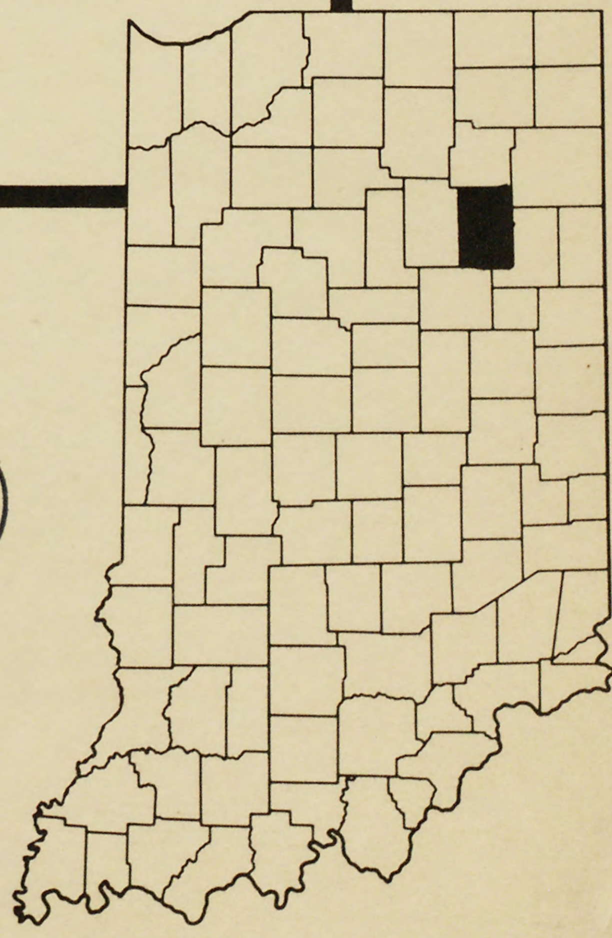
UNITED STATES

DEPARTMENT OF THE INTERIOR

GEOLOGICAL SURVEY

A ONE-DIMENSIONAL, STEADY-STATE, DISSOLVED-OXYGEN MODEL AND WASTE-LOAD ASSIMILATION STUDY FOR WABASH RIVER, HUNTINGTON COUNTY, INDIANA



Open-File Report 80-75

Prepared in cooperation with
Indiana State Board of Health

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By Charles G. Crawford, William G. Wilber, and James G. Peters

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METRIC CONVERSION FACTORS

The inch-pound units used in this report can be converted to the metric system of units as follows:

Multiply inch-pound unit	By	To obtain metric unit
inch (in.)	2.540	centimeter (cm)
foot (ft)	0.3048	meter (m)
foot per mile (ft/mi)	0.1894	meter per kilometer (m/km)
foot per second (ft/s)	0.3048	meter per second (m/s)
cubic foot per second (ft^3/s)	0.0283	cubic meter per second (m^3/s)
mile (mi)	1.609	kilometer (km)
square foot (ft^2)	0.0929	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
million gallon per day (Mgal/d)	3,785	cubic meter per day (m^3/d)
pound (lb)	0.4540	kilogram (km)

ABBREVIATIONS

Abbreviation	Description
BOD	Biochemical-oxygen demand.
CBOD	Carbonaceous biochemical-oxygen demand.
°C	Degree Celsius.
d	Day.
DO	Dissolved oxygen.
e	Base of the natural logarithm, 2.71828.
ft	Foot.
ft^3/s	Cubic foot per second.
$(g/m^2)/d$	Gram per square meter per day.
in.	Inch.
ISBH	Indiana State Board of Health.
K_a	Atmospheric reaeration rate.
K_d	Deoxygenation rate for CBOD.
K_n	First-order kinetics deoxygenation rate for NBOD.
K_r	Stream decay rate for CBOD.
(lb/d)/d	Pound per day per day.
ln	The natural logarithm, base e.
Mgal/d	Million gallons per day.
mg/L	Milligram per liter.
(mg/L)/d	Milligram per liter per day.
mi	Mile.
mL	Milliliter.
NBOD	Nitrogenous biochemical-oxygen demand.
NPDES	National Pollution Discharge Elimination System.
$Q_{7,10}$	Average low flow over a 7-day period with a recurrence interval of 10 years.
RM	River mile.
spec. cond.	Specific conductance.
sta. no.	Station number.
t	Traveltime down the stream.
Temp.	Temperature.
µmho/cm	Micromho per centimeter.

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ABSTRACT

The Indiana State Board of Health is developing a State water-quality management plan that includes establishing limits for wastewater effluents discharged into Indiana streams. A digital model calibrated to conditions in the Wabash River in Huntington County, Ind., was used to predict alternatives for future waste loadings that would be compatible with Indiana stream water-quality standards defined for two critical hydrologic conditions, summer and winter low flows.

The major point-source waste load affecting the Wabash River in Huntington County is the Huntington wastewater-treatment facility.

The most significant factor potentially affecting the dissolved-oxygen concentration during summer low flows is nitrification. However, nitrification should not be a limiting factor on the allowable nitrogenous and carbonaceous waste loads for the Huntington wastewater-treatment facility during summer low flows if the ammonia-nitrogen toxicity standard for Indiana streams is met.

This wasteload assimilation study is not based on a verified model. The changes in stream water quality predicted by the model represent only possible stream response to different effluent conditions.

The dissolved-oxygen standard for Indiana streams, an average of 5.0 milligrams per liter, should be met during summer and winter low flows if the National Pollution Discharge Elimination System's 5-day, carbonaceous biochemical-oxygen demands of a monthly average concentration of 30 milligrams per liter and a maximum weekly average of 45 milligrams per liter are not exceeded.

INTRODUCTION

To meet the goals of section 208 of the Federal Water Pollution Control Act, Amendments of 1972, Public Law 92-500, the ISBH (Indiana State Board of Health) is developing a State water-quality-management plan. A key element of the plan is establishing effluent-discharge limits under the NPDES (National Pollution Discharge Elimination System). These limits for Indiana are designed to maintain the following in-stream water-quality standards:

1. Average DO (dissolved-oxygen) concentrations of at least 5.0 mg/L (milligrams per liter) per calendar day and not less than 4.0 mg/L at any time.

2. Maximum ammonia-nitrogen concentrations of 2.5 mg/L for June-August (based on a 96-hour median lethal concentration of 0.05 mg/L unionized ammonia nitrogen) and 4.0 mg/L for November through March.

3. A maximum concentration for toxic substances of one-tenth the 96-hour median lethal concentration for important indigenous aquatic species (Indiana State Board of Health, 1977, p. 6).

In the past, point-source discharge limitations were based on arbitrary assumptions and "best engineering estimates." In the current approach, a digital model is used to link a stream's water quality with effluent discharges. Once calibrated and verified to the specific stream conditions, the model can be used to predict the effect of varying waste load, streamflow, and stream temperature. This capability is essential to proper waste-load allocation.

The objectives of this study were to (1) use data provided by ISBH for calibrating and verifying a one-dimensional, steady-state, dissolved-oxygen model for the Wabash River in Huntington County, Ind., and (2) use the verified model for determining alternatives for future waste loadings that will ensure that the stream meets Indiana water-quality standards defined for two critical hydrologic conditions, summer and winter low flows. The critical-condition rationale is useful for water-quality planning and management (Hines and others, 1975, p. B5-B6).

BASIN DESCRIPTION

The Wabash River, the major drainage in Huntington County (fig. 1), drains much of northern Indiana as it flows westward across the northern one-third of the State. The watershed, within the Tipton till plain of Indiana, is characterized by low topography (Schneider, 1966, p. 49). Variation in elevation is about 100 ft (Malott, 1922, p. 81). Unconsolidated

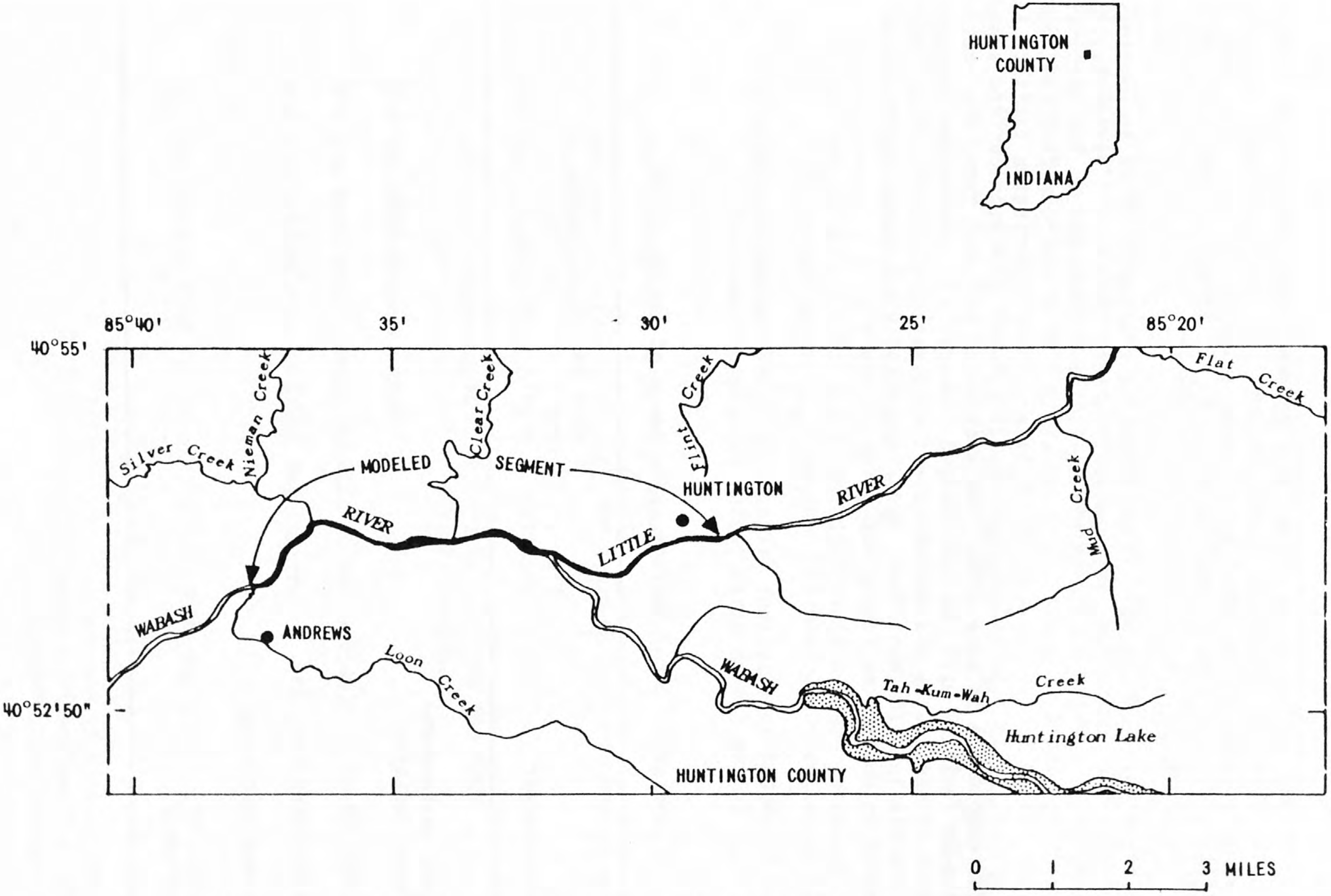

Figure 1.-- Location of modeled segment on the Little and Wabash Rivers, Huntington County, Ind.

valley train and alluvial deposits are present over most of the county. Soils in the area are primarily Blount-Pewano and Morley-Blount associations, whose permeabilities are typically low (Ulrich, 1966, p. 57). Annual precipitation for the area is approximately 33 in. (Schaal, 1966, p. 157). Land use in the county is primarily agricultural.

The major point-source waste load impacting the Wabash River in Huntington County is the Huntington wastewater-treatment facility. Wastewater at this facility is treated by the trickling-filter process before the water is discharged into the Little River approximately 1 mile upstream from the confluence of Little River with the Wabash River. The Huntington facility is to be replaced by an activated-sludge facility under construction and scheduled for completion in late 1979. The wastewater effluent from the activated-sludge facility will be discharged immediately downstream from the confluence of the Little and Wabash Rivers at river mile 405.82. The NPDES-permit restrictions for municipal and industrial dischargers in the Wabash River basin in Huntington County are given in table 1.

Table 1.--NPDES restrictions for municipal and industrial dischargers in the Wabash River basin, Huntington County, Ind.

[Source of data, Indiana State Board of Health, 1978]

Discharger	Flow (Mgal/d)	Five-day BOD (mg/L)	Suspended solids (mg/L)	pH[1]
Huntington wastewater-treatment facility	3.8	[2]30/45	[2]35/525	6/9
Ashland Oil Co.	--	--	--	6/9
Indiana Moulding Co., Inc.	--	[3]10/15	[3]10/15	6/9
Kraftco-Sealtest Corp.	--	--	--	6/9
Sun Oil Co. of Pennsylvania	--	--	--	6/9

[1] Daily minimum/daily maximum.
[2] Monthly average/weekly average.
[3] Daily average/daily maximum.

MODEL DESCRIPTION

A steady-state, one-dimensional, segmented water-quality model developed by Bauer and others (1979) was used in this study. The modeling approach assumes that the various flows, loads, and other factors used do not vary significantly with time. The model uses a modified Streeter-Phelps equation that incorporates nitrogenous, benthic, photosynthetic, and respiratory effects on the DO balance. The dissolved-oxygen balance is represented in the model by the following equation:

$$\text{Zero} = -\frac{1}{A}\frac{\partial (QD)}{dx} - K_a D + K_d L + K_n N - P + R + B \qquad (1)$$

where

- A is stream cross-sectional area,
- D the DO deficit defined as the difference between saturated DO concentration (C_s) and the observed DO concentration (C),
- Q the streamflow,
- x the downstream distance,
- K_a the atmospheric reaeration rate,
- K_d the deoxygenation rate for CBOD (carbonaceous biochemical-oxygen demand),
- L the ultimate CBOD,
- K_n the first-order kinetics deoxygenation rate for NBOD (nitrogenous biochemical-oxygen demand),
- N the NBOD concentration,
- P the mean daily photosynthetic DO production,
- R the oxygen used by plant respiration,

and

- B the oxygen used by the stream-bottom deposits.

By integration, the dissolved-oxygen deficit becomes the sum of the following components:

$$D_o e^{-K_a t} \quad \text{the initial DO deficit,} \tag{2}$$

$$\frac{K_d L_o}{K_a - K_r} (e^{-K_r t} - e^{-K_a t}) \quad \text{the deficit due to CBOD,} \tag{3}$$

$$\frac{K_n N_o}{K_a - K_n} (e^{-K_n t} - e^{-K_a t}) \quad \text{the deficit due to NBOD,} \tag{4}$$

$$\frac{R}{K_a} (1 - e^{-K_a t}) \quad \text{the deficit due to plant respiration,} \tag{5}$$

$$\frac{B}{K_a} (1 - e^{-K_a t}) \quad \text{the deficit due to bottom deposits, and} \tag{6}$$

$$\frac{-P}{K_a} (1 - e^{-K_a t}) \quad \text{the deficit due to mean daily photosynthetic production,} \tag{7}$$

where

D_o is the DO deficit at some initial time, t_o,

t the traveltime down the stream,

L_o the ultimate CBOD concentration at some initial time, t_o,

K_r the stream decay rate for CBOD,

N_o the ultimate NBOD concentration at some initial time, t_o,

and

e the base of the natural logarithm, 2.71828.

DATA COLLECTION

The model segment drains a 367-mi^2 area and extends from RM 2.29 on the Little River at Huntington to RM 401.68 on the Wabash River at Andrews (fig. 2). Twelve stations were sampled in two water-quality surveys during low-flow conditions, August 5, 1977, and September 29, 1978.

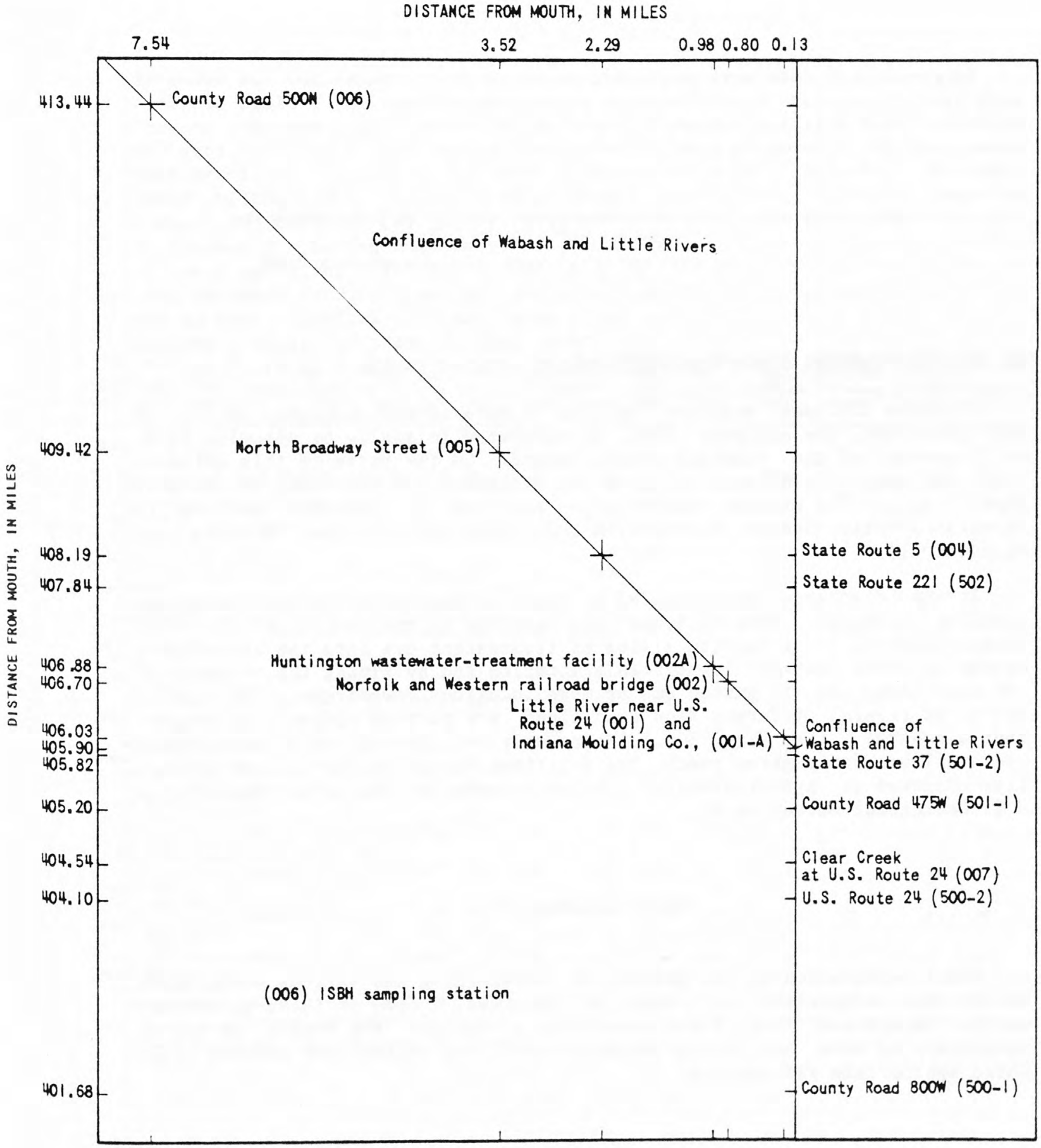

Figure 2. -- Locations of sampling stations in the Wabash River basin, Huntington County, Ind.

Water-quality data were collected every 4 to 6 hours for 24 hours at each site by the ISBH Water-Quality Surveillance Section. Field measurements included dissolved-oxygen concentration, temperature, specific conductance, and pH. Composite samples were analyzed by the ISBH Laboratory for 5-day BOD (biochemical-oxygen demand), ammonia nitrogen, total Kjeldahl nitrogen, and nitrite nitrogen plus nitrate nitrogen. Additionally, three long-term CBOD determinations from the first survey and two from the second were observed periodically throughout the incubation period. A summary of water-quality data collected by the ISBH is presented in tables 2 and 3. The annual laboratory performance evaluation by the U.S. Environmental Protection Agency in 1978 indicated that water-quality analyses done by the Indiana State Board of Health Laboratory were accurate to within 1 percent (S. R. Kin, Indiana State Board of Health, written commun., 1979).

Ultimate CBOD was measured by the Elmore method (Ludzack, 1966). In this procedure, the ultimate CBOD is estimated initially by assuming it to be 30 percent of the chemical-oxygen demand. On the basis of this calculation, the sample is diluted to give an estimated ultimate CBOD of not more than 4 mg/L. The diluted samples were analyzed by standard methods in American Public Health Association and others (1976). (See Results and Discussion.)

Stream discharge was measured by the U.S. Geological Survey during the sampling periods. Time of travel was measured by the Geological Survey by instantaneously injecting a slug of flourescent dye into the stream at a bridge or other easily identifiable location and by timing the movement of the dye cloud as it passed one or more locations downstream. The time of travel at several different flow conditions are plotted against the concurrent discharge. (See figs. 3-6.) Where only one time of travel measurement was available for a given reach, the relationship was estimated from velocities observed in hydrologically similar reaches of the river (unpublished U.S. Geological Survey data).

MODEL CALIBRATION

Model calibration is the process of determining the various model parameters used to describe the system of interest. Some of these parameters may be determined from field measurements, whereas, for others, it may be neccessary to make an initial estimate until the calibration process indicates appropriate refinements.

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Wabash River basin, Huntington County, Ind., August 5, 1977

[Water-quality data collected by Indiana State Board of Health; discharge measured by U.S. Geological Survey]

Location and station in the Wabash River basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Field pH range	Average temp.[1] (°C)	Average spec. cond.[1] (μmho/cm at 25°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Twenty-day CBOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)	Chloride (mg/L)
Little River at County Road 500N (006)	7.54	17	--	25.3	881	38	7.8	2.6	--	<0.1	0.5	0.1	52
Little River at North Broadway Street (005)	3.52	19.2	--	25.8	887	26	7.5	2.4	--	<.1	--	<.1	53
Little River at State Route 5 (004)	2.29	--	--	26.0	890	44	7.2	3.0	--	<.1	.6	<.1	64
Huntington wastewater-treatment facility (002-A)	.98	3.9	--	23.6	757	34	4.6	47.0	76.0	3.9	8.3	2.4	82
Little River at Norfolk and Western Railroad bridge (002)	.80	--	--	26.0	860	28	5.4	22.0	--	2.6	6.0	1.1	94
Indiana Moulding Company, Inc. (001-A)	.13	--	--	--	--	3	--	<1	--	<.1	--	.3	27
Wabash River at State Route 221 (502)	407.84	23	--	26.6	582	34	7.5	3.0	--	<.1	.9	1.5	44
Wabash River at State Route 37 (501-2)	405.82	--	--	26.2	736	57	6.0	7.1	--	.4	--	1.0	53
Wabash River at County Road 475W (501-1)	405.20	--	--	26.2	736	46	6.1	6.4	--	.2	--	1.0	51
Clear Creek at U.S. Route 24 (007)	404.54	.9	--	24.7	750	32	6.7	8.7	--	1.9	--	.3	88
Wabash River near U.S. Route 24 (500-2)	404.10	--	--	26.0	737	32	6.2	5.5	9.2	.2	1.3	.8	53
Wabash River at County Road 800W (500-1)	401.68	--	--	27.0	725	92	7.5	4.3	--	<.1	1.3	1.0	62

[1] Average of data collected every 4 to 6 hours for 24 hours.

Table 3.--Water-quality analyses and discharge measurements for sampling stations in the Wabash River basin, Huntington County, Ind., September 29, 1978

[Water-quality data collected by Indiana State Board of Health; discharge measured by U.S. Geological Survey]

Location and station in the Wabash River basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Field pH range	Average temp.[1] (°C)	Average spec. cond.[1] (µmho/cm at 25°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Twenty-day CBOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)	Chloride (mg/L)
Little River at North Broadway Street (005)	3.52	--	6.4-7.2	14.3	--	15	8.4	1.2	--	0.1	0.4	0.5	69
Little River at State Route 5 (004)	2.29	--	6.5-7.3	15.5	--	13	8.7	1.5	--	<.1	.4	.8	64
Huntington wastewater-treatment facility (002-A)	.98	--	6.5-7.1	17.6	--	30	5.8	62.1	101.3	4.4	8.9	4.5	140
Little River at Norfolk and Railroad bridge (002)	.80	--	6.8-7.3	15.2	--	19	4.9	13.0	--	1.1	2.5	1.1	87
Indiana Moulding Company, Inc. (001-A)	.13	--	--	--	--	17	6.7	--	--	<.1	.2	.2	34
Little River near U.S. Route 24 (001)	.13	16.2	6.8-7.2	15.5	--	18	2.6	12.0	19.0	1.0	2.4	.8	82
Wabash River at State Route 221 (502)	407.84	18.4	6.7-7.4	16.3	--	12	9.1	2.9	5.5	<.1	1.0	1.1	53
Wabash River at State Route 37 (501-2)	405.82	38.2	6.7-7.7	15.7	--	15	6.4	5.8	--	.6	1.6	.9	54
Wabash River at County Road 475W (501-1)	405.20	--	6.9-7.4	15.8	--	12	6.8	5.3	--	.4	1.4	.9	65
Clear Creek at U.S. Route 24 (007)	404.54	.32	7.0-7.4	7.9	--	15	7.9	7.0	--	1.4	2.2	.2	43
Wabash River near U.S. Route 24 (500-2)	404.10	--	6.9-7.6	7.5	--	11	7.5	5.7	10.8	.3	1.2	.8	65
Wabash River at County Road 800W (500-1)	401.68	47.7	6.9-7.5	8.6	--	15	7.6	4.2	--	.2	1.0	.8	64

[1] Average of data collected every 4 to 6 hours for 24 hours.

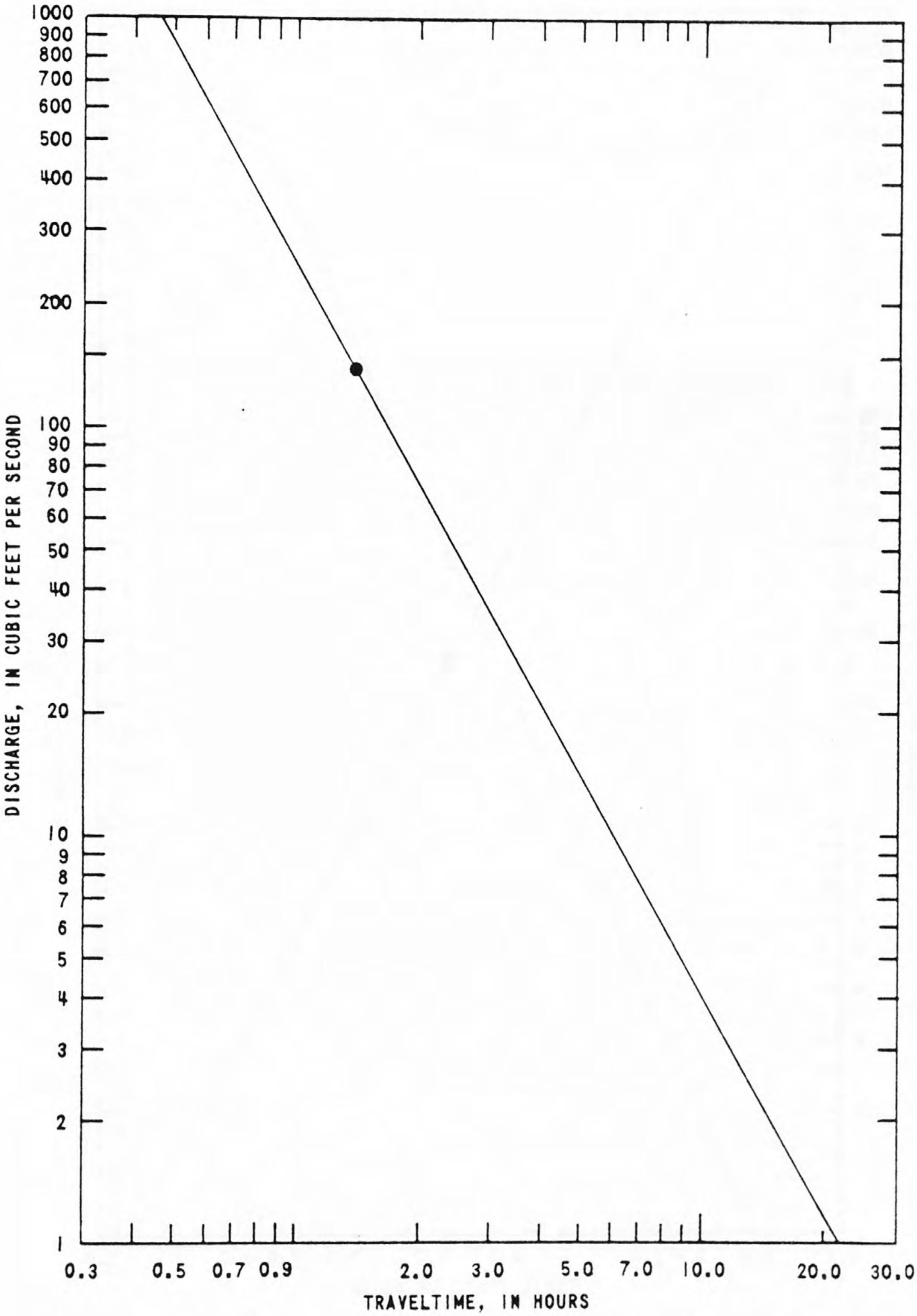

Figure 3.-- Relation of discharge to estimated traveltime of the peak dye concentration for the Little River, river miles 2.29 to 0.98.

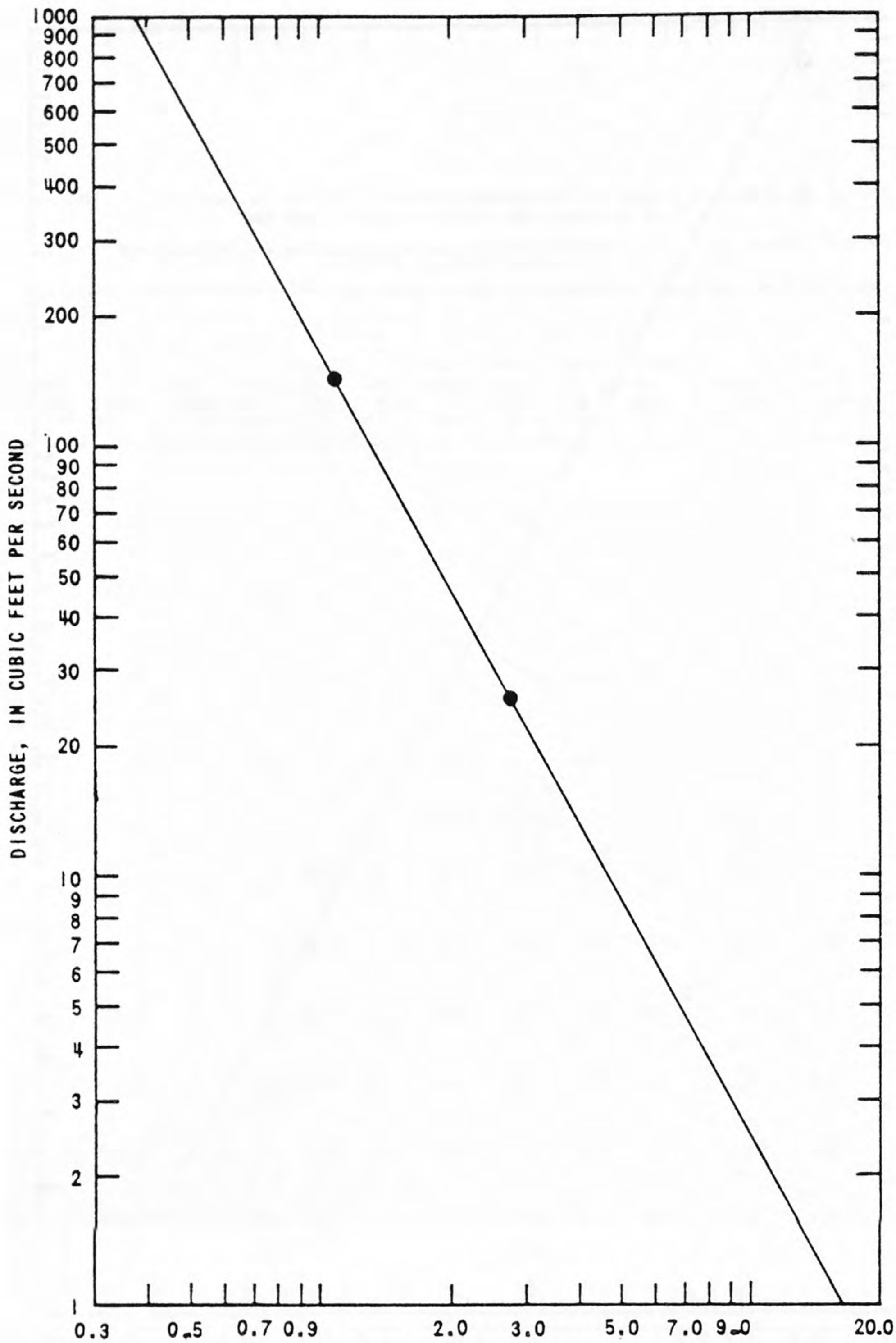

Figure 4.-- Relation of discharge to traveltime of the peak dye concentration for the Little River from river mile 0.98 to the Wabash River at river mile 405.82.

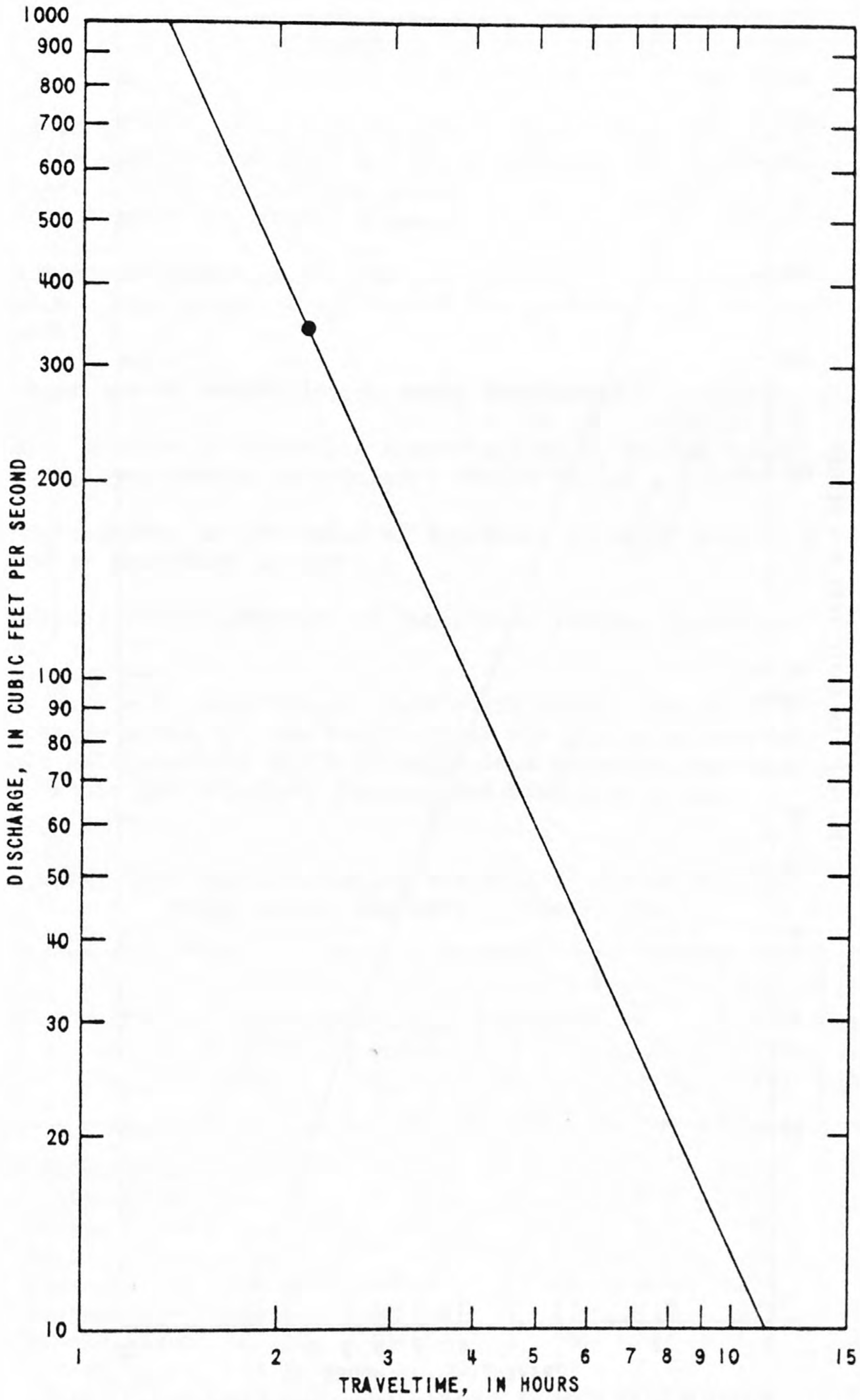

Figure 5.-- Relation of discharge to estimated traveltime of the peak dye concentration for the Wabash River, river miles 405.82 to 404.10.

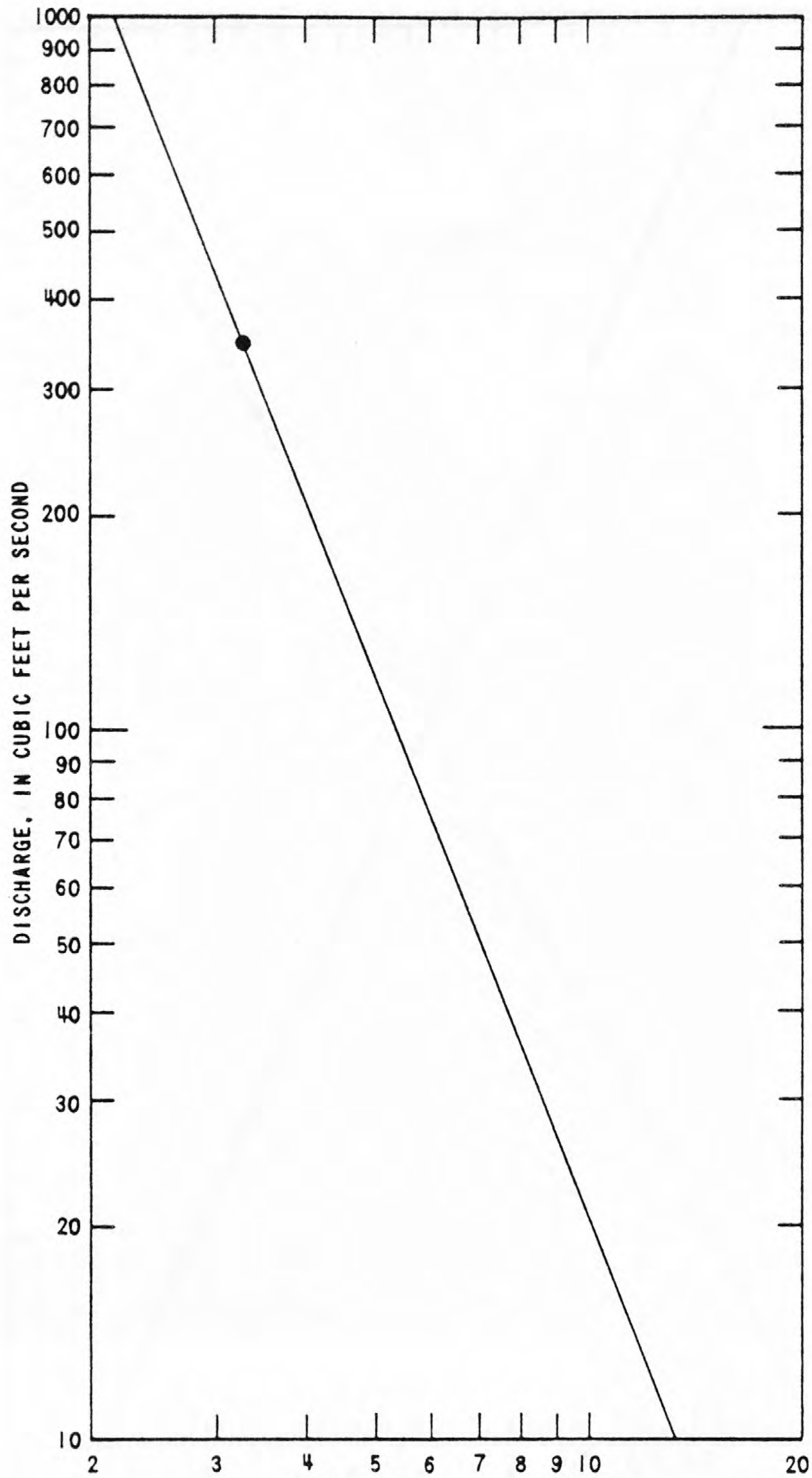

Figure 6.-- Relation of discharge to estimated traveltime of the peak dye concentration for the Wabash River, river miles 404.10 to 401.68.

Parameter Estimation

For modeling, the stream segment used in the study must be divided into reaches. The number of reaches and the boundaries are determined by the program user. Bauer and others (1979, p. 15-16) suggested the following criteria for dividing the stream segment:

1. Each point-source waste flow and tributary should start a reach. Discharge should enter the upstream end of the reach.

2. Linear runoff should define reach boundaries.

3. Major changes in hydraulic characteristics, stream temperature, or reaction coefficients should define a reach.

4. Future inputs or increases or decreases in input should be used to determine a reach.

The physical characteristics of the stream reaches studied are presented in table 4.

The two sets of model-input parameters used in the calibration of the dissolved-oxygen model for the Wabash River are presented in tables 5 and 6. Included are water-quality and hydrologic data representing initial upstream conditions, waste and tributary inputs, and main-stem sites.

Table 4.--Physical characteristics for modeled stream reaches, Wabash River basin, Huntington County, Ind.

Reach	River	Starting river mile	Ending river mile	Length of reach	Slope (ft/mi)	Average[1] width (ft)	Average[1] depth (ft)
1	Little River	2.29	0.98	1.31	7.6	60	0.7
2	Little River	.98	.80	.18	6.1	60	.7
3	Little River	.80	.13	.67	8.2	60	.7
4	Little River	.13	.0	.13	10.9	60	.7
5	Wabash River	405.90	405.82	.08	4.2	100	.9
6	Wabash River	405.82	405.20	.62	4.2	100	1.0
7	Wabash River	405.20	404.54	.66	3.4	100	1.0
8	Wabash River	404.54	404.10	.44	3.4	100	1.0
9	Wabash River	404.10	401.68	2.42	3.4	100	1.0

[1] Average of the estimates for the two water-quality surveys, August 5, 1977, and September 29, 1978.

Table 5.--Model input for the Little and Wabash Rivers, Huntington County, Ind., August 5, 1977

[Water-quality data collected by Indiana State Board of Health; discharge measured by U.S. Geological Survey]

Reach	Upstream boundary of modeled reach (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Linear runoff (ft^3/s)	Time of travel to next site (hours)	Ultimate CBOD (mg/L)	NBOD (mg/L)	DO (mg/L)	DO deficit (mg/L)	Temp. (°C)	Mean daily photosynthetic DO production [(mg/L)/d]	Benthic-oxygen demand [$(g/m^2)/d$]	K_r (day^{-1})	K_d (day^{-1})	K_n (day^{-1})	K_a (day^{-1})
1	Little River at State Route 5, RM 2.29 (004)----------	408.19	19.2	0.0	4.3	5.0	0.4	7.2	0.8	26	0.0	0.0	0.20	0.20	0.5	10.8
2	Huntington wastewater-treatment facility, RM 0.98 (002-A)--------	406.88	3.9	.0	.5	80	16.9	4.6	3.8	26	.0	14.6	12.4	.20	2.5	12.3
3	Little River at Norfolk and Western Railroad bridge, RM 0.80 (002)----------	406.70	.5	.0	1.9	420	50	4.6	3.8	26	.0	8.0	1.45	.20	2.5	12.3
4	Little River near U.S. Route 24, RM 0.13 (001)----------	406.03	--	.2	.4	--	--	--	--	26	.0	8.0	1.45	.20	2.5	12.3
5	Wabash River at confluence-----	405.90	23.2	.1	.2	5.0	.4	7.5	.4	26	-2.5	2.0	.98	.15	2.5	6.8
6	Wabash River at State Route 37 (501-2)--------	405.82	--	1.4	1.9	--	--	--	--	26	-2.1	2.0	.98	.15	2.5	6.3
7	Wabash River at County Road 475W (501-1)--------	405.20	--	1.5	2.0	--	--	--	--	26	-2.1	2.0	.98	.15	2.5	5.3
8	Clear Creek (007)----------	404.54	.9	1.0	1.4	14.5	8.2	6.2	2.0	26	-2.1	.4	.98	.15	2.5	5.3
9	Wabash River near U.S. Route 24 (500-2)--------	404.10	--	5.4	6.8	--	--	--	--	27	-1.1	.3	1.02	.156	2.7	6.9

Table 6.--Model input for the Little and Wabash Rivers, Huntington County, Ind., September 29, 1978

[Water-quality data collected by Indiana State Board of Health; discharge measured by U.S. Geological Survey]

Reach	Upstream boundary of modeled reach (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Linear runoff (ft^3/s)	Time of travel to next site (hours)	Ultimate CBOD (mg/L)	NBOD (mg/L)	DO (mg/L)	DO deficit (mg/L)	Temp. (°C)	Mean daily photosynthetic DO production [(mg/L)/d]	Benthic-oxygen demand [(g/m^2)/d]	K_r (day^{-1})	K_d (day^{-1})	K_n (day^{-1})	K_a (day^{-1})
1	Little River at State Route 5, RM 2.29 (004)----------	408.19	12.4	0.0	5.4	2.4	0.4	8.7	1.1	15	0.0	0.0	0.12	0.12	0.20	8.6
2	Huntington wastewater-treatment facility, RM 0.98 (002-A)--------	406.88	3.7	.0	.6	102	19.1	5.8	3.7	15	.0	7.3	7.47	.12	.98	10.5
3	Little River at Norfolk and Western Railroad bridge, RM 0.80 (002)----------	406.70	--	.0	2.3	--	--	--	--	15	.0	2.6	.87	.12	.98	10.5
4	Little River at RM 0.13 (001)----------	406.03	--	.2	.4	--	--	--	--	15	.0	2.6	.87	.12	.98	10.5
5	Wabash River at confluence--	405.90	21.7	.1	.3	5.6	.4	9.1	.7	16	-4.3	.7	.62	.09	1.06	5.4
6	Wabash River at State Route 37 (S01-2)--------	405.82	--	1.4	2.2	--	--	--	--	16	-2.4	.8	.62	.09	1.06	5.3
7	Wabash River at County Road 475W (S01-1)--------	405.20	--	1.5	2.3	--	--	--	--	16	-2.4	.8	.62	.09	1.06	5.3
8	Clear Creek (007)----------	404.54	.3	1.0	1.5	13.5	6.1	7.9	2.3	16	-2.4	.8	.62	.09	1.06	4.5
9	Wabash River at U.S. Route 24 (S00-2)--------	404.10	--	5.4	7.4	--	--	--	--	16	-.9	.8	.62	.09	1.06	6.1

Hydrology

The flow data collected during the August 1977 and September 1978 surveys represent steady-state, low-flow conditions. Flow in the Wabash River upstream from its confluence with Little River is regulated by the Huntington Reservoir and is maintained by the U.S. Army Corp of Engineers.

Contributions of water from other than known sources was attributed to ground-water inflow. Increases in flow of this type were assumed to be added to the river linearly with distance. The largest unaccounted for inflow occurred immediately downstream from RM 405.82. Concentrations of CBOD, NBOD, and DO of the ground-water inflow were assumed to be 2.5, 0.4, and 2.0 mg/L, respectively, unless otherwise noted.

Average depth of reach was estimated by the following equation:

$$D = \frac{Q}{V \cdot W} \tag{8}$$

where

- D is the average depth of reach, in feet,
- Q the average discharge of reach, in cubic feet per second,
- W the average width of reach, in feet,

and

- V the average velocity of reach, in feet per second.

Carbonaceous Biochemical-Oxygen Demand

Long-term CBOD measurements were made for only a small percentage of the samples collected during the two water-quality surveys. Five-day BOD measurements were made for the other samples. The authors used the ratio of ultimate CBOD to 5-day CBOD, from the long-term CBOD measurements, to estimate the ultimate CBOD concentrations for samples where only a 5-day BOD determination was made. (See tables 7 and 8 and fig. 7.) This method gives reliable estimates of ultimate CBOD (Stamer and others, 1979). For the August 1977 data, the ultimate CBOD of all samples was assumed to be 1.7 times that of the 5-day BOD. A ratio of 1.6 was determined for the September 1978 data. These ratios were used to estimate the ultimate CBOD concentrations where only a 5-day BOD was determined.

The deoxygenation rate for CBOD, K_d, was determined by plotting the percentage of long-term CBOD remaining against time. The least squares method was then used to find the best estimate of the CBOD deoxygenation rate, K_d (Nemerow, 1974, p. 93). The calculated stream deoxygenation rate was 0.17 day^{-1} at 20°C for the August 1977 data and ranged from 0.22 to

Table 7.--Carbonaceous biochemical-oxygen-demand data for sampling stations in the Wabash River basin, Huntington County, Ind., August 5, 1977

[Day, number of days after beginning of analysis; data collected and analyzed by Indiana State Board of Health]

Location and station: Huntington wastewater-treatment facility *(002-A)*			Location and station: Wabash River at U.S. Route 24 *(001)*		
Day	CBOD (mg/L)	Percentage remaining	Day	CBOD (mg/L)	Percentage remaining
2	29	64	2	2.9	69
4	45	44	4	4.7	49
5	47	41	5	5.5	40
7	55	31	7	6.6	28
10	62	23	10	7.2	22
14	70	13	14	8.4	9
18	74	8	17	9.2	0
20	76	5	20	9.2	0
Ultimate CBOD = 80 mg/L K_d (base e) = 0.14 day^{-1}			Ultimate CBOD = 9.2 mg/L K_d (base e) = 0.17 day^{-1}		

Table 8.--Carbonaceous biochemical-oxygen-demand data for sampling stations in the Wabash River basin, Huntington County, Ind., September 29, 1978

[Day, number of days after beginning of analysis; data collected and analyzed by Indiana State Board of Health]

Location and station: Huntington wastewater-treatment facility (002-A)			Location and station: Little River at U.S. Route 24 (001)		
Day	CBOD (mg/L)	Percentage remaining	Day	CBOD (mg/L)	Percentage remaining
1	13.4	87	1	4.5	77
2	28.0	73	2	8.6	56
3	51.1	50	3	10.0	49
5	62.1	39	5	12.0	38
6	68.2	33	6	13.6	30
8	76.6	25	8	15.0	23
10	80.6	21	10	16.1	17
11	84.8	17	11	16.3	16
15	96.4	6	15	17.7	9
20	101.3	1	20	19.0	2
Ultimate CBOD = 102 mg/L K_d (base e) = 0.18 day^{-1}			Ultimate CBOD = 19.4 mg/L K_d (base e) = 0.15 day^{-1}		

Table 8.--Carbonaceous biochemical-oxygen-demand data for sampling stations in the Wabash River basin, Huntington County, Ind., September 29, 1978--Continued

Location and station: Wabash River at State Route 221 (502)			Location and station: Wabash River near U.S. Route 24 (500-2)		
Day	CBOD (mg/L)	Percentage remaining	Day	CBOD (mg/L)	Percentage remaining
2	1.3	77	2	2.0	82
3	2.0	64	3	5.0	55
5	2.9	48	5	5.7	48
6	3.6	36	6	6.6	40
8	4.4	21	8	7.0	36
10	4.8	15	10	7.4	33
11	5.0	11	11	7.8	29
15	5.6	0	15	9.4	15
20	5.5	0	20	10.8	2
Ultimate CBOD = 5.6 mg/L; K_d (base e) = 0.22 day^{-1}			Ultimate CBOD = 11.0 mg/L; K_d (base e) = 0.11 day^{-1}		

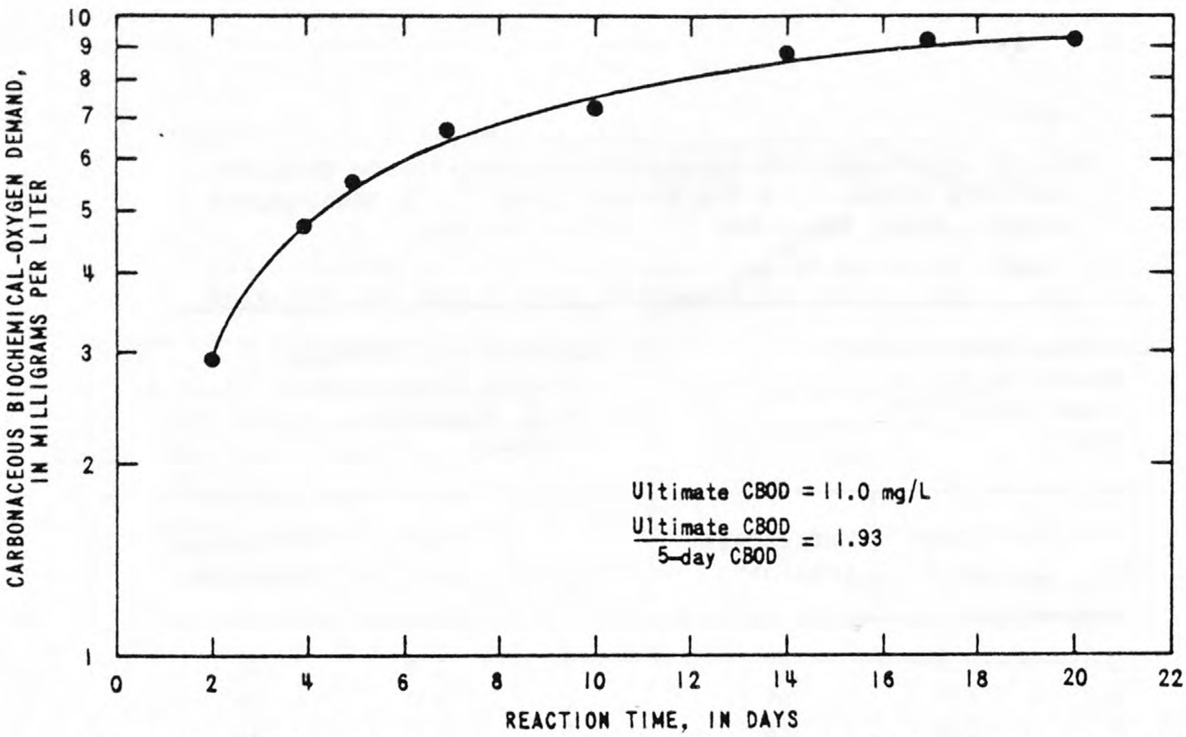

Figure 7.-- Relation of carbonaceous biochemical-oxygen demand to time at river mile 404.10, Wabash River, August 5, 1977.

0.11 day^{-1} at 20°C for the September 1978 data. In the model calibration, an average rate of 0.15 day^{-1} at 20°C was used for reaches on the Little River, and an average rate of 0.11 day^{-1} at 20°C was used for reaches on the Wabash River.

The stream decay rate for CBOD, K_r, was calculated on the basis of CBOD load, rather than concentration, so that changes in concentration due to dilution would be included (Thomann, 1972, p. 96).

$$K_r = \ln \left[\frac{C_d}{C_u} \right] t^{-1} \qquad (9)$$

where

K_r is the stream decay rate for CBOD, in day^{-1},

C_d and C_u the loads of CBOD at downstream and upstream sites, respectively, in pounds per day,

t the time of travel between the two sites, in days,

and

ln the natural logarithm, base e.

Values of K_r, for reaches downstream from the Huntington wastewater-treatment facility, in September 1978 were 9.4 and 1.1 day^{-1} at 20°C. These values were also assumed for the August 1977 model because insufficient data were collected during the August 1977 water-quality survey to calculate K_r for these reaches. Values of K_r, downstream from the confluence of the Little and Wabash Rivers, were 0.48 day^{-1} at 20°C for the August 1977 survey and 0.74 day^{-1} at 20°C for the September 1978 survey.

There are several anomalies in the CBOD data for the two surveys. The increase in observed CBOD downstream from the Huntington wastewater-treatment facility in Little River (station 002, RM 0.80) during the August 1977 survey cannot be accounted for solely by the CBOD load in the wastewater effluent. This anomaly could be due to several factors. First, the wastewater and river water at station 002 may not have been well mixed. As a result, a sample taken from the wastewater plume would yield an unrepresentatively high value. Transect samples were not collected at this site. Second, an underestimation of the effluent CBOD concentration would also result in this discrepancy. Third, sediment having an oxygen demand may have been scoured between the wastewater outfall and the sampling station. Finally, two sewer overflows and one wastewater-treatment facility overflow enter Little River between station 004 (RM 2.29), sampled to determine dissolved headwater constituents, and station 002 (RM 0.80). The wastewater-treatment facility overflow is at RM 0.80. Owing to design problems in the sewer system and a wastewater-treatment plant that is overloaded, untreated wastewater from the sewer overflows, and the wastewater-treatment facility is discharged into the Little River on virtually a daily basis (Kay Walters, Huntington Water Pollution Control Plant, oral commun., 1979). None of these discharges was sampled during either water-quality survey. Therefore, the influences of the three waste loads on the stream dissolved-oxygen balance cannot be assessed.

The increase in CBOD load was attributed to the sewer overflow at RM 0.80. The CBOD concentration of this wastewater effluent was assumed to be the same as the BOD concentration of the wastewater flowing into the wastewater-treatment facility. This value was 258 mg/L, 5-day BOD (Kay Walters, Huntington Water Pollution Control Plant, oral commun., 1979). On the basis of the assumed CBOD concentration, a discharge of 0.5 ft^3/s was estimated for the overflow by mass balance.

During the September 1978 survey, an increase in CBOD load between RM 405.20 and RM 404.10 could not be accounted for by known point-source CBOD loads. The increase may have been a result of the scouring of oxygen-demanding sediment, or nonpoint sources, and was attributed to nonpoint sources for the model calibration. To account for the increase, the authors assumed that ultimate CBOD concentrations of 26 mg/L were entering the river in linear runoff. In addition, the ultimate CBOD concentration entering the river from Clear Creek was assumed to be 13.5 mg/L instead of the estimated 11.2 mg/L. These assumptions were based on a mass balance of the CBOD in the stream at RM 405.20 and the concentration of CBOD required in the linear runoff to equal the observed CBOD at RM 404.10.

The deoxygenation rate for CBOD, K_d, and the stream decay rate for CBOD, K_r, were adjusted for temperature by the following equation:

$$(K)_T \quad \text{Carbonaceous} = (K)_{20°C}(1.047^{T-20°C}) \tag{10}$$

where

(K) is the base-e reaction constant at temperature T, in day^{-1},

1.047 a constant, applicable over a typical field-temperature range,

and

T the temperature, in degrees Celsius.

Nitrogenous Biochemical-Oxygen Demand

Because of the complex nature of nitrification and its application to the Wabash River, a brief description of the process is appropriate. In the broadest sense, nitrification is the biologically mediated increase in the oxidation state of reduced organic or inorganic forms of nitrogen. A narrower definition restricts nitrification to the autotrophic oxidation of ammonia to nitrate; nitrite is an intermediate compound. The two-step oxidation by nitrifying bacteria is as follows:

Ammonia oxidation

$$NH_4^+ + \frac{3}{2}\ O_2 = NO_2^- + 2H^+ + H_2O + \text{energy} \tag{11}$$

Nitrite oxidation

$$NO_2^- + \frac{1}{2}\ O_2 = NO_3^- + \text{energy} \tag{12}$$

The amount of dissolved oxygen consumed by the two-step oxidation can be significant. Wezernak and Gannon (1967) found experimentally that 3.22 mg of molecular oxygen is needed to convert 1 mg of ammonia nitrogen (NH_4 - N) to nitrite nitrogen (NO_2 - N) and that 1.11 mg is needed to convert 1 mg of nitrite nitrogen to nitrate nitrogen [NO_2 - N to NO_3 - N]. In a typical non-nitrified secondary effluent, the ammonia-nitrogen concentration may range from 12 to 50 mg/L (Metcalf and Eddy, 1972, p. 231). The potential oxygen demand placed on the receiving stream in this example would range from 52 to 217 mg of oxygen per liter of effluent.

The factors that should be evaluated in determining the significance of nitrification in an aquatic environment are habitat suitability, changes in loads of nitrogen species, and nitrifier populations. Other environmental factors necessary for nitrification are sufficient ammonia loads, water temperatures greater than 20°C, near neutral or slightly alkaline pH, and an environment free of toxic substances such as manganese.

The environmental conditions necessary for nitrification in the Little and Wabash Rivers were favorable during the August 1977 water-quality survey. During this survey, stream temperatures averaged 26°C. Values of pH were at or near neutrality. Ammonia-nitrogen concentration in the Little and Wabash Rivers downstream from the Huntington wastewater-treatment facility ranged from <0.1 to 2.6 mg/L. Conditions during the September 1978 survey were somewhat less than optimal because stream temperature was near 15°-16°C for all reaches. Ammonia-nitrogen concentration of the Little and Wabash Rivers downstream from the Huntington wastewater-treatment facility ranged from 0.2 to 1.1 mg/L. However, significant losses of ammonia nitrogen were still observed.

The nitrification process in streams is difficult to model. One of the reasons for this is that the rate of change of nitrogen compounds is dependent not only on nitrification but also on other processes that are involved in the nitrogen cycle. For example, nitrification causes a decrease in the ammonia-nitrogen concentration and an increase in the nitrate-nitrogen concentration. However, the ammonia-nitrogen concentration may increase in the stream as a result of the hydrolysis of organic nitrogen or the reduction of nitrate nitrogen or may decrease as a result of aquatic plant consumption or conversion to organic nitrogen for cell synthesis by heterotrophic bacteria. Ruane and Krenkel (1977) discussed several factors other than nitrification effecting a change in nitrate-nitrogen concentration in a stream, including denitrification, respiratory reduction, and assimilatory reduction. Consequently, because of the difficulty in estimating the significance of these processes, a mass-balance for nitrogen species from upstream to downstream locations is not always achieved.

The decision to include nitrification in the model was based on significant losses of ammonia nitrogen in the streams during the two water-quality surveys. Nitrification was simulated in the Little and Wabash Rivers by a BOD-equivalent model. More sophisticated models (for example, a sequential-reaction model) may be used to estimate the effect of nitrification on stream dissolved oxygen. However, insufficient data were collected to use them. Thomann and others (1971) stated that the BOD-equivalent model is adequate to describe nitrification if the difference between ammonia-nitrogen and organic-nitrogen concentrations is not great.

The deoxygenation rate for NBOD, K_n, was determined on the basis of change in ammonia-nitrogen load rather than concentration. Ammonia nitrogen was chosen over nitrate nitrogen because increases in nitrate nitrogen were greater than could be accounted for solely from the conversion of ammonia nitrogen.

$$K_n = \frac{1}{t} \ln \left[\frac{C_u}{C_d} \right] \tag{13}$$

where

C_u and C_d are the loads of ammonia nitrogen at upstream and downstream sites, respectively, in pounds per day,

t the time of travel between the two sites, in days, and

ln the natural logarithm.

Sufficient data were collected on the Little River to determine a value of K_n for only one reach during one of the two water-quality surveys. A value of 1.5 day^{-1} at 20°C was determined for the Little River from data collected at sampling stations 002 and 001 during the September 1978 survey. This value was used in both sets of data for the model simulations for the Little River downstream from the Huntington wastewater-treatment facility. An average value of 0.3 day^{-1} at 20°C (Thomann, 1972, p. 97) was used for reaches upstream from the wastewater-treatment facility during both model simulations.

Values of K_n at 20°C, calculated for reaches on the Wabash River, ranged from 1.2 to 3.3 day^{-1} during the two water-quality surveys. Average values were 1.4 day^{-1} at 20°C for the August 1977 survey and 2.2 day^{-1} at 20°C for the September 1978 survey. The rate calculated for the August 1977 survey corresponds well with the rate calculated for the Little River during the September 1978 survey. Therefore, the value 1.4 day^{-1} at 20°C was used for all stream reaches downstream from the Huntington wastewater-treatment facility for both model simulations. This rate is outside the normal range of nitrification rates reported for rivers (Thomann, 1972, p. 97). However, it is approximately equivalent to the rate reported for hydrologically similar reaches of the Willamette River by Hines and others (1975).

Nitrogenous decay coefficients were adjusted for temperature by the following equation:

$$(K)_T \text{ nitrogenous} = K_{20°C}(1.09^{T-20°C}) \qquad (14)$$

where

(K) is the base-e reaction constant, in day^{-1},

1.09 a constant, applicable over a typical field-temperature range,

and

T the temperature, in degrees Celsius.

Benthic-Oxygen Demand

The in-stream CBOD decay coefficient, K_r, determined for the two water-quality surveys suggests that CBOD is settling from the water column in some reaches in addition to being removed by biochemical oxidation. Although this condiiton suggests a benthic-oxygen demand would be present, no attempt was made to measure benthic-oxygen demand in the field during the surveys.

In some large rivers, benthic-oxygen demand can be omitted as a significant oxygen demand. However, in shallow rivers and streams, benthic-oxygen demand can become one of the most important factors affecting the dissolved-oxygen dynamics. According to Thomann (1972, p. 104) an average benthic-oxygen demand of sludge deposits is 4 $(g/m^2)/d$ (grams per square meter per day). This demand in a river 10 ft deep consumes 1.3 (mg/L)/d dissolved oxygen. The same benthic-oxygen demand in a shallow stream or river, 1 ft deep, consumes 13.1 (mg/L)/d dissolved oxygen.

Benthic-oxygen demand is generally assumed to be dependent on the relation between accumulated deposits and the oxygen demand exerted by the deposits. The accumulation of CBOD on the stream bottom is related to the period of stable hydrologic conditions when CBOD may settle. A velocity of 0.6 ft/s is generally assumed to be the critical velocity at which organic solids may deposit (Velz, 1970, p. 162). The oxygen demand of the deposit is a function of accumulation and the rate of consumption. For short periods of time after a scour has removed previous deposits, the level of accumulation would be low.

If the rate of settling is greater than the rate of consumption, CBOD will accumulate much faster than it is consumed. Additionally, only the upper 1 to 2 inches of the benthic deposit is assumed to be active in the stream deoxygenation process. CBOD underlying this upper layer is utilized by anaerobic decomposition (Phelps, 1944, p. 122). This means that CBOD can be removed from the water without exerting an oxygen demand until it is resuspended during scouring.

Phelps (1944, p. 125) reported a method for estimating the effect of deoxygenation of benthic deposits on stream dissolved oxygen. The method estimates the amount of CBOD accumulation on the stream bottom in a given reach as follows:

$$L_d = \frac{P}{K^1} (1 - e^{-K^1 t}) \tag{15}$$

where

L_d is the accumulated deposit of CBOD, in pounds,

P the daily deposit of CBOD, in pounds,

K^1 the base-e reaction-rate coefficient for benthic-oxygen demand, in day^{-1},

and

t the elapsed time of accumulation, in days.

The authors assumed that the daily deposit of CBOD, P, is the difference between the amount of CBOD removed in a reach and that estimated to be oxidized in the water column in the reach.

The oxygen demand associated with the accumulated CBOD deposit for a given reach can be estimated by:

$$D = K^1 L_d \tag{16}$$

where

D is the oxygen demand, in pounds per day.

Benthic-oxygen demand, B, is customarily reported in grams per square meter per day. Pounds per day can be converted to grams per square meter per day by the following equation:

$$B = 454\ D/A \tag{17}$$

where

B is the benthic-oxygen demand, in grams per square meter per day,

and

A the area over which D is exerted, in square meters.

Streeter (1935) indicated that the reaction rate coefficient for benthic-oxygen demand, K^1, is approximately equivalent to the deoxygenation rate for CBOD, K_d, for fresh benthic deposits (for example, those created by untreated wastewater). For aged deposits the reaction rate coefficient may be somewhat less. Velz (1970, p. 167) reported that the range of values of K^1 (converted to base e) is 0.02 to 0.12 day^{-1} at 25°C. Values of benthic-oxygen demand calculated by Phelps method (1944, p. 125) are presented in table 9.

A drawback of Phelps' method is that it only estimates oxygen demand resulting from microbial activity. Other processes by which benthic deposits remove oxygen from the overlying water are (1) oxygen consumption by the soluble end products of aerobic benthic decomposition (Fair and others, 1941), (2) oxygen consumption by gases that are produced during anaerobic decomposition (Baity, 1938), and (3) CBOD added to the water column by bottom scour (Streeter, 1935; Edwards and Rolley, 1965). Bottom scour can suspend organic sediment, which, after being redeposited downstream, can again exert an oxygen demand downstream.

Several investigators have indicated that type of biological population influences the nature and magnitude of benthic-oxygen demand. Edwards and Rolley (1965), working with river muds, found that 40 percent of the total oxygen demand of the muds can be attributed to macroinvertebrates, primarily tubificids. McDonnell and Hall (1969) working with benthal deposits from an eutrophic stream and obtaining results similar to those of Edwards and Rolley, attributed more than 50 percent of the total oxygen demand to invertebrate respiration.

Further complicating estimation of benthic-oxygen demand is the dependence of invertebrate respiration on the stream-oxygen concentration (Knowles, Edwards and Briggs, 1962; Edwards and Rolley, 1965; Hargrave, 1969; McDonnell and Hall, 1969). Microbial respiration does not seem to be inhibited if the dissolved-oxygen concentration of the overlying water is more than 1.5 mg/L (Baity, 1938; Hanes and Irvine, 1968).

Estimates of benthic-oxygen uptake of various sediment materials are given in table 10. The wide variation in these values represents the influence of differing environmental conditions such as the nature and magnitude of invertebrate populations, dissolved-oxygen concentration, and sediment depths. In a study of very clean to severely degraded northeastern Illinois streams, Butts and Evans (1978) found a benthic-oxygen demand greater than or within the range from 0.7 to 1.5 $(g/m^2)/d$ at 20°C in 95 percent of the sites studied. Thus, the benthic-oxygen demand in most streams is probably within the preceding range. Values of benthic-oxygen demand used in the model calibrations ranged from 0.2 to 10 $(g/m^2)/d$ at 20°C, well within the ranges reported in table 10.

Benthic-oxygen demand was adjusted for temperature by the following equation:

$$B_T = B_{20°C}(1.065^{T-20°C}) \quad (18)$$

where

B is the benthic-oxygen demand, in grams per square meter per day,

1.065 a constant, applicable over a typical field-temperature range,

and

T the temperature, in degrees Celsius.

Table 9.--Estimated values of benthic-oxygen demand for model reaches on the Little and Wabash Rivers, Huntington County, Ind.

Reach	Benthic-oxygen demand [$(g/m^2)/d$ at 20°C] August 5, 1977	Benthic-oxygen demand [$(g/m^2)/d$ at 20°C] September 29, 1978
1	0	0
2	10	10
3	5.5	3.5
4	5.5	3.5
5	1.4	1
6	1.4	1
7	1.4	1
8	.2	1
9	.2	1

Table 10.--Values of benthic-oxygen uptake by various sediments

Sediment type	Range of benthic oxygen uptake [$(g/m^2)/d$ at 20°C]	References
Sewage sludge	0.5-4.6	Baity (1938).
Paperwaste sludge	.2-3.3	Hanes and Irvine (1968).
River mud	1-24	Edwards and Rolley (1965).
Stream mud	1.4-6.1	McDonnell and Hall (1969).
Lake mud	.3-1.1	Brewer and others (1977).
Sand	.2-1.0	Thomann (1972, p. 104).

Reaeration

Reaeration is generally the most important single parameter used in describing a stream's ability to assimilate biodegradable material. Many of the common empirical or semi-empirical equations used to predict reaeration [for example, those developed by O'Connor and Dobbins (1958); Owens and others (1964); Thackston and Krenkel (1969); and Bennett and Rathbun (1972)] assume that gaseous exchange varies directly with stream velocity and inversely with stream depth and that reaeration increases with decreasing flow. However, channel morphology should also be considered in the determination of reaeration. Langbein and Durum (1967) indicated that gaseous exchange rates increase with decreasing flow in riffles but decrease in pools. The low slopes of most Indiana streams usually cause the pooled condition to predominate. In addition, several investigators using the radioactive-tracer technique for direct determination of reaeration found a strong correlation between reaeration and channel slope (Tsivoglou and Neal, 1976; Foree, 1976). Foree (1976), in his study on small streams in Kentucky, also reported a general tendency for reaeration to decrease with decreasing specific discharge (discharge per unit drainage area).

The equation used initially to predict reaeration in this study is the energy-dissipation model developed by Tsivoglou and Neal (1976).

$$K_a = 16.36(0.110SV) \quad \text{when } 1 \le Q \le 10 \text{ ft}^3/\text{s} \tag{19}$$

$$K_a = 16.36(0.054SV) \quad \text{when } 25 \le Q \le 3{,}000 \text{ ft}^3/\text{s} \tag{20}$$

where

K_a is the base-e reaeration rate, at 20°C, in day^{-1},

0.110 and 0.054 the gaseous-escape coefficients for equations 19 and 20, respectively, in feet^{-1}, S the stream channel slope, in feet per mile,

V the stream velocity, in feet per second,

and

Q the stream discharge, in cubic feet per second.

Grant (1976), using the radioactive-tracer technique for small streams in Wisconsin (less than 37 ft^3/s discharge), determined the escape coefficient to be 0.81/ft $\pm$ 0.014/ft at 20°C. These values approximate the escape coefficient determined by Tsivoglou and Neal (1976) for small streams.

Wilson and Macleod (1974) concluded from a statistical evaluation of predictive reaeration equations that equations containing slope give better results than equations based on velocity and depth. In a similar study, Rathbun (1977) found the energy-dissipation model of Tsivoglou and Neal (1976) to be the best overall equation when compared to observed data obtained by the tracer method for direct determination of reaeration.

Initial attempts at model calibration with the energy-dissipation model of Tsivoglou and Neal (1976) yielded unsatisfactory results. Reaeration rates underestimated stream dissolved oxygen. A possible explanation for this may be that the value of the escape coefficient is too low. Tsivoglou and Neal (1976) reported that the escape coefficient varies from stream to stream. An attempt at direct measurement of stream reaeration in the Wabash River by ISBH by the modified-tracer technique was unsuccessful. Additional measurements of stream reaeration for defining the escape coefficient would prove useful in the prediction of reaeration in the Little and Wabash Rivers.

The following reaeration equations were also tested:

Bennett and Rathbun (1972)

$$K_a = 20.17\ V^{0.607}/H^{1.689} \qquad (21)$$

Cadwallader and McDonnell (1969)

$$K_a = 336(VS/5280)^{0.5}/H \qquad (22)$$

Churchill and others (1962)

$$K_a = 11.6\ V/H^{1.67} \qquad (23)$$

O'Connor and Dobbins (1958)

$$K_a = 12.9\ V^{0.5}/H^{1.5} \qquad (24)$$

Owens and others (1964)

$$K_a = 21.6\ V^{0.67}/H^{1.85} \tag{25}$$

where

H is the average stream depth, in feet,

and

K_a, S, and V are as previously defined.

The preceding equations are based on either theory or estimates of reaeration by the oxygen balance or disturbed equalibrium methods. Several investigators (Bennett and Rathbun, 1972; Lau, 1972; and Tsivoglou and Neal, 1976) suggested that uncertainties in the estimation of factors affecting the dissolved-oxygen balance limit the accuracy of reaeration estimates and, therefore, predictive reaeration equations determined in this manner.

Reaeration coefficients were adjusted for temperature by the following equation:

$$(K)_T \text{ reaeration} = (K)_{20°C}(1.021^{T-20°C}), \tag{26}$$

where

(K) is the base-e reaction constant, in day^{-1},

1.021 a constant, applicable over a typical field-temperature range,

and

T the temperature, in degrees Celsius.

The normalized-mean errors for dissolved-oxygen concentration determined by reaeration equations 20-25 are presented in table 11. (See Results and Discussion.) On the basis of the normalized-mean error and August 1977 data, the equation developed by Cadwallader and McDonnell (1969) gives the best fit for the model. For the September 1978 data, the energy-dissipation equation of Tsivoglou and Neal (1976) gives the best fit for the model.

The largest source of error for all but the equation of Tsivoglou and Neal (1976) is for model reaches in the Little River. The normalized-mean errors calculated for observed points in the Little River are presented in table 12. None of the equations (20-25) adequately describes reaeration in the Little River for the September 29, 1978, model. Several factors may account for this inadequacy, although the actual reason is not known. First, insufficient data were collected to estimate the effect of net photosynthetic-oxygen production for the Little River. O'Connell and Thomas (1965) indicated that, if benthic algae are the predominant algal population in a stream, algal respiration will normally exceed oxygen production. Odum (1956) suggested that the stream respiratory metabolism far exceeds production in stream reaches downstream from a wastewater outfall. This possibility was tested by using equations 20-25 to determine the rates of respira-

Table 11.--Normalized-mean errors for calculated dissolved-oxygen concentrations for the Little and Wabash Rivers, Huntington County, Ind.

Equation	Reference	Normalized-mean error (percent)	
		August 5, 1977	September 29, 1978
20	Tsivoglou and Neal (1976)	-19.7	26.4
21	Bennett and Rathbun (1972)	9.7	63.0
22	Cadwallader and McDonnell (1969)	.9	52.7
23	Churchill and others (1962)	-1.9	49.8
24	O'Connor and Dobbins (1958)	4.8	57.5
25	Owens and others (1964)	10.2	63.9

Table 12.--Normalized-mean errors for calculated dissolved-oxygen concentrations for the Little River, Huntington County, Ind.

Equation	Reference	Normalized-mean error (percent)	
		August 5, 1977	September 29, 1978
20	Tsivoglou and Neal (1976)	0.6	65.5
21	Bennett and Rathbun (1972)	9.0	264.1
22	Cadwallader and McDonnell (1969)	5.4	99.0
23	Churchill and others (1962)	4.7	94.6
24	O'Connor and Dobbins (1958)	6.7	104.1
25	Owens and others (1964)	9.4	206.8

tion necessary to account for the observed dissolved-oxygen deficit. Net photosynthetic-oxygen-production values determined in this manner ranged from -26 $(g/m^2)/d$ for the energy-dissipation equation of Tsivoglou and Neal (1976) to -55 $(g/m^2)/d$ for the equation developed by Owens and others (1964). These values are greater than those normally found. However, Odum (1956), using the data of Butcher and others (1930) and Denham (1938), estimated net community respiration rates of 52 $(g/m^2)/d$ and 29 $(g/m^2)/d$ for two polluted rivers.

Another possible explanation for the discrepancy between the observed and calculated dissolved-oxygen concentrations is the inability of any of the equations to predict reaeration adequately in the Little River, probably because of theoretical shortcomings. This possibility was examined by determining the reaeration rate in the Little River by an oxygen-balance method. In this method, reaeration rates are adjusted in the model until the observed and calculated dissolved-oxygen concentrations are equal. This method indicates that under the existing stream conditions, assuming all model coefficients other than reaeration are accurately estimated, reaeration rates of less than 0.01 day^{-1} would still not account for the observed dissolved-oxygen deficit. Reaeration rates of this magnitude are much lower than would be expected for a surface-active river such as the Little River.

Finally, the stream dissolved-oxygen concentration may be severely affected by an oxygen-demanding waste from an unknown source between station 004 (RM 2.29) and the Huntington wastewater-treatment facility. This waste would result in a lower upstream dissolved-oxygen concentration at the mixing zone of wastewater-treatment facility effluent than the model predicted for known inputs.

On the basis of overall performance and the anomaly in dissolved-oxygen concentration downstream from the Huntington wastewater-treatment facility, and because Rathbun (1977) found the equation of Churchill and others (1962) to be the second best composite equation when tested against directly determined reaeration measurements, the equation of Churchill and others (1962) was chosen for use in this study.

Photosynthesis

Mean-daily, net-photosynthetic, dissolved-oxygen production was evaluated by the two-station (upstream-downstream) method of Odum (1956) as modified by Stephens and Jennings (1976). The two-station method generates a relationship describing the rate of change of dissolved oxygen between two stations at discrete sampling intervals. The model assumes that oxygen production may occur only during daylight hours and that any change in dissolved oxygen during this period, after corrections have been made for diffusion, is due to photosynthetic oxygen production. Any change in dissolved-oxygen concentration during hours of darkness, after corrections for diffusion have been made, is attributed to respiration.

Diel dissolved-oxygen fluctuations ranged from 1.5 to 7.5 and 1.4 to 9.6 mg/L for the August 1977 and September 1978 surveys, respectively. Calculated mean-daily, net-photosynthetic, dissolved-oxygen-production values for reaches on the Wabash River are presented in table 13. These values are within the range of values reported by several investigators (Odum, 1956 and Hoskin, 1959). All net-photosynthetic, dissolved-oxygen-production values calculated for the Wabash River are less than zero, and, therefore, deplete the stream dissolved oxygen. Insufficient data were collected for estimating net-photosynthetic, dissolved-oxygen production for reaches on the Little River.

Table 13.--Mean-daily, net-photosynthetic, dissolved-oxygen production for model reaches on the Wabash River, Huntington County, Ind.

Model reach	Mean daily, net-photosynthetic, dissolved-oxygen production [(g/m^2/d)]	
	August 5, 1977	September 29, 1978
6	-0.7	-1.2
7	- .6	- .7
8	- .6	- .7
9	- .3	- .3

Model-Calibration Results

The calculated and observed CBOD, NBOD, dissolved oxygen, and calculated flows are presented in figures 8-15. Several observations of these results follow.

The normalized-mean error was used as the criterion for comparison. The normalized-mean error is defined by Wilson and Macleod (1974) as:

$$\text{Normalized-mean error} = \left[\sum_{i=1}^{N} \frac{(A_{calc} - A_{obs})}{A_{obs} N} \right] 100 \text{ percent} \qquad (27)$$

where

A_{calc} and A_{obs} are the calculated and observed values, respectively, and

N the number of observations.

The normalized-mean error for CBOD was 1.1 percent for the August 5, 1977, model and 1.8 percent for the September 29, 1978, model. The corresponding normalized-mean errors for NBOD were 24 percent for the August 5, 1977, model and 2.8 percent for the September 29, 1978, model. The large normalized-mean error for NBOD for the August 5, 1977, model results principally from the large discrepancy between the observed and calculated values at RM 0.80 (station 002). This discrepancy can probably be attributed to the uncertainty about the effect of the sewer overflow at this site. The normalized-mean error for dissolved oxygen was 1.9 percent for the August 5, 1977, model and 49.8 percent for the September 29, 1978, model.

MODEL VERIFICATION

The purpose of the model-verification process is to measure the degree of validity of the model by comparing the difference between simulated results and prototype-measured values for several different time periods with an established set of criteria. If the difference between simulated results and prototype-measured values is within an acceptable range, then the model can be considered to be verified.

The range of conditions for which a model is applicable depends on the design constraints of the mathematical formulation, the range of data available for its calibration, the stability of the parametric coefficients, and the degree of accuracy required. If a model is based on well-defined, fundamental processes or concepts that are valid throughout a wide range of conditions, then one can safely assume that the model may be used throughout this range of conditions.

The Wabash River model is to be used for determining alternatives for future waste loadings that will be compatible with Indiana water-quality standards defined for summer and winter low flows. However, time and budgetary constraints for the study prohibited adequate data collection for model calibration and verification.

For the Wabash River study, two sets of water-quality data were collected. Stream discharge during both surveys was not significantly greater than the $Q_{7,10}$ flow. Therefore, extrapolating the model to critical low-flow conditions from flow conditions during the two water-quality surveys should not represent a large source of error.

The same values of the coefficients K_r, K_d, and K_n used in the model for the two sets of water-quality data adequately predicted stream conditions during both model simulations. (See section "Model Calibration Results.") This conclusion, however, should be considered with the discrepancy noted between the observed and calculated dissolved-oxygen concentrations in the Little River and the possible explanations for it.

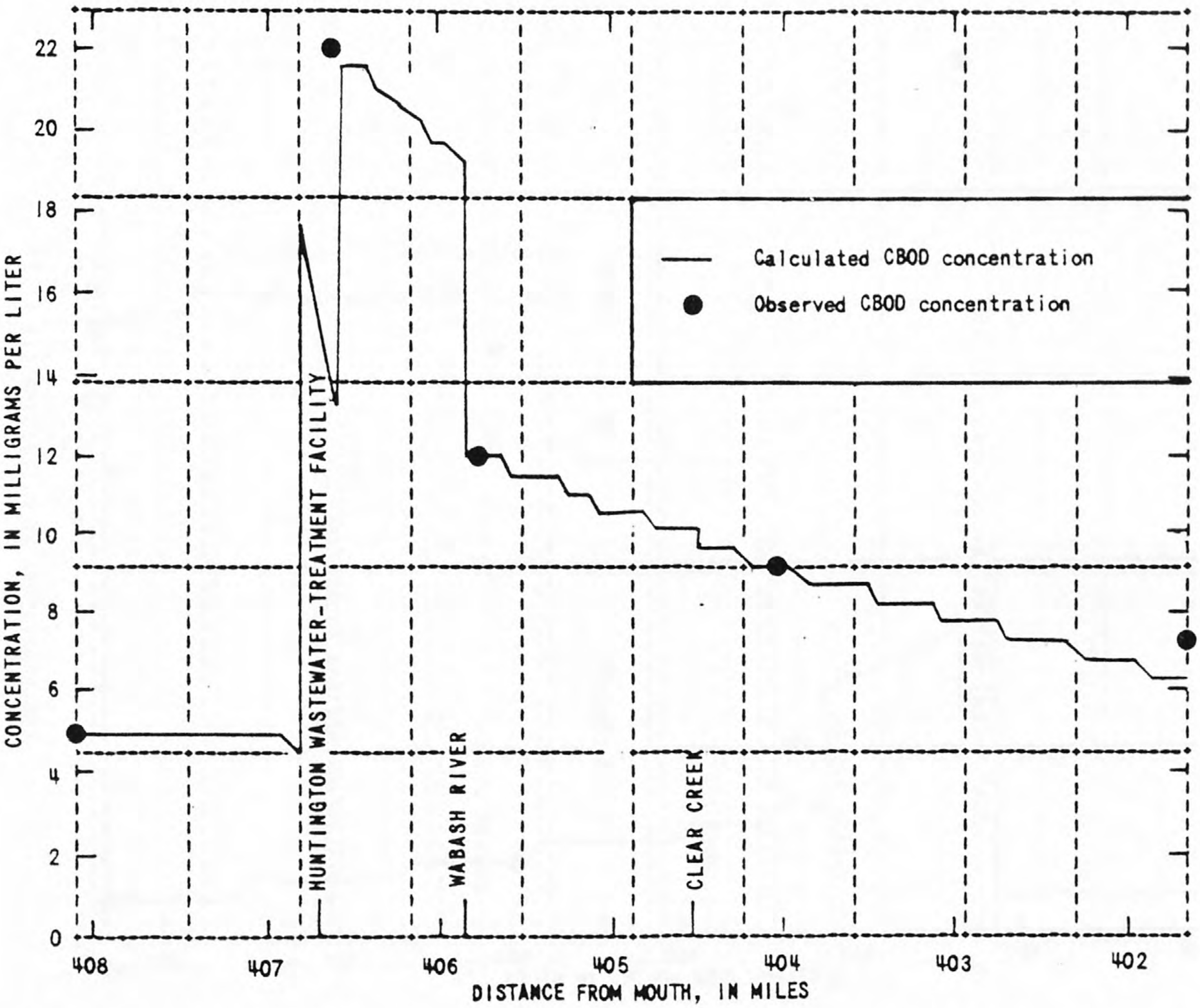

Figure 8.-- Calculated and observed carbonaceous biochemical-oxygen-demand concentrations in the Little and Wabash Rivers, August 5, 1977.

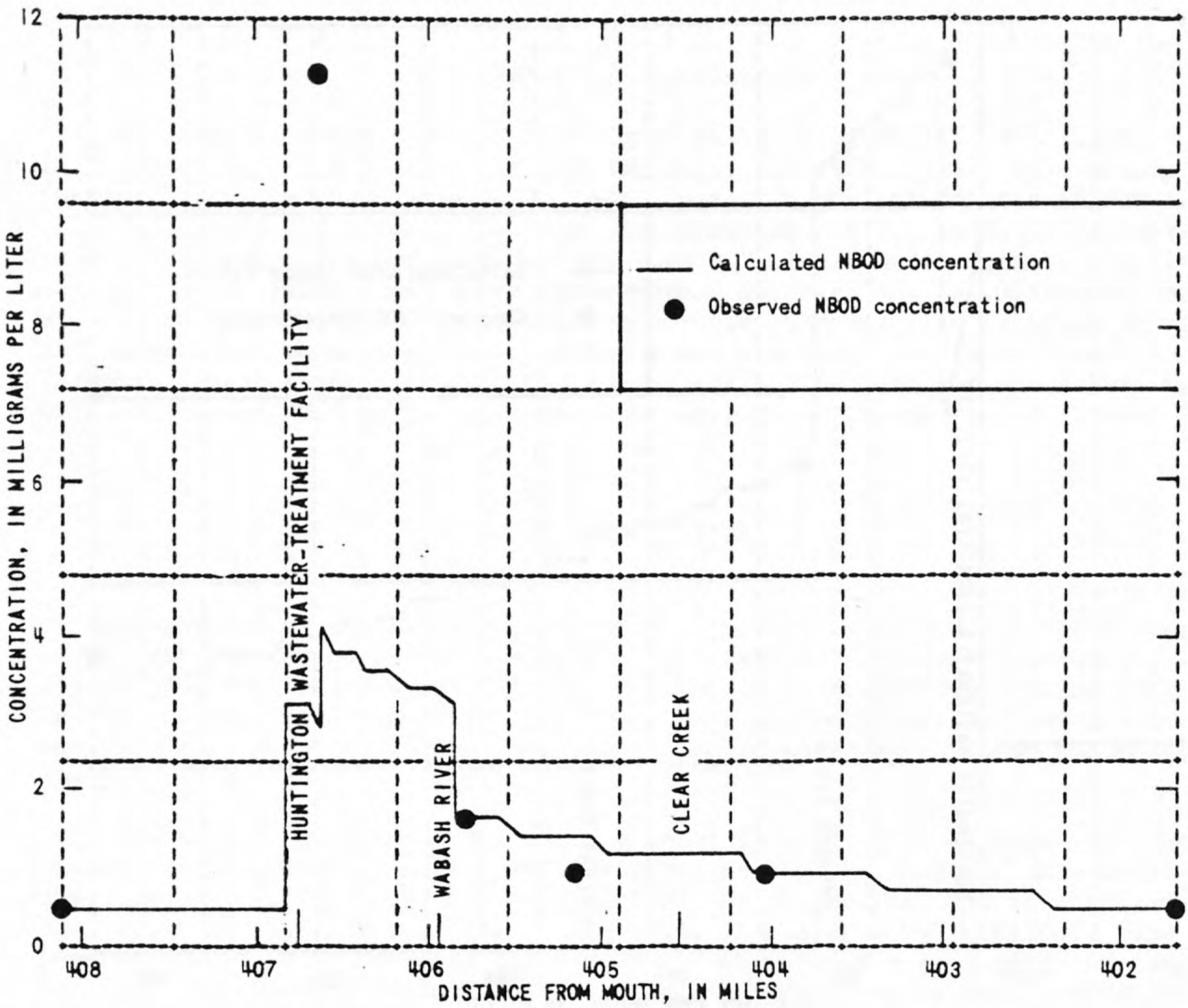

Figure 9.-- Calculated and observed nitrogenous biochemical-oxygen-demand concentrations in the Little and Wabash Rivers, August 5, 1977.

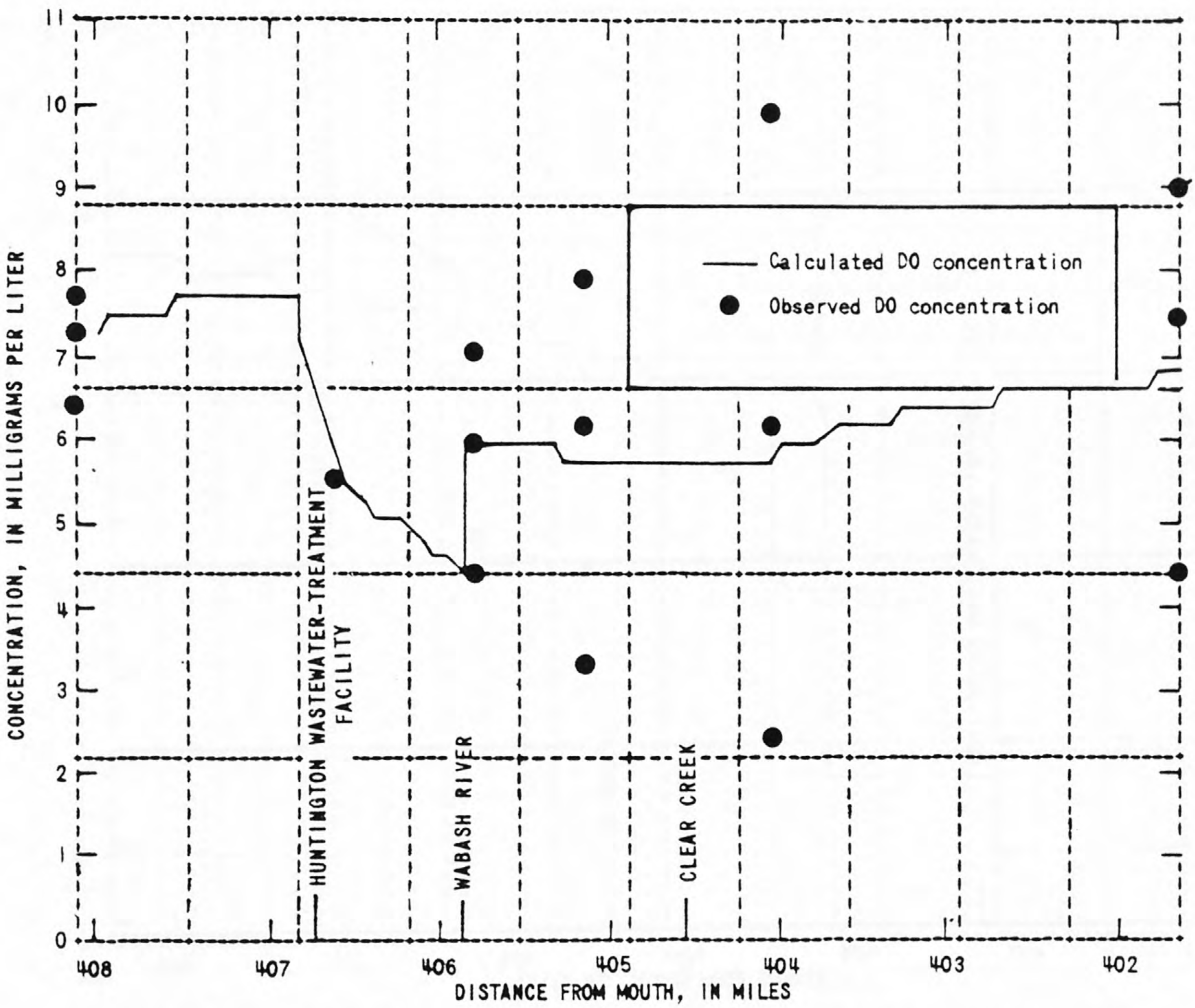

Figure 10.-- Calculated and observed dissolved-oxygen concentrations in the Little and Wabash Rivers, August 5, 1977.

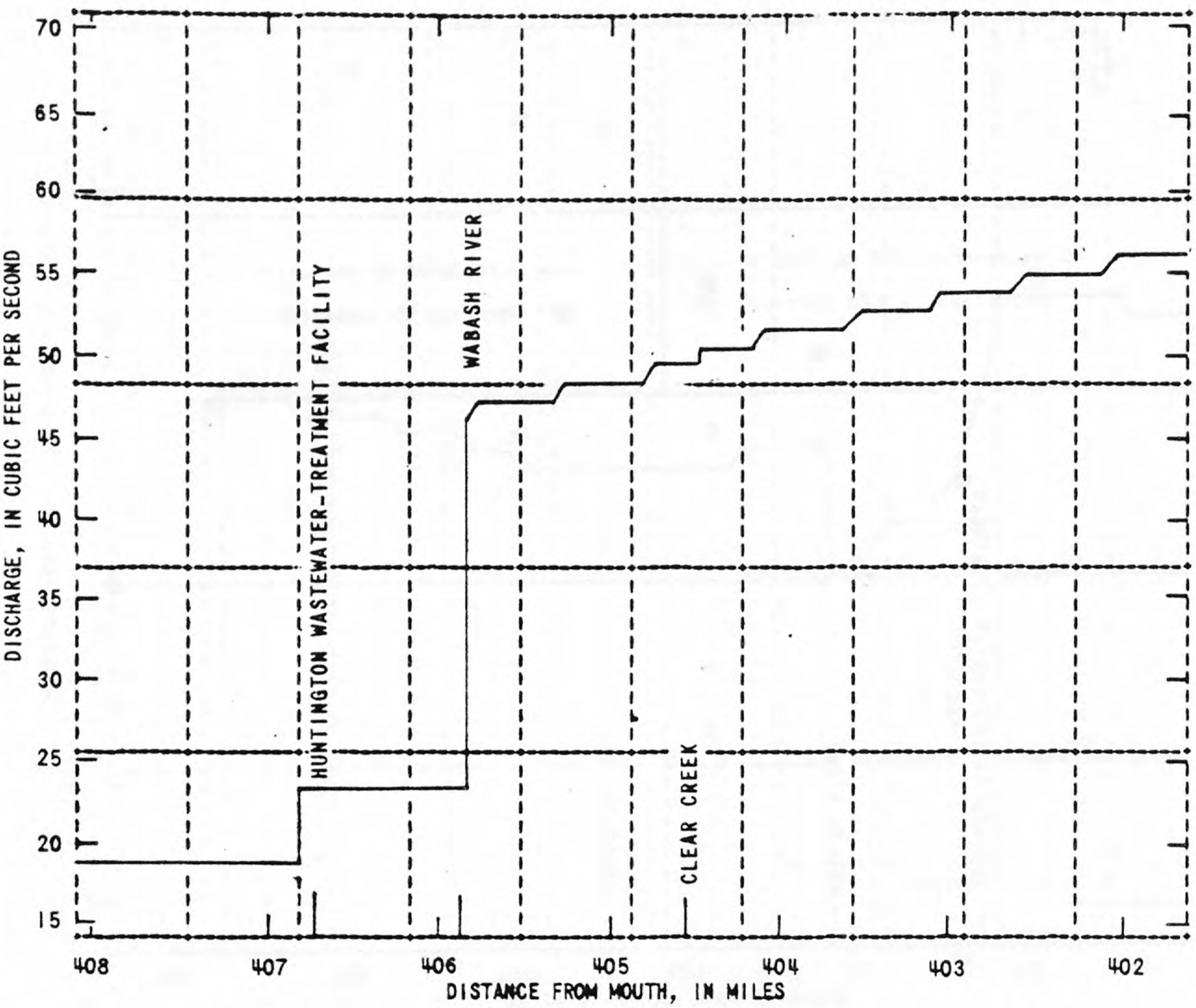

Figure 11.-- Discharge in the Little and Wabash Rivers, August 5, 1977.

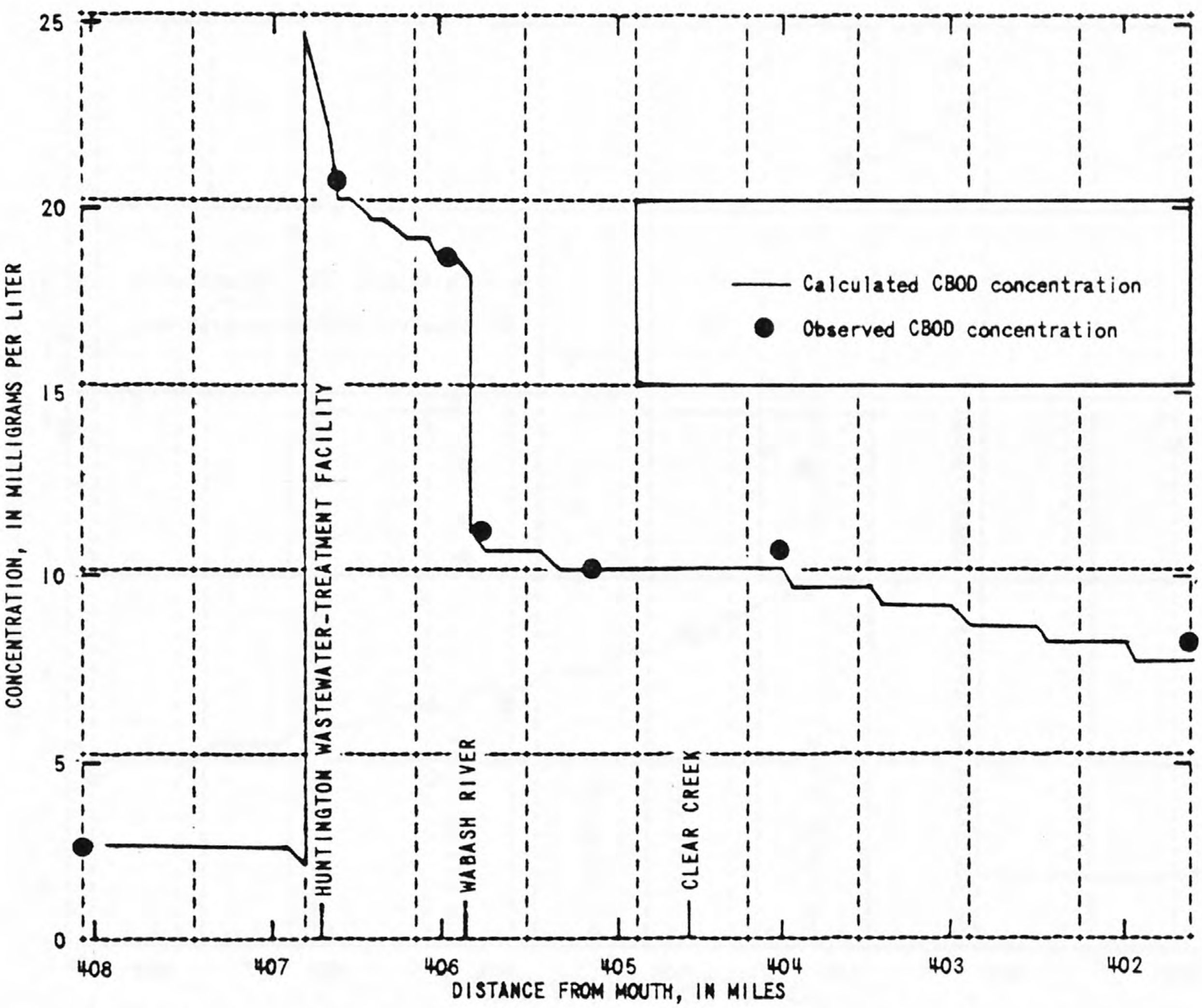

Figure 12.-- Calculated and observed carbonaceous biochemical-oxygen-demand concentrations in the Little and Wabash Rivers, September 29, 1978.

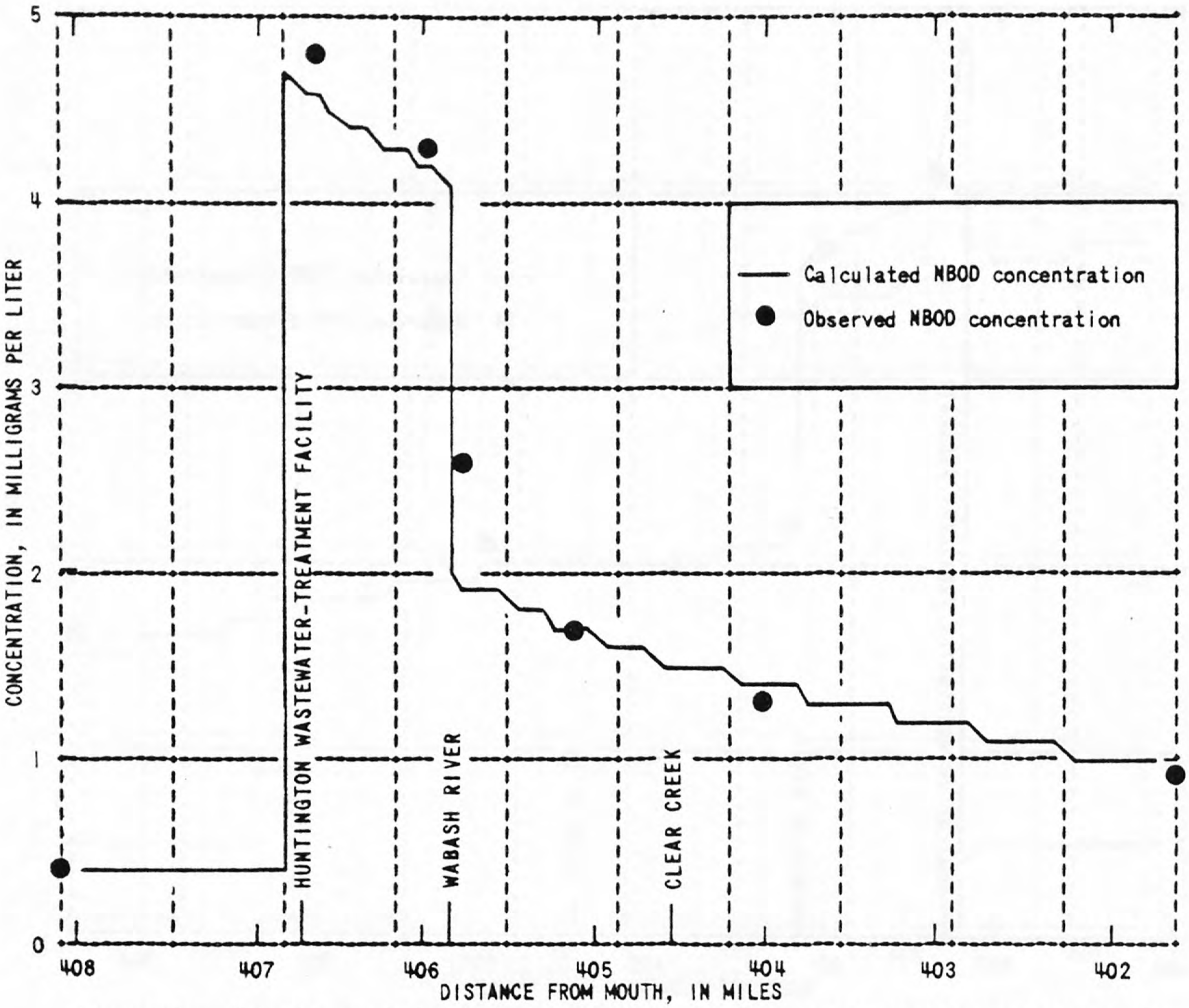

Figure 13.-- Calculated and observed nitrogenous biochemical-oxygen-demand concentrations in the Little and Wabash Rivers, September 29, 1978.

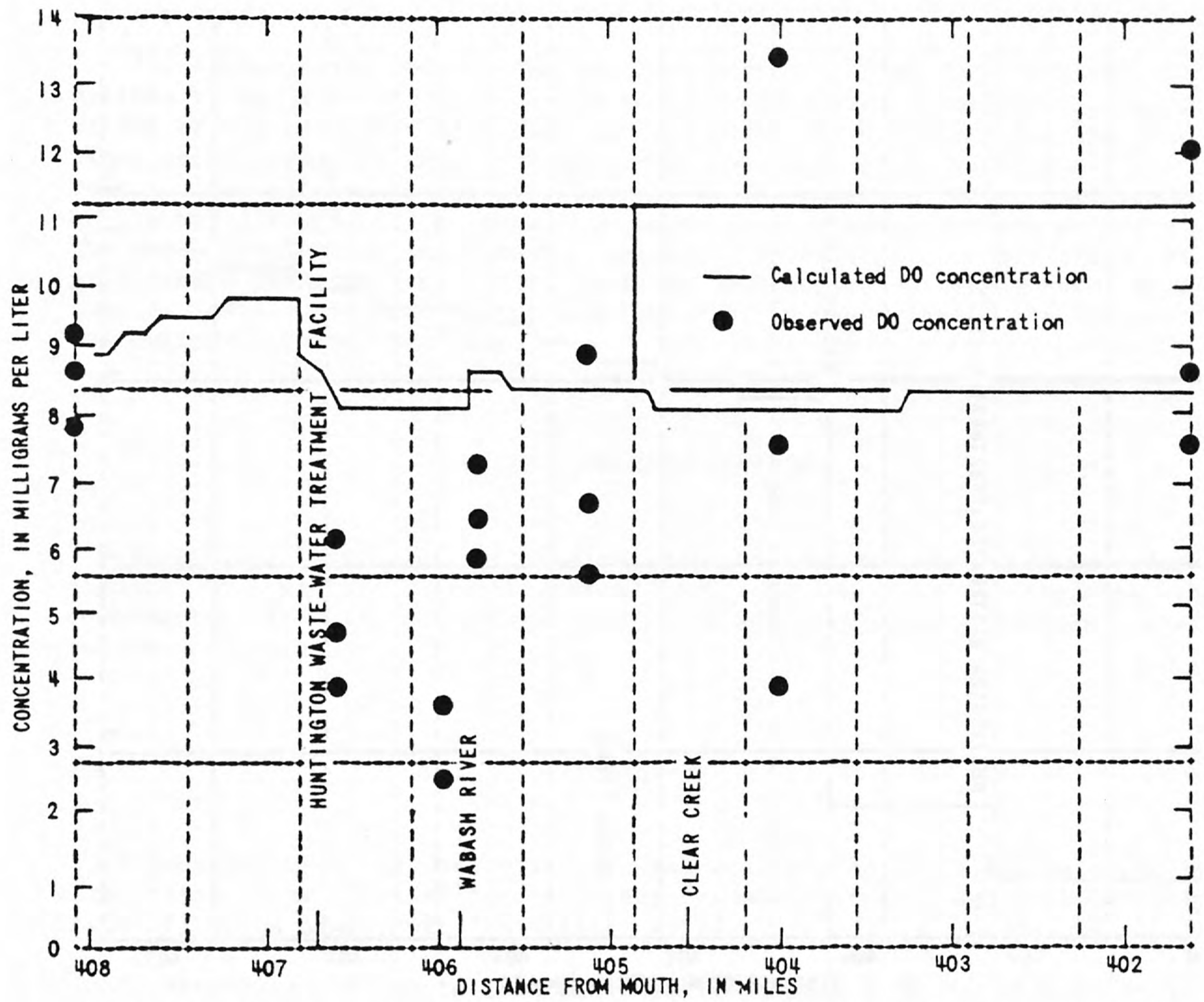

Figure 14.-- Calculated and observed dissolved-oxygen concentrations in the Little and Wabash Rivers, September 29, 1978.

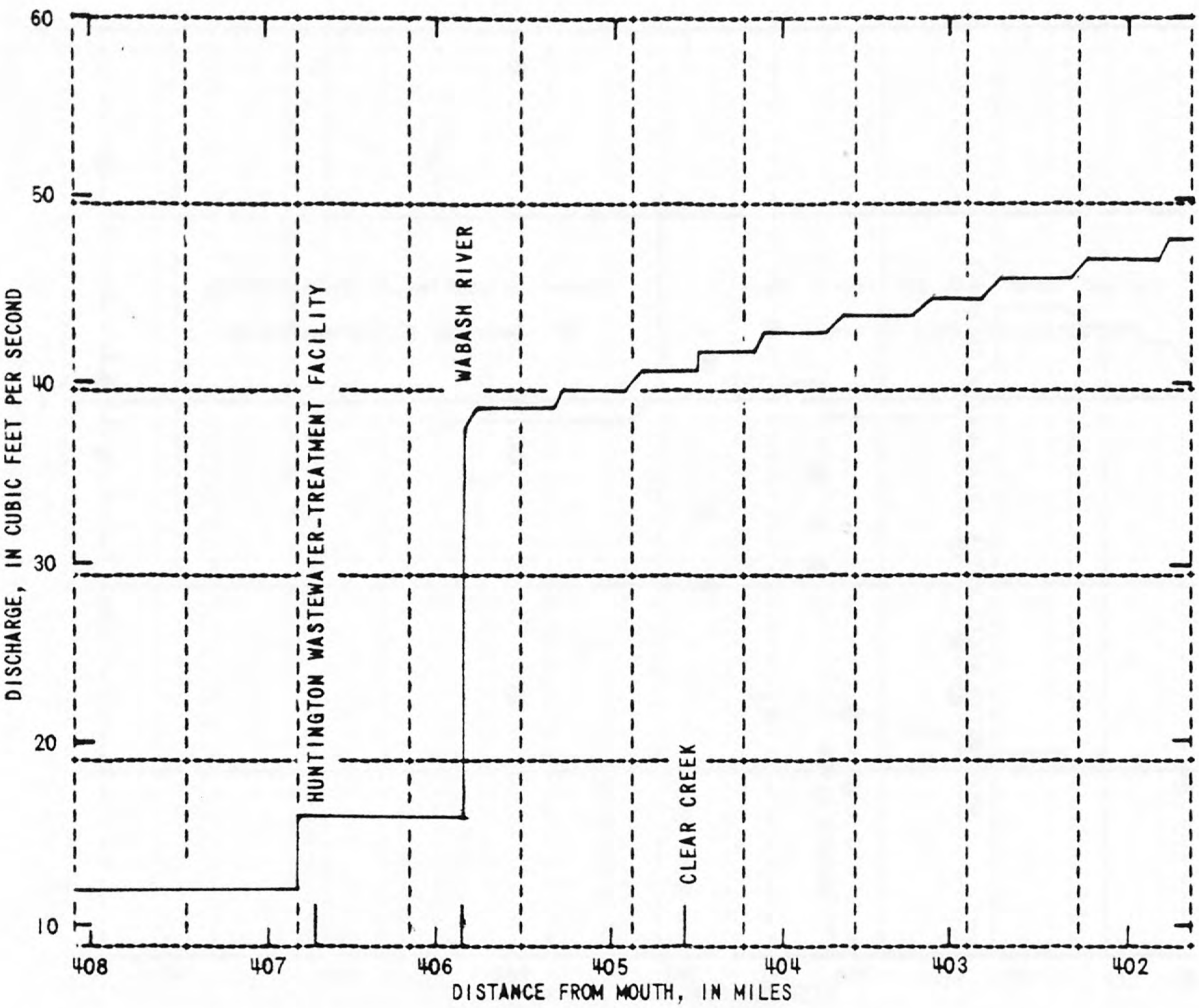

Figure 15.-- Discharge in the Little and Wabash Rivers, September 29, 1978.

The Wabash River model cannot be considered verified for the stream conditions to be simulated. Both the type of wastewater treatment and the location of the effluent discharge on the river are different for the conditions to be simulated than for the conditions during the water-quality surveys. (See section "Basin Description.") The effect of these differences on the model's predictive capability is unknown. Therefore, the accuracy of the model predictions is unknown. However, the model is still useful as a preliminary planning tool until it can be updated and revised with water-quality data collected after completion of improvements to the Huntington wastewater-treatment facility.

WASTE-LOAD ASSIMILATION

Waste-load assimilation studies were made for both the summer (June-August) and winter (November-March) critical low flows to determine the combination of waste loadings that would meet the current Indiana water-quality standards.

Procedures

Procedures used in establishing the waste-load assimilation capacity were taken from "Guidelines for Waste-Load Allocation and Maximum Daily Loads" (Indiana State Board of Health, 1977).

To determine the waste-load assimilative capacity of the stream, as defined by water-quality standards, the authors initially used maximum daily loadings (twice the monthly average) for the Huntington wastewater-treatment facility.

The concentrations of ammonia nitrogen in the wastewater effluent during the two water-quality surveys ranged from 3.9 to 4.4 mg/L. These low concentrations may have been due to nitrification in the trickling filter, owing to an attached population of nitrifiers, or dilution by ground water. However, an activated-sludge process facility is being constructed to replace the trickling filter. Nitrification in the activated-sludge process is usually insignificant because the detention times required for nitrifiers to proliferate are too short. Jenkins and Garrison (1968) found that, for a domestic wastewater treated by the activated-sludge process at a temperature of 21° to 22°C, a mean-cell residence time of at least 10 days is needed to ensure nitrification. According to Metcalf and Eddy (1972, p. 231) the ammonia-nitrogen concentration of untreated domestic sewage ranges from 12 to 50 mg/L. Consequently, the concentration of ammonia nitrogen in effluent may be significantly higher than that observed during the two water-quality surveys. Therefore, the effluent ammonia-nitrogen concentration for the wastewater-treatment facility was assumed to be 15 mg/L or 65 mg/L NBOD (Aolad Hossain, Indiana State Board of Health, oral commun., 1978).

The observed dissolved-oxygen concentration for the Huntington wastewater-treatment facility averaged 59 percent of saturation during the two water-quality surveys. Although the future dissolved-oxygen concentration of the effluent of the activated-sludge process facility under construction is not known, effluent dissolved-oxygen concentrations in this type of facility are typically low. If the dissolved-oxygen concentrations of the wastewater effluents had been assumed to be as low as 59 percent of saturation for the waste-assimilation study, the amount of wastes that could have been discharged without violating the stream water-quality standards would have been severely limited. The aeration of wastewater effluents before discharge is relatively inexpensive compared to additional waste treatment. Therefore, the dissolved-oxygen concentration of the wastewater was assumed to be 80 percent of saturation (Aolad Hossain, Indiana State Board of Health, 1978).

Where the model predicted that the current water-quality standards would be exceeded, the CBOD and NBOD loads for the wastewater-treatment facility were reduced until the appropriate standards were met. The determination of alternative CBOD and NBOD loads that would just meet the Indiana water-quality standards defined the assimilative capacity of the stream.

For the Little River, Rohne (1972, p. 62) lists the summer $Q_{7,10}$ as 6.1 ft^3/s and the annual $Q_{7,10}$ as 3.3 ft^3/s. Flow in the Wabash River is regulated by the Huntington Reservoir operated by the U.S. Army Corps of Engineers. The minimum flow maintained in the Wabash River below the dam, 20 ft^3/s, was used for the summer and annual $Q_{7,10}$ flows.

Temperatures of 24° and 25°C were used in the summer $Q_{7,10}$ model for the Little and Wabash Rivers, respectively. These temperatures were based on the mean daily water temperatures that were exceeded 10-20 percent of the time for the months June through August. Temperatures of 7.8° and 8.7°C were used in the winter $Q_{7,10}$ model for the Little and Wabash Rivers, respectively. These temperatures were the average of the daily mean water temperatures for November, the month with the lowest flow (Horner, 1976, p. 70-74), required by ISBH procedures (Indiana State Board of Health, 1977).

Temperatures for the Little and Wabash Rivers were estimated by the following equation:

$$T = M + A\ [\sin\ (0.172\underline{d} + C]\quad \text{(Shampine, 1977, p. 13)} \tag{28}$$

where

T is the temperature at a given site on a specific day, in degrees Celsius,

M the mean annual stream temperature, in degrees Celsius,

A the stream temperature amplitude, in degrees Celsius,

$\underline{d}$ the Julian date,

and

C the angle-phase coefficient, in radians.

Values of M, A, and C for the Little River at Huntington are 12.15°C, 12.01°C, and 4.30 radians, respectively (Shampine, 1977, p. 44). Values of M, A, and C for the Wabash River at Huntington are 13.12°C, 12.26°C, and 4.30 radians, respectively (Shampine, 1977, p. 49).

Concentrations of CBOD, NBOD, and dissolved oxygen for the Wabash River upstream from its confluence with the Little River were the average values of data collected by the ISBH for the years 1971-75. Data used to determine the average concentrations of chemical constituents were limited to those collected at flows five times the $Q_{7,10}$ or less. Average concentrations for the Wabash River upstream from its confluence with the Little River for the summer $Q_{7,10}$ were based on data collected between April and October. Average concentrations for the winter $Q_{7,10}$ were based on data collected between October and April. Concentrations of CBOD, NBOD, and dissolved oxygen upstream from the wastewater-treatment facility on the Little River were assumed to be the average of concentrations determined during the two water-quality surveys.

Reaction-rate coefficients for CBOD and NBOD used in the waste-load assimilation study were the same as those used in the model calibration. Reaeration rates were computed by the equation developed by Churchill and others (1962) as discussed in the section "Reaeration."

Algal effects were not included in the waste-load assimilation study. These effects under various environmental conditions cannot be accurately predicted.

Benthic-oxygen demand was computed as discussed in the section "Benthic-Oxygen Demand." The impact of the benthic-oxygen demand in the waste-load allocation was included in the model calibration because of the large difference between the stream decay rate for CBOD, K_r, and the stream deoxygenation rate for CBOD, $\tilde{K}_r$. If this factor had not been included, large amounts of CBOD would have been removed from the water column without exerting an oxygen demand.

All model coefficients were corrected for temperature as discussed in the appropriate sections of "Model-Parameter Estimation."

Results and Discussion

The model used in the waste-load assimilation study has not been verified. Therefore, interpretations of the results should be limited to qualitative uses only. A quantitative model would require calibration and verification with data collected after changes to effluent discharges have been made.

According to the Indiana State Board of Health (1977), part of the assimilative capacity of a stream should be reserved for future population growth and development. Two modeling approaches are suitable for this purpose. First, a percentage of the assimilative capacity of the stream can be left as a reserve. This capacity should be no greater than 30 percent of the assimilative capacity and, ideally, should be the capacity required to assimilate probable future growth for the planning period. The size of the reserve is dependent on the rate of growth and the length of time of the planning period. Second, the waste loads and flows for the existing dischargers (except industries) may be projected at 5-year intervals to the end of the planning period (year 2000) and used as input into the calibrated waste-load allocation model.

According to Vince Sommers (Indiana State Board of Health, written commun., 1979) projected waste flows are not available at 5-year intervals to the end of the planning period. Therefore, a modification of the second technique was used. The average flow of 2.7 Mgal/d for the present Huntington wastewater-treatment facility at RM 0.98, which discharges into the Little River, and three additional effluent flows (3.0, 4.0, and 5.0 Mgal/d) were used in the waste-load assimilation study. These flows should represent effluent flows after the completion of the wastewater-treatment facility currently under construction. The design flow of the new wastewater-treatment facility is 5.0 Mgal/d. Therefore, the three effluent flows should adequately define the effect of changes in effluent flows during the planning period. The goals were an average dissolved-oxygen concentration of 5.0 mg/L and maximum ammonia-nitrogen concentrations of 2.5 and 4.0 mg/L during summer and winter low flows, respectively. Waste loadings for the wastewater-treatment facility that will meet Indiana water-quality standards are presented in figure 16 and in tables 14 and 15.

The model indicates that nitrification is potentially the most significant factor affecting the stream dissolved-oxygen concentration during summer low flows. However, if the ammonia-nitrogen toxicity standard is not exceeded, nitrification should not become a limiting factor on the allowable carbonaceous and nitrogenous waste loads. The Indiana stream dissolved-oxygen standard should be met during the winter low flow if the current NPDES permit restrictions are not exceeded. The model predicts that the minimum dissolved-oxygen concentration of the $Q_{7,10}$ low flow will occur at approximately RM 403. Therefore, the impact of future growth will be least downstream from this point.

Future water-quality studies to verify the model after completion of improvements at the wastewater-treatment facility and clarify the effect of various factors on the dissolved-oxygen dynamics include: (1) Additional water-quality surveys at steady-state, low-flow conditions on the Wabash River that emphasize sampling all municipal and industrial outfalls and overflows immediately downstream from the Huntington wastewater-treatment facility outfall; (2) samples collected in equal-width increments across the stream transect and composited so that the volume from each increment is proportional to the discharge; (3) a study of the fate of nitrogen species and their effect on dissolved-oxygen concentration, where sufficient data are collected to enable use of a more sophisticated model for nitrification, such as a sequential reaction model; (A model of this type may better describe the process of nitrification in the Wabash River than the BOD equivalent model used in this study.) (4) enumeration of nitrifier populations of both the bottom materials and the water column downstream from the wastewater-treatment facility to determine if significant populations of nitrify-

Note: The information presented in this figure is not based on a verified model and should be used only for qualitative interpretation.

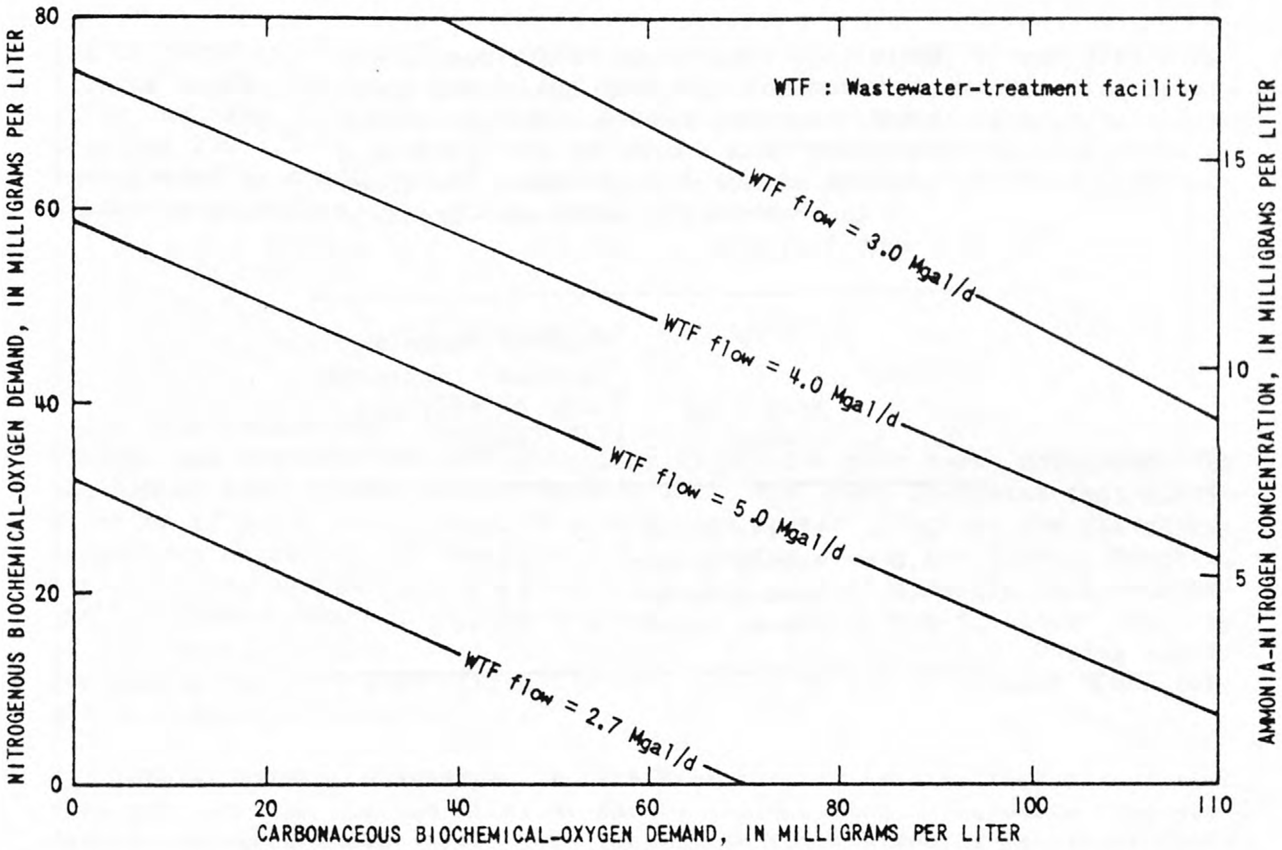

Figure 16.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Huntington wastewater-treatment facility that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows.

Table 14.--Total ammonia-nitrogen concentrations that will provide a maximum in-stream ammonia-nitrogen concentration of 2.5 mg/L in the Little and Wabash Rivers during summer low flows at the Huntington wastewater-treatment facility

Effluent flow (Mgal/d)	Receiving stream	Maximum ammonia-nitrogen concentration of effluent (mg/L)
2.7	Little River	6.0
3.0	Wabash River	15.7
4.0	Wabash River	12.3
5.0	Wabash River	10.4

Table 15.--Total ammonia-nitrogen concentrations that will provide a maximum in-stream ammonia-nitrogen concentration of 4.0 mg/L in the Little and Wabash Rivers during winter low flows at the Huntington wastewater-treatment facility

Effluent flow (Mgal/d)	Receiving stream	Maximum ammonia-nitrogen concentration of effluent (mg/L)
2.7	Little River	9.7
3.0	Wabash River	26.1
4.0	Wabash River	20.4
5.0	Wabash River	17.2

ing bacteria are available to exert an oxygen demand; (5) direct determination of benthic-oxygen demand; (6) additional direct determination of reaeration at the low-flow condition for adequately defining reaeration at the critical low flow; and (7) collection of additional hydrologic data during low-streamflow conditions for better definition of time of travel, stream-discharge relations, and stream width and depth.

SUMMARY AND CONCLUSIONS

A one-dimensional, steady-state, dissolved-oxygen model has been calibrated, and alternatives for waste-load allocation have been developed for the Wabash River in Huntington County, Ind. The model indicates that nitrification is potentially the most significant factor affecting the dissolved-oxygen concentration in the Wabash River during summer low flows. However, nitrification should not be a limiting factor on the allowable carbonaceous- and nitrogenous-waste loads for the Huntington wastewater-treatment facility if the ammonia-nitrogen toxicity standard is not exceeded. During winter low flows, the dissolved-oxygen standard should be met if present NPDES permit restrictions are met.

Natural stream discharge in the Wabash River is significantly greater than that in the Little River at the $Q_{7,10}$ condition. Therefore, the predicted allowable waste loads for the Huntington wastewater-treatment facility increase substantially after the effluent discharge is moved from the Little River to the Wabash River. The model predicts that the impact of the wastewater-treatment facility is greatest at RM 403. Therefore, the impact of future growth will be least downstream from this point.

This waste-load assimilation study is not based on a verified model. The changes in stream water quality predicted by the model represent only possible stream responses to different effluent conditions.

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