approximate density of vegetation must also be taken into consideration in order to give the correct volume of water. If available area is a limiting factor in the design, the system must be designed to receive the appropriate influent rate in order to achieve a hydraulic retention time. For example, in order to achieve a removal efficiency of 80%, the hydraulic retention time should be approximately 10 days. The required influent rate can be determined based on the available area and required depth for the system, or conversely, given limitations on flow rates, the required volume of water in order to give a hydraulic retention time of 10 days may be specified. However, as previously noted, the area should be maximized in order to provide a greater surface area for the formation of biofilm. Also, the greater the depth and surface area, the greater the influent rate can be.

The length to width ratio should also be maximized in order to impose greater flow rates through the wetland resulting in greater removal of BOD through the biofilm. The effect of length to width ratio should be investigated more thoroughly, however, a greater reduction in equilibrium concentration was seen when the ratio was increased from 5 to 10 than was seen from 10 to 15. This may indicated a threshold past which significant reductions are no longer seen.

For a given removal efficiency, a range of organic loading rates were seen. For example, a removal efficiency of 80% was achieved with organic loading rates ranging from 0.45 - 1.33 kg/ha-hr. At lower influent concentrations (10 and 20 mg/l), small changes in organic loading rate had significant effects on removal efficiency. The greater influent concentrations (40 and 60 mg/l) showed more gradual changes in removal efficiency. Given a know influent concentration and desired removal efficiency, based on the data obtained from this model, the appropriate organic loading rate can be determined.

Recommendations

The model developed in support of this research is the result of a first attempt at identifying wetland processes relevent to BOD degradation, and simulating those processes through the use of a system dynamics model. The model has not been validated with field data to determine its accuracy in simulating those processes. Further research may reveal other significant processes that should be incorporated in the model, or substantiate the need for different assumptions than those made here. The following recommendations for further research suggest work that may add to the validity of the model and provide a future user with a better tool for optimizing the design of a constructed wetland for the purpose of stormwater manangement, specifically with respect to BOD degradation aspects.

1. Obtain field data from an existing constructed wetland receiving stormwater runoff and validate the model using this data.

2. Include in the model simulation of the processes that affect the dissolved oxygen concentration in the water column.

3. Determine the significance of water depth on density of vegetation and incorporate an appropriate factor in the model.

4. Determine effects of several successive storm events on wetland BOD removal efficiency.

5. Determine the effects of nutrient availability on degradation rates, and incorporate into the model.

6. Do not assume the wetland behaves as a completely mixed reactor. Account for BOD transport in the direction of flow.

7. Run the model assuming different types of BOD exist in the stormwater, each with a different diffusion coefficient in water.

Appendix A: Runoff Coefficients

Description of Area	Runoff Coefficients
Business	
Downtown areas	0.70 - 0.95
Neighborhood areas	0.50 - 0.70
Residential	
0.40 - 0.60 Single-family areas	0.30 - 0.50
Multiunits, detached	0.60 - 0.75
Multiunits, attached	0.25 - 0.40
Residential (Suburban)	0.25 -0.40
Apartment dwelling areas	0.50 - 0.70
Industrial	
Light areas	0.50 - 0.80
Heavy areas	0.60 - 0.90
Parks, cemeteries	0.10 - 0.25
Playgrounds	0.20 - 0.35
Railroad yard areas	0.20 - 0.35
Unimproved areas	0.10 - 0.30

Character of Surface	Runoff Coefficients
Streets	
Asphaltic and concerete	0.70 - 0.95
Brick	0.70 - 0.85
Roofs	0.75 - 0.95
Lawns; sandy soil	
Flat, 2%	0.05 - 0.10
Average, 2 to 7%	0.10 - 0.15
Steep, 7%	0.15 - 0.20
Lawns, heavy soil	
Flat, 2%	0.13 - 0.17
Average, 2 to 7%	0.18 - 0.22
Steep, 7%	0.25 - 0.35

(Haan et al., 1994:84)

Appendix B: Constructed Wetland Initial (Baseline) Physical Paramters

Physical Parameter	*Initial Value*	*Range*
Constructed Wetland Surface Area (m^2)	14164	7587-15175
Design Surface Water Depth (m)	.5	.1-.9
Length to Width Ratio	10:1	3:1-15:1
Porosity due to Vegetation	.75	.6-.9
Influent Rate (m^3/hr)	15	15-90
Influent BOD Concentration (mg/liter)	20	20-40

Appendix C: The Constructed Wetland Model

Equations by Sector

Hydrology

CW_volume(t) = CW_volume(t - dt) + (CW_influent_rate - CW_evapotranspiration_&_precip_rate - CW_effluent_rate) * dt

INIT CW_volume = 8498.4
DOCUMENT: The total volume of water in the wetland (subsurface and surface water) in meters cubed. Initial volume is at maximum capacity, assuming bed media porosity = .45 and porosity due to vegetation = .75.

CW_influent_rate = IF(DP_volume>DP_min_volume)THEN(150)ELSE(0)
DOCUMENT: The constructed wetland influent rate is the specified detention pond effluent rate in meters cubed per hour.

CW_evapotranspiration_&_precip_rate = (8*.8*.001*(1/24)-(storm_intensity*.001))*CW_surface_area
DOCUMENT: The evapotranspiration rate & precipitation rate for a wetland is represented by .8 times Class A pan evaporation from an adjacent open site (Hammer: 26) plus the rainfall that falls directly on the wetland. A Class A pan evaporation rate of 8 mm/day is converted to m/h. The storm intensity input is converted from mm/hr to m/hr. The quantity is converted to meters cubed per hour based on the surface area of the wetland.

CW_effluent_rate = IF((CW_influent_rate-CW_evapotranspiration_&_precip_rate)>=0) AND(CW_volume>Initial_CW_Volume)THEN(CW_influent_rate-CW_evapotranspiration_&_precip_rate) ELSE(0)
DOCUMENT: The constructed wetland effluent rate is assumed managed by a water control structure so that a constant volume can be maintained. Using a water balance which takes into consideration the influent rate from the detention pond, precipitation, evapotranspiration and the desired change in volume with time equal to zero, the effluent rate is then determined. There is no effluent until the surface water volume equals or exceeds the initial surface water volume. Units are meters cubed/hr.

DP_volume(t) = DP_volume(t - dt) + (runoff_rate - CW_influent_rate - DP_evapotranspiration_&_precip_rate) * dt

INIT DP_volume = 9633
DOCUMENT: The current volume of water in the detention pond in meters cubed. The initial volume is alos the permanent pool volume and the pond should not be drained below that volume.
runoff_rate = 4.013*runoff_coefficient*storm_intensity*watershed_area
DOCUMENT: The rate of runoff in meters cubed per hour, entering the detention pond.

CW_influent_rate = IF(DP_volume>DP_min_volume)THEN(150)ELSE(0)
DOCUMENT: The constructed wetland influent rate is the specified detention pond effluent rate in meters cubed per hour.

DP_evapotranspiration_&_precip_rate = (8*.7*.001*(1/24)-(storm_intensity*.001))*DP_area
DOCUMENT: The evapotranspiration rate & precipitation rate is based upon Class A pan evaporation from a nearby site times a factor of .7 and the rainfall that falls directly on the detention pond. A Class A pan evaporation rate of 8 mm/day (conversion to meters per hour is included) is used and precipitation is the storm intensity input, converted from mm/hr to meters per hour. The quantity is then converted to meters cubed per hour based on the surface area of the detention pond.

hydraulic_loading_rate = (CW_effluent_rate+CW_influent_rate)/(2*CW_surface_area)*10000
DOCUMENT: The hydraulic loading rate is the average flow rate per unit surface area of the wetland in meters cubed/(hectare-hr). One study suggests a maximum of 200 meters cubed/ha-day.

hydraulic_retention_time = IF(CW_effluent_rate+CW_influent_rate>0) THEN(Surface_Water_Volume/(24*(CW_effluent_rate+CW_influent_rate)/2)) ELSE(0)
DOCUMENT: The hydraulic retention time is the surface water volume of the CW divided by the average flow rate. When there is no flow through the wetland, retention time is set to zero. Conversion is made from hours to days.

runoff_coefficient = .8
DOCUMENT: The runoff coefficient is a unitless value characteristic of the surface of the watershed. The value chosen here may be typical of a paved heavy industrial area or even an airfield.

watershed_area = 100
DOCUMENT: The watershed is the area in acres that is affected by the flow of stormwater.

storm_intensity = GRAPH(TIME)
(0.00, 0.00), (2.00, 0.00), (4.00, 0.00), (6.00, 0.00), (8.00, 0.00), (10.0, 0.00), (12.0, 0.00), (14.0, 0.00), (16.0, 0.00), (18.0, 0.00), (20.0, 0.00), (22.0, 0.00), (24.0, 3.00), (26.0, 3.00), (28.0, 3.00), (30.0, 3.00), (32.0, 0.00), (34.0, 0.00), (36.0, 0.00), (38.0, 0.00), (40.0, 0.00), (42.0, 0.00), (44.0, 0.00), (46.0, 0.00), (48.0, 0.00), (50.0, 0.00), (52.0, 0.00), (54.0, 0.00), (56.0, 0.00), (58.0, 0.00), (60.0, 0.00), (62.0, 0.00), (64.0, 0.00), (66.0, 0.00), (68.0, 0.00), (70.0, 0.00), (72.0, 0.00), (74.0, 0.00), (76.0, 0.00), (78.0, 0.00), (80.0, 0.00), (82.0, 0.00), (84.0, 0.00), (86.0, 0.00), (88.0, 0.00), (90.0, 0.00), (92.0, 0.00), (94.0, 0.00), (96.0, 0.00), (98.0, 0.00), (100, 0.00), (102, 0.00), (104, 0.00), (106, 0.00), (108, 0.00), (110, 0.00), (112, 0.00), (114, 0.00), (116, 0.00), (118, 0.00), (120, 0.00), (122, 0.00), (124, 0.00), (126, 0.00), (128, 0.00), (130, 0.00), (132, 0.00), (134, 0.00), (136, 0.00), (138, 0.00), (140, 0.00), (142, 0.00), (144, 0.00), (146, 0.00), (148, 0.00), (150, 0.00), (152, 0.00), (154, 0.00), (156, 0.00), (158, 0.00), (160, 0.00), (162, 0.00), (164, 0.00), (166, 0.00), (168, 0.00), (170, 0.00), (172, 0.00), (174, 0.00), (176, 0.00), (178, 0.00), (180, 0.00), (182, 0.00), (184, 0.00), (186, 0.00), (188, 0.00), (190, 0.00), (192, 0.00), (194, 0.00), (196, 0.00), (198, 0.00), (200, 0.00), (202,

0.00), (204, 0.00), (206, 0.00), (208, 0.00), (210, 0.00), (212, 0.00), (214, 0.00), (216, 0.00), (218, 0.00), (220, 0.00), (222, 0.00), (224, 0.00), (226, 0.00), (228, 0.00), (230, 0.00), (232, 0.00), (234, 0.00), (236, 0.00), (238, 0.00), (240, 0.00), (242, 0.00), (244, 0.00), (246, 0.00), (248, 0.00), (250, 0.00), (252, 0.00), (254, 0.00), (256, 0.00), (258, 0.00), (260, 0.00), (262, 0.00), (264, 0.00), (266, 0.00), (268, 0.00), (270, 0.00), (272, 0.00), (274, 0.00), (276, 0.00), (278, 0.00), (280, 0.00), (282, 0.00), (284, 0.00), (286, 0.00), (288, 0.00), (290, 0.00), (292, 0.00), (294, 0.00), (296, 0.00), (298, 0.00), (300, 0.00), (302, 0.00), (304, 0.00), (306, 0.00), (308, 0.00), (310, 0.00), (312, 0.00), (314, 0.00), (316, 0.00), (318, 0.00), (320, 0.00), (322, 0.00), (324, 0.00), (326, 0.00), (328, 0.00), (330, 0.00), (332, 0.00), (334, 0.00), (336, 0.00), (338, 0.00), (340, 0.00), (342, 0.00), (344, 0.00), (346, 0.00), (348, 0.00), (350, 0.00), (352, 0.00), (354, 0.00), (356, 0.00), (358, 0.00), (360, 0.00), (362, 0.00), (364, 0.00), (366, 0.00), (368, 0.00), (370, 0.00), (372, 0.00), (374, 0.00), (376, 0.00), (378, 0.00), (380, 0.00), (382, 0.00), (384, 0.00), (386, 0.00), (388, 0.00), (390, 0.00), (392, 0.00), (394, 0.00), (396, 0.00), (398, 0.00), (400, 0.00), (402, 0.00), (404, 0.00), (406, 0.00), (408, 0.00), (410, 0.00), (412, 0.00), (414, 0.00), (416, 0.00), (418, 0.00), (420, 0.00), (422, 0.00), (424, 0.00), (426, 0.00), (428, 0.00), (430, 0.00), (432, 0.00), (434, 0.00), (436, 0.00), (438, 0.00), (440, 0.00), (442, 0.00), (444, 0.00), (446, 0.00), (448, 0.00), (450, 0.00), (452, 0.00), (454, 0.00), (456, 0.00), (458, 0.00), (460, 0.00), (462, 0.00), (464, 0.00), (466, 0.00), (468, 0.00), (470, 0.00), (472, 0.00), (474, 0.00), (476, 0.00), (478, 0.00), (480, 0.00), (482, 0.00), (484, 0.00), (486, 0.00), (488, 0.00), (490, 0.00), (492, 0.00), (494, 0.00), (496, 0.00), (498, 0.00), (500, 0.00)
DOCUMENT: The storm intensity is a graphical input of one or more storm events with intensity given in mm/hr.

<u>Wetland Physical Parameters and Related Calculations</u>

CW_surface_area = 14164
DOCUMENT: The surface area of the constructed wetland in meters squared must be defined. Square meters.

DP_area = 3*(DP_surcharge_volume/3.3)
DOCUMENT: The area of the detention pond is determined from the design surcharge volume, assuming the maximum depth is 2.4 meters (permanent pool depth of 1.3 m and max surcharge volume depth of 1.1 m) and the length to width ratio is 3:1

DP_max_volume = DP_min_volume+DP_surcharge_volume
DP_min_volume = DP_area*1.3
DOCUMENT: The minimum volume is the volume of the permanent pool, based on the assumption that the depth of the detention pond should not dip below 1.3 meters

DP_surcharge_volume = 8151.4
DOCUMENT: This is the surcharge design volume of the detention pond in meters cubed determined by assuming 13 mm of runoff must be collected plus a factor 1.25 to accomodate sediment accumulation.
Ground_Water_Volume = CW_surface_area*.5*.45
DOCUMENT: The ground water volume is essentially constant, assuming conditions are such that the wetland does not dry out. Therefore, the ground water volume is determined by the depth and area of the bed and porosity of the media. Porosity is assumed to be .45, depth = .5 meters. Units are in meters cubed.

Initial_CW_Volume =
(CW_surface_area*SW_Design_Depth*Plant_porosity)+(CW_surface_area*.5*.45)
DOCUMENT: The initial volume of water in the constructed wetland above the bed media, based on design water depth, defined surface area and porosity due to vegetation. Units are cubic meters.

LW_Ratio = 10
DOCUMENT: The constructed wetland length to width ratio.

Plant_Population = Plant_Volume/(PI*(.007^2)*Surface_Water_Depth)
DOCUMENT: Estimates the number of plant stalks based on volume of surface water and porosity due to vegetation. Used to estimate surface area available for microbial population associated with the vegetation. Units are meters squared.

Plant_porosity = .75
DOCUMENT: The porosity of the surface water compartment due to the presents of vegetation. It is the ratio of surface water volume to total volume in the surface water compartment, to include volume taken up by vegetation. Unitless.

Plant_Surface_Area = 2*PI*.007*Surface_Water_Depth*Plant_Population
DOCUMENT: The estimated plant stalk surface area available for microbial population changes with the surface water volume and depth. It is based on an average plant stalk diameter of .014 meters. Units are meters squared.

Plant_Volume = (Surface_Water_Volume/Plant_porosity)-Surface_Water_Volume
DOCUMENT: The volume of vegetation taking up space in the surface water portion of the wetland. It is a funciton of the porosity of the vegetation, and the current surface water volume. Units are meters squared.

Surface_Water_Depth = (Surface_Water_Volume+Plant_Volume)/CW_surface_area
DOCUMENT: The surface water depth changes with surface water volume. The depth of surface water is important as it defines the area of plant stalks available for microbial population. Units are in meters.

Surface_Water_Volume = CW_volume-Ground_Water_Volume
DOCUMENT: The surface water volume will vary based on the CW total volume of water since the ground water volume remains constant. The surface water volume is then equal to the total water volume minus the ground water volume. Units are in meters cubed.

SW_Design_Depth = .5
DOCUMENT: The maximum desired depth of the surface water. Units are meters.

SW_Xsectional_area = Surface_Water_Depth*SQRT(CW_surface_area/LW_Ratio)
DOCUMENT: The cross sectional area of the surface water compartment. This changes as the depth of surface water changes. Units are in meters squared.

removal_efficiency = (Biofilm_Flux+SW_Degradation)/BOD_influent

Degradation Rate Constant Calculations

K20_biofilm = 6.3
DOCUMENT: The assumed first order rate constant applicable to the biofilm at 20 degrees celsius. (Polprasert and Argawalla, 1994: 728). Units are 1/hr.

K20_surface_water = .056*(1-.083*LOG10(67.2/(Total_Organic_Loading_Rate*24))/.056)/24
DOCUMENT: The first order rate constant for degradation of BOD in the surface water at 20 degrees celsius. It is a function of the organic loading rate and the standard first order rate constant equal to 0.056 1/day as reported by Polprasert and Argarwalla. Units are 1/hr.

Kt_biofilm = K20_biofilm*(1.1^(Water_Temperature-20))
DOCUMENT: The first order rate constant in the biofilm at given water temperature. A function of the rate constant at 20 degrees celsius, and the given temperature. Units are 1/hr

Kt_surface_water = IF(K20_surface_water<0)THEN(0)ELSE(K20_surface_water*(1.1^(Water_Temperature-20)))
DOCUMENT: The first order degradation rate in the bulk liquid phase at the given wetland water temperature. It is a function of the rate constant at 20 degrees celsius, and a temperature coefficient assumed to equal 1.1 (US EPA, 1988: 19) Units are 1/hr.

Total_Organic_Loading_Rate = Added_Organic_Loading_Rate+((Macrophyte_DOM_Production+Water_Column_DOM_Production)/(CW_surface_area*100))
DOCUMENT: The total organic load on the system to include macrophyte and water column generation of DOM. Units are kg/ha-hr

Water_Temperature = 18
DOCUMENT: The temperature of the wetland water in degrees celsius. This must be known as it affects the degradation rate.

BOD Concentration in Surface Water and Biofilm Compartments

Surface_Water_BOD(t) = Surface_Water_BOD(t - dt) + (BOD_influent - BOD_effluent - Biofilm_Flux - SW_Degradation) * dt
INIT Surface_Water_BOD = 37605420
DOCUMENT: The mass of BOD in the surface water given in mg.

BOD_influent = (Influent_BOD_concentration*1000*CW_influent_rate)+Macrophyte_DOM_Production+Water_Column_DOM_Production
DOCUMENT: The mass of BOD entering the wetland from stormwater runoff plus that produced by the macrophyte population. This is calculated from the influent BOD concentration provided in mg/liter and the constructed wetland influent flow rate plus the quantity produced within the wetland by macrophytes. Units are in mg/hr.

BOD_effluent = Surface_water_BOD_concentration*1000*CW_effluent_rate
Biofilm_Flux = beta*Conc_at_Interface*Biofilm_Surface_Area*1000
DOCUMENT: Rate of diffusion from the stagnant liquid film into the aerobic biofilm. Units are in mg/hr.

SW_Degradation =
Kt_surface_water*Surface_water_BOD_concentration*Surface_Water_Volume*1000

Added_Organic_Loading_Rate =
CW_influent_rate*Influent_BOD_concentration*10/CW_surface_area
DOCUMENT: The added organic loading rate is the mass of BOD (kg)/ha-hr that the wetland receives due to input from stormwater runoff. A performance data base for wastewater treatment shows that removal efficiencies are near 70% or more at mass loading rates of up to 280 kg/ha/day. (Moshiri: 49). Units are kg/ha-hr.
Influent_BOD_concentration = 60
DOCUMENT: The influent BOD concentration is a measured value given in mg/liter. It may be a function that varies with time.

Macrophyte_DOM_Production = (8+7)*.3*.5*11.42*CW_surface_area
DOCUMENT: The rate of macrophyte DOM production is assumed 30% of the macrophyte annual net primary production. It is assumed that 1/2 the wetland is populated by cattail, and 1/2 by bulrush. . Units of production are mg/hr.

Surface_water_BOD_concentration =
Surface_Water_BOD/(Surface_Water_Volume*1000)
DOCUMENT: The concentration of BOD in the surface water is determined from the mass of BOD in the surface water and the volume of surface water. Units are converted to mg/liter

Water_Column_DOM_Production = .67*Macrophyte_DOM_Production*.10
DOCUMENT: The contribution of the water column primary producers to dissolved organic matter in the system is assumed to be 10% of their net primary productivity. The net primary productivity of the entire system is assumed 60% due to macrophyte productivity, and 40% due to water column primary producers. Based on assumed NPP of the macrophytes, NPP of water column and therefore DOM release can be determined. Units are mg/hr.

Relevant BOD Flux Calculations

Biofilm_Effective_Diffusivity = Diffusivity_Ratio*Diffusivity_in_Water

Biofilm_Surface_Area =
((Biofilm_Thickness+.007)*2*PI*Surface_Water_Depth*Plant_Population)+CW_surface_area+(2*Surface_Water_Depth*(SQRT(LW_Ratio*CW_surface_area)+(CW_surface_area/SQRT(LW_Ratio*CW_surface_area))))
DOCUMENT: The biofilm surface area is the surface area available for biofilm to form and have contact with the surface water. This includes the surface area associated with vegetation stalks and the surface of the constructed wetland bed and walls. Units are meters squared.

Biofilm_Thickness = ((-1)*(Diffusivity_Ratio*Liquid_Film_Thickness)+SQRT((Diffusivity_Ratio*Liquid_Film_Thickness)^2+(Diffusivity_Ratio*Diffusivity_in_Water*Surface_Water_DO_Concentration/(1800))))
DOCUMENT: The thickness of the aerobic biofilm on vegetation stalks and constructed wetland bed surfaces. It is dependant on the stagnant liquid film layer thickness, the dissolved oxygen concentration in the surface water and the aerobic degradation rate. Units are in meters.

Diffusivity_in_Water = 2.54*10^(-6)
DOCUMENT: Approximate diffusivity of the BOD molecule in water at 18 degrees C. Units are meters squared/hr. The value chosen here is that of Toluene, determined by the equation for diffusivity in water of a molecule of given molar volume given in chapter 3.

Diffusivity_Ratio = .3305
DOCUMENT: Ratio of effective diffusivity of the BOD molecule in the biofilm, to the diffusivity in the bulk liquid (water). Data is obtained from Zhang and Bishop. This ratio represents the average ratio over the biofilm. A thin biofilm (< 500 micrometers) is assumed.

Energy_dissipation_rate = IF(CW_influent_rate+CW_effluent_rate>0)
THEN(9.81*(CW_influent_rate+CW_effluent_rate)/(2*3600*SW_Xsectional_area))
ELSE(10^(-4))
DOCUMENT: The energy dissipation rate is a factor which determines the thickness of the stagnant liquid film on the biofilm surface, which inturn is used to determine the mass transfer coefficient for transfer of BOD from the surface water to the biofilm. It is a function of the velocity of the water flow and thegradient of the CW bed. Units are in meters squared per second cubed.

Liquid_Film_Thickness = ((Energy_dissipation_rate/(10^(-18)))^(-.25))
DOCUMENT: The thickness of the stagnant liquid layer that surrounds the aerobic biofilm on the plant stalks. This thickness is a function of the energy dissipation rate and the viscosity of the liquid (water). Units are in meters.

Surface_Water_DO_Concentration = 8
DOCUMENT: The dissolved oxygen concentration in the surface water. Units are in mg/L (or g/meters cubed).

<u>Biofilm Flux Calculations 2</u>

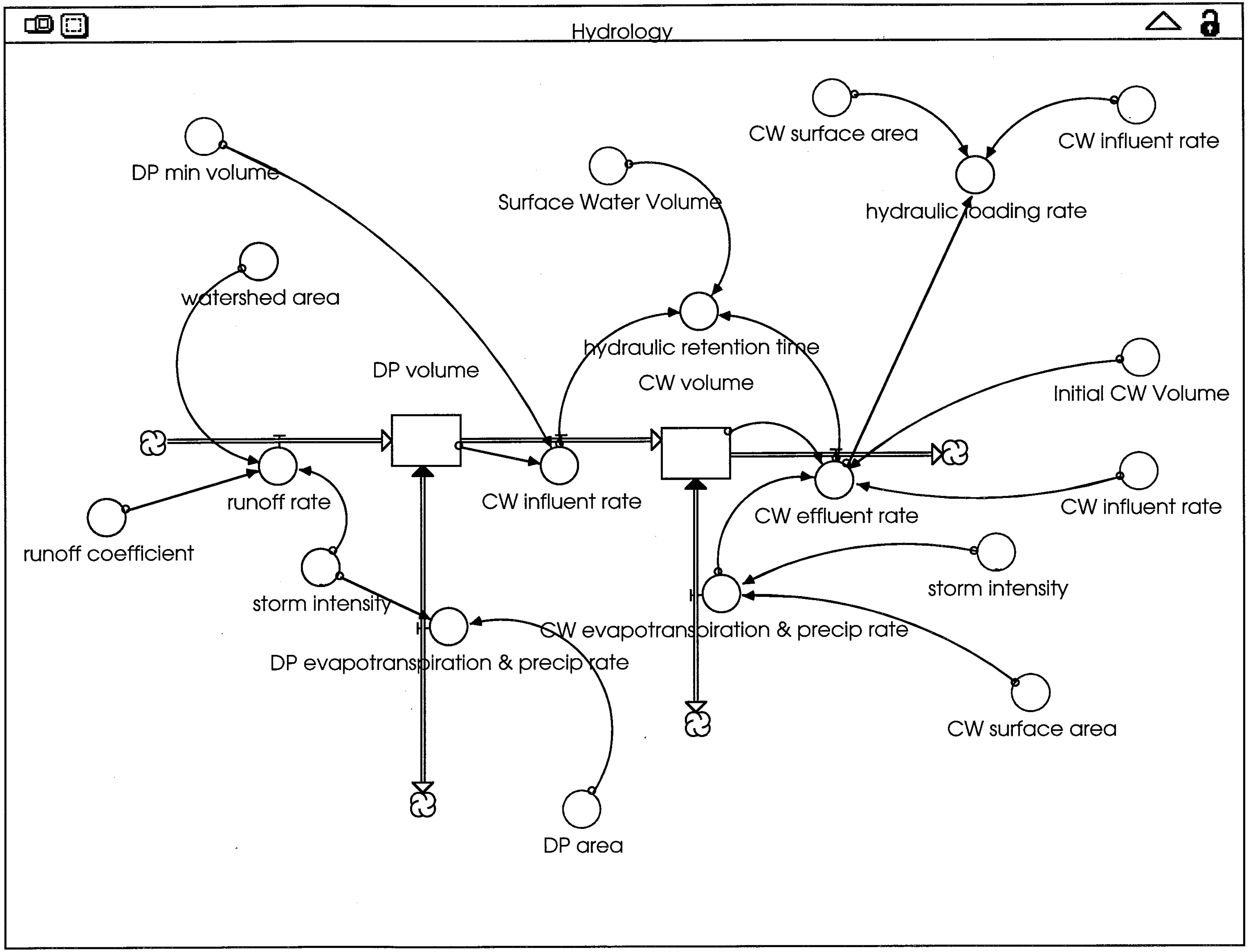
alpha = Diffusivity_in_Water/Liquid_Film_Thickness
DOCUMENT: Units are meters/hr.

beta = (tanhphi/phi)*Kt_biofilm*Biofilm_Thickness
DOCUMENT: Units are meters/hr.

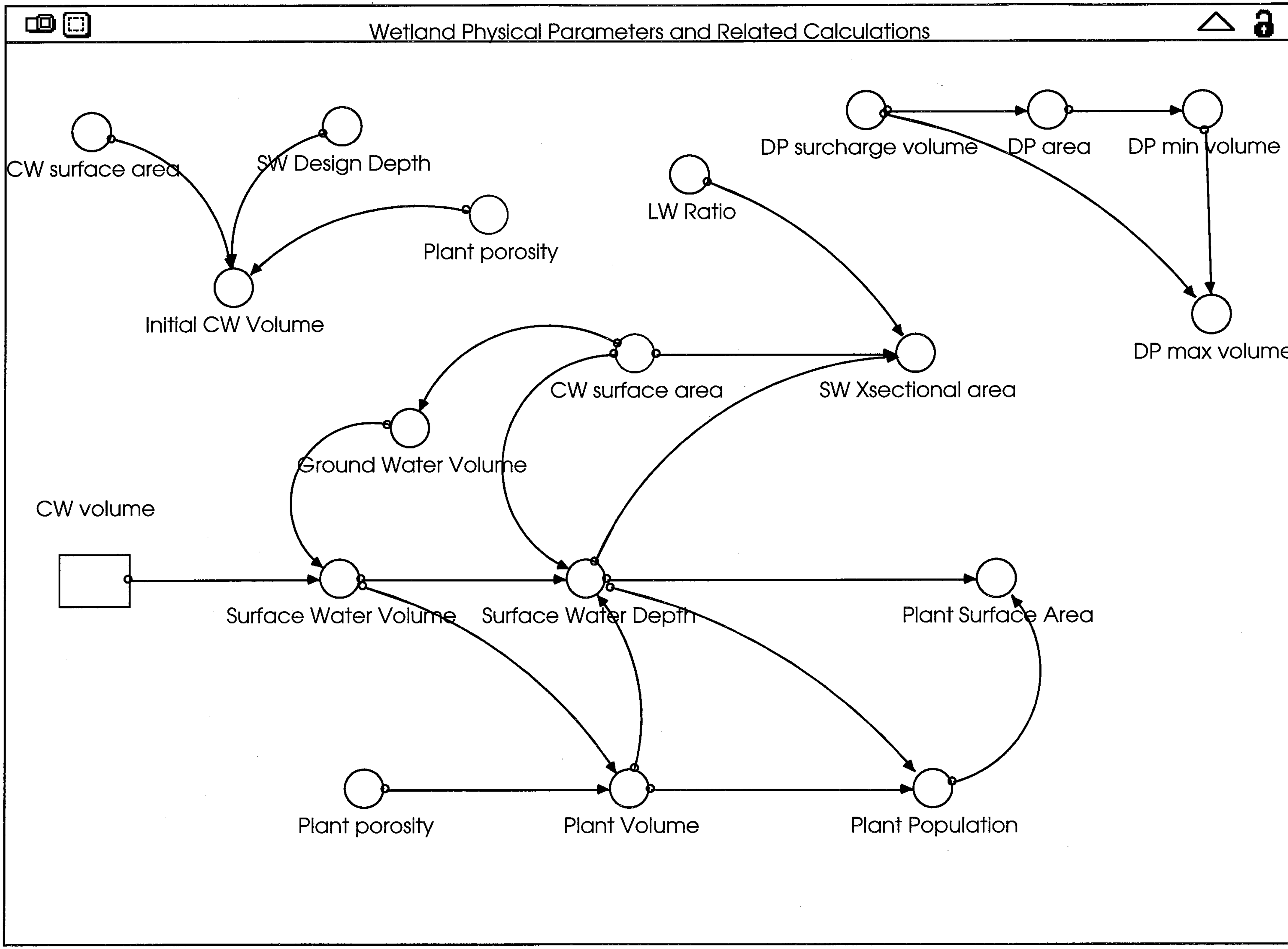
Conc_at_Interface = (alpha/(alpha+beta))*Surface_water_BOD_concentration
DOCUMENT: The concentration of BOD at the interface of the liquid film layer and the biofilm. Units are mg/liter.

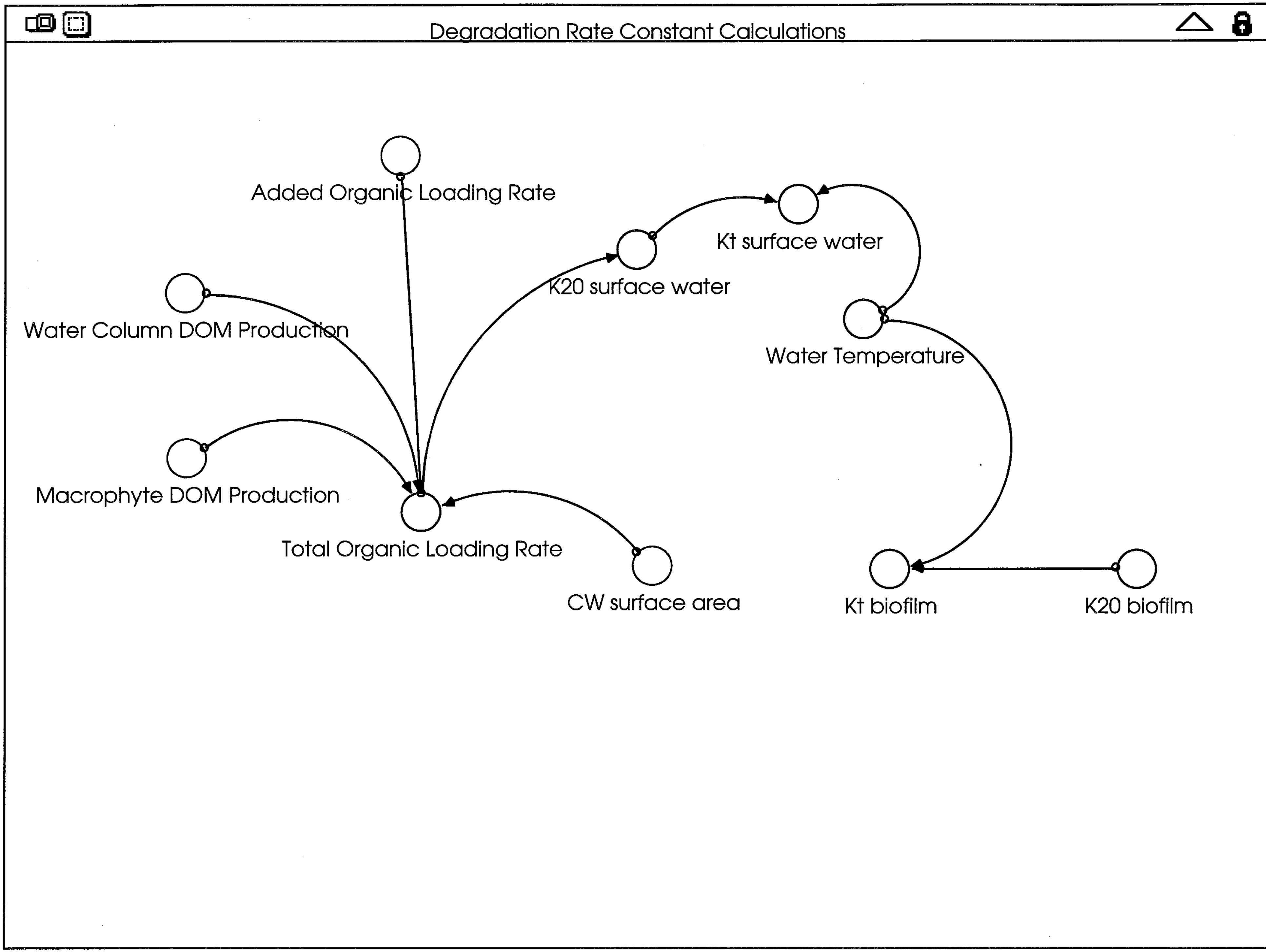
phi = SQRT(Kt_biofilm*(Biofilm_Thickness^2)/Biofilm_Effective_Diffusivity)
DOCUMENT: Unitless characteristic biofilm parameter.

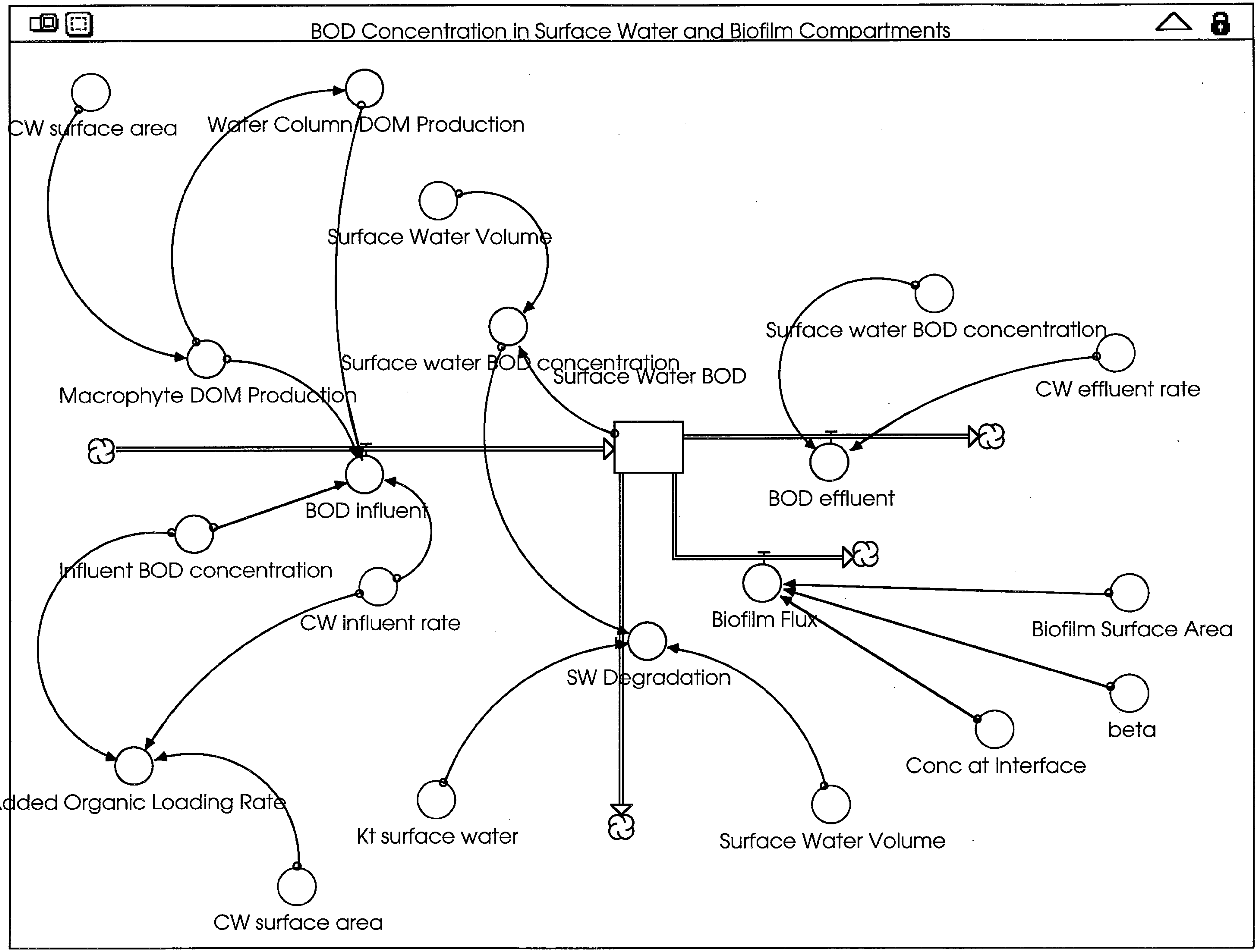
tanhphi = (EXP(phi)-EXP(-phi))/(EXP(phi)+EXP(-phi))

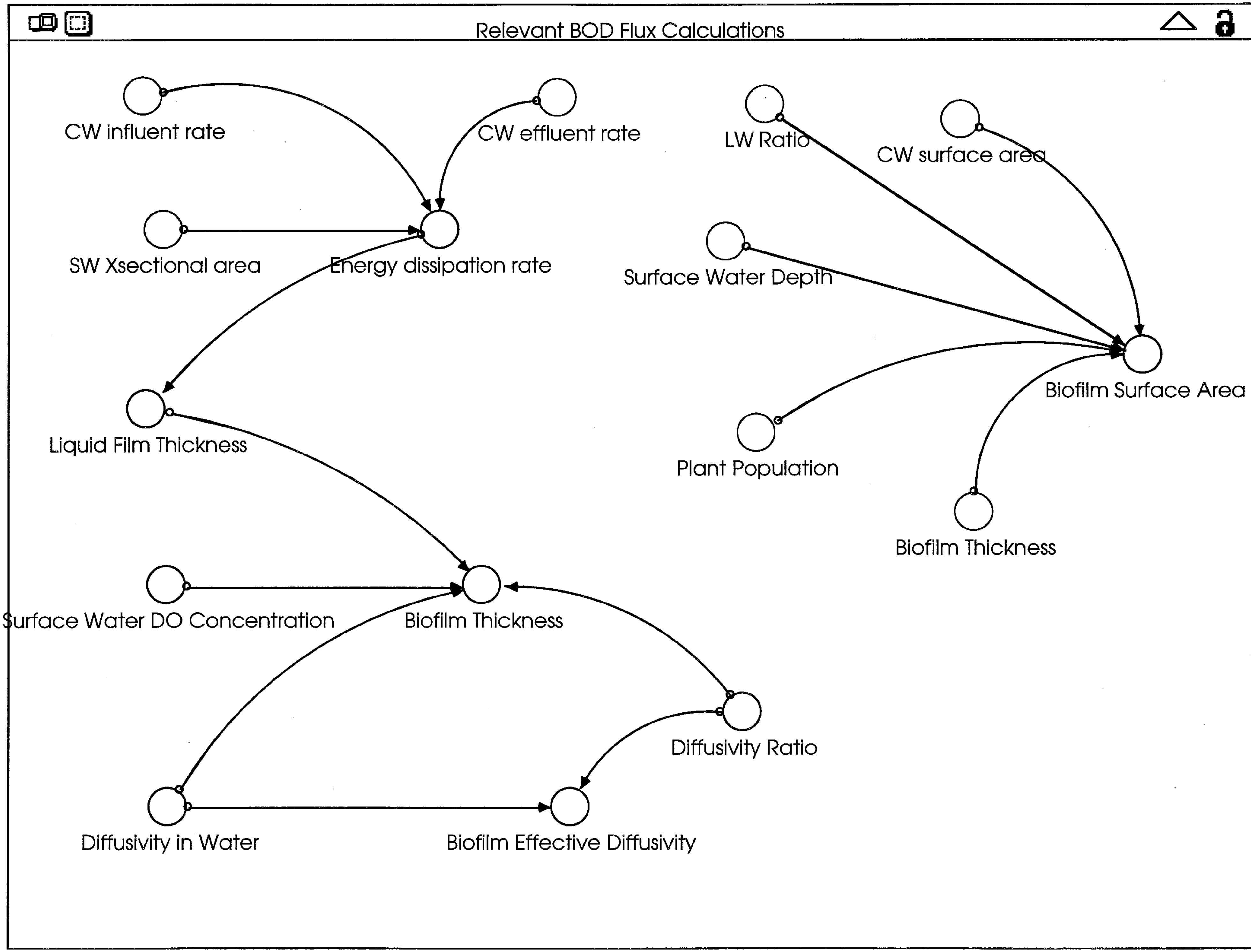


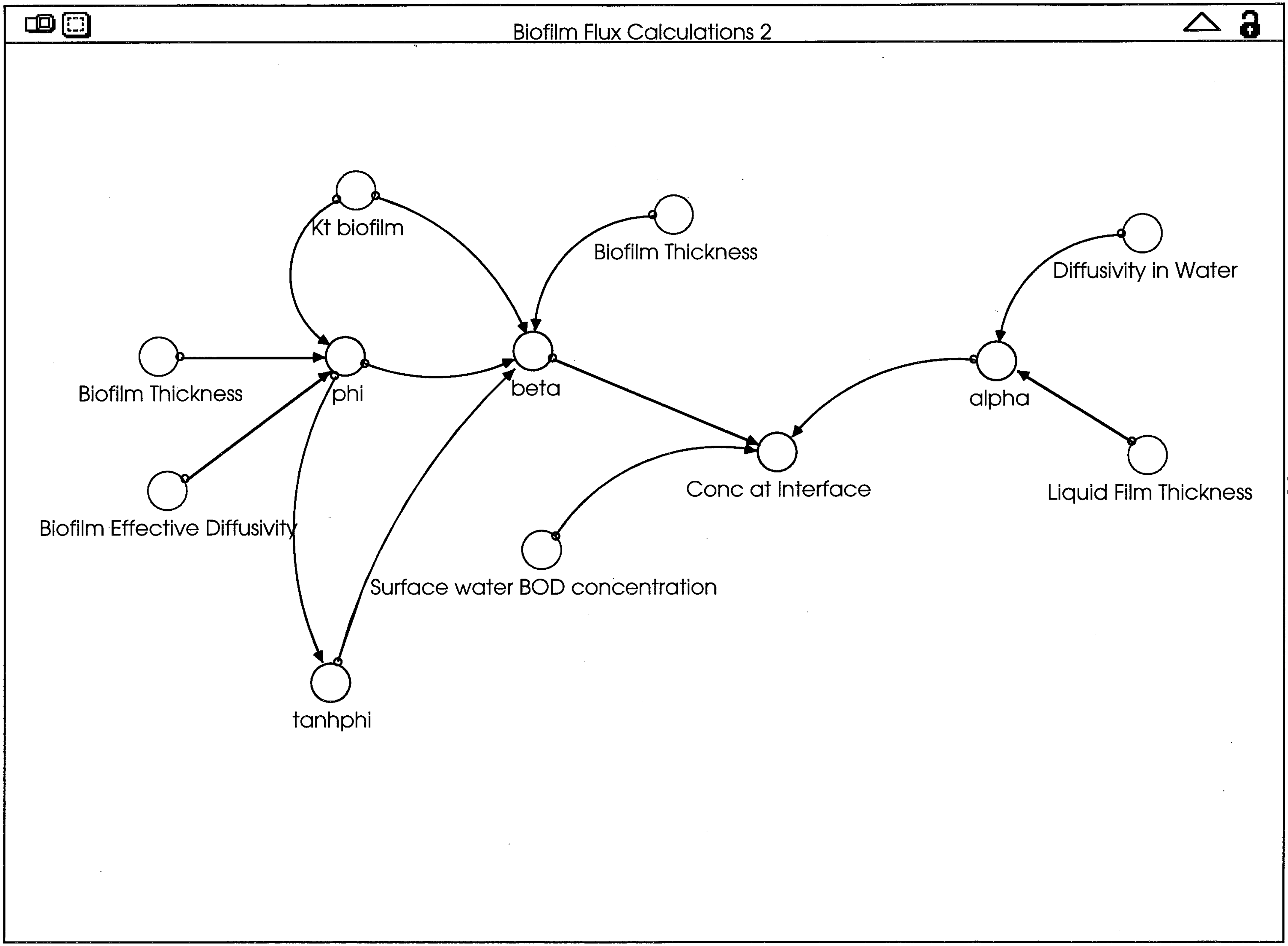
Hydrology
DP min volume
watershed area
runoff rate
runoff coefficient
storm intensity
DP volume
DP evapotranspiration & precip rate
DP area
CW influent rate
Surface Water Volume
hydraulic retention time
CW volume
CW surface area
CW influent rate
hydraulic loading rate
Initial CW Volume
CW effluent rate
CW influent rate
storm intensity
CW evapotranspiration & precip rate
CW surface area


Wetland Physical Parameters and Related Calculations
CW surface area
SW Design Depth
Plant porosity
Initial CW Volume
LW Ratio
DP surcharge volume
DP area
DP min volume
DP max volume
CW surface area
SW Xsectional area
Ground Water Volume
CW volume
Surface Water Volume
Surface Water Depth
Plant Surface Area
Plant porosity
Plant Volume
Plant Population


Degradation Rate Constant Calculations
Added Organic Loading Rate
Water Column DOM Production
Macrophyte DOM Production
Total Organic Loading Rate
K20 surface water
Kt surface water
Water Temperature
CW surface area
Kt biofilm
K20 biofilm

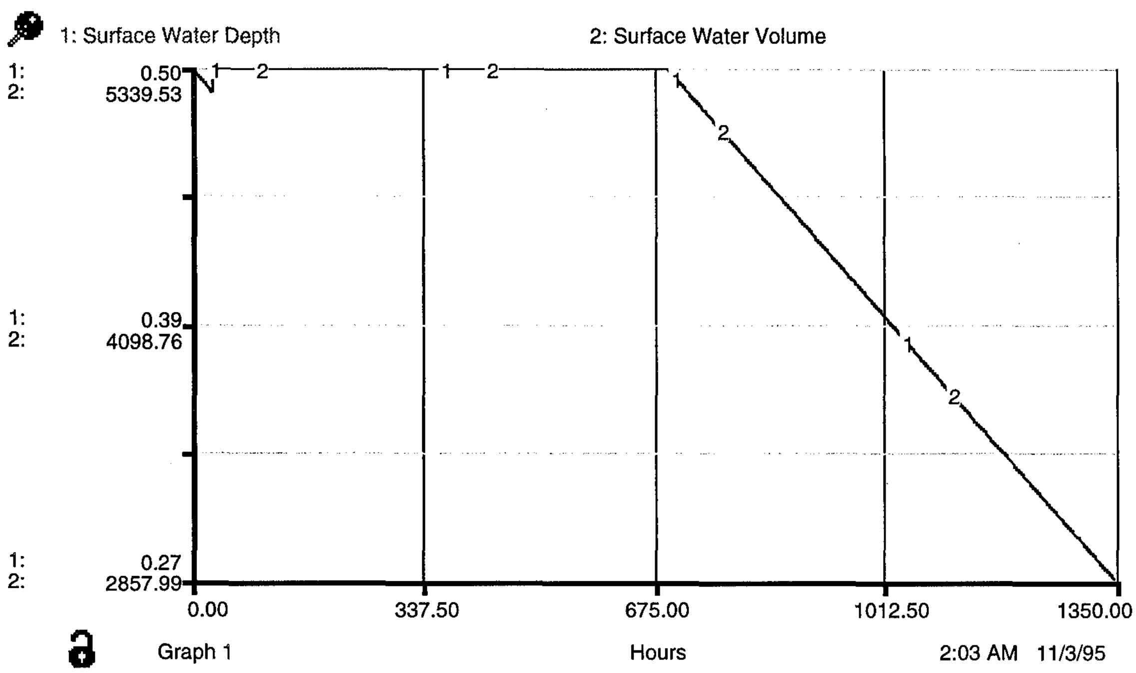

BOD Concentration in Surface Water and Biofilm Compartments
CW surface area
Water Column DOM Production
Surface Water Volume
Macrophyte DOM Production
Surface water BOD concentration
Surface Water BOD
BOD influent
Influent BOD concentration
CW influent rate
Added Organic Loading Rate
CW surface area
Kt surface water
SW Degradation
Surface Water Volume
Surface water BOD concentration
CW effluent rate
BOD effluent
Biofilm Flux
Biofilm Surface Area
beta
Conc at Interface


Relevant BOD Flux Calculations
CW influent rate
CW effluent rate
SW Xsectional area
Energy dissipation rate
Liquid Film Thickness
Surface Water DO Concentration
Biofilm Thickness
Diffusivity Ratio
Diffusivity in Water
Biofilm Effective Diffusivity
LW Ratio
CW surface area
Surface Water Depth
Biofilm Surface Area
Plant Population
Biofilm Thickness

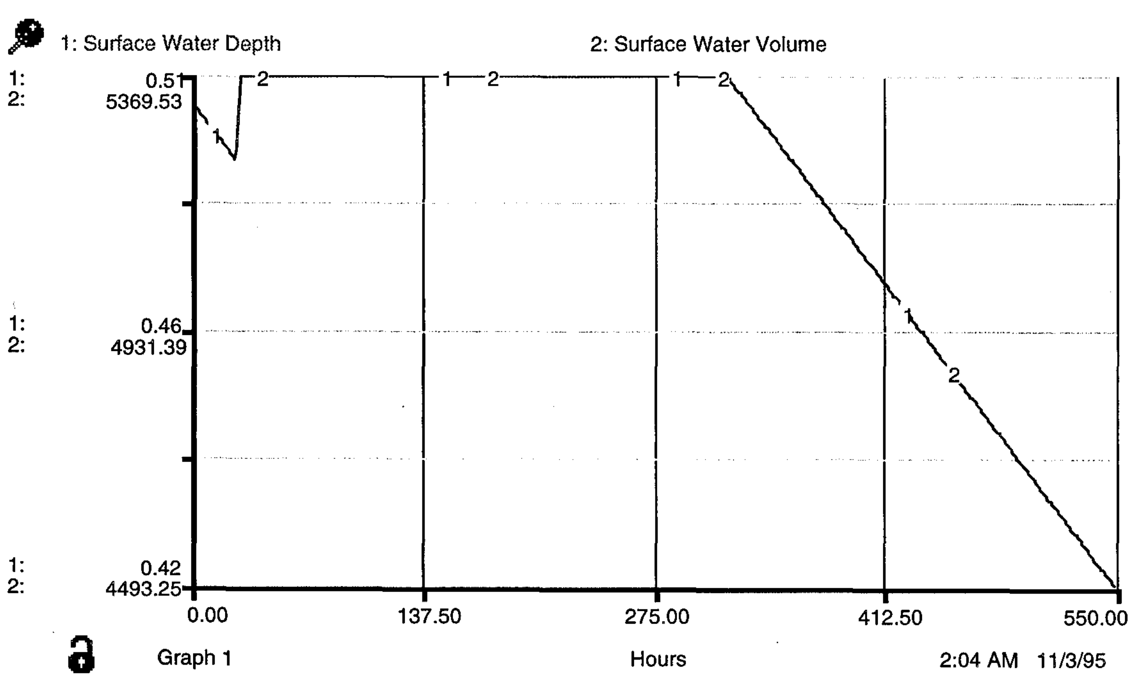

Biofilm Flux Calculations 2
Kt biofilm
Biofilm Thickness
Diffusivity in Water
Biofilm Thickness
phi
beta
alpha
Biofilm Effective Diffusivity
Conc at Interface
Liquid Film Thickness
Surface water BOD concentration
tanhphi

Appendix D: Water Budget for Various Influent Rates

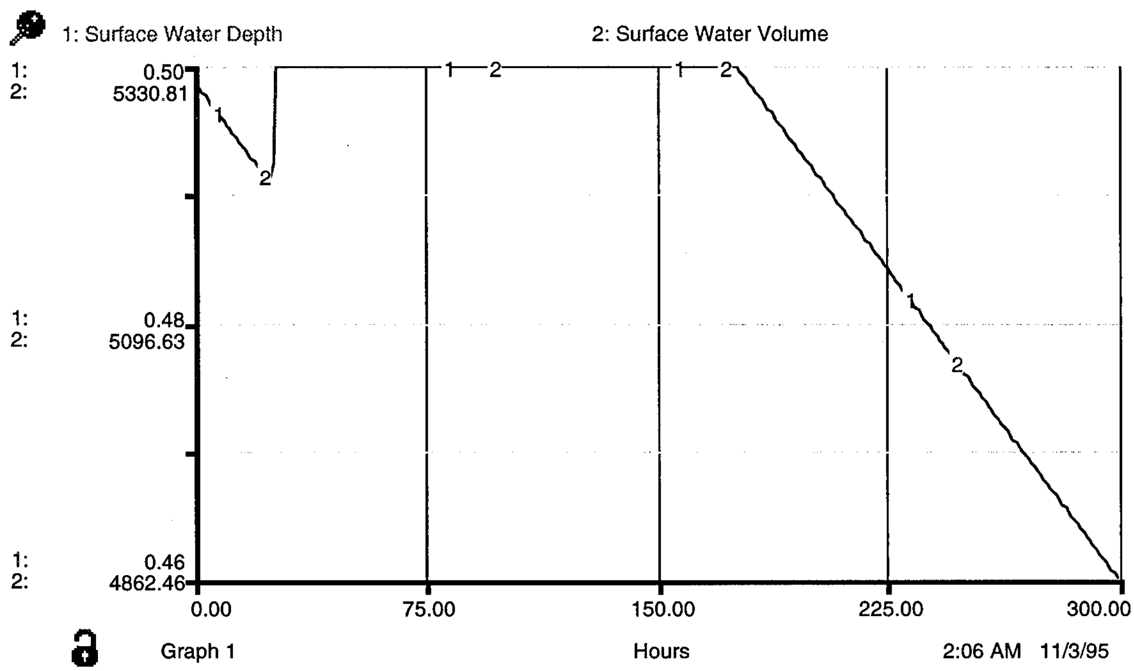
Constructed Wetland Influent Rate 10 m^3/hr



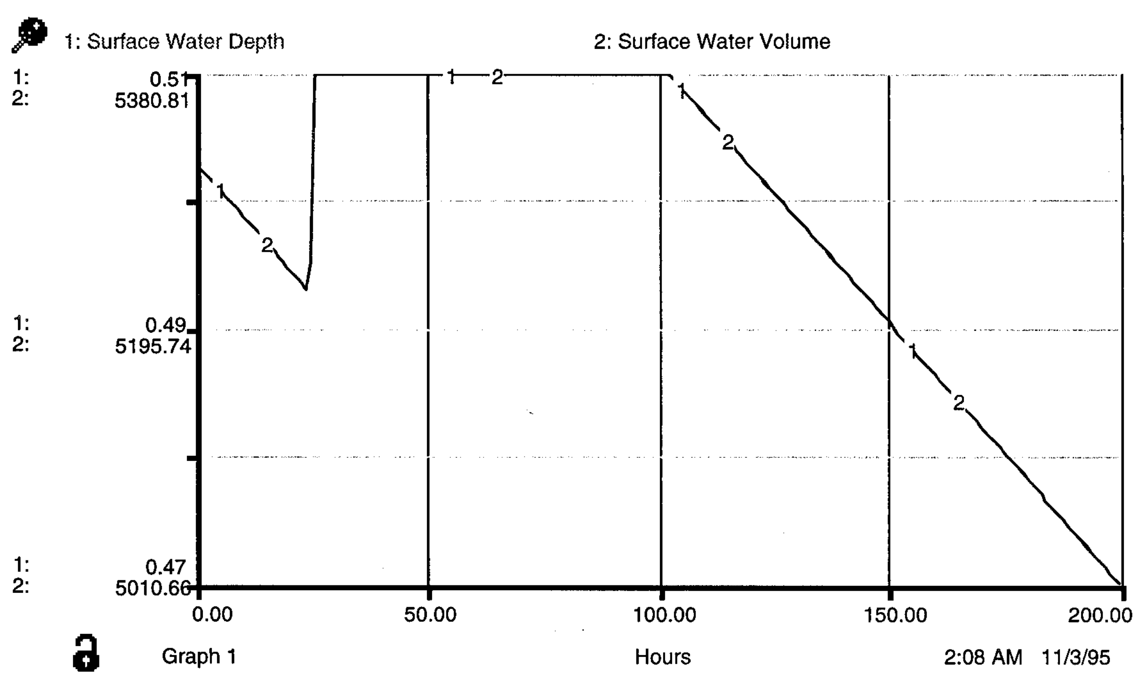
Constructed Wetland Influent Rate 25 m^3/hr



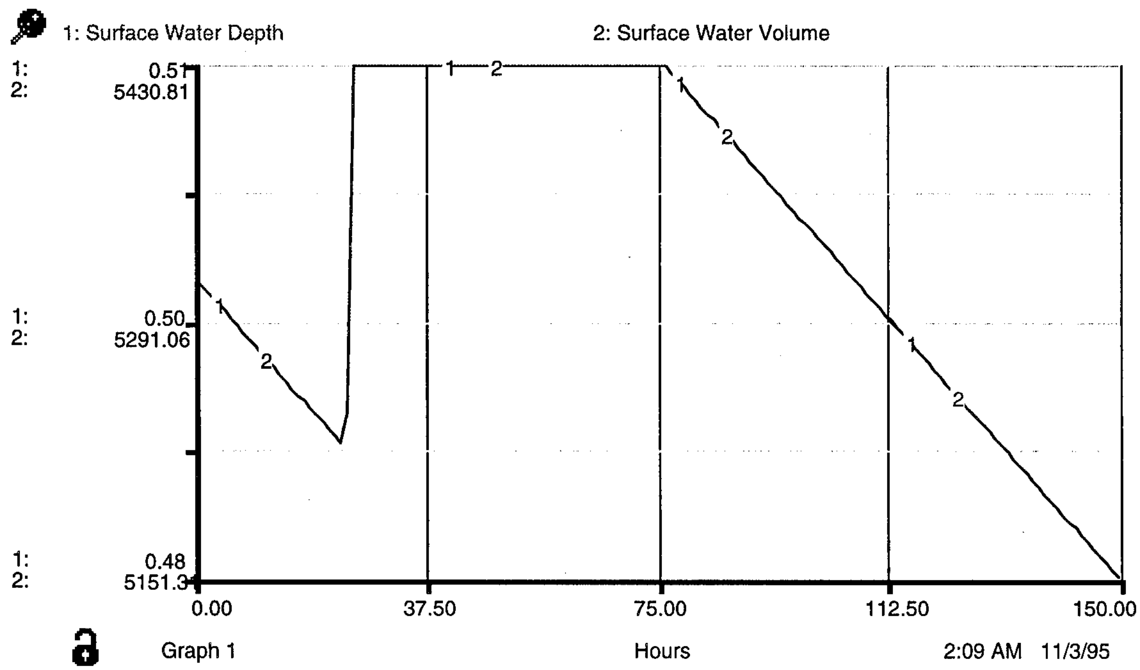
Constructed Wetland Influent Rate 50 m^3/hr



Constructed Wetland Influent Rate 100 m^3/hr



Constructed Wetland Influent Rate 150 m^3/hr


1: Surface Water Depth
2: Surface Water Volume
1: 0.51
2: 5430.81
1: 0.50
2: 5291.06
1: 0.48
2: 5151.31
0.00
37.50
75.00
112.50
150.00
Graph 1
Hours
2:09 AM 11/3/95

Appendix E: Results from Constant Volume Conditions

Constant Volume Conditions, no storm event.
Organic load due to macrophyte and algae generation only.
Baseline parameters established, each varied individually (**Bolded**).
Objectives: Establish equilibrium BOD concentration in wetland and note trends due to varying physical paramters.

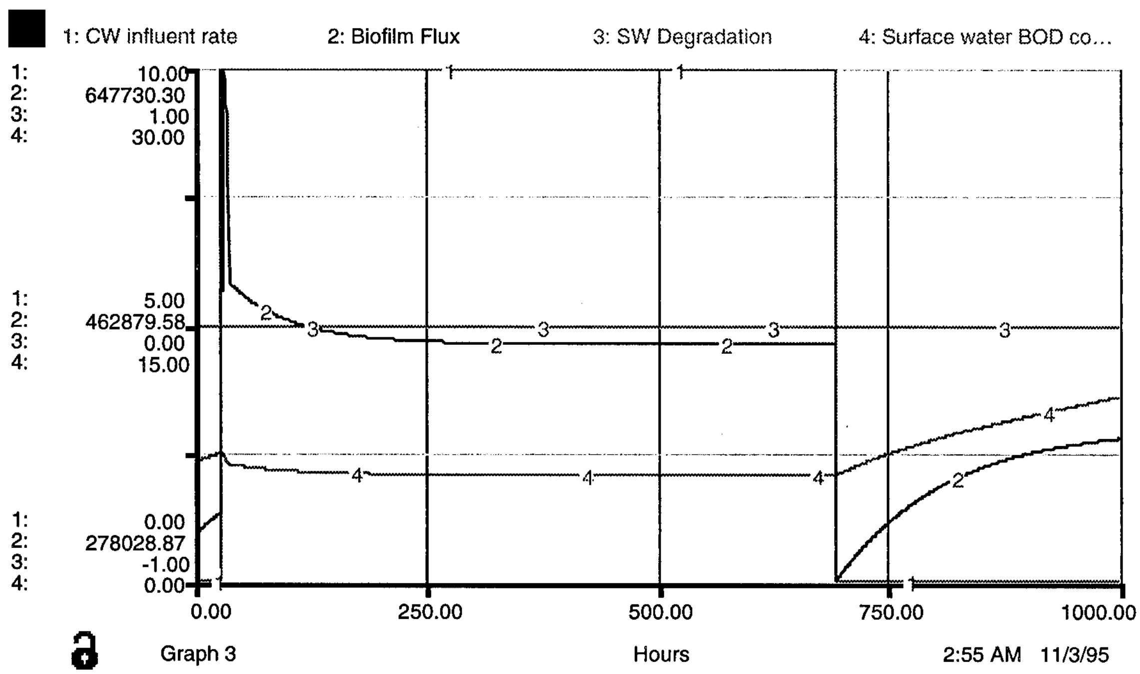
Parameter				
CW Surface Area (square meters):	**7578**	**10000**	**14164**	**15175**
Surface Water Initial Depth (m):	0.5	0.5	0.5	0.5
CW Initial Volume (cubic meters):	4546.8	6000	8498.4	9105
Length to Width Ratio :	10	10	10	10
Influent Rate (cubic meters/hr):	EVAPOTRANSPIRATION RATE			
Influent BOD Concentration (mg/liter):	0	0	0	0
Vegetative Porosity:	0.75	0.75	0.75	0.75
Dissolved Oxygen Concentration (mg/liter):	8	8	8	8
Initial BOD Concentration (mg/liter):	0	0	0	0
Observed Data				
Run Time (hours):	700	700	700	700
Hydraulic Retention Time (days):	117.19	117.19	117.19	117.19
Hydraulic Loading Rate (cm/ha-day):	1.33	1.33	1.33	1.33
Total Organic Loading Rate (kg/ha-hr):	0.27	0.27	0.27	0.27
Final BOD Concentration (mg/liter):	**7.54**	**7.33**	**7.08**	**7.03**

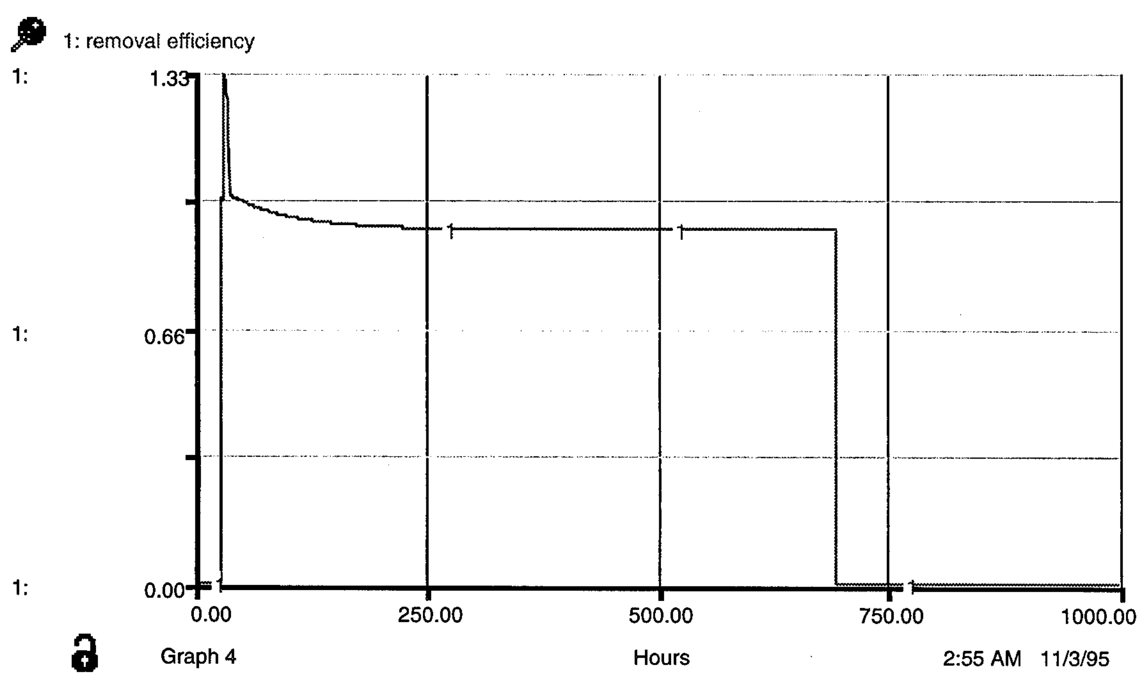
Parameter				
CW Surface Area (square meters):	14164	14164	14164	14164
Surface Water Initial Depth (m):	**0.25**	**0.5**	**0.75**	**0.9**
CW Initial Volume (cubic meters):	5842.65	8498.4	11154.15	12747.6
Length to Width Ratio :	10	10	10	10
Influent Rate (cubic meters/hr):	EVAPOTRANSPIRATION RATE			
Influent BOD Concentration (mg/liter):	0	0	0	0
Vegetative Porosity:	0.75	0.75	0.75	0.75
Dissolved Oxygen Concentration (mg/liter):	8	8	8	8
Initial BOD Concentration (mg/liter):	0	0	0	0
Observed Data				
Run Time (hours):	700	700	700	700
Hydraulic Retention Time (days):	58.59	117.19	175.78	210.94
Hydraulic Loading Rate (cm/ha-day):	1.33	1.33	1.33	1.33
Total Organic Loading Rate (kg/ha-hr):	0.27	0.27	0.27	0.27
Final BOD Concentration (mg/liter):	**12.06**	**7.08**	**5.17**	**4.49**

Parameter				
CW Surface Area (square meters):	14164	14164	14164	14164
Surface Water Initial Depth (m):	0.5	0.5	0.5	0.5
CW Initial Volume (cubic meters):	8498.4	8498.4	8498.4	8498.4
Length to Width Ratio :	**3**	**5**	**10**	**15**
Influent Rate (cubic meters/hr):	EVAPOTRANSPIRATION RATE			
Influent BOD Concentration (mg/liter):	0	0	0	0
Vegetative Porosity:	0.75	0.75	0.75	0.75
Dissolved Oxygen Concentration (mg/liter):	8	8	8	8
Initial BOD Concentration (mg/liter):	0	0	0	0
Observed Data				
Run Time (hours):	700	700	700	700
Hydraulic Retention Time (days):	117.19	117.19	117.19	117.19
Hydraulic Loading Rate (cm/ha-day):	1.33	1.33	1.33	1.33
Total Organic Loading Rate (kg/ha-hr):	0.27	0.27	0.27	0.27
Final BOD Concentration (mg/liter):	**8.01**	**7.6**	**7.08**	**6.8**

Appendix F: BOD Concentrations and Removal Efficiencies for the Baseline CW

Influent Rate 10 m^3/hr
Influent Concentration 10 mg/l





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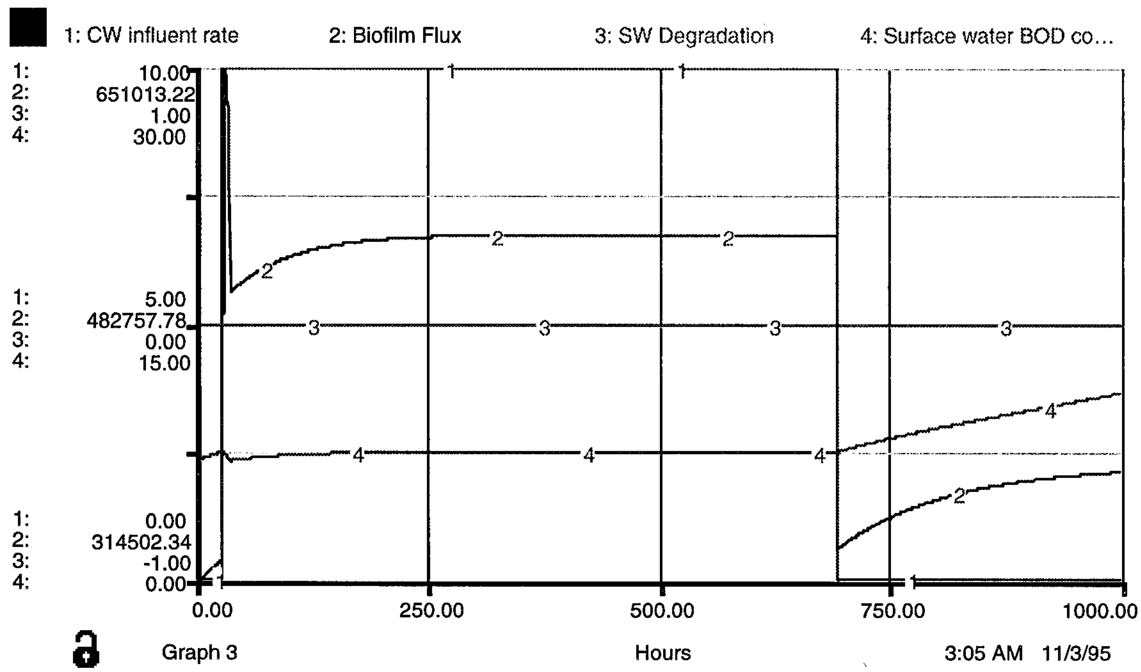

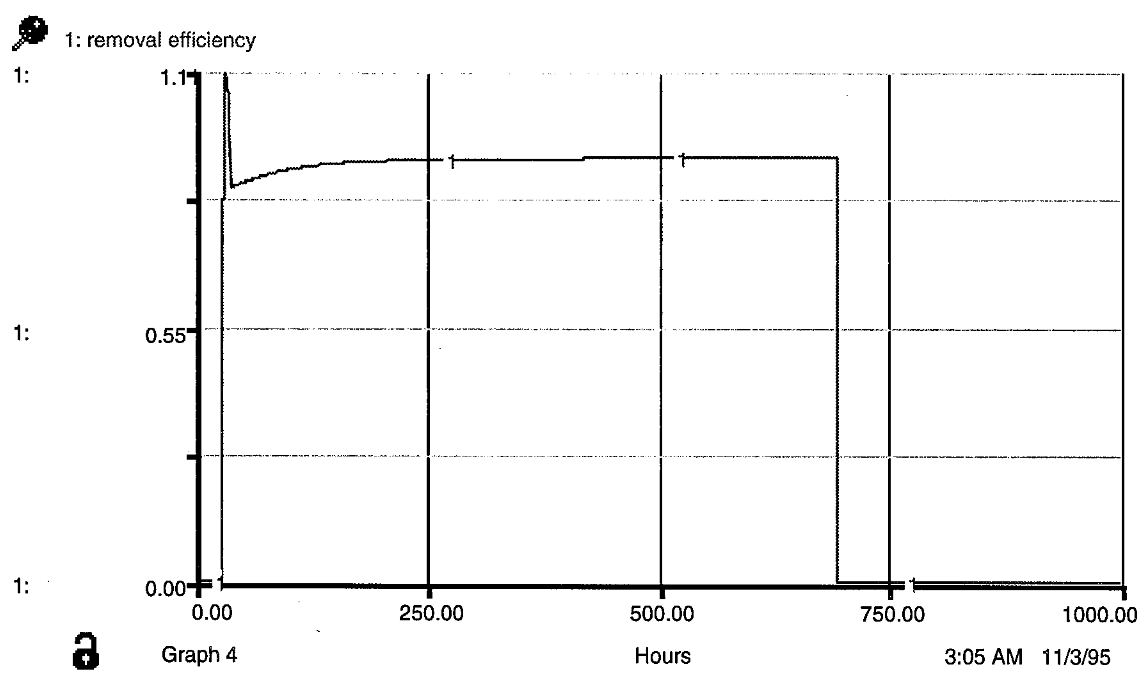

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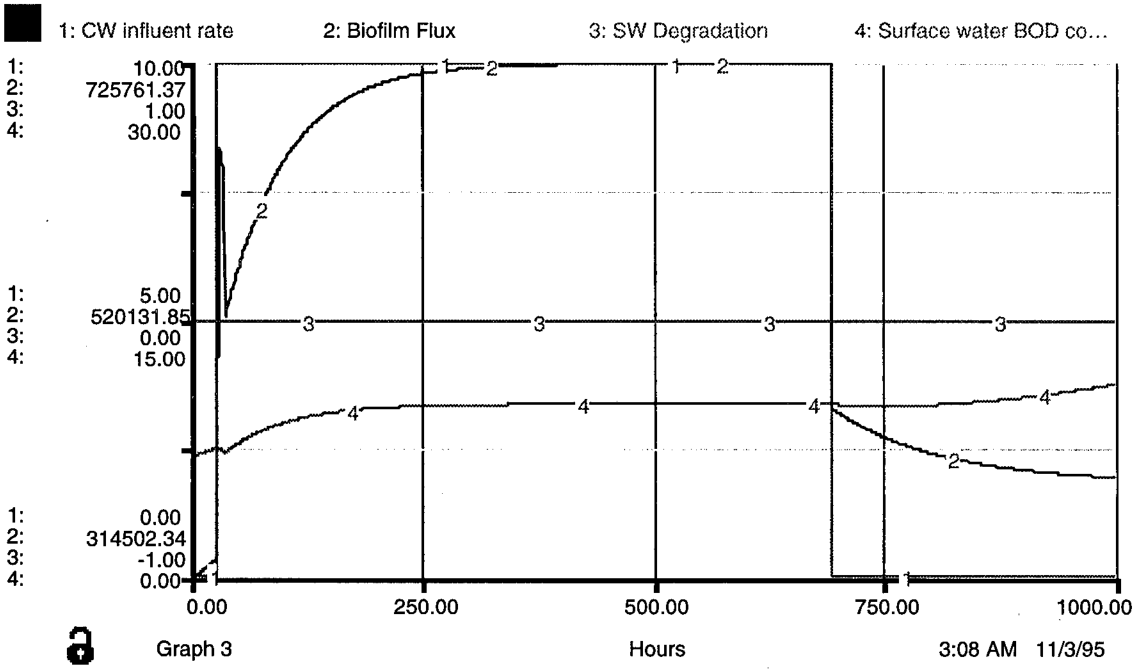

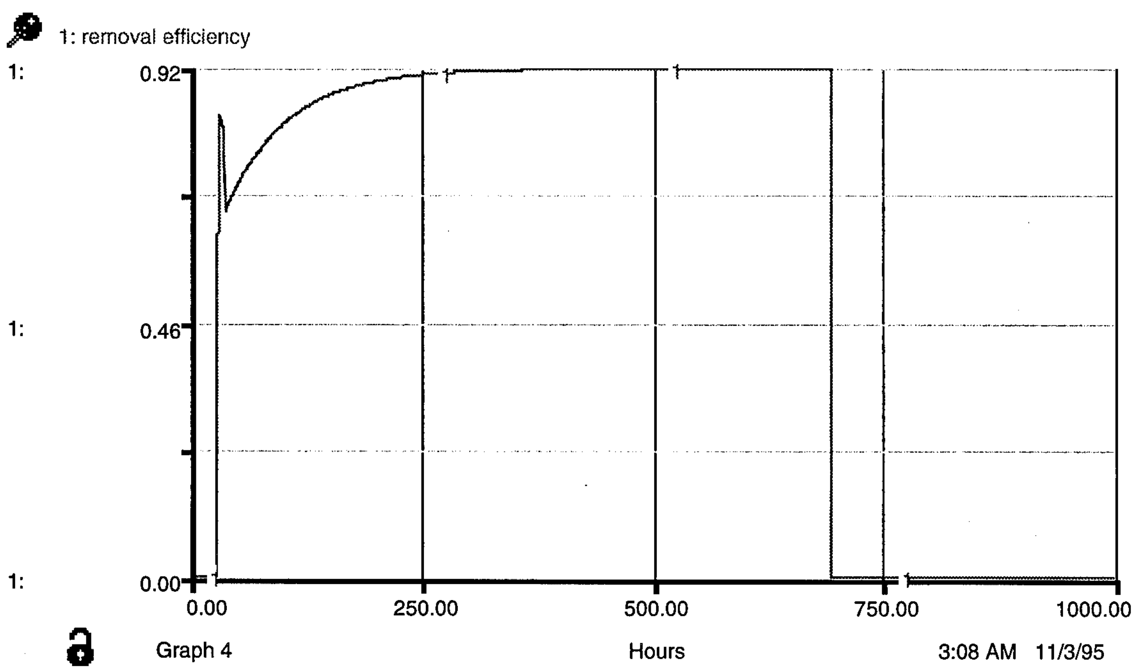

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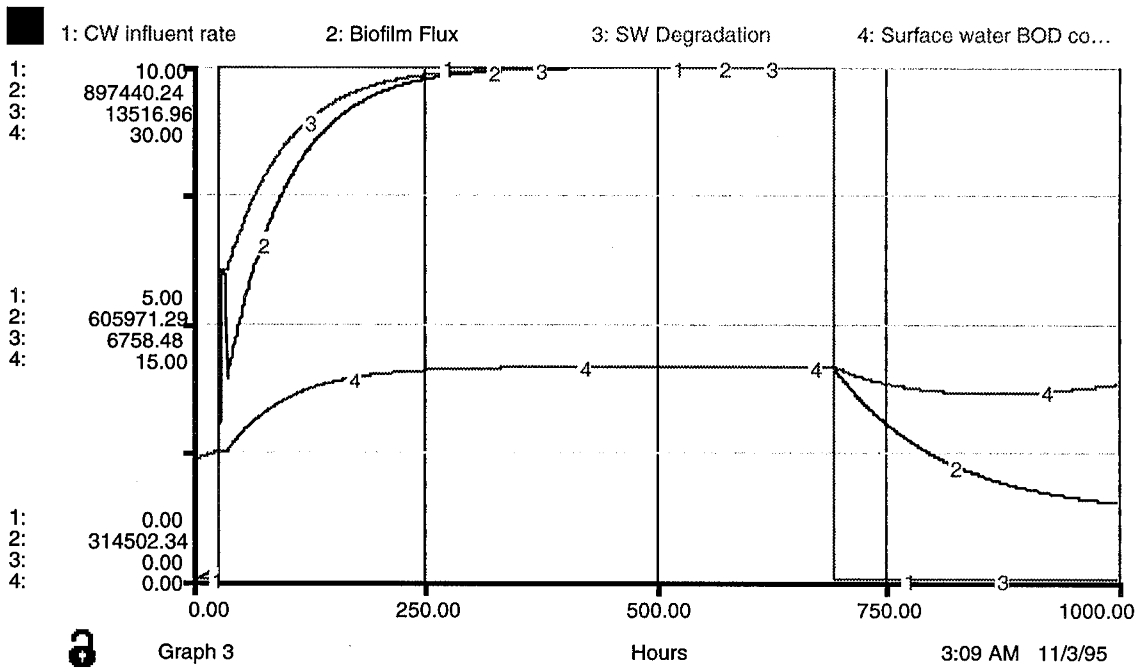

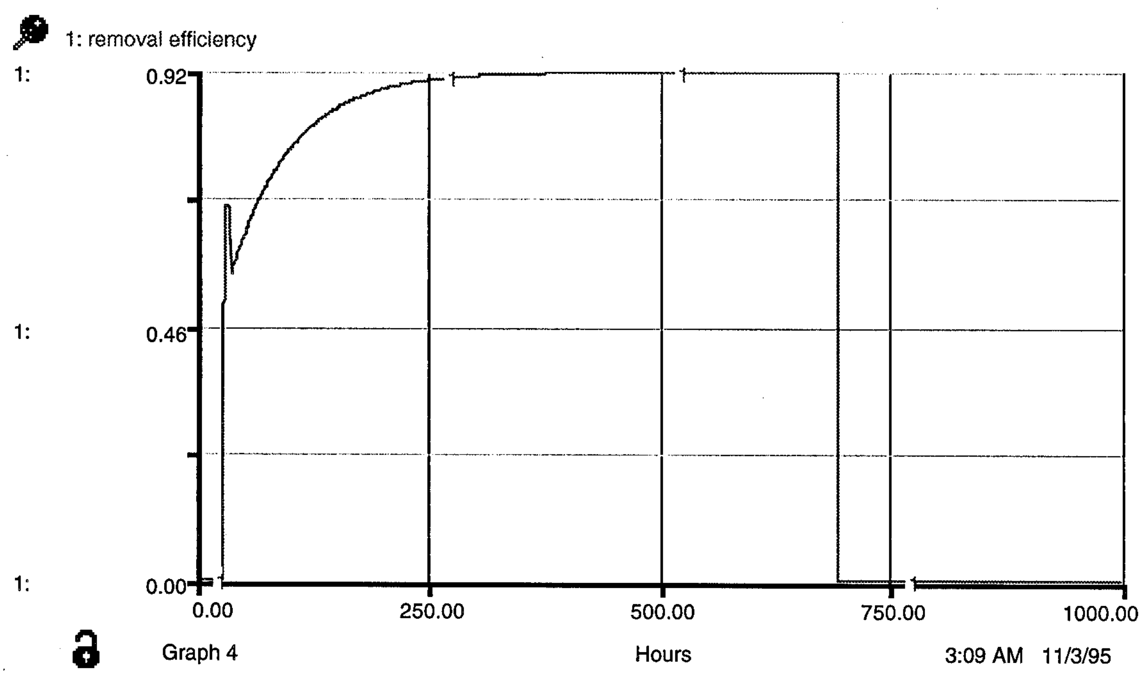

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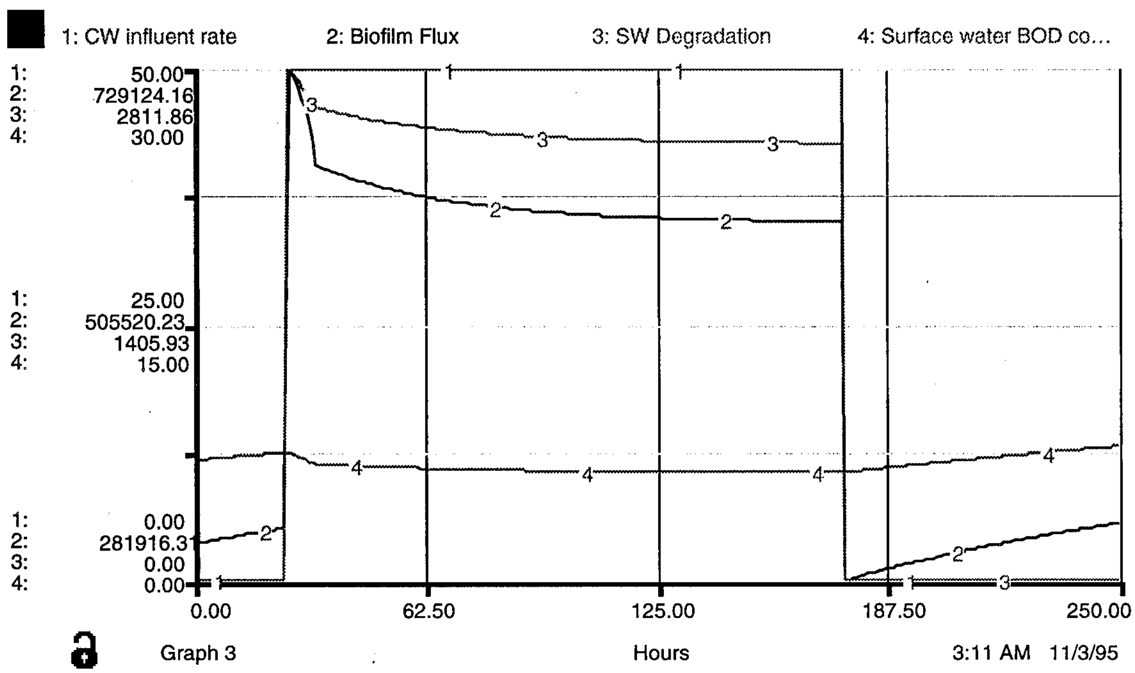

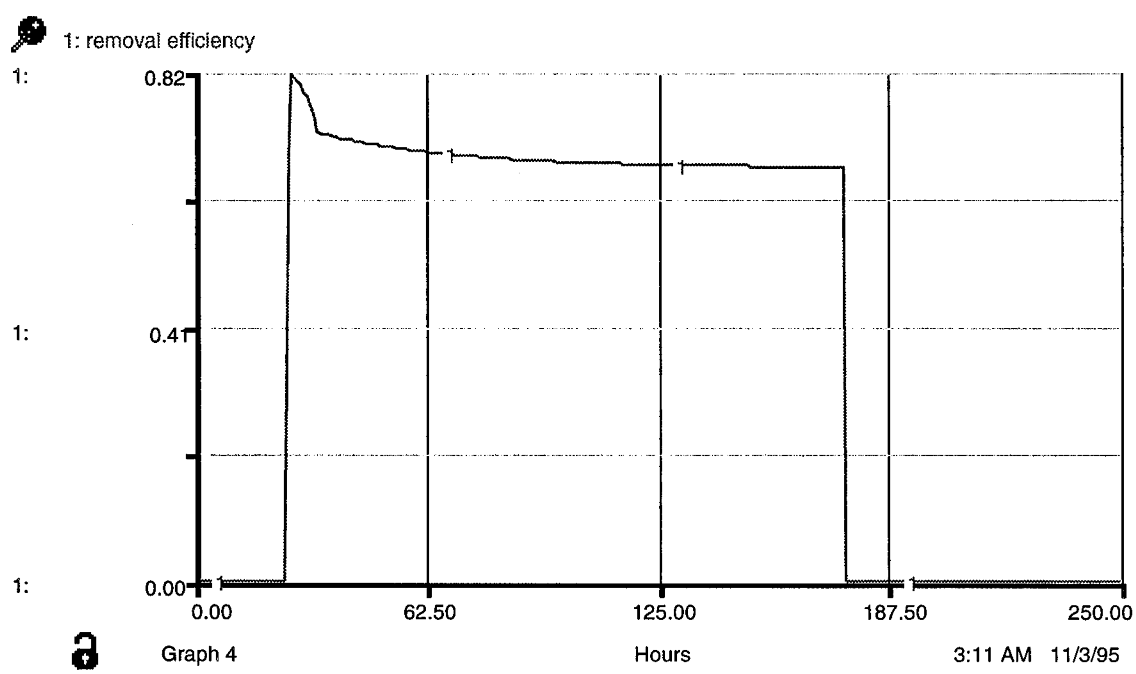

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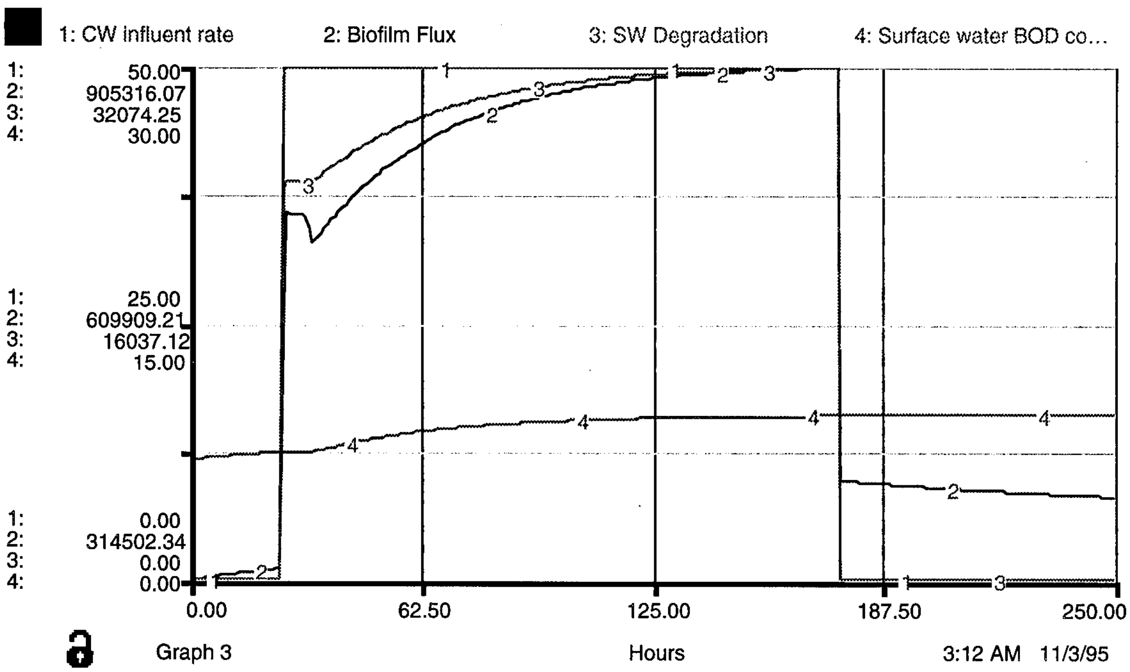

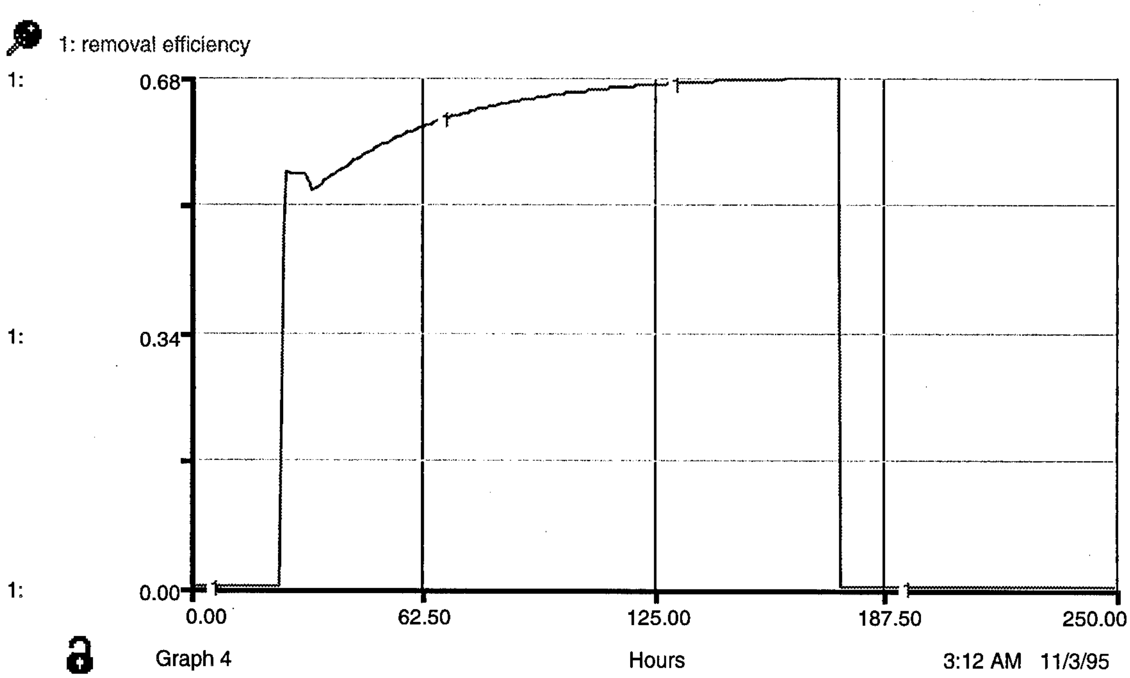

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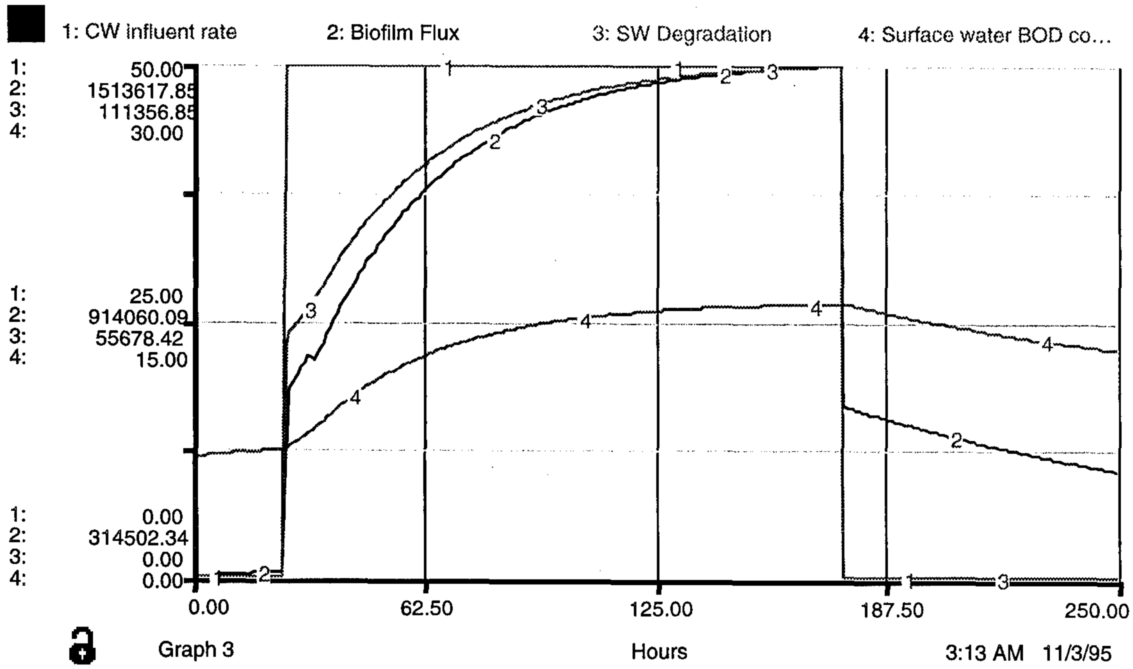

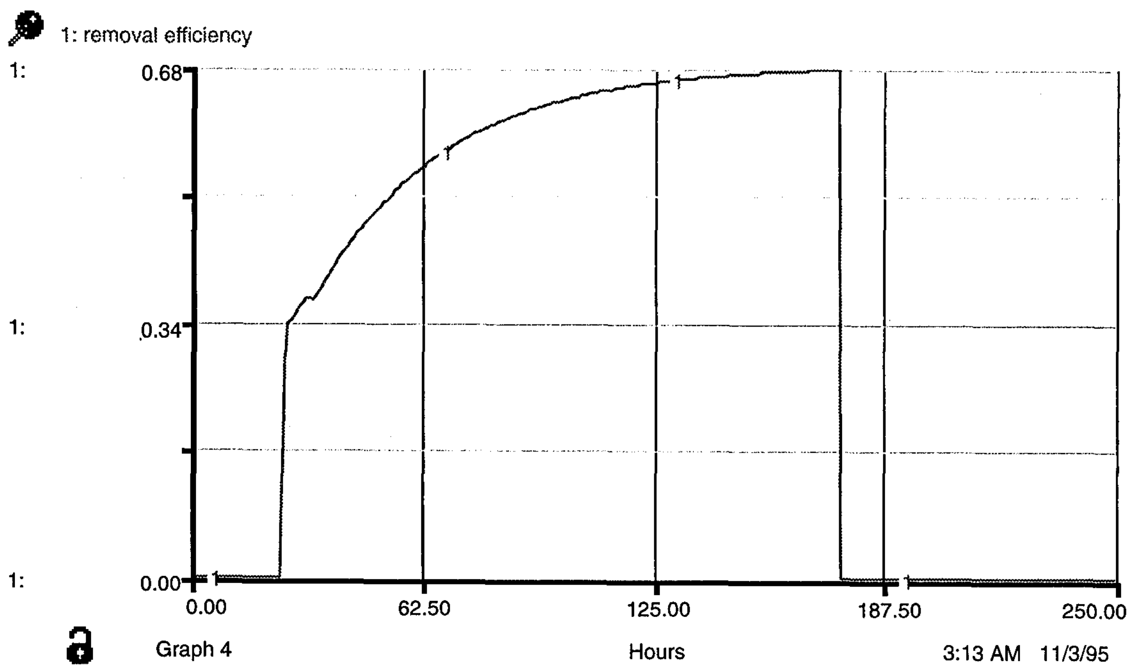

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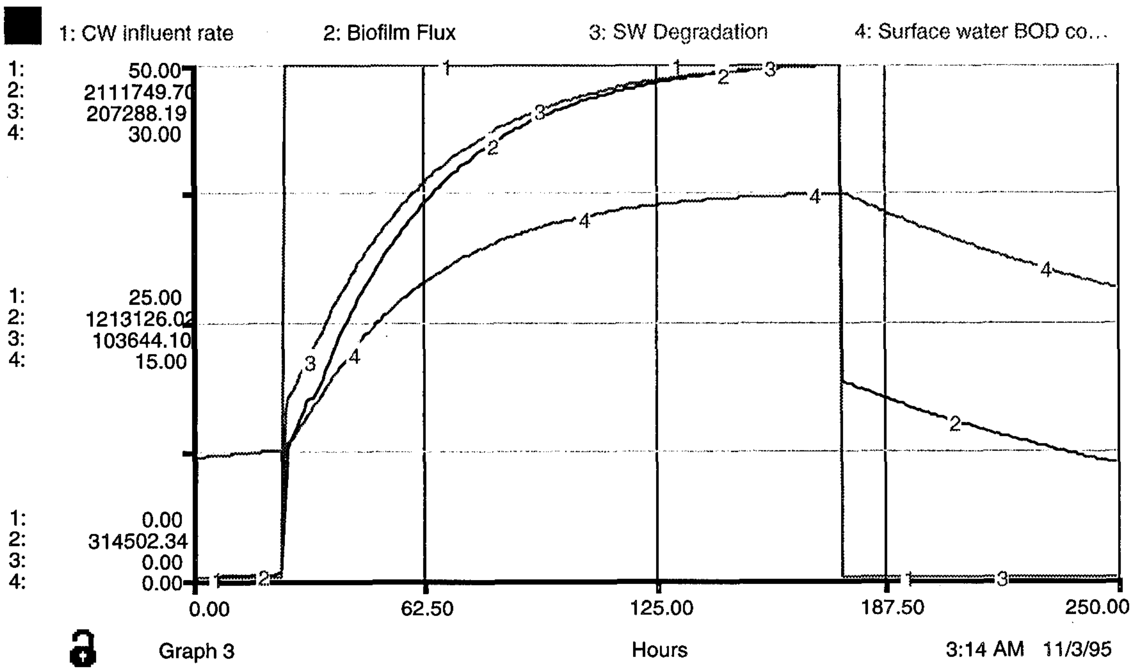

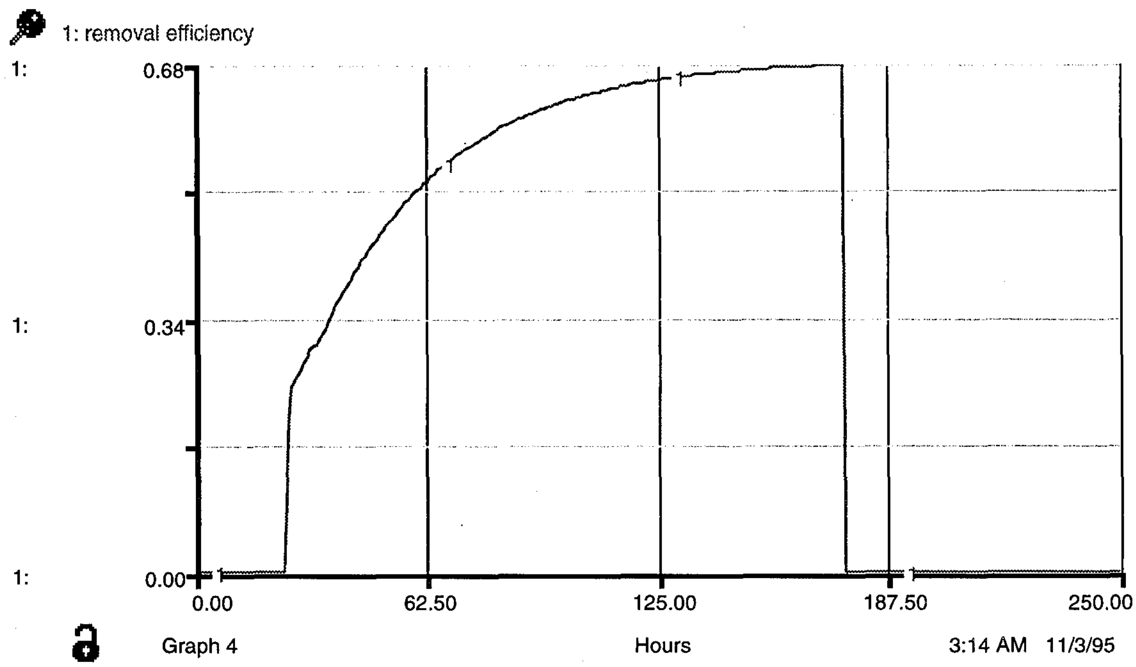

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The objective of this research was to develop a tool to aid the Air Force Environmental Manager in the identification of the design parameters of a constructed wetland system that may be optimized to provide a desired biochemical oxygen demand (BOD) removal efficiency during the treatment of Air Force stormwater runoff. The objective is achieved through the development and use of a system dynamics model which simulates the hydrological functions of a constructed wetland as well as the processes within the wetland responsible for degradation of BOD. Based on literature review, the primary mechanism responsible for the degradation of BOD within a constructed wetland system is degradation due to microbial populations in the form of both suspended biomass and biofilm found on the surface of vegetation and the wetland floor. The model was run for constructed wetlands of various surface areas, each subjected to a range of stormwater influent rates and influent concentrations. The hydraulic retention times, organic loading rates and BOD removal efficiencies were determined for each case. Scatter plots of both hydraulic retention times and organic loading rates vs. removal efficiency indicated a clear relationship between both hydraulic retention time and removal efficiency as well as organic loading rate and removal efficiency. Several runs of the model also indicated that larger surface areas, greater length to width ratios and depths contributed to lower BOD concentrations in the water column.

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