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A MODEL TO ENHANCE WETLAND DESIGN AND OPTIMIZE NONPOINT SOURCE POLLUTION CONTROL[1]

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ABSTRACT: A dynamic, compartmental, simulation model (WETLAND) was developed for the design and evaluation of constructed wetlands to optimize nonpoint source (NPS) pollution control. The model simulates the hydrologic, nitrogen, carbon, dissolved oxygen (DO), bacteria, vegetative, phosphorous, and sediment cycles of a wetland system. Written in Fortran 77, the WETLAND models both free-water surface (FWS) and subsurface flow (SSF) wetlands, and is designed in a modular manner that gives the user the flexibility to decide which cycles and processes to model. WETLAND differs from many existing wetland models in that the interactions between the different nutrient cycles are modeled, minimizing the number of assumptions concerning wetland processes. It also directly links microbial growth and death to the consumption and transformations of nutrients in the wetland system. The WETLAND model is intended to be utilized with an existing NPS hydrologic simulation model, such as ANSWERS or BASINS, but also may be used in situations where measured input data to the wetland are available. The model was calibrated and validated using limited data from a FWS wetland located at Benton, Kentucky. The WETLAND predictions were not statistically different from measured values for of five-day biochemical oxygen demand (BOD_5), suspended sediment, nitrogen, and phosphorous. Effluent DO predictions were not always consistent with measured concentrations. A sensitivity analysis indicated the most significant input parameters to the model were those that directly affected bacterial growth and DO uptake and movement. The model was used to design a hypothetical constructed wetland in a subwatershed of the Nomini Creek watershed, located in Virginia. Two-year simulations were completed for five separate wetland designs. Predicted percent reductions in BOD_5 (4 to 45 percent), total suspended solids (85 to 100 percent), total nitrogen (42 to 56 percent), and total phosphorous (38 to 57 percent) were similar to levels reported by previous research.
(KEY TERMS: constructed wetlands; simulation models; nonpoint source pollution.)

INTRODUCTION

Nonpoint source (NPS) pollution accounts for more than 50 percent of the nation's water quality problems (Novotny and Olem, 1994) and over 65 percent of the total pollutant load to inland surface waters (U.S. EPA, 1993). The U.S. EPA mandates reductions in nonpoint source pollution associated with urban and agricultural storm water runoff, and promotes the development of new and improved methods for reducing degradation of water quality. Constructed wetlands are relatively new effective systems used to mitigate NPS pollution (Raisin and Mitchell, 1995), but have been shown to be effective management practices (Daukas *et al.,* 1989).

The constructed wetland's design plays a vital role in determining its effectiveness in controlling NPS pollution. Models that accurately represent processes within the wetland system would help optimize NPS pollution control by allowing evaluation of the effects of alternative management strategies prior to their implementation. These evaluations would be used to help with the design of an appropriate wetland system for meeting specific water quality goals.

Numerous attempts to model wetland systems have been made by various investigators. Existing wetland models can simulate various cycles, ranging from modeling only the wetland hydrologic cycle (Hammer and Kadlec, 1986; Duever, 1988; Walton *et al.,* 1996), to modeling a combination of hydrologic, nitrogen (N), phosphorous (P), and vegetative cycles (Brown, 1988; Kadlec and Hammer, 1988; Dorge, 1994). Different models have been developed to simulate dissolved oxygen (DO) (Wynn and Liehr, 2001),

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bacteria (Thomas, 1993; Gidley, 1995), fish interactions (Jorgensen, 1976), and contaminants (Hearn *et al.*, 1991).

Applications of these models are limited for various reasons. Most of existing models do not adequately describe pollutant removal processes in wetlands, are site-specific, limited to modeling either free-water surface (FWS) or subsurface flow (SSF), or are too complex for practical engineering applications. A model is needed that can be applied widely to various conditions encountered in the design and evaluation of constructed wetland systems. The objective of this study was to develop, calibrate, and validate a user-friendly, dynamic, long-term simulation model (WETLAND) to assist in the design of constructed wetlands for optimizing NPS pollution control.

MODEL DESCRIPTION

The WETLAND model is designed as a continuous stirred-tank reactor; therefore, the model assumes that all incoming nutrients are evenly mixed throughout the entire volume. The model allows for both FWS and SSF wetlands to be modeled and, although developed to help with the design of constructed wetlands, WETLAND model may also be applied to model the functions of natural wetlands.

The WETLAND model simulates the hydrologic, N, carbon (C), bacteria, DO, vegetative, P, and sediment cycles of a wetland system. The N, C, DO, and bacteria cycles are linked and cannot be run independently. These cycles are referred to as the "NCOB" cycle to denote their dependence. The WETLAND model is designed so that the hydrologic component may run independently or concurrently with any or all of the NCOB, sediment, vegetative, and P cycles, with a few restrictions. These restrictions include: (1) the P cycle is dependent on the sediment cycle; (2) neither the sediment nor P cycles may be simulated with the SSF wetland option; and (3) any simulation run that includes modeling of the NCOB, sediment, or P cycles requires that vegetative growth be simulated and vice versa.

NCOB relations are based on the work of Wynn and Liehr (2001) and Parnas (1975), whereas P and sediment components are similar to those presented by Christensen *et al.* (1994). Figure 1 presents the relationships between the model's main code and its submodels.

The WETLAND model is designed to accept two forms of data input. The first option requires daily input values for hydrologic and nutrient parameters, either from measured data or output from a NPS model such as ANSWERS (Bouraoui and Dillaha, 2000) or HSPF (Donigian *et al.*, 1993). The second option for data input is based on the SCS curve number method (SCS, 1968). The SCS method determines the amount of daily runoff from the watershed, which is then multiplied by a runoff concentration coefficient for each respective nutrient parameter to determine nutrient inflow to the wetland system. An option to include point source contributions may be used with either input format and is based on daily input values to the model.

Modeling of the processes that occur in FWS and SSF wetlands is fairly similar, except for diffusion and particulate movement that are accounted for in FWS wetlands and not in SSF wetlands due to the specific characteristics of these systems. As Figure 2 shows, the water level in the system is above substrate level for FWS wetlands and below substrate level for SSF wetlands.

The WETLAND model is based on several options, allowing for some flexibility by the user. Depending upon which options are chosen, the input requirements and modeling approaches will vary. The model is written in Fortran 77 to facilitate linkage with existing NPS models, and is designed such that a main program calls and manages the submodels and options that need to be simulated. Daily time steps are managed within season and time-period loops.

Brief descriptions of WETLAND submodels are included in the following sections. Because the processes modeled for FWS and SSF wetlands are similar, only the FWS systems are described here. For simplification, a process that occurs only in the free water surface water pool is designated with a "W," in substrate water, an "S," and in both pools, a "B."

Hydrology Submodel

The surface storage of the hydrologic cycle is accounted for by using an adapted dynamic water budget approach (Kadlec and Knight, 1996):

$$\frac{dV}{dt} = Q_c + Q_p - Q_{out} + \Delta BMV + \Delta SDV + (P - PI - ET) * A \tag{1}$$

where dV/dt is the change in surface storage (m^3/day); Q_c is the watershed catchment runoff additions (m3/day); Q_p is the addition from point sources (m^3/day); Q_{out} is the daily outflow rate (m^3/day); ΔBMV is the change in living biomass volume in the surface water (m^3/day); ΔSDV is the change in standing dead plant volume in the surface water (m^3/day);

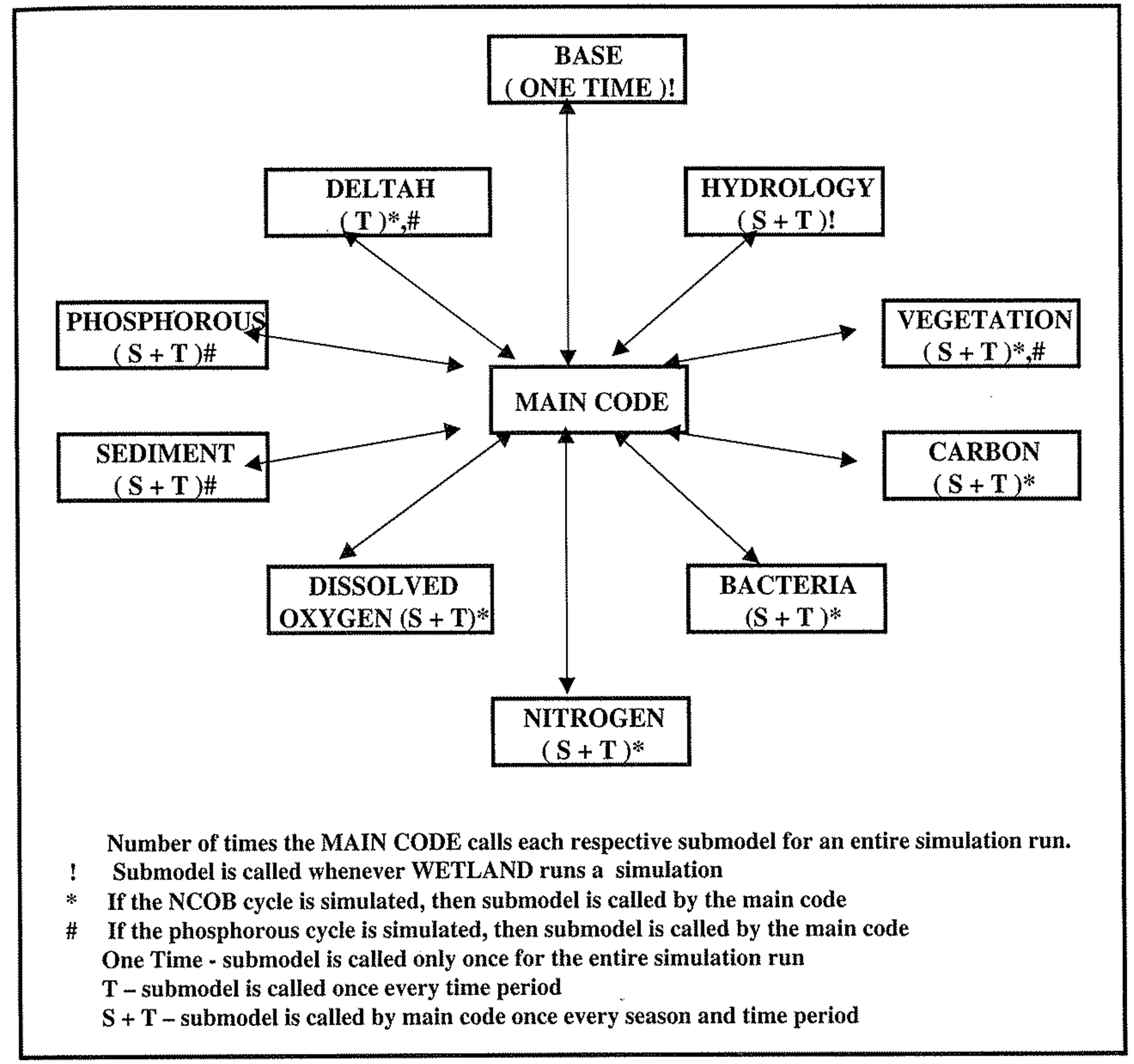


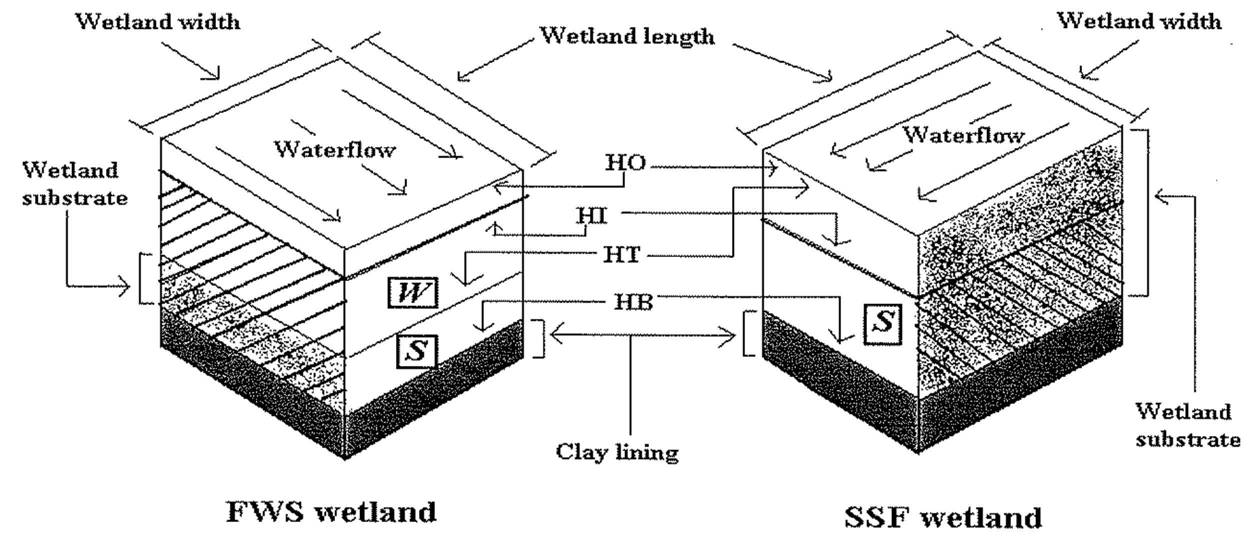
Figure 1. Relationships of WETLAND Main Code to Its Submodels.

P is the daily precipitation rate (m/day); PI is the percolation/infiltration rate (m/day); ET is the evapotranspiration rate (m/day); and A is the wetland surface area (m^2). The water volume in the wetland substrate is determined by multiplying the wetland area by the soil thickness and the soil porosity.

Outflow from the system may be modeled using six different outlet options, five of which are for FWS wetlands (Lee, 1999). Evapotranspiration (ET) may be modeled with either the Pan method or Thornthwaite's method (Thornthwaite and Mather, 1955). In the Thornthwaite's method, daily values are estimated by using the daily average temperature, instead of monthly average values, and then dividing by 30 days (Hann *et al.*, 1982; Wynn and Liehr, 2001). To improve outflow computation, it was necessary to divide the daily time step into hourly values (by dividing the daily values by 24 hours), so the surface water level could be determined more frequently.

Vegetation Submodel

Biomass is modeled with a simple plant model, similar to the one used by Wynn and Liehr (2001) and Hammer (1984). At the beginning of the growing



W is surface water; **S** is wetland substrate water; **HB** is bottom of wetland substrate; **HT** is top of wetland substrate; **HI** is initial water height; and **HO** is the crest of the wetland cell
Striped area represents volume occupied by water
Shading represents wetland substrate

Figure 2. Wetland Description for FWS and SSE Wetlands.

season, biomass growth rate is assumed to increase linearly for up to 20 days until the maximum rate for the growing season is reached. Biomass grows at this constant rate until the end of the growing season, when the growth rate linearly decreases to zero over a ten-day period. Vegetative senescence starts at the end of the growing season and it is assumed that 99 percent of the biomass becomes standing dead material in the first ten days of winter.

NCOB CYCLE

Unlike most wetland models, the WETLAND model explicitly accounts for the effects of biomass and microbial dynamics in the wetland system using the dynamically linked NCOB cycle. The WETLAND model also accounts for carbon/nitrogen (C:N) ratios in the system and does not automatically assume that C amounts in the wetland are sufficient. It does not rely exclusively on empirical relationships to determine nutrient transformations in the system; rather it models microbial growth with Monod kinetics to link transformations such as nitrification and denitrification (Lee, 1999).

Carbon Submodel

The carbon cycle consists of five state variables: biomass C, standing dead C, particulate organic C [POC (B)], dissolved organic C [DOC (B)], and refractory C.

The biomass C and standing dead C components are linked with the vegetative submodel. Biomass C is determined by multiplying a biomass C concentration by the existing biomass in the wetland. The amount of C in the standing dead is similarly determined, but is also affected by DOC leaching and physical degradation. Physical degradation is an input source to the POC variable.

For FWS wetlands, the mass balance for POC is dependent on the particulate BOD influx (W), microbial death (B), peat accumulation (B), and POC mineralization (B). For DOC, the mass balance is dependent on the soluble BOD influx (W), DOC mineralization (B), DOC leaching (S), and diffusion (B).

Whether DOC or POC (along with dissolved organic N (DON), particulate organic N (PON), and NH_4^+ discussed later) is utilized for microbial growth and energy is determined by the required substrate C:N ratio, as defined by the microbe total C:N ratio. The proportion of the microbe total C:N ratio is directly related to the DO concentration (Parnas, 1975). There

is an increase in the required C:N ratio as a result of anaerobic conditions, where reactions are less efficient and therefore require more C for equal amounts of cell growth (Wynn and Liehr, 2001).

POC, DOC, PON, DON, and NH_4^+ are all available for microbial growth. Utilization of each nutrient is dependent on the ratio of the total organic C (TOC) to total organic N (TON), in relation to the MICTCN (ratio of carbon to nitrogen in microbial cells). WETLAND uses the following relationships to simulate DOC mineralization/immobilization (DOC min/imob) and POC mineralization/immobilization (POC min/imob):

$$\text{DOC min/imob} = \text{DON/TON} * \text{HTGROW/HTYIELD} \quad (2a)$$

(when TOC/TON > MICTCN, N is limiting microbial growth)

$$\text{DOC min/imob} = \text{DOC/TOC} * \text{HTGROW/HTYIELD} \quad (2b)$$

(when TOC/TON < MICTCN, C is limiting microbial growth), and

$$\text{POC min/imob} = \text{PON/TON} * \text{HTGROW/HTYIELD} \quad (3a)$$

(when TOC/TON > MICTCN, N is limiting microbial growth)

$$\text{POC min/imob} = \text{POC/TOC} * \text{HTGROW/HTYIELD} \quad (3b)$$

(when TOC/TON < MICTCN, C is limiting microbial growth)

where HTGROW is the heterotrophic (HT) growth rate (g microbes/day), and HTYIELD is the yield of HT microbes (g microbes/g C degraded).

Nitrogen Submodel

Processes that are always modeled by the N submodel include ammonification (B), immobilization (B), nitrification (B), denitrification (B), and peat accumulation (B). Inclusion of NH3 volatilization (W), atmospheric deposition (W) and N fixation (W) in the modeling of the overall N cycle is optional. Sorption of NH_4^+ to the soil and organic matter is not modeled because it is assumed that sorbed NH_4^+ is still available to attached microbes (Wynn and Liehr, 2001). The state variables for the N cycle are DON (B), PON (B), ammonia and ammonium N (NH_4, B), nitrate N (NO_3, B), immobilized N (B), and refractory N (B).

Influent DON enters the wetland from four sources: catchment runoff, point sources, percolation (seepage), or atmospheric deposition (dry and wet). DON also accumulates, by N leaching of standing dead, due to the physical degradation of standing dead biomass. A part of DON is reincorporated into microbial biomass during the degradation of organic C, while the rest is converted to NH_4^+ (when carbon is limiting microbial growth). An optional model component that increases the DON amounts in the wetland system is N fixation. N fixation is modeled with a zero-order equation. PON dynamics are similar to those for DON except that PON is also assumed to accumulate in peat as refractory N.

NH_4^+ accumulates in wetlands through NH_4^+ influent (catchment runoff, point sources, seepage, and atmospheric deposition), and PON and DON ammonification. PON ammonification is modeled similar to DON ammonification. Ammonium uptake is the sum of the plant uptake (minus NO_3^- uptake) and microbial uptake. Biomass is assumed to prefer NO_3^- to NH_4^+ as a N source, thus NH_4^+ utilization only occurs when NO_3^- amounts are insufficient. Ammonium is also used by autotrophic (AT) bacteria as an electron source, converting it to NO_3^- through nitrification. This is modeled as the quotient of Nitrosomonas (NS) growth and NS yield. Volatilization is an optional model component that will affect NH_4^+ concentrations, and is modeled as a first order equation that considers pH values.

Immobilized N is the sum of DON and PON immobilization, NO_3^- uptake and NH_4^+ uptake. Immobilized N eventually returns to PON through the death of microbes and biomass. These values are the product of the respective death rates and N contents, as discussed in the microbial and C submodels.

The last N form is NO_3^-. NO_3^- increases due to nitrification and influent contributions, and decreases by plant NO_3^- uptake and denitrification. Nitrate uptake is modeled as the product of biomass growth and biomass C:N ratio. Anaerobic heterotrophs use NO_3^- as an electron acceptor, converting the NO_3^- to N_2 gas, which is eventually lost to the atmosphere. It is assumed that the movement of the N_2 gas from the soil system is instantaneous; therefore, the diffusion of the N_2 gas is not modeled. Denitrification is modeled as the quotient of anaerobic HT growth and HT nitrate yield.

Dissolved Oxygen Submodel

The oxygen budget consists of the single state variable, DO (B). Oxygen is added to the wetland by influent runoff, influent point sources, precipitation, biomass flux, and reaeration. Reaeration with the atmosphere is modeled only for FWS wetlands. It is assumed that oxygen transfer with the atmosphere is negligible in SSF systems because the water surface is below the substrate (Kadlec and Knight, 1996).

Biomass flux is the product of the biomass oxygenation rate (g O_2/m^{2}*day) and the wetland surface area. It is assumed there is a uniform vegetation stand throughout the wetland system and that plants transport oxygen to the wetland bottom at a constant rate throughout the growing season. HT and NS respiration are assumed to be proportional to microbial growth (Wynn and Liehr, 2001).

Bacteria Submodel

The bacteria submodel accounts for all of the microbial growth and activity in the model. Both AT and HT bacteria groups are examined by the WETLAND model. The two microbial groups are modeled in similar manners.

The state variable NITROSOMONAS (NS, B) represents the AT population within the wetland. Changes in the population of NS are due to NS growth and death. Growth rate of NS is described using Monod dual substrate limitation kinetics (Wynn and Liehr, 2001):

$$\mu = \mu_{max} * \left(\frac{NH_4}{NH_4 + K_{NH_4}}\right) * \left(\frac{DO}{DO + K_{DO}}\right) \qquad (4)$$

where μ is the actual NS specific growth rate (day^{-1}); (max is the NS maximum growth rate (day^{-1}); NH_4 is the NH_4^+ concentration (mg/l); K_{NH4} is the NS NH_4 half-saturation constant (mg/l); DO is the wetland dissolved oxygen concentration (mg/l); and K_{DO} is the NS DO half-saturation constant. This expression modifies the NS growth rate when oxygen (the electron acceptor) or NH_4^+ (the electron donor) is limiting.

Besides substrate limitations, temperature and pH also limit microbial growth. Since wetlands tend to drive pH toward neutrality (pH = 7) (Wynn and Liehr, 2001), a pH factor is not included in the WETLAND model. Subsequently, the NS growth is calculated as:

$$NSGROW_j = \mu * NSTEMPF * NITRSOMONAS_{j\text{-}1} \qquad (5)$$

where NSGROW is the amount of NS growth (g microbes/day); NSTEMPF is the NS temperature factor; and NITROSOMONAS is the amount of NS microbes (g microbes/day). Since there is little information concerning the factors controlling microbial die-off, it is modeled as a first-order reaction.

The Monod models used for the HT bacteria are similar to that of the NS (Equation 5), except the growth is dependent on TOC instead of NH_4^+. Some HT bacteria are facultative, meaning they can survive under aerobic or anaerobic conditions. WETLAND determines the fraction of the HT bacteria which utilize either aerobic or anaerobic conditions, based on the DO concentrations in the system (Wynn and Liehr, 2001). The optimum temperature and pH for NS is from 15 to 33°C and 6 to 9, respectively. The same optimum temperature and pH rules apply to HT bacteria as they do for the AT bacteria.

Sediment Submodel

The sediment cycle models sediment particles and any other desired suspended solid in the wetland system. The sediment model has been designed to accept up to five different sediment classifications. The processes modeled in the sediment cycle include inflow, outflow, deposition, resuspension, and decomposition. Sediment inflow depends on the input option chosen, while sediment outflow is based on the fall rate, resuspension and the total amount suspended in the wetland water. If the total fall distance (m) for the day exceeds the free water surface level (m) of that day, then outflow is negligible, otherwise, a removal ratio is used, based on the hydrologic outflow and the water volume in the system. Due to the desire for the sediment, P and NCOB cycles to operate independently, the decomposition component in the sediment submodel does not run concurrently with the NCOB cycle. Decomposition of sediment is modeled with a simple, user-defined, first-order rate equation.

Phosphorous Submodel

The P submodel is based on the assumption that all of the suspended sediment particles provide surface area to which P can be attached and consequently, settled, resuspended, or transformed. There are four pools for the P cycle; particulate and dissolved, for both the surface and bottom of the wetland. Processes modeled include mineralization and additions from biomass decomposition.

Incoming P amounts are modeled using either the Freundlich isotherm, the linear isotherm, or as direct

input. If using either the linear or Freundlich isotherms, the input dissolved P (DP) concentrations are used to determine the particulate P concentrations. Using the input sediment diameter, the surface area for each particle class is determined. Particulate P is divided among the particle classes according to the relative surface area for each class.

Mineralization is modeled with first-order equations for each particle class. Resuspension and settling of particulate P correspond to the amount of sediment particles and are related to the ratio of each category that resuspends and settles. The contribution of P made by physical degradation is based on the plant biomass:phosphorous ratio and plant biomass at the time of its death. After each time step, the partition of particulate P to each sediment category is redistributed; this repartitioning is dependent upon the number of particles in both the water and soil (Christensen *et al.*, 1994).

MODEL EVALUATION

The WETLAND model's performance in simulating the functions of FWS wetlands was evaluated with data collected at a wetland site in Benton, Kentucky. This was not an ideal data set because the Benton wetlands were designed to treat municipal waste. The model's performance for SSF wetlands was not evaluated; however, previous research by Wynn and Liehr (2001) examined WETLAND's basic approach towards modeling SSF wetlands. Model evaluation procedures included calibrating and validating the model, performing two types of statistical analyses, conducting a sensitivity analysis, and using WETLAND model to demonstrate its application for wetland design.

Study Area

As described by Choate *et al.* (1990) and summarized by Kadlec and Knight (1996), the Benton wetlands were designed to polish municipal effluent from an existing lagoon for 5,000 people. The examined wetland cell consists of a substrate of a native clay and impermeable clay lining of 3.0 to 4.5 m thickness, which eliminated infiltration or percolation. It was planted with *Scripus cyperinus (L.) Kunth* (woolgrass bulrush), *Scripus validius Vahl* (softstem bulrush), and *Typha latiofolia L.* (cattail). Influent and effluent samples were collected either monthly or bimonthly for a variety of parameters. Listed in Table 1 is the data set used as input for the calibration and validation of the WETLAND model.

Incoming daily water flow, nutrient, and sediment values were determined by linear interpolation between the data points. This was the case for all daily parameters, excluding the daily weather data

TABLE 1. Measured Inflow Values to Wetland Cell 2 in Benton, Kentucky, Used for Validation and Calibration of SET-WET Model (Choate *et al.*, 1990).

Date	Flow (m^3/d)	DO (mg/l)	BOD_5 (mg/l)	NH_3-N (mg/l)	NO_3-NO_2 (mg/l)	Org-N (mg/l)	TKN (mg/l)	TSS (mg/l)	Dis-P (mg/l)	Tot-P (mg/l)
April 27, 1988	213.2	2.4	30.0	12.0	0.0	5.0	17.0	30.0	3.0	4.5
May 25, 1988	567.9	8.1	34.0	12.0	0.0	8.0	20.0	49.0	5.5	6.8
June 29, 1988	339.0	2.0	18.0	5.2	0.0	11.0	16.2	63.0	6.8	7.8
July 27, 1988	350.4	8.3	19.0	3.1	0.0	12.3	15.4	110.0	4.1	5.6
August 30, 1988	472.5	16.8*	2.0	0.1	0.1	9.9	10.0	–	5.4	9.7
September 28, 1988	403.6	2.4	14.0	4.1	0.0	9.6	13.7	59.0	5.6	5.7
October 25, 1988	478.0	6.2	23.0	9.8	0.4	7.1	16.9	54.0	–	6.6
November 29, 1988	1263.1	10.8	22.0	3.8	0.7	6.5	10.3	31.0	2.3	3.4
December 13, 1988	523.9	19.1*	38.0	4.8	0.4	6.3	11.1	54.0	2.9	3.3
January 24, 1989	877.1	13.0	24.0	3.1	0.5	1.5	4.6	31.0	0.7	3.7
February 22, 1989	1816.1	10.0	23.0	1.0	0.6	4.3	5.3	9.0	0.9	1.4
March 28 1989	1016.4	8.3	24.0	2.0	0.3	6.0	8.0	29.0	1.1	2.6
April 26,1989	633.0	9.0	26.0	3.0	0.2	12.0	15.0	53.0	2.3	2.9

*Data removed; physically impossible.
–Analysis incomplete.

(precipitation and air temperature) which were obtained from the National Oceanographic and Atmospheric Administration records, at a site located in Padukah, Kentucky (NOAA, 1988; NOAA, 1989).

The data set used to calibrate and validate the WETLAND model was not ideal, as the use of linear interpolation between the monthly observed data points contradicts the idea of NPS pollution's randomness and dependence upon climatic occurrences. However, since the Benton wetland site was developed for municipal waste treatment, this problem may have been mitigated. The ideal data set would consist of daily data points for all concerned hydrologic and nutrient values over a minimum of two years, but no such data were available. This type of data set would allow the seasonal, as well as long term, comparison of model performance.

Model Calibration and Validation Procedures

Data collected from April 27, 1988, to April 26, 1989, were used to calibrate and validate the model. The data were divided into three groups, two of which were used for calibration (April 27, 1988, to July 27, 1988, and January 24, 1989, to April 26, 1989), and one of which (July 27, 1988, to January 24, 1989) was used for validation. These time periods were selected to evaluate how the model performed during warm and cold time periods.

The hydrologic component of the model was calibrated first, followed by the NCOB, the sediment and the P components, respectively. The initial input parameter values for the model were estimated using various reference data. The bacterial parameters were estimated from resources including Grady and Lim (1980), Henze *et al.* (1986), Tchobangolous and Burton (1991), and Wynn and Liehr, (2001). The initial amounts of nutrients in the water were determined from the effluent concentrations of the wetland cell on the first sampling date, while the initial amounts of nutrients in the wetland soil were determined through calibration and assumptions derived from Reed (1994), Johnston (1991), and Wynn and Liehr, (2001).

The calibrated values did not deviate significantly from published values, except for parameter values directly involving bacterial growth and oxygen transfer. The predicted calibrated values mostly followed the trends of their respective measured values. Ammonium concentrations were over-predicted for the first calibration period, while the NO_3^-, organic N, and DO effluent concentration predictions did not completely match the observed value trends. For the second calibration period, the DO concentrations were overpredicted. The overpredictions may be attributable to the few data points, where each influent measurement is amplified in importance. Additionally, there were no bacterial counts measured in the system; thus it is unknown if the simulated bacteria counts reflected actual conditions.

In the validation process, the initial amounts of each nutrient in the system were determined in the same manner as the calibration procedure. The input parameter values were determined from the average of the two calibrated parameter values. Figures 3, 4(a-d) and 5(a-d) display the model predictions compared with the observed data for the outflow rate and the effluent concentrations of NH_4^+, NO_3^-, organic N, DO, BOD_5, TSS, DP and TP for the validation period. Table 2 lists the measured values, predicted values, and the difference between the two values for the outflow rate, and the NH_4^+, NO_3^-, organic N, DO, BOD_5, TSS, DP, and TP effluent concentrations.

As evident in Figures 3, 4(a-d), and 5(a-d), the predicted trends match the observed values fairly well, except for the NH_4^+ (Figure 4a) and DO (Figure 4d) predictions. The trend for predicted NH_4^+ concentrations does not follow the measured values for the period of mid-August 1988 to the end of October 1988. During this period, the model's predictions are low, indicating that the conversion of organic N through mineralization is not sufficiently handled by the model. However, there is not an overestimation of organic N concentration (Figure 4c); therefore, the model may lack accounting for additions to organic N from biomass degradation.

The inability of the model to match DO concentrations is a cause for concern since many of the bacterial growth and uptake rates are directly and indirectly affected by oxygen concentrations in the system. The high DO concentrations predicted by the model indicate that the input bacterial parameter values that control growth and nutrient uptake may have been too low.

Statistical Analyses

Statistical analyses were performed on the differences between the observed measurements and the model's predicted values for the hydrology and nutrient values. The SAS software package "proc univariate" procedure was used to perform the Wilcoxon signed rank test on the data (Ott, 1993). For this test, the null hypothesis (H_0) was that the difference between the observed and predicted values is equal to zero, while the alternative hypotheses (H_1) was that there was a significant difference between the two values. Using a two-sided test, with an alpha value of

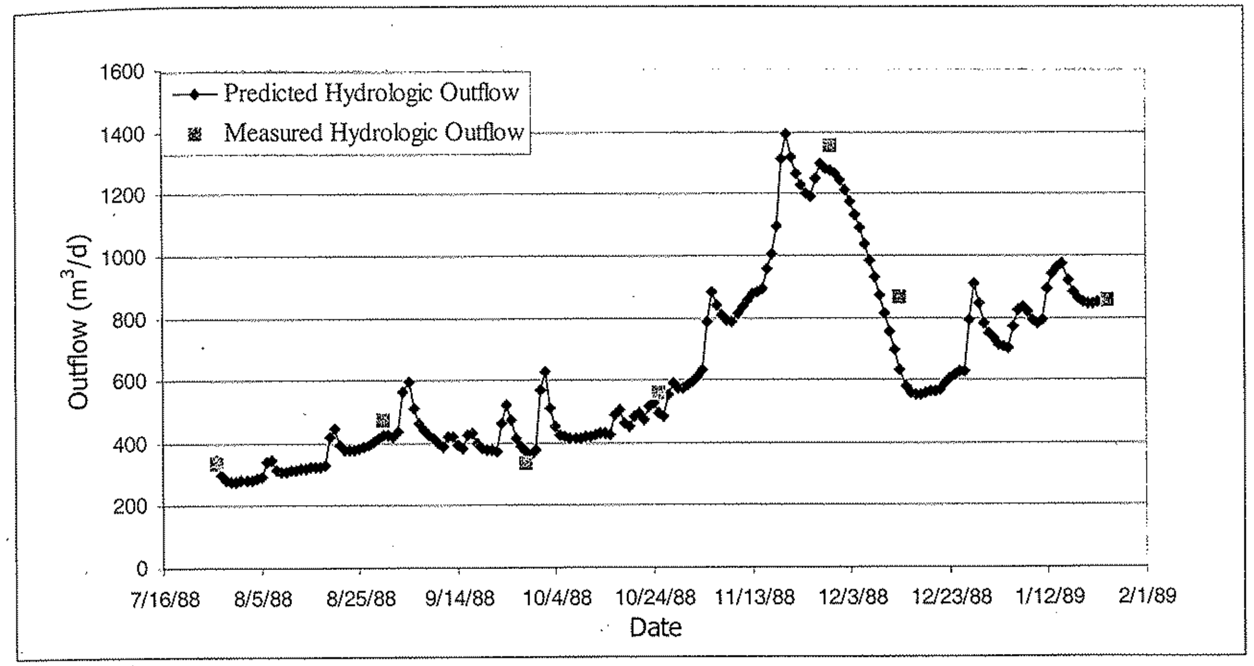


Figure 3. Observed and Validated Predicted Values (July 27, 1988, to January 24, 1989) for Hydrologic Outflow From the Wetland at Benton, Kentucky.

0.10, it was determined whether the model predictions were statistically similar to the measured values.

For the hydrology, NO_3^-, NH_4^+, organic N, BOD_5, TSS, DP and TP outflows, we failed to reject the null hypothesis (p-values were greater than 0.10, indicating the values were statistically similar) (Table 3). DO was the only parameter for which the null hypothesis was rejected, with a p-value of 0.016. These results must be examined closely because there were only seven values in the data set used for each comparison.

Linear regression analysis was also performed on the output values. The null hypothesis (H_0) was that the slope coefficient between the observed and predicted values is 0, whereas the alternative hypothesis (H_1) was that the slope coefficient is not equal to 0. The ideal situation would be a line that crosses the y intercept (B_0) at 0, has a slope (B_1) of 1.0, an R^2 close to 1.0 and the p value for the slope parameter is less than 0.05. Independently, the p-values tell us little, but if analyzed in conjunction with the slope coefficient and R^2 value, we may be able to tell which parameters were statistically similar. The parameters that follow these constraints are hydrology, NO_3^-, DO, TSS, DP, and TP. The other parameters either have p-values above 0.05 (Organic N, NH_4^+, BOD_5), slope coefficients below 1.0 (NH_4^+, organic N), or low R^2 values (NH_4^+, BOD_5).

The linear regression results were used to obtain the 95 percent confidence intervals (C.I.) for further evaluation of model performance. As seen in Table 4, in the expected operating range, C.I. predictions overlap 0 for B_0 and 1.0 for B_1, for all nine concentrations. The C.I. for the parameters are extremely large and the fact that they overlap the idealized 1:1 line for all nine parameters is attributable to the large variance in the small number of data points.

Sensitivity Analysis

A sensitivity analysis was conducted to determine which parameters would require the most scrutiny in future simulations. Each parameter was adjusted a total of six times, with changes of (+/-) 10 percent, (+/-) 25 percent, and (+/-) 50 percent, to determine the relative sensitivity (S_r) of each parameter (Heatwole *et al.*, 1987):

$$S_r = \left(\frac{O - O_b}{Pv - Pv_b}\right) * \left(\frac{Pv_b}{O_b}\right) \tag{6}$$

where O is the model output variable of interest; Pv is the parameter value; and b is a subscript denoting the base scenario.

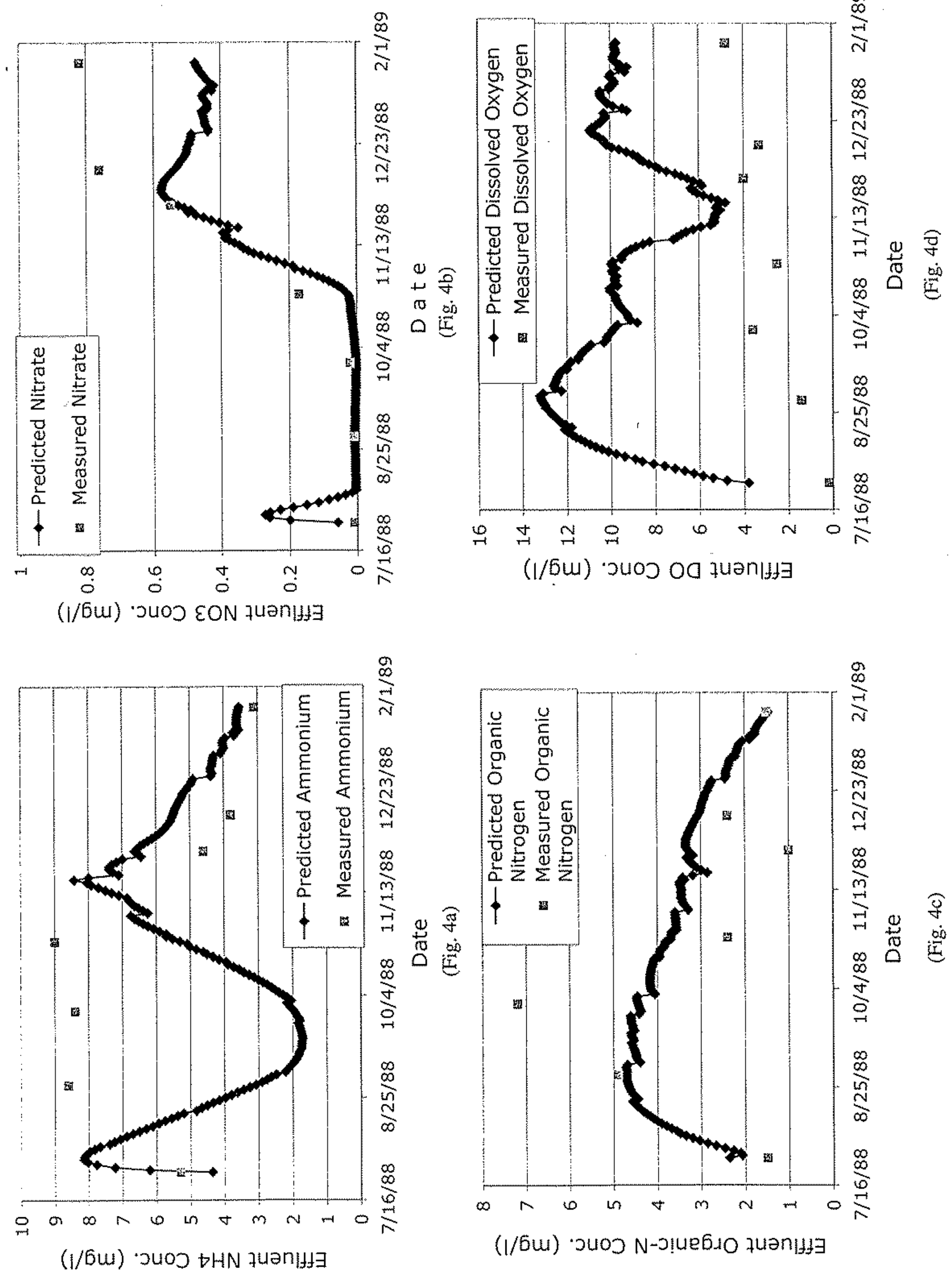


Figure 4. Observed and Predicted Values (July 27, 1988, to January 24, 1989) for (a) Ammonium, (b) Nitrate, (c) Organic Nitrogen, and (d) Dissolved Oxygen Effluent Concentrations.

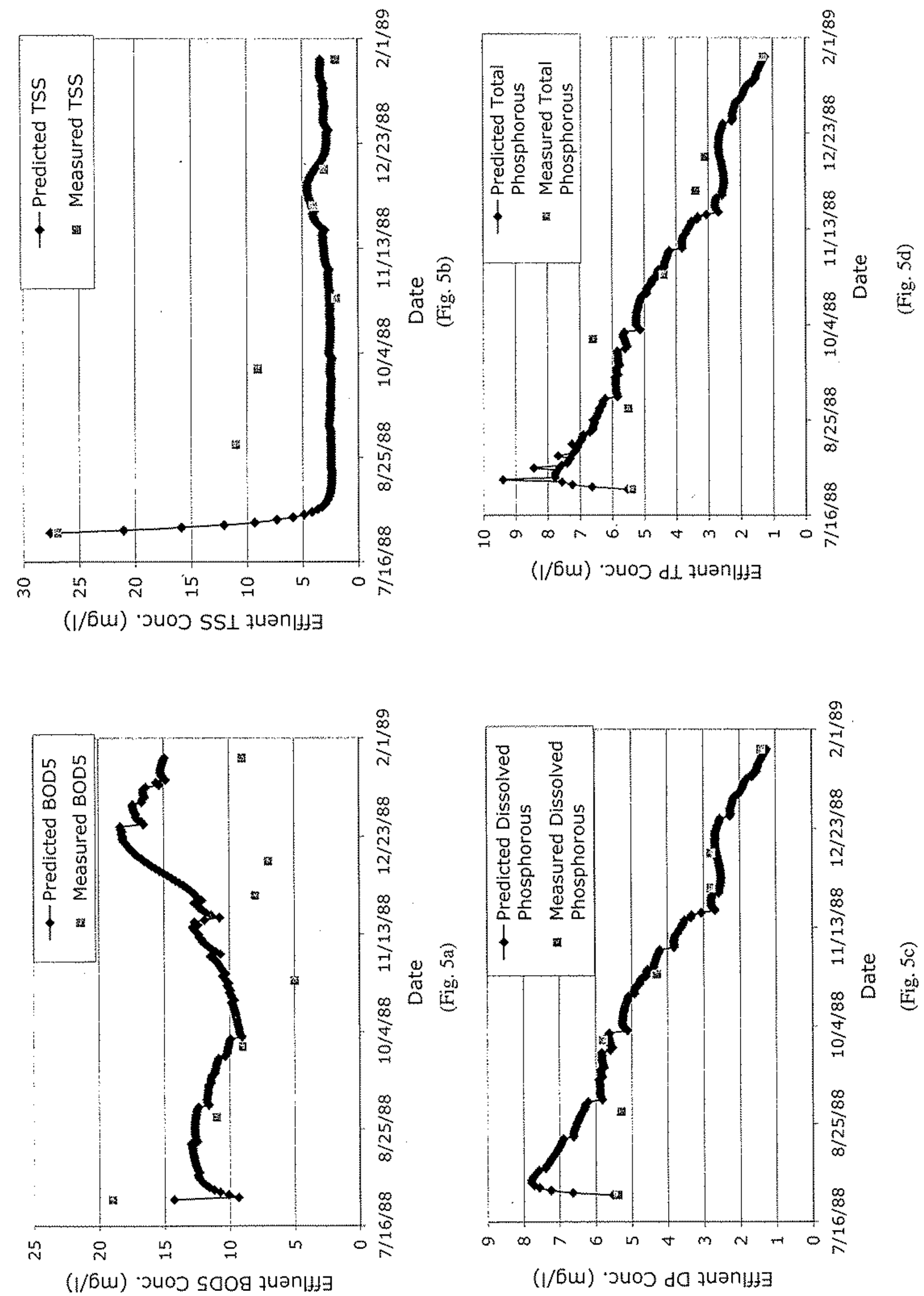


Figure 5. Observed and Predicted Values (July 27, 1988, to January 24, 1989) for (a) BOD_5, (b) TSS, (c) Dissolved Phosphorus, and (d) Total Phosphorus Effluent Concentrations.

TABLE 2. Measured, Predicted, and Difference Between the Measured and Predicted Values for the Hydrology, and Various Wetland Effluent Concentrations for the Validated, Predicted Values.

Date	Hydrologic Outflow (m^3)			NH_4 Effluent Concentration (mg/l)		
	Measured	Predicted	Difference	Measured	Predicted	Difference
July 27, 1988	332.33	345.64	-13.31	5.30	4.36	0.94
August 30, 1988	473.75	424.41	49.34	8.60	3.08	5.52
September 28, 1988	332.33	371.21	-38.88	8.40	1.98	6.42
October 25, 1988	560.24	492.28	67.96	9.00	5.30	3.70
November 29, 1988	1355.81	1274.87	80.94	4.60	6.62	-2.02
December 13, 1988	864.91	631.60	233.31	3.80	5.50	-1.70
January 24, 1989	858.29	856.48	1.81	3.10	3.54	-0.44

Date	NO_3-N Effluent Concentration (mg/l)			Organic-N Concentration (mg/l)		
	Measured	Predicted	Difference	Measured	Predicted	Difference
July 27, 1988	0.01	0.06	-0.05	1.50	2.36	-0.86
August 30, 1988	0.01	0.01	0.00	4.90	4.71	0.19
September 28, 1988	0.02	0.00	0.02	7.20	4.44	2.76
October 25, 1988	0.17	0.03	0.14	2.40	3.69	-1.29
November 29, 1988	0.55	0.53	0.02	1.00	3.29	-2.29
December 13, 1988	0.76	0.54	0.22	2.40	3.06	-0.66
January 24, 1989	0.82	0.48	0.34	1.50	1.46	0.04

Date	DO Effluent Concentration (mg/l)			BOD_5 Effluent Concentration (mg/l)		
	Measured	Predicted	Difference	Measured	Predicted	Difference
July 27, 1988	0.20	3.82	-3.62	19.00	14.28	4.72
August 30, 1988	1.40	13.16	-11.76	11.00	12.62	-1.62
September 28, 1988	3.60	9.91	-6.31	9.00	10.10	-1.10
October 25, 1988	2.50	9.96	-7.46	5.00	10.23	-5.23
November 29, 1988	4.00	6.52	-2.52	8.00	12.63	-4.63
December 13, 1988	3.30	10.16	-6.86	7.00	16.48	-9.48
January 24, 1989	4.80	9.74	-4.94	9.00	14.93	-5.93

Date	TSS Effluent Concentration (mg/l)			DP Effluent Concentration (mg/l)		
	Measured	Predicted	Difference	Measured	Predicted	Difference
July 27, 1988	27.00	27.62	-0.62	5.40	5.52	-0.12
August 30, 1988	11.00	2.47	8.53	5.30	6.36	-1.06
September 28, 1988	9.00	2.53	6.47	5.80	5.62	0.18
October 25, 1988	2.00	2.60	-0.60	4.30	4.59	-0.29
November 29, 1988	4.00	4.15	-0.15	2.80	2.56	0.24
December 13, 1988	3.00	3.89	-0.89	2.80	2.64	0.16
January 24, 1989	2.00	3.34	-1.34	1.40	1.24	0.16

The cumulative average effluent value for the base scenario was compared with the average change in model response for the hydrology and nutrient parameters. From the S_r value it can be determined if there is an indirect (-) or direct (+) relationship between the parameter and the output (Heatwole *et al.,* 1998). For the WETLAND model, the results of the sensitivity analysis indicated that most of the S_r values approached zero, implying that the effects on model predictions are minimal.

Generally, the NCOB cycle is insensitive to changes in a single parameter due to the complexity and number of interactions in the wetland system; however, there are exceptions. The NCOB cycle is most sensitive to changes that affect microbial growth and oxygen, which is expected, because the WETLAND model is driven by the interactions and processes of the bacteria cycle. Because the DO concentrations actively interact with the bacteria cycle the model is sensitive to changes in the DO values. The simulated BOD_5 values were sensitive to changes involving the C cycle parameters, while the NH_4^+ and NO_3^- outputs were sensitive to changes associated with bacteria and NO_3^- parameters. Since the NCOB, sediment and P cycles were designed to run independently, changes in associated parameter values had little effect upon the predictions of the other cycles. The model is generally less sensitive to changes affecting sediment or P parameters. Complete results of the sensitivity analyses of the WETLAND model are presented by Lee (1999).

TABLE 3. P-Values and Results of the Wilcoxon Signed Rank Test Procedure for Differences Between the Measured and Validated, Predicted Values of Wetland Effluent in Benton, Kentucky.

Outflow Parameter	P-Value	H_0
Hydrology	0.156	Fail to Reject
NO_3-N	0.375	Fail to Reject
NH_4-N	0.297	Fail to Reject
Org-N	0.938	Fail to Reject
DO	0.016	Reject
BOD_5	0.109	Fail to Reject
TSS	0.938	Fail to Reject
DP	0.999	Fail to Reject
TP	0.688	Fail to Reject

Model Application

To examine the potential use of WETLAND for constructed wetland design, it was applied to a hypothetical situation where a constructed wetland might be appropriate. The objectives of this analysis was to evaluate the potential use of WETLAND for long-term simulations, to analyze the design capabilities of the WETLAND model, and to determine its strengths and weaknesses. The wetland designs emphasized the capture of NPS pollutants from a watershed area with the goal of optimizing pollutant removal.

Data collected by Mostaghimi *et al.* (1998) in the Nomini Creek watershed were used as input for the simulation. Located in Westmoreland County, Virginia, the Nomini Creek watershed is 80 km northeast of Richmond. The entire watershed is 1463 ha, but a 214 ha subwatershed, QN2, was used for the analysis.

Data for a two-year period (March 26, 1992, to March 25, 1994) were used for the simulations. Collection of the hydrologic (streamflow, precipitation) and nutrient (NH_3-N, NO_3-N, DON, PON, DP, particulate P, TSS) data occurred on a weekly basis as well as for each storm-event. These values were extended to daily input points by linear interpolation between the weekly and storm values. Water temperature was estimated from the measured average daily temperature, while BOD influent was estimated from periodic COD measurements. The Thornthwaite method was used to estimate ET. It was assumed that the wetland was lined, there were no point source additions, the wetland was fully established, and direct ground water interactions were negligible.

TABLE 4. Linear Regression Data for Observed (y-axis) and Predicted (x-axis) Wetland Effluent.*

	B_0	B_1	R^2	P-Value	Lower 95 Percent C.I. B_0	Upper 95 Percent C.I. B_0	Lower 95 Percent C.I. B_1	Upper 95 Percent C.I. B_1
Hydrology	9.820	1.071	0.944	0.001	-199.292	218.928	0.773	1.369
NH_4	8.814	-0.623	0.159	0.376	1.269	16.365	-2.270	1.024
NO_3	0.0258	1.320	0.903	0.001	-0.142	0.194	0.822	1.819
Org-N	-1.908	1.489	0.565	0.050	-7.093	3.277	-0.013	2.991
DO	1.627	0.132	0.063	0.588	-3.950	7.206	-0.457	0.723
BOD_5	3.419	0.483	0.062	0.581	-24.431	31.270	-1.624	2.589
TSS	2.478	0.873	0.812	0.006	-2.770	7.726	0.389	1.356
DP	0.535	0.843	0.956	0.001	-0.393	1.464	0.634	1.051
TP	0.693	0.869	0.887	0.002	-0.898	2.285	0.512	1.226

*B_0 = y intercept; B_1 = slope; R^2 = correlation coefficient; C.I. = confidence interval.

Five simulations were conducted for the model application. The first analysis was based on the recommendation that the desired minimum hydraulic residence time in the wetland system should be five days with a maximum water depth of 50 cm (Mitsch and Reeder, 1991). This resulted in a wetland size of 2.7 ha and is referred to as such in Table 5. Once the proper hydrologic design was designated, WETLAND was used to model the NCOB, sediment and P cycles.

The second wetland design was based on the recommendation that a wetland be at least 2 hectares in size for every 1000 m^3 of inflow per day (Mitsch and Reeder, 1991). This would require the wetland design to be at least 5.4 ha in size and is referred to as such in Table 5.

The final three model applications were intended to examine the design parameters of a constructed wetland, including important considerations such as construction costs and size of the wetland. For this reason, a simulation with a smaller area and deeper water depth ("Smaller," in Table 5); a simulation that examined the amount of biomass in the system ("Plants," in Table 5); and one which examined substrate depth ("Substrate," in Table 5) were considered.

Model Application Results

Table 5 lists the results of the model simulations for the five separate applications. Listed are the influent total, effluent total, and reduction efficiency of nutrients for the simulation runs. The influent and effluent totals are based on the entire two-year time frame, with the reduction efficiency based on the retention of nutrients over this period. A negative reduction efficiency value indicates that the amount of nutrient that has left the wetland is greater than that which entered. In addition, Table 5 lists the average hydraulic residence times (HRT), hydraulic loading rate (HLR) and water depth for the simulations.

All of the designed wetlands reduced the percentage of influent BOD_5 (39.9 to 45.5 percent), TSS (84.5 to 99.9 percent), TN (41.9 to 56.4 percent), and TP (37.7 to 56.5 percent) to levels reported by previous investigators (Bastian and Hammer, 1993). For all simulations, the NH_4^+ concentration (-41.1 to -338.2 percent) increased considerably through the wetland. This is not an unreasonable result because the loading rates of NH_4^+ to the system were small when compared to the loading rates of the organic N and NO_3^- to the system. The 5.4 ha wetland was twice as large, but retained only 6 percent more TN and 10 percent more TP, compared with the 2.7 ha wetland. A wetland with shallower average water depth (0.52 m, ~99.9 percent retention) allowed more particulate matter to settle when compared to the one with deeper water depth (0.87 m depth, ~ 85 percent retention). This retention may also be attributable to the larger wetland volume associated with the shallower wetland designs. Although a system with more vegetation ('Plants') retained 44 percent TN and 42 percent TP, the increase in retention was only 3 to 5 percent. Another interesting note is that the thickness of the wetland substrate was negligible; in fact, the retention in 'Substrate' was greater for NH_4^+, TKN, and TN. Thus, the model suggests that depending on the location of the wetland material, the designer has flexibility with the amount of substrate that needs to be placed in the system, when considering only nutrient retention. Depending on the retention requirements of the wetland, an 'optimal' wetland can be designed for the area by using the information gained from the five simulations. If an increase in TN retention was needed, it can be seen from Table 5 that TN retention increased with an increase in wetland size, plant amounts, and a decrease in substrate thickness. These design parameters can be modified until a desirable retention is attained. Conversely, if a high retention of TSS is the objective of the wetland, it is apparent that a larger wetland with a smaller water depth is more effective for sediment retention.

A drawback of modeling the system with a continuously stirred tank reactor design is that one can not determine the effect of the systems shape, length and width on total retention by the system. This problem can be addressed with a series of CSTRs, but it is unknown if the increase in data input requirements make this feasible, since the model would be more difficult to calibrate, validate and apply.

TABLE 5. Influent, Effluent, and Percent Reduction for Various Nutrients for a Two-Year Period of Wetland Simulations for QN2 Subwatershed. Unless otherwise noted, all parameters are in Mg (1000 Kg).

	NH_4-N	NO_3-N	DON	PON	TKN	TN	BOD_5	TSS	DP	TP	HRT (days)	HLR (cm/d)	Water Depth (cm)
2.7 Hectare													
Total In	0.139	3.77	1.82	5.36	7.33	11.10	108.00	510.00	0.943	1.47			
Total Out	0.231	3.07	2.00	2.16	2.44	5.52	60.10	1.14	0.788	0.79	4.96	10.92	0.52
Percent Reduction	65.8	18.5	-9.4	96.0	66.7	50.3	44.6	99.8	16.4	46.3			
5.4 Hectare													
Total In	0.139	3.77	1.82	5.36	7.33	11.10	108.00	510.00	0.943	1.47			
Total Out	0.196	2.46	2.06	1.22	2.38	4.84	59.10	5.91	0.639	0.639	9.84	5.49	0.53
Percent Reduction	-41.1	34.7	-12.9	97.7	67.5	56.4	45.5	99.9	42.2	56.5			
Smaller*													
Total In	0.139	3.77	1.82E	5.36	7.33	11.10	108.00	510.00	0.943	1.47			
Total Out	0.610	3.38	195E	0.509	3.07	6.45	64.00	76.10	0.900	0.911	3.46	26.72	0.87
Percent Reduction	-338.2	10.3	-6.8	90.5	58.1	41.9	41.0	85.1	4.5	37.9			
Plant**													
Total In	0.139	3.77	1.82	5.36	7.33	11.10	108.00	510.00	0.943	1.47			
Total Out	0.536	3.20	1.95	0.53	3.01	6.21	65.10	79.00	0.839	0.849	3.40	26.72	0.87
Percent Reduction	-285.6	15.2	-6.7	90.2	58.9	41.1	39.9	84.5	11.0	42.1			
Substrate***													
Total In	0.139	3.77	1.82	5.36	7.33	11.10	108.00	510.00	0.943	1.47			
Total Out	0.305	3.44	1.95	0.536	2.82	6.26	64.20	76.10	0.903	0.914	3.46	26.72	0.87
Percent Reduction	-119.5	8.8	-7.0	89.5	61.5	43.6	40.8	85.1	4.2	37.7			

*"Smaller" is 1.1 hectares in size.
**"Plant" has two times the biomass amount of "Smaller."
***"Substrate" has 50 percent the substrate thickness of "Smaller."

SUMMARY AND CONCLUSIONS

The WETLAND model is a user-friendly, dynamic, simulation model developed for design and evaluation of constructed wetlands to optimize NPS pollution control. The model simulates the hydrologic, N, C, bacteria, DO, vegetative, P, and sediment cycles within a wetland system. The model allows for either free water surface (FWS) or subsurface flow (SSF) wetland simulations, and is designed in a modular manner; thus, it gives the user the flexibility to concentrate on simulations of specific parameters. The season/time period breakdown accounts for seasonal variation by allowing the user to change parameter values in the middle of a simulation run. Designed as a continuously stirred tank reactor, the model assumes that all incoming nutrients are evenly mixed throughout the entire volume.

Due to the lack of data collected for NPS pollution control with FWS wetlands, the WETLAND model was evaluated with data collected from a wetland built for municipal wastewater treatment in Benton, Kentucky. Model evaluation included the calibration and validation of the model, performance of two statistical analyses, performance of a sensitivity analysis and the design of a hypothetical constructed wetland to illustrate the application of WETLAND.

A nonparametric Wilcoxon Rank Sum statistical analysis indicated eight out of nine examined outflow predictions were not statistically different from the measured observations. Linear regression analysis indicated that six out of nine examined parameter values were statistically similar. A sensitivity analysis showed the most significant input parameters to the model are those which directly affect bacterial growth and oxygen uptake and movement. WETLAND was applied to a hypothetical simulation for the QN2 subwatershed in the Nomini Creek watershed, located in Virginia. Various designs were examined to determine which design parameters had the largest control on sediment and nutrient retention at the site. These results demonstrated that an improvements in site design could be made based on the WETLAND output and constraints from the site. Selecting one 'optimal' wetland design would depend on many factors, but when considering nutrient retention only, WETLAND may assist in the optimization of wetland design.

The WETLAND model takes a novel approach in its attempt to model the constructed wetland system. It varies from similar models due to its modularity, generality, and the fact that it models both FWS and SSF wetlands. In many ways, this flexibility is the model's strongest asset, because it provides the model user the ability to evaluate a number of designs for constructed wetlands.

Results of the first simulation runs were promising. However, due to the limited data used for evaluating the model's performance, the WETLAND model should be tested more rigorously once more comprehensive data sets become available.

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