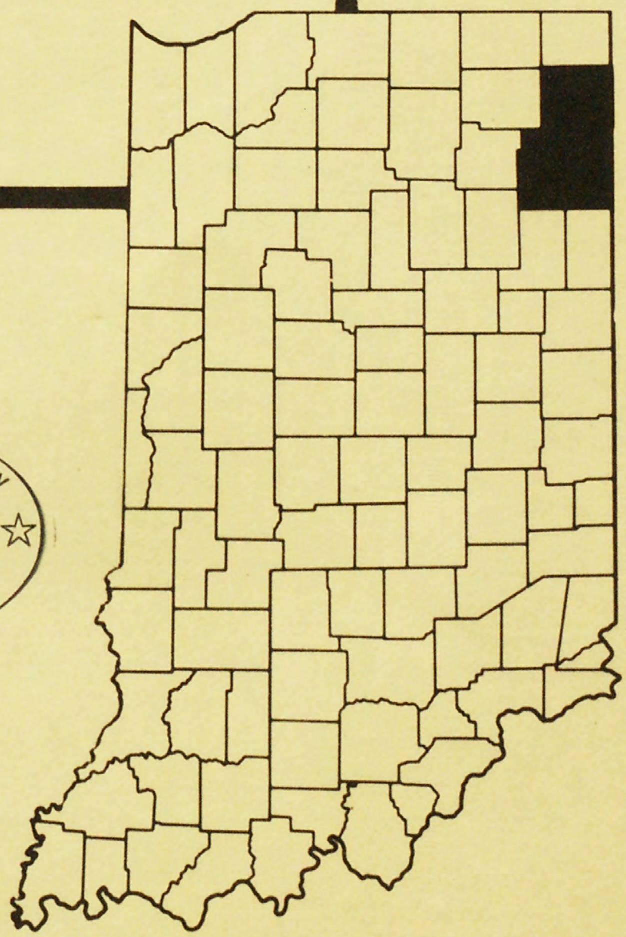
UNITED STATES
DEPARTMENT OF THE INTERIOR
GEOLOGICAL SURVEY

A ONE-DIMENSIONAL, STEADY-STATE DISSOLVED-OXYGEN MODEL AND WASTE-LOAD ASSIMILATION STUDY FOR CEDAR CREEK, DEKALB AND ALLEN COUNTIES, INDIANA



Open-File Report 79-1062

Prepared in cooperation with
Indiana State Board of Health

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By William G. Wilber, James G. Peters, Mark A. Ayers, and Charles G. Crawford

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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)
square foot (ft^2)	0.0929	square meter (m^2)
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 mile (mi^2)	2.590	square kilometer (km^2)
million gallons per day (Mgal/d)	3,785	cubic meters per day (m^3/d)

ABBREVIATIONS

Abbreviation	Description
BOD	Biochemical-oxygen demand.
CBOD	Carbonaceous biochemical-oxygen demand.
°C	Degree Celsius.
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.
h	Hour.
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.
$K_{n,zero}$	Zero-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.
sta. no.	Station number.
t	Traveltime down the stream.
Temp.	Temperature.
umho/cm	Micromho per centimeter.
USGS	U.S. Geological Survey.

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ABSTRACT

The Indiana State Board of Health is developing a State water-quality management plan that includes the establishing of limits for wastewater effluents discharged into Indiana streams. A digital model calibrated to conditions in Cedar Creek was used to develop 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. All point-source waste loads affecting Cedar Creek are in the four incorporated municipalities of Auburn, Garrett, Huntertown, and Waterloo, in a primarily agricultural area. Avilla, because of its distance from Cedar Creek, does not significantly affect the water quality of the modeled segment.

The model indicates that the dissolved-oxygen concentration of the Auburn wastewater effluent and nitrification are the most significant factors affecting the dissolved-oxygen concentration in Cedar Creek during summer low flows. The observed dissolved-oxygen concentration of the Auburn wastewater effluent was low, and averaged 30 percent of saturation. Whether the effluent is aerated before discharge will ultimately define the waste-load assimilative capacity of Cedar Creek. Projected nitrogenous biochemical-oxygen demand loads, from the Indiana State Board of Health, for the Auburn and Waterloo wastewater-treatment facilities will result in violations of the current instream dissolved-oxygen standard (5 milligrams per liter), even with an effluent dissolved-oxygen concentration of 80 percent saturation.

Natural streamflow for Cedar Creek upstream from the confluence of Willow and Little Cedar Creeks is small compared with the waste discharge, so benefits of dilution for Waterloo and Auburn are minimal. Stream reaeration capacity is not sufficient to maintain an average dissolved-oxygen concentration of at least 5 milligrams per liter, the State's water-quality standard for streams.

The model also indicates that, during winter low flows, ammonia toxicity, rather than dissolved oxygen, is the limiting water-quality criterion in the reach of Cedar Creek downstream from the wastewater-treatment facility at Auburn and the confluence of Garrett ditch. Ammonia-nitrogen concentrations predicted for 1978 through 2000 downstream from the Waterloo wastewater-treatment facility do not exceed Indiana water-quality standards for streams.

Calculations of the stream's assimilative capacity indicate that future waste discharge in the Cedar Creek basin will be limited to the reaches between the Auburn wastewater-treatment facility and County Road 68.

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 the establishing of effluent-discharge limits under the NPDES (National Pollution Discharge Elimination System). These limits for Indiana are designed to maintain the following instream water-quality standards:

1. Average dissolved-oxygen (DO) 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. A maximum ammonia-nitrogen concentration 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 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) calibrate and verify a one-dimensional, steady-state, dissolved-oxygen model for Cedar Creek in Dekalb and Allen Counties, Ind., and (2) use the calibrated model for determining alternatives for future waste loadings that would result in the stream meeting Indiana water-quality standards defined for critical summer and winter low-flow conditions. The critical-condition rationale is useful in water-quality planning and management (Hines and others, 1975, p. B5-B6).

BASIN DESCRIPTION

Cedar Creek drains about half of Dekalb County in northeastern Indiana and flows generally south to the St. Joseph River at Cedarville in Allen County (fig. 1). The watershed lies within the Steuben morainal lake area and is characterized by low rolling topography (Schneider, 1966, p. 40). Variation in elevation is about 50 ft (feet). Johnson and Keller (1972) indicated that unconsolidated valley train and alluvial deposits, as much as 100 ft thick, overlie Antrim shale of Devonian age. Soils in the area are primarily Blount-Pewamo and Morley Blount associations, whose permeabilities are typically low (Ulrich, 1966, p. 57). Annual precipitation for the area is approximately 33 inches (Schaal, 1966, p. 157). The average annual runoff is 10 inches (Hoggatt, 1962, p. 9).

Land use in Dekalb County is 70 percent agricultural, 8 percent forested, and 22 percent urban (U.S. Department of Commerce, 1976).

Virtually all point-source waste loads impacting Cedar Creek are in the five incorporated municipalities of Auburn, Avilla, Garrett, Huntertown, and Waterloo (table 1). Each of these municipalities has a sanitary sewer system and at least secondary waste treatment. Advanced waste treatment (rapid sand filtration) was added to the Garrett wastewater-treatment facility in 1976. Sanitary sewage from industrial plants is discharged into this facility, and industrial process and cooling waters are discharged directly into streams. Several unincorporated communities and a rural population are served by septic systems.

Public and semi-private water supplies in the basin are exclusively ground water.

MODEL DESCRIPTION

A steady-state, one-dimensional, segmented water-quality model developed by Bauer and others (1979) was used in this study. One of the objectives of the study was to determine the waste-load assimilation capacity of Cedar Creek at the critical low-flow condition. The use of a steady-state, one-dimensional model is consistent with this objective. The modeling approach assumes that the various flows, loads, and other factors used do not vary significantly with time for a given simulation. The model uses a modified Streeter-Phelps equation that also incorporates nitrogenous, benthic, photosynthetic, and plant respiratory effects on the DO balance. The dissolved-oxygen balance is represented in the model by the following equation:

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

Table 1.--NPDES restrictions for municipalities and industries in the Cedar Creek basin, Dekalb and Allen Counties, Ind.

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

Discharger	Municipality	Flow (Mgal/d)	Five-day BOD (mg/L)	Suspended solids (mg/L)	pH[1]
Auburn waste-water treatment facility	Auburn	2.0	[2]30/45	[2]30/45	6/9
Avilla waste-water treatment facility	Avilla	.16	[2]30/45	[2]30/45	6/9
Garrett waste-water treatment facility	Garrett	.80	[2]10/15	[2]10/15	6/9
Huntertown waste-water treatment facility	Huntertown	.14	[2]30/45	[2]30/45	6/9
Waterloo waste-water treatment facility	Waterloo	.18	[2]30/45	[2]30/45	6/9
Kitchen Quip	Waterloo	-----	-----	[3]20/30	6/9
Borg Warner	Auburn	-----	[3]10/15	[3]10/15	6/9
Standyne Stanscrew	Garrett	-----	-----	[3]10/15	6/9
Cooper Industrial Products	Auburn	-----	[3]10/15	[3]10/15	6/9
County Line Cheese Co.	Auburn	-----	[4]20/30	[4]20/30	6/9
Dana Corp.	Auburn	-----	-----	-----	6/9
Rieke Corp.	Auburn	-----	-----	[3,5]23/34	6/9
Modonair Corp.	Waterloo	-----	-----	-----	6/9
Tenth St. storm sewer	Auburn	-----	-----	-----	---
Altoona storm sewer	Auburn	-----	-----	-----	---

[1]Daily low/daily high.
[2]Monthly average/weekly maximum.
[3]Daily average/daily maximum.
[4]Monthly average/daily maximum.
[5]In pounds.

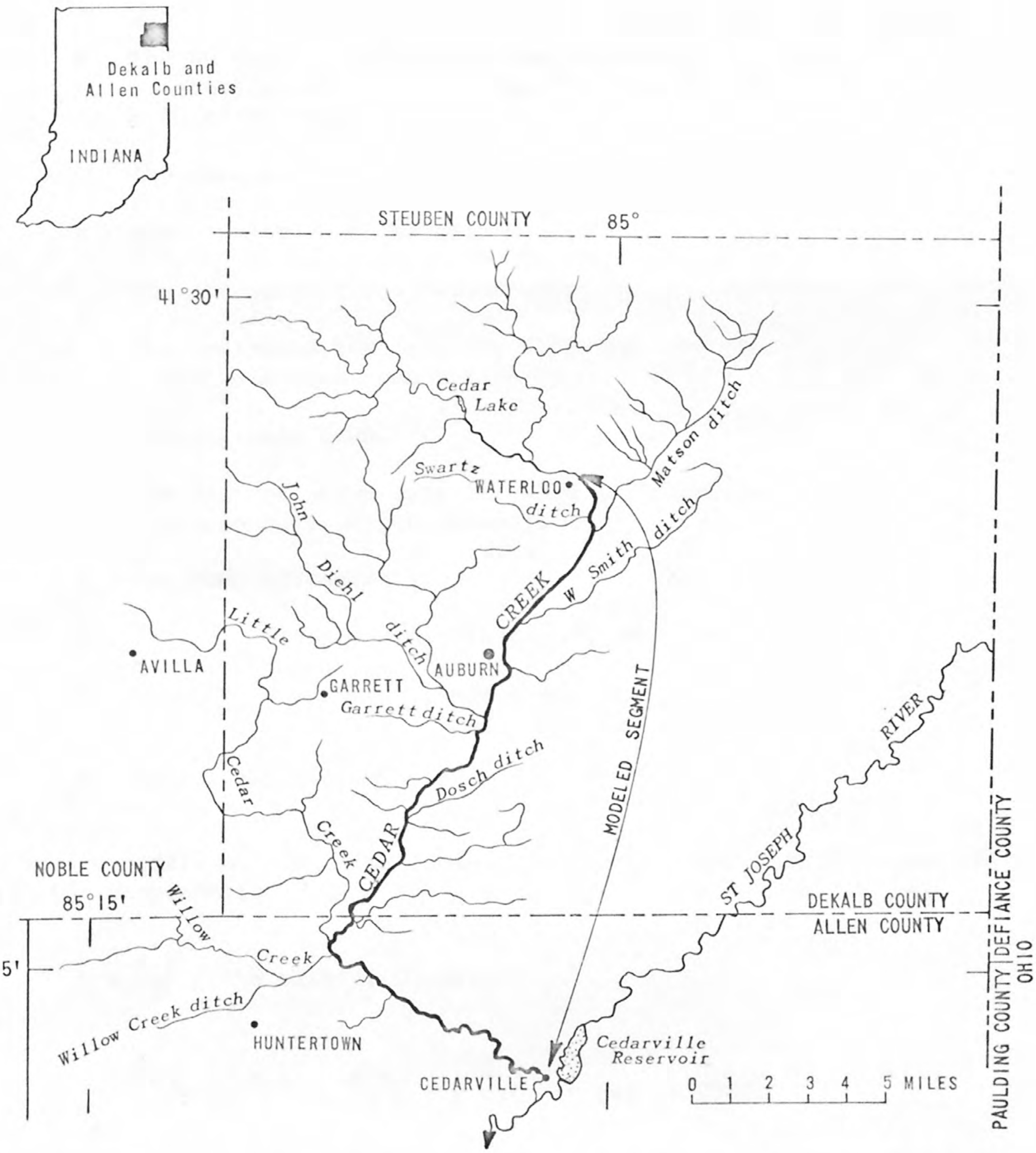

Figure 1.-- Location of modeled segment on Cedar Creek, Dekalb and Allen Counties, Ind.

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 deoxygenation rate for NBOD (nitrogenous biochemical-oxygen demand),

N the NBOD concentration,

P the mean daily photosynthetic DO production,

R the oxygen used by respiration,

and

B the oxygen used by the stream-bottom deposits.

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

$D_o e^{-K_a t}$ the initial DO deficit, (2)

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

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

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

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

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

If the deoxygenation rate for NBOD is assumed to follow zero-order kinetics, equation 1 becomes:

$$\text{Zero} = -1/A \; \partial \frac{(QD)}{dx} - K_a D + K_d L + K_{n,\ zero} - P + R + B, \tag{8}$$

and component 4 becomes :

$$\frac{K_{n,\ zero}}{K_a} \; (1 - e^{-K_a t}) \text{ the deficit due to NBOD,} \tag{9}$$

where $K_{n,\ zero}$ is zero-order kinetics deoxygenation rate for NBOD.

DATA COLLECTION

The modeled segment of Cedar Creek drains a 222 mi^2 area and extends from RM 27.02 at Waterloo to RM 0.77 at Cedarville (fig. 2). Twenty-six sampling stations were selected for the initial survey in August 1976. The number of stations was increased to 34 for the July 1977 survey.

Water-quality data were collected every 3 to 4 h (hours) for 24 h at each station by the Indiana State Board of Health Water-Quality Surveillance Section. Field measurements included dissolved-oxygen concentration, temperature, specific conductance, and pH. Time-weighted composite samples were analyzed by the Indiana State Board of Health Laboratory for 5-day BOD (5-day biochemical-oxygen demand), ammonia nitrogen, and nitrite nitrogen

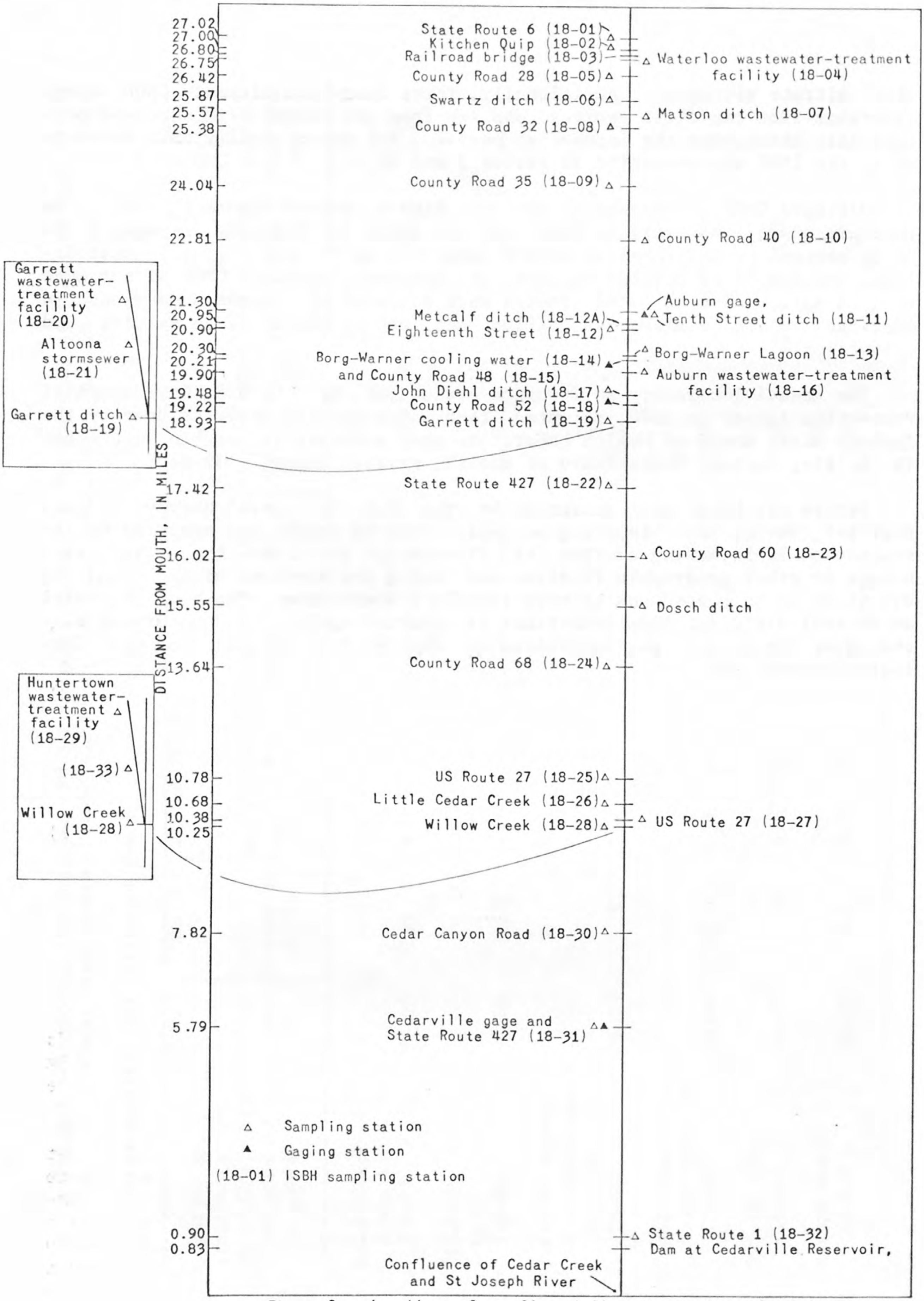

Figure 2.-- Locations of sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind.

plus nitrate nitrogen. Additionally, three long-term ultimate CBOD determinations from the first survey and two from the second were observed periodically throughout the incubation period. All water-quality data collected by the ISBH are presented in tables 2 and 3.

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.)

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).

Stream discharge was measured by the U.S. Geological Survey, Indiana District, during the sampling periods. Time of travel was measured by instantaneously injecting a slug of fluorescent dye into the stream at a bridge or other geographic location and timing the movement of the resulting dye cloud as it passed one or more locations downstream. The time of travel at several different flow conditions is plotted against the concurrent discharge at the nearest gaging station in figures 3-11 (unpublished U.S. Geological Survey data).

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976

[Water-quality data collected and discharge measured by ISBH (Indiana State Board of Health)]

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	Average temp.[1] (°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)
Cedar Creek at State Route 6 (18-01)	27.02	22	8.0	22	----	8.1	3.1	0.4	1.3	1.0
Kitchen Quip[2] (18-02)	27.00	.20	----	----	----	---	---	---	---	---
Waterloo waste-water-treatment facility (18-04)	26.75	.30	7.2	21	----	6.7	3.0	.2	1.6	18.0
Cedar Creek at County Road 28 (18-05)	26.42	22	7.8	22	----	7.6	2.5	.2	1.2	1.2
Swartz ditch (18-06)	25.87	.60	8.2	21	----	9.3	1.5	.2	1.3	.4
Matson ditch (18-07)	25.57	3.0	7.7	21	----	7.3	1.8	.2	1.1	1.4

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976--Continued

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	Average temp.[1] (°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)
Cedar Creek at County Road 32 (18-08)	25.38	----	----	----	----	----	----	----	----	----
Cedar Creek at County Road 35 (18-09)	24.04	29	----	----	----	----	----	----	----	----
Cedar Creek at County Road 40 (18-10)	22.81	----	8.0	21	----	8.0	2.9	0.4	2.9	1.4
Tenth Street ditch (18-11)	21.30	1.8	7.0	19	----	2.7	120	15	17	.1
Metcalf ditch (18-12A)	20.95	----	7.5	18	----	6.0	1.9	.4	.7	.2
Borg Warner cooling water (18-14)	20.30	----	9.4	24	----	8.9	8.0	.6	2.0	.1

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976--Continued

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	Average temp.[1] ($^{\circ}C$)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)
Cedar Creek at County Road 48 (18-15)	20.21	----	7.9	20	----	7.2	2.5	0.3	0.7	7.8
Auburn waste-water treatment facility (18-16)	19.90	2.2	7.2	21	----	3.6	1.7	4.9	5.0	4.8
John Diehl ditch (18-17)	19.48	9.0	7.8	19	----	8.3	2.6	0.3	0.4	0.6
Cedar Creek at County Road 52 (18-18)	19.22	----	7.8	20	18	7.1	3.2	.5	1.5	2.1
Garrett ditch (18-19)	18.93	1.6	8.0	22	26	8.8	6.4	2.4	1.6	4.3
Garrett waste-water treatment facility (18-20)	18.93	----	6.4	21	6	5.8	9.3	12.0	12	10.0

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976--Continued

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	Average temp.[1] (°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)
Altoona Storm Sewer (18-21)	18.93	----	6.7	18	14	4.1	13.0	3.9	4.7	1.7
Cedar Creek at County Road 66 (18-24A)	14.36	----	6.4	20	19	6.7	2.3	.4	1.3	2.1
Cedar Creek at U.S. Route 27 (18-25)	10.78	----	7.0	20	26	7.1	2.1	1.8	1.9	2.0
Little Cedar Creek (18-26)	10.68	15	6.5	19	41	9.5	1.6	.2	1.0	.7
Willow Creek at State Route 27 (18-28)	10.25	6.5	6.5	20	32	7.6	1.8	0.2	1.0	1.7
Huntertown waste-water treatment facility (18-29)	----	----	6.8	25	32	10.0	26.0	12	14.0	.2

Table 2.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976--Continued

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	Average temp.[1] (°C)	Suspended solids (mg/L)	Average DO[1] (mg/L)	Five-day BOD (mg/L)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)
Cedar Creek at Cedar Canyon Road (18-30)	7.82	----	6.7	20	31	8.0	2.2	0.2	1.0	1.5
Cedar Creek at State Route 1 (18-32)	.90	80	6.7	21	35	7.9	4.4	.2	1.1	1.2

[1]Average based on water-quality data collected every 3 to 4 hours for 24 hours.

[2]The use of brand names in this report is for identification purposes only and does not imply endorsement by the U.S. Geological Survey.

Table 3.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., July 20, 1977

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

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	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)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)	Chloride (mg/L)	Fluoride (mg/L)	Total phosphorus (mg/L)	Fecal coliform (colonies per 100 mL)	Oil and grease (mg/L)
Cedar Creek at State Route 6 (18-01)	27.02	-----	7.2	27	515	20	6.7	2.6	<0.1	---	2.3	24	---	0.05	1,200	---
Kitchen Quip (18-02)	27.00	0.20	7.0	20	605	20	7.9	11	.5	---	12	10	---	1.4	7,500	---
Cedar Creek at railroad bridge (18-03)	26.80	----	7.3	27	540	26	6.3	2.6	<.1	---	2.4	14	---	.08	1,200	---
Waterloo waste-water treatment facility (18-04)	26.75	.30	7.6	25	1,130	6	5.0	5.5	2.8	---	20	190	---	2.8	2,300	3.5
Cedar Creek at County Road 28 (18-05)	26.42	-----	7.5	26	535	28	6.0	2.8	<.1	---	3.5	27	---	.18	1,700	3.5
Swartz ditch (18-06)	25.87	1.2	7.4	24	640	80	6.9	3.5	<.1	---	2.9	38	---	.21	21,000	---
Matson ditch (18-07)	25.57	3.3	7.5	26	700	15	6.9	2.1	<.1	---	2.7	24	---	.09	36	---
Cedar Creek at County Road 32 (18-08)	25.38	----	7.5	25	610	24	6.4	2.3	.1	---	2.9	29	---	.14	2,000	---
Cedar Creek at County Road 35 (18-09)	24.04	13	7.6	26	570	26	7.7	2.1	<.1	---	2.7	29	---	.14	1,900	---
Cedar Creek at County Road 40 (18-10)	22.81	-----	7.4	26	640	32	5.7	1.9	<.1	---	2.9	31	---	.15	5,900	---
Tenth Street ditch (18-11)	21.30	22	7.5	24	650	10	6.0	4.3	.2	---	8.7	14	---	.04	8,600	3.6
Cedar Creek at Eighteenth Street (18-12)	20.90	-----	7.4	26	690	26	5.9	2.6	<.1	---	3.8	36	---	.15	14,000	2.7
Borg Warner lagoon (18-13)	20.30	-----	9.0	31	585	210	8.4	80	<.1	---	.1	11	---	2.7	10	7.9
Borg Warner cooling water (18-14)	20.21	-----	8.0	26	565	6	6.7	<1.0	<.1	---	.2	4	---	<.03	100	---
Cedar Creek at County Road 48 (18-15)	20.21	-----	---	26	680	24	5.4	1.9	.1	---	3.4	37	---	.17	8,600	---
Auburn waste-water treatment facility (18-16)	19.90	2.6	7.3	25	960	13	2.0	3.5	7.1	---	1.0	85	---	.40	340	2.2
John Diehl ditch (18-17)	19.48	15	7.4	25	600	120	6.5	2.4	<.1	---	1.9	23	---	.17	2,500	---

Table 3.--Water-quality analyses and discharge measurements for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., July 20, 1977--Continued

Location and station in the Cedar Creek basin (ISBH sta. no. in parens)	River mile	Discharge (ft^3/s)	Median pH	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)	Ammonia nitrogen (mg/L)	Total Kjeldahl nitrogen (mg/L)	Nitrite plus nitrate nitrogen (mg/L)	Chloride (mg/L)	Fluoride (mg/L)	Total phosphorus (mg/L)	Fecal coliform (colonies per 100 mL)	Oil and grease (mg/L)
Cedar Creek at County Road 52 (18-18)	19.22	-----	---	25	695	56	5.3	3.7	0.5	---	2.2	34	---	0.19	4,700	---
Garrett ditch (18-19)	18.93	1.3	7.5	28	---	26	6.8	3.5	.6	1.4	3.3	140	0.8	.15	20,000	---
Stanadyne-Stanscrew (18-19a)	18.93	-----	---	31	---	20	---	26	4.7	---	3.4	700	4.0	.42	<10	10
Garrett waste-water treatment facility (18-20)	18.93	-----	---	23	---	2	6.8	4.6	1.8	---	8.2	240	.7	.56	------	3.9
Altoona storm sewer (18-21)	18.93	-----	---	20	---	12	5.6	5.9	1.3	---	.8	49	.4	.24	40,000	3.9
Cedar Creek at State Route 427 (18-22)	17.42	-----	7.6	24	---	64	5.7	3.4	<.1	1.1	3.1	37	.5	.18	2,200	---
Cedar Creek at County Road 60 (18-23)	16.02	-----	7.5	25	---	64	5.9	3.4	.3	1.2	2.4	37	.5	.17	2,600	---
Cedar Creek at County Road 68 (18-24)	13.64	-----	7.6	24	---	52	6.3	2.5	.1	1.1	2.2	35	.5	.13	3,800	---
Cedar Creek at U.S. Route 27 (18-25)	10.78	-----	---	25	---	44	6.4	3.4	.2	1.0	2.1	37	.5	.16	6,400	---
Little Cedar Creek (18-26)	10.68	38	7.8	24	560	88	6.7	2.6	.2	1.3	1.8	26	.4	.18	160	---
Cedar Creek at U.S. Route 27 (18-27)	10.38	-----	---	24	610	68	6.7	2.5	.2	---	1.9	35	.4	.12	1,300	---
Willow Creek at State Route 27 (18-28)	10.25	-----	7.8	23	740	9	6.9	2.3	<.1	.6	.4	36	.5	.05	190	---
Huntertown waste-water treatment facility (18-29)	-----	-----	---	28	2,500	64	10.6	30	8.2	---	.1	860	.4	4.3	700	74
Willow Creek at Huntertown (18-33)	-----	-----	---	24	940	---	5.7	7.7	1.4	---	.3	73	---	.12	3,700	---
Cedar Creek at Cedar Canyon Road (18-30)	7.82	-----	7.4	24	635	36	7.4	2.4	<.1	.6	1.6	40	.4	.10	1,500	---
Cedar Creek at State Route 427 (18-31)	5.79	84	7.8	24	630	26	7.4	2.1	<.1	1.7	1.1	36	.4	.08	4,000	---
Cedar Creek at State Route 1 (18-32)	.90	-----	---	25	660	22	8.3	2.0	<.1	1.5	.6	32	.4	.07	2,000	---

[1] Average based on water-quality data collected every 3 to 4 hours for 24 hours.

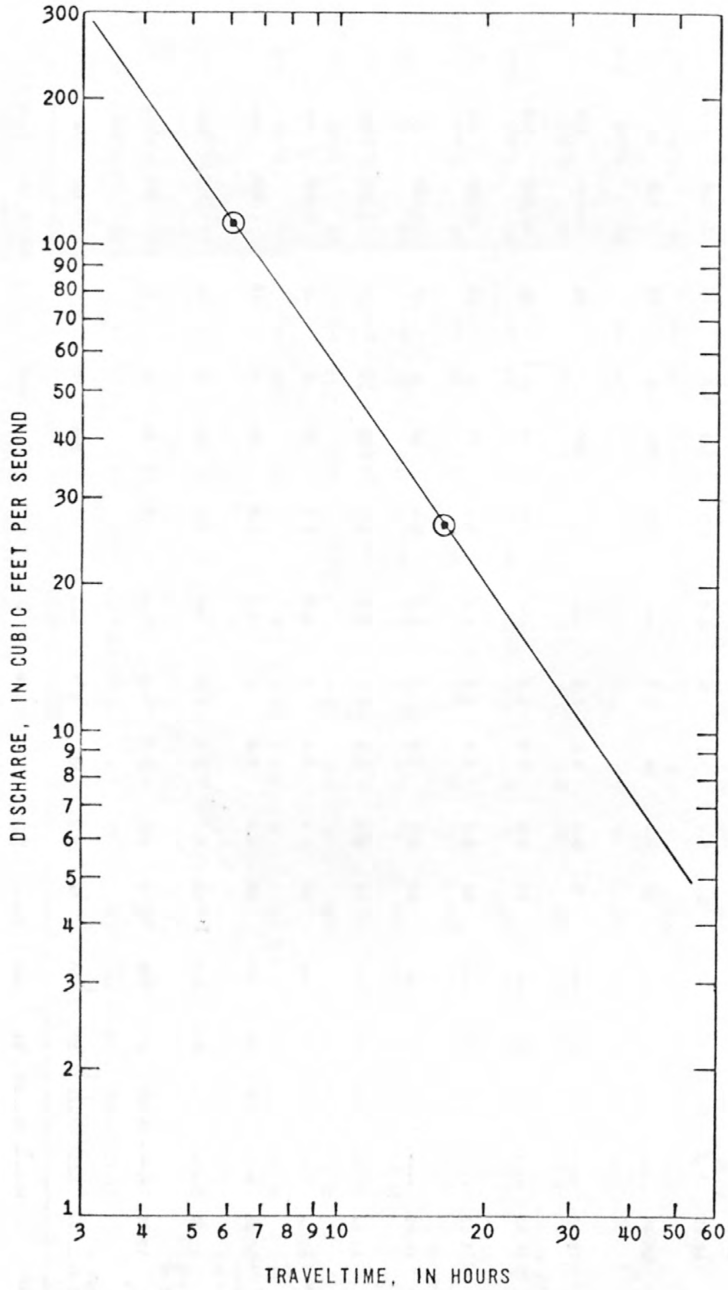

Figure 3.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 28.57 to 24.04.

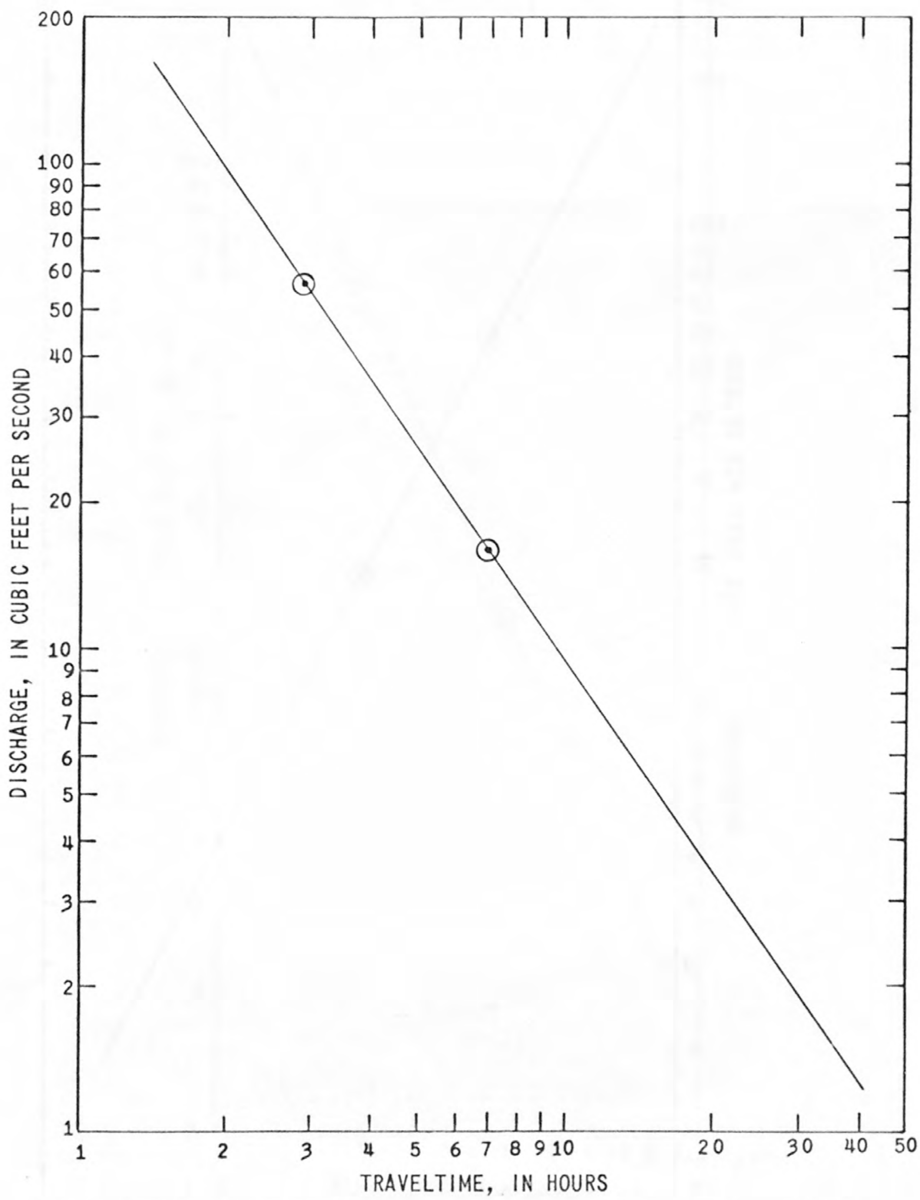

Figure 4.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 24.04 to 21.34.

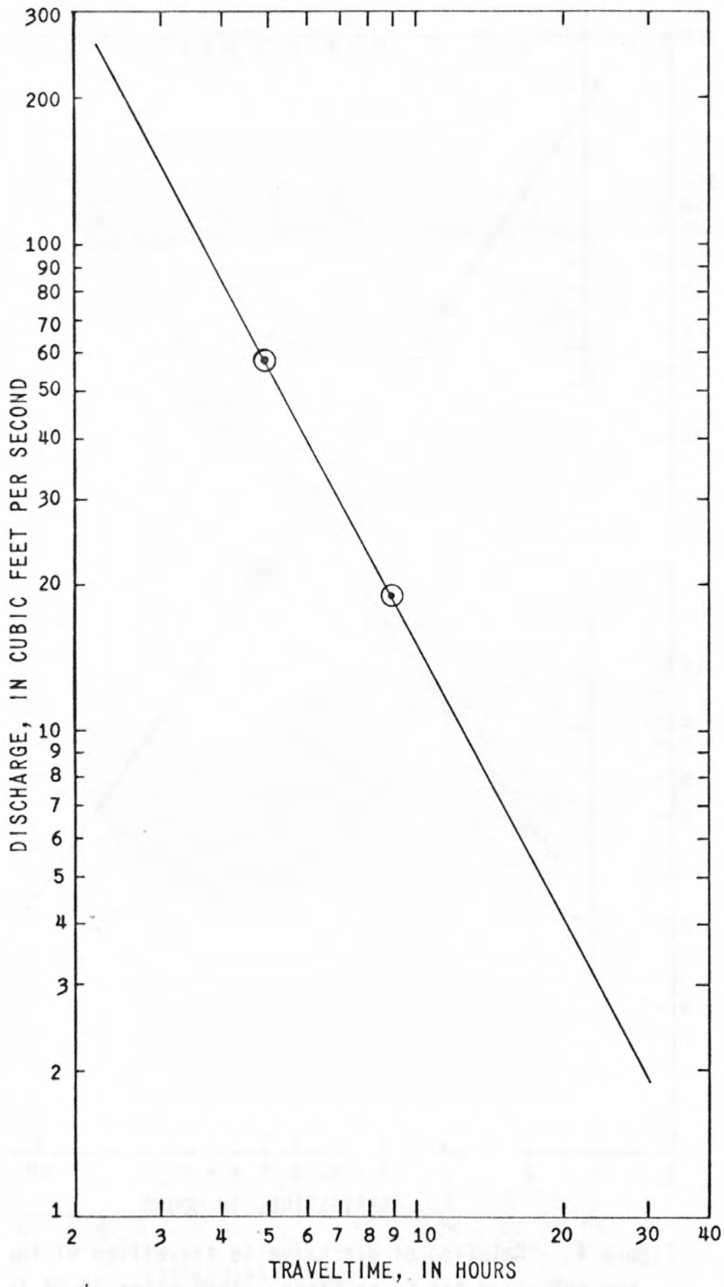

Figure 5.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 21.34 to 17.42.

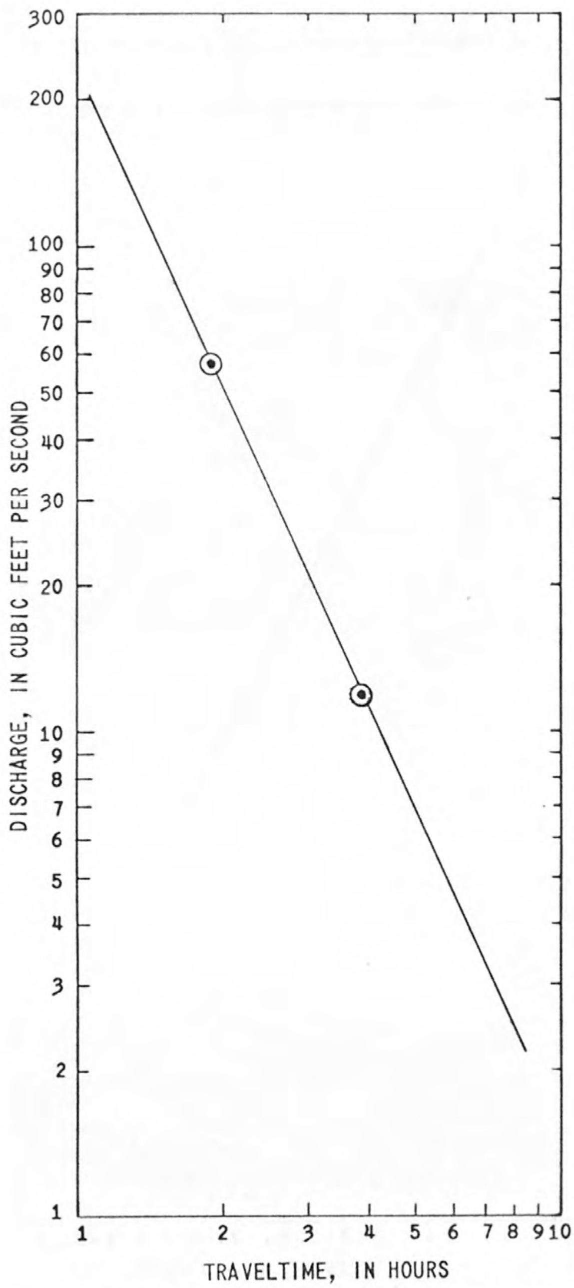

Figure 6.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 17.42 to 16.02.

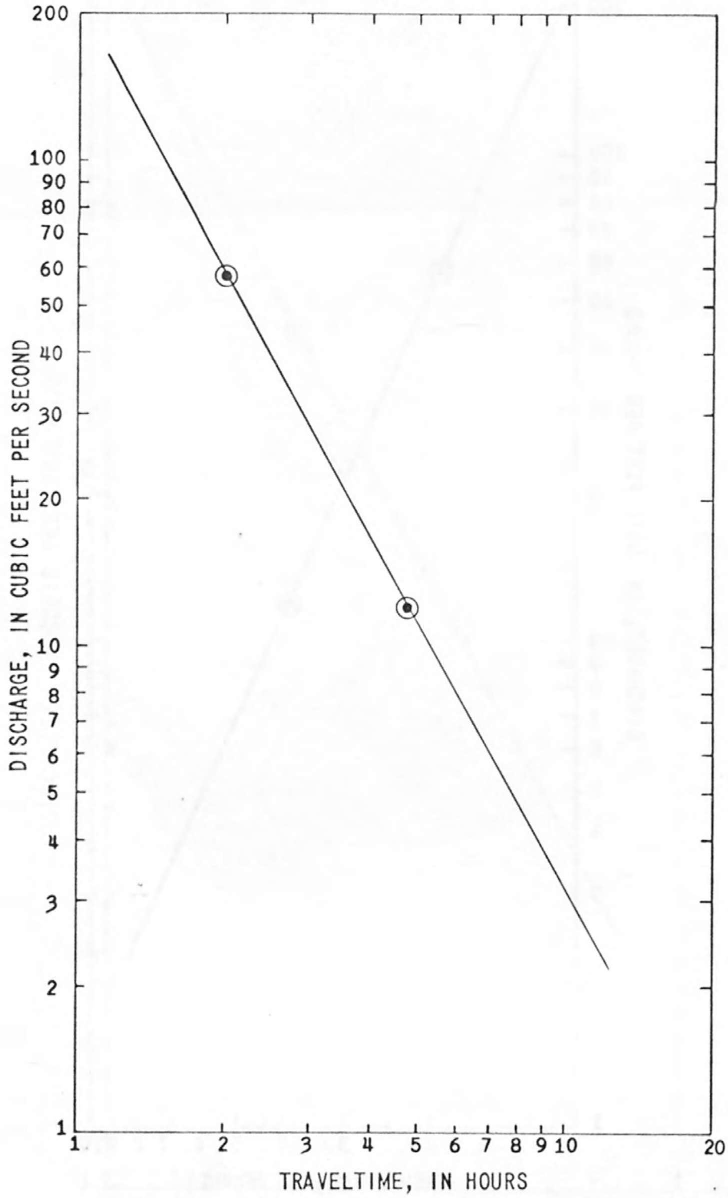

Figure 7.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 16.02 to 14.36.

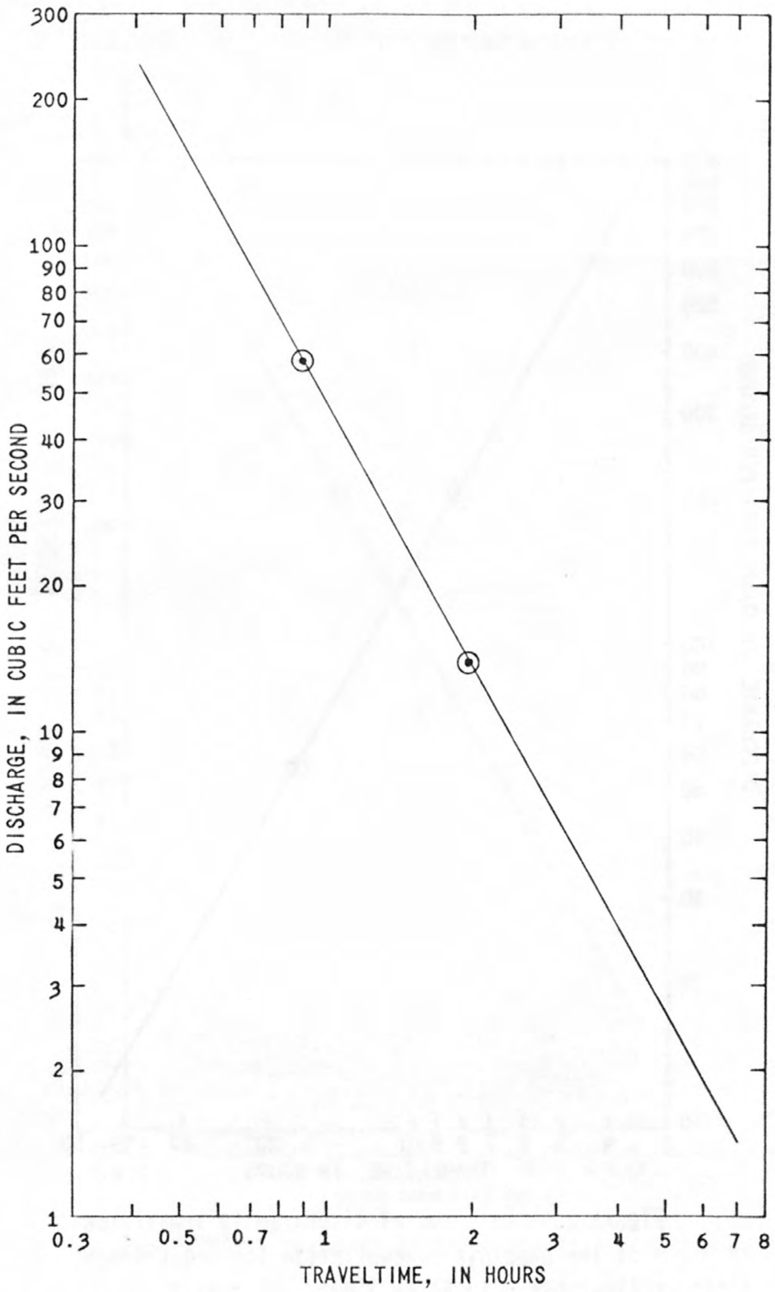

Figure 8.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 14.36 to 13.64.

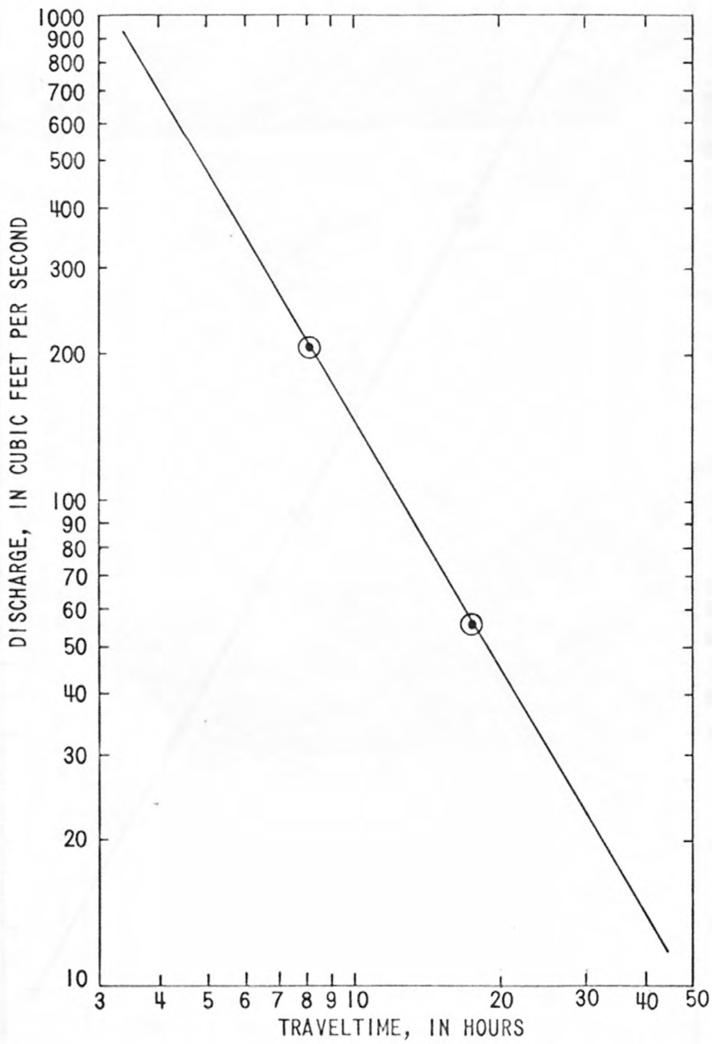

Figure 9.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 13.64 to 7.82.

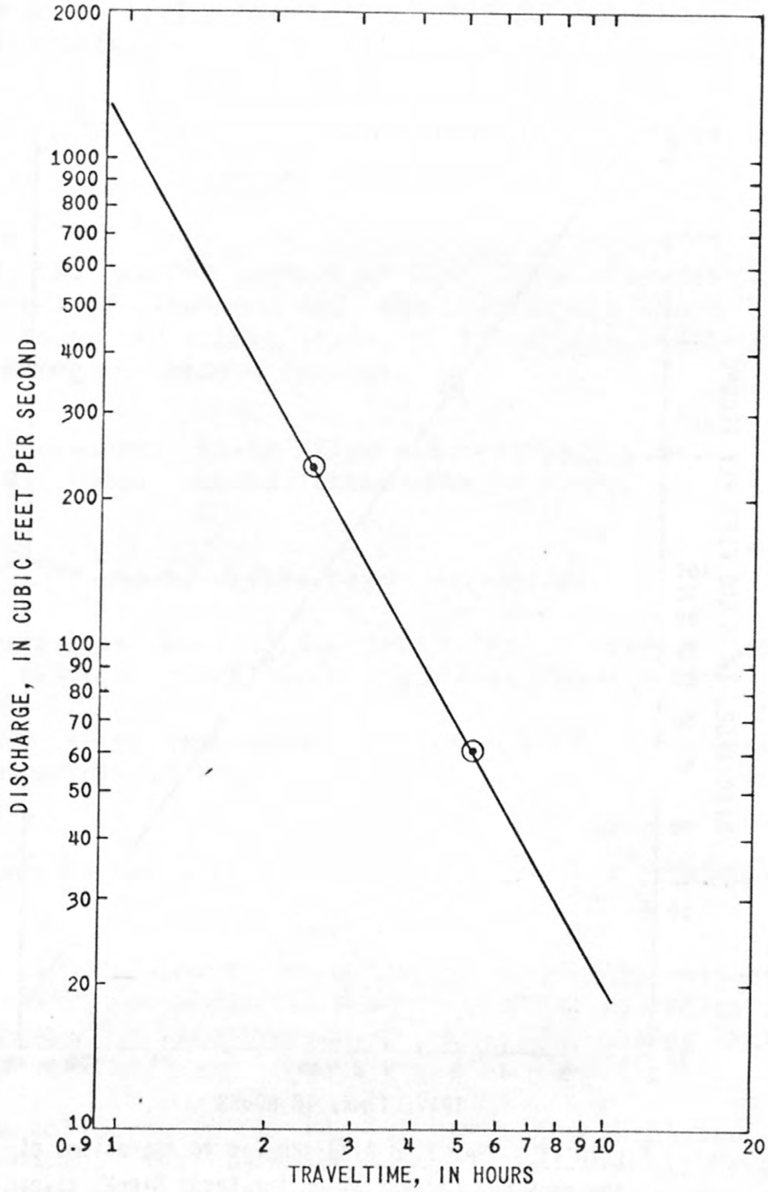

Figure 10.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 7.82 to 5.79.

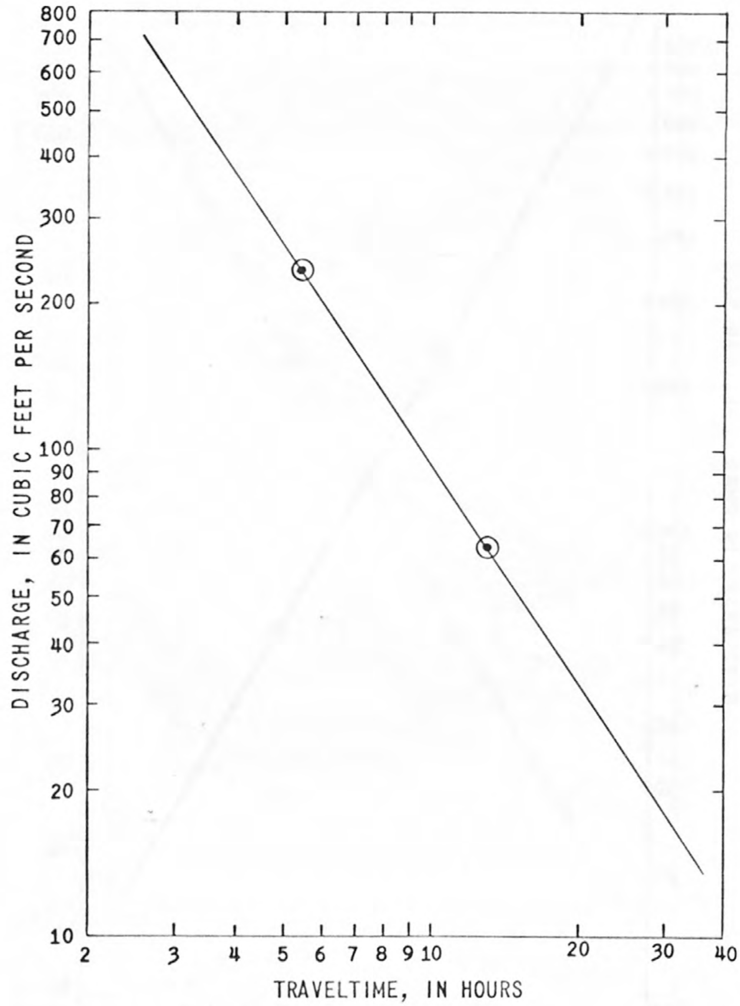

Figure 11.-- Relation of discharge to traveltime of the peak dye concentration for Cedar Creek, river miles 5.79 to 0.77.

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 necessary to make an initial estimate until the calibration process indicates appropriate refinements.

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 Cedar Creek 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.

The flow data collected during July 20, 1977 (table 6) indicate that the survey was done during the receding stages of a small hydrograph. The change in flow during the survey was approximately 11 percent at the U.S. Geological Survey gage at Cedarville. Hydrologic conditions during August 11, 1976 (table 5) were considered to be more representative of the steady-state conditions assumed in the model than the data in table 6. However, because only two sets of water-quality data were collected for Cedar Creek, both sets were needed for determining the model-input parameters.

Table 4.--Physical characteristics for modeled stream reaches, Cedar Creek basin, Dekalb and Allen Counties, Ind.

Reach	Starting river mile	Ending river mile	Length of reach (mi)	Average slope (ft/mi)	Average[1] width (ft)	Average[1] depth (ft)
1	27.02	27.00	0.02	10.0	15.5	2.6
2	27.00	26.75	.25	5.2	15.5	2.6
3	26.75	26.42	.33	13.6	15.5	2.6
4	26.42	25.87	.55	3.6	19.5	2.1
5	25.87	25.57	.30	3.7	19.5	2.3
6	25.57	25.38	.19	3.7	19.5	2.8
7	25.38	24.04	1.34	4.5	22.5	2.5
8	24.04	22.81	1.23	6.1	22.5	2.5
9	22.81	21.30	1.51	3.1	17.5	1.9
10	21.30	20.30	1.00	5.5	20.3	1.8
11	20.30	19.90	.40	3.3	25.8	1.5
12	19.90	19.48	.42	4.8	25.8	1.6
13	19.48	18.93	.55	4.9	25.8	2.2
14	18.93	17.42	1.51	5.5	25.8	2.3
15	17.42	15.55	1.87	3.7	32.2	1.8
16	15.55	14.22	1.33	3.5	32.2	1.8
17	14.22	13.64	.58	7.8	28.3	2.1
18	13.64	10.78	2.86	3.7	46.0	1.3
19	10.78	10.68	.10	3.0	53.0	1.6
20	10.68	10.25	.43	4.2	59.0	2.1
21	10.25	7.82	2.43	5.0	59.0	2.3
22	7.82	5.79	2.03	2.7	59.0	2.3
23	5.79	.90	4.89	3.1	59.0	2.0

[1]Values represent average conditions during the two water-quality surveys, August 11, 1976, and July 20, 1977.

Difference in streamflow between two points that could not be attributed to effluent discharge or tributaries by mass-balance analysis was accounted for by linear runoff, which was assumed to be proportional to the number of miles in a modeled reach.

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 those samples in which only a 5-day BOD concentration had been determined (figs. 12 and 13 and tables 7 and 8). This method gives reliable estimates of ultimate CBOD (Stamer and others, 1979). For the August 1976 data, the ultimate CBOD of all samples was assumed to be 2.0 times that of the 5-day CBOD. This relationship was also used for estimating ultimate CBOD for wastewater-treatment facilities and industries from the July 1977 data. A ratio of 1.6 (ultimate CBOD/5-day CBOD) was determined for stream conditions for the July 20, 1977, survey. Long-term CBOD measurements were made at different sampling sites for the two surveys. Consequently, the variation in the ratio of ultimate CBOD/5-day CBOD with sampling dates could not be determined.

The following assumptions were made in estimating ultimate CBOD for several dischargers and tributaries to Cedar Creek. The ultimate CBOD for Kitchen Quip (18-02) for the August 1976 calibration was estimated from the Monthly Report of Operations (unpublished data, Indiana State Board of Health, Indianapolis, Ind.). Ultimate CBOD concentrations for Dosch and Schmadel ditches, which were not sampled in the August 1976 survey, were assumed to be the same as the ultimate CBOD for Little Cedar Creek (18-26).

The measured 5-day BOD concentration at the Tenth Street ditch (18-11), August 11, 1976, survey was 120 mg/L. This value was significantly higher than other measured values determined by the Indiana State Board of Health (2.8 mg/L, 6/17/76; 17 mg/L, 6/29/77; and 4.3 mg/L, 7/20/77; Sammy C. Gibson, Indiana State Board of Health, oral commun., 1978) and did not agree with observed in-stream CBOD values. Consequently, the ultimate CBOD values for the Tenth Street ditch for the 1976 and 1977 surveys were determined by mass-balance analysis.

The 5-day BOD concentrations for the Auburn wastewater-treatment facility were 1.7 and 3.5 mg/L for the August 1976 and July 1977 surveys, respectively. These values did not agree with the 5-day BOD values reported by the Auburn wastewater-treatment facility, which were 32 and 19 mg/L for the August 1976 and July 1977 surveys, respectively. Consequently, the CBOD value for the Auburn wastewater-treatment facility was estimated by mass-balancing observed instream CBOD values.

Table 5.--Model input for Cedar Creek, Dekalb and Allen Counties, Ind., August 11, 1976

[All rates corrected to observed stream temperatures); 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	Average depth (ft)	Discharge (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})	Linear runoff: Discharge (ft^3/s)	Linear runoff: Ultimate CBOD (mg/L)	Linear runoff: NBOD (mg/L)	Linear runoff: DO (mg/L)
1	Cedar Creek at State Route 6 (18-01)	27.02	3.1	22.0	0.06	6.2	1.8	8.1	----	22	-----	0.3	0.12	0.12	0.4	4.21	---	---	---	---
2	Kitchen Quip (18-02)	27.00	3.1	.2	.80	12.2	3.0	7.9	0.82	22	-----	.3	.12	.12	.4	2.19	---	---	---	---
3	Waterloo waste-water treatment facility (18-04)	26.75	3.1	.3	1.06	6.0	4.0	6.7	2.20	22	-----	.3	5.48	.12	[1]357	5.75	---	---	---	---
4	Cedar Creek at County Road 28 (18-05)	26.42	2.5	----	1.76	----	---	---	----	21	-----	.3	.21	.12	.3	1.50	---	---	---	---
5	Swartz ditch (18-06)	25.87	2.6	.6	.96	3.0	1.0	9.3	-.40	21	-----	.3	.10	.12	.3	1.51	0.6	2.0	0.4	2.0
6	Matson ditch (18-07)	25.57	3.0	3.0	.61	3.6	1.0	7.3	1.60	21	-----	.3	.10	.12	.3	1.52	.6	25.0	.4	2.0
7	Cedar Creek at County Road 32 (18-08)	25.38	2.6	----	4.61	----	---	---	----	21	-----	.3	.10	.12	.3	1.86	.7	25.0	.4	2.0
8	Cedar Creek at County Road 35 (18-09)	24.04	2.7	----	2.23	----	---	---	----	21	-----	.3	.10	.12	.3	4.44	1.0	2.0	.4	2.0
9	Cedar Creek at County Road 40 (18-10)	22.81	2.0	----	2.74	----	---	---	----	20	-----	.3	.10	.11	.3	2.22	1.0	2.0	.4	2.0
10	Tenth Street ditch (18-11)	21.30	1.9	1.8	1.73	72.0	17.0	2.7	6.58	20	-----	.3	7.94	.11	.3	4.10	4.0	2.0	.4	2.0
11	Borg Warner (18-14)	20.30	1.7	.5	.69	16.0	2.7	8.9	-.69	20	-----	.3	7.94	.11	.3	2.42	.1	2.0	.4	2.0
12	Auburn waste-water treatment facility (18-16)	19.90	1.8	2.2	.73	60.0	21.3	3.6	5.30	20	-----	.3	.68	.11	[1]357	3.55	---	---	---	---
13	John Diehl ditch (18-17)	19.48	2.2	9.0	.95	5.4	1.4	8.3	.98	20	-3.48	.3	.68	.11	[1]300	3.66	---	---	---	---
14	Garrett ditch (18-19)	18.93	2.3	1.6	2.62	12.8	10.5	8.8	.29	20	-3.48	.3	.68	.11	[1]300	4.10	.8	2.0	.4	2.0

Table 5.--Model input for Cedar Creek, Dekalb and Allen Counties, Ind., August 11, 1976--Continued

Reach	Upstream boundary of modeled reach (ISBH sta. no. in parens)	River mile	Average depth (ft)	Discharge (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})	Linear runoff: Discharge (ft^3/s)	Linear runoff: Ultimate CBOD (mg/L)	Linear runoff: NBOD (mg/L)	Linear runoff: DO (mg/L)
15	Cedar Creek at State Route 427 (18-22)	17.42	1.9	----	3.17	----	---	---	----	20	-3.48	0.9	0.68	0.11	[1]300	2.85	1.0	2.0	0.4	2.0
16	Dosch ditch (18-23A)	15.55	1.9	0.3	2.24	3.2	1.0	7.7	1.39	20	-3.48	.9	.68	.11	.3	2.69	.5	2.0	.4	2.0
17	Schmadel ditch	14.22	2.1	.5	1.01	3.2	1.0	7.7	1.39	20	-3.48	.3	.25	.11	.3	5.78	.7	2.0	.4	2.0
18	Cedar Creek at County Road 68 (18-24)	13.64	1.4	----	7.62	----	---	----	----	20	-1.66	.8	.25	.11	.3	1.82	---	---	---	---
19	Cedar Creek at U.S. Route 27 (18-25)	10.78	1.8	----	.27	----	---	----	----	20	-1.66	1.0	.25	.11	.3	1.46	---	---	---	---
20	Little Cedar Creek (18-26)	10.68	2.1	15.0	1.15	3.2	1.0	7.5	1.78	20	-1.66	1.2	.10	.11	.3	2.04	---	---	---	---
21	Willow Creek at State Route 27 (18-28)	10.25	2.3	6.5	6.47	3.6	1.0	7.6	1.49	20	-----	.3	.10	.11	.3	2.44	-3.5	2.0	.4	2.0
22	Cedar Creek at Cedar Canyon Road (18-30)	7.82	2.2	----	5.00	----	---	----	----	20	-----	.3	.10	.11	.3	1.42	-3.0	2.0	.4	2.0
23	Cedar Creek at State Route 427 Cedarville gage (18-31)	5.79	2.0	----	12.86	----	---	----	----	21	-----	.3	.10	.12	.3	1.54	12.0	2.0	.4	2.0

[1]Zero-order kinetic removal rate for ammonia nitrogen in (lb/d)/d.

Table 6.--Model input for Cedar Creek, Dekalb and Allen Counties, Ind., July 20, 1977

[All rates corrected to observed stream temperatures; 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	Average depth (ft)	Discharge (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})	Linear runoff: Discharge (ft^3/s)	Linear runoff: Ultimate CBOD (mg/L)	Linear runoff: NBOD (mg/L)	Linear runoff: DO (mg/L)
1	Cedar Creek at State Route 6 (18-01)	27.02	2.1	11.4	0.08	4.2	0.5	6.7	0.88	27	-----	0.3	0.83	0.19	0.5	3.57	----	---	---	---
2	Kitchen Quip (18-02)	27.00	2.2	.2	1.05	22.0	2.4	7.9	.83	26	-3.48	1.0	.79	.18	.5	1.82	----	---	---	---
3	Waterloo waste-water-treat-ment facility (18-04)	26.75	2.2	.2	1.38	11.0	12.3	5.0	2.89	25	-3.48	1.0	.76	.18	[1]50	4.67	0.2	2.0	0.4	2.0
4	Cedar Creek at County Road 28 (18-05)	26.42	1.8	----	2.31	----	----	---	----	25	-----	.3	.63	.18	.5	1.24	.5	2.0	.4	2.0
5	Swartz ditch (18-06)	25.87	2.0	1.2	1.26	5.6	.4	6.9	1.15	25	-----	.3	.63	.18	.5	1.25	.5	2.0	.4	2.0
6	Matson ditch (18-07)	25.57	2.6	3.3	.80	3.4	.4	6.9	.84	25	-----	.3	.63	.18	.5	1.26	.5	2.0	.4	2.0
7	Cedar Creek at County Road 32 (18-08)	25.38	2.3	----	6.04	----	----	---	----	26	-----	.3	.66	.18	.5	1.57	.5	2.0	.4	2.0
8	Cedar Creek at County Road 35 (18-09)	24.04	2.4	----	2.64	----	----	---	----	26	-3.48	4.7	.66	.18	.5	4.17	.5	2.0	.4	2.0
9	Smith ditch	22.81	1.8	2.2	3.24	4.5	.4	6.9	.99	26	-3.48	.3	.66	.18	.5	2.13	.1	2.0	.4	2.0
10	Tenth Street ditch (18-11)	21.30	1.6	1.0	2.14	35.0	2.2	6.0	2.05	26	-3.48	.3	.66	.18	.5	3.77	.3	2.0	.4	2.0
11	Borg Warner (18-14)	20.30	1.3	.5	.86	51.2	.4	7.0	.89	26	-3.48	.3	.66	.18	.5	2.23	-.6	2.0	.4	2.0
12	Auburn waste-water-treat-ment facility (18-16)	19.90	1.4	2.6	.90	20.0	31.0	2.0	5.89	25	-3.48	.3	.63	.18	[1]357	3.19	-1.0	2.0	.4	2.0
13	John Diehl ditch (18-17)	19.48	2.2	15.0	1.18	3.8	.4	6.5	1.39	25	-3.48	.3	.63	.18	[1]357	3.29	-1.0	2.0	.4	2.0
14	Garrett ditch (18-19)	18.93	2.3	1.8	3.23	5.6	2.7	6.8	.64	24	-3.48	.3	.48	.17	[1]300	3.62	----	---	---	---

Table 6.--Model input for Cedar Creek, Dekalb and Allen Counties, Ind., July 20, 1977--Continued

Reach	Upstream boundary of modeled reach (ISBH sta. no. in parens)	River mile	Average depth (ft)	Discharge (ft³/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/m²/d)]	K_r (day^{-1})	K_d (day^{-1})	K_n (day^{-1})	K_a (day^{-1})	Linear runoff: Discharge (ft³/s)	Linear runoff: Ultimate CBOD (mg/L)	Linear runoff: NBOD (mg/L)	Linear runoff: DO (mg/L)
15	Cedar Creek at State Route 427 (18-22)	17.42	1.8	----	3.88	---	---	---	----	25	-3.48	0.3	0.50	0.18	0.5	2.59	-0.4	2.0	0.4	2.0
16	Dosch ditch (18-23A)	15.55	1.8	0.6	2.76	4.5	0.4	6.9	0.99	24	-1.24	.3	.48	.17	.4	2.39	----	---	---	---
17	Schmadel ditch	14.22	2.1	1.0	1.21	4.5	.4	6.9	.99	24	-1.24	.3	.48	.17	.4	5.25	.4	2.0	.4	2.0
18	Cedar Creek at County Road 68 (18-24)	13.64	1.3	----	6.73	---	---	---	----	25	-1.02	.3	.38	.18	.5	2.29	.6	2.0	.4	2.0
19	Cedar Creek at U.S. Route 27 (18-25)	10.78	1.3	----	.23	---	---	---	----	24	-1.02	.3	.36	.17	.4	1.79	----	---	---	---
20	Little Cedar Creek (18-26)	10.68	2.2	38.0	1.01	4.2	1.0	6.7	1.35	24	-1.02	.3	.48	.17	.4	2.51	----	---	---	---
21	Willow Creek at State Route 27 (18-28)	10.25	2.4	11.0	5.72	3.7	.4	6.9	1.31	24	-----	.3	.48	.17	.4	3.01	-3.0	2.0	.4	2.0
22	Cedar Creek at Cedar Canyon Road (18-30)	7.82	2.3	----	4.40	---	---	---	----	24	-----	.3	.36	.17	.4	1.76	-4.1	2.0	.4	2.0
23	Cedar Creek at State Route 427 Cedarville gage (18-31)	5.79	2.1	----	10.88	---	---	---	----	25	-----	.3	.13	.18	.5	1.99	----	---	---	---

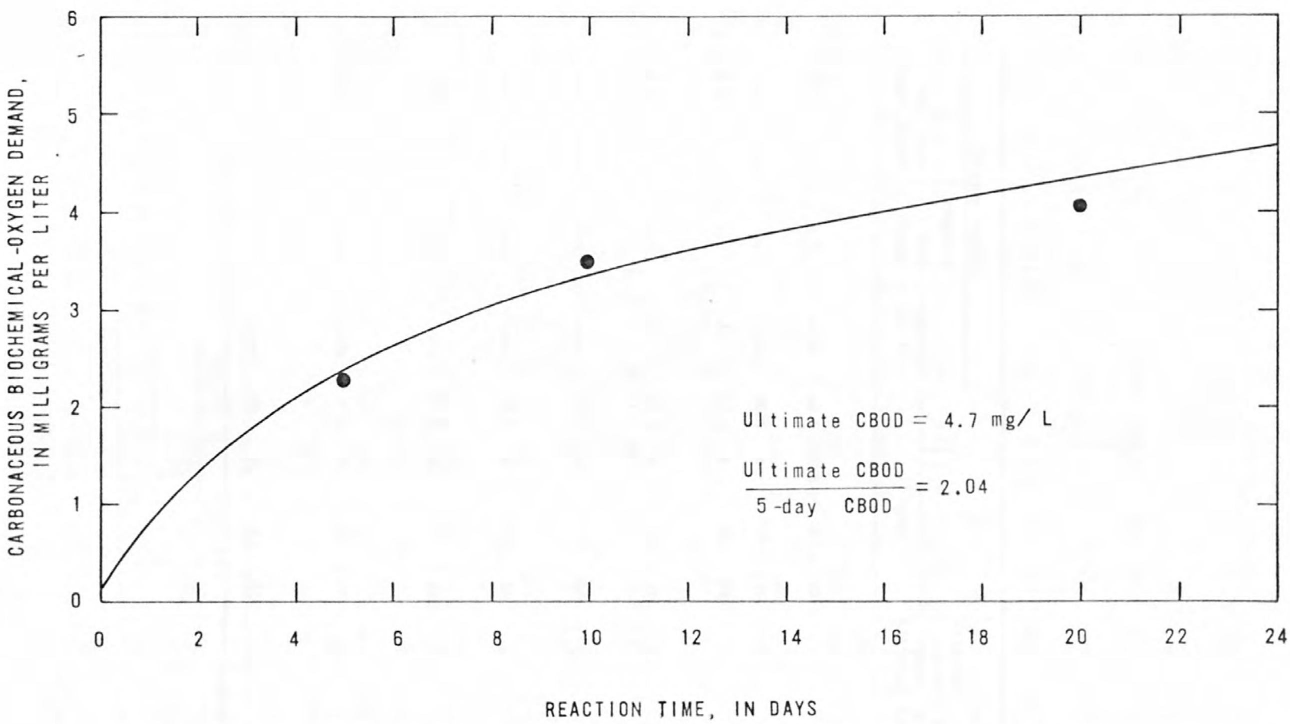

Figure 12.--Relation of carbonaceous biochemical-oxygen demand to time at river mile 13.64, Cedar Creek, August 11, 1976.

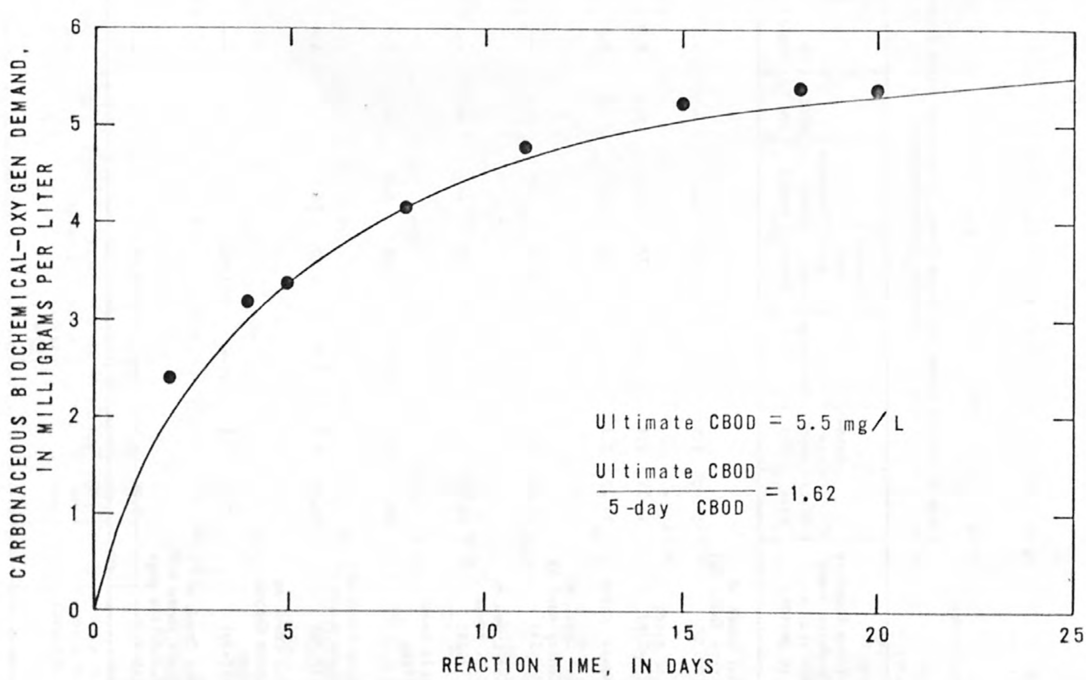

Figure 13.—Relation of carbonaceous biochemical-oxygen demand to time at river mile 10.78, Cedar Creek, July 20, 1977.

Table 7.--Carbonaceous biochemical-oxygen demand data for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., August 11, 1976

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

Location and station: Auburn wastewater-treatment facility (18-16)			Location and station: Cedar Creek at County Road 52 (18-18)			Location and station: Cedar Creek at County Road 66 (18-24A)		
Day	CBOD (mg/L)	Percent remaining	Day	CBOD (mg/L)	Percent remaining	Day	CBOD (mg/L)	Percent remaining
5	1.7	79	5	3.2	63	5	2.3	51
10	---	--	10	4.6	47	10	3.5	26
20	3.9	51	20	6.9	21	20	4.1	13
30	5.7	29	30	8.3	5	30	4.5	4
Ultimate CBOD = 8.0 mg/L			Ultimate CBOD = 8.7 mg/L			Ultimate CBOD = 4.7 mg/L		
K_d (base e) = 0.06 day^{-1}			K_d (base e) = 0.09 day^{-1}			K_d (base e) = 0.11 day^{-1}		

Table 8.--Carbonaceous biochemical-oxygen demand data for sampling stations in the Cedar Creek basin, Dekalb and Allen Counties, Ind., July 20, 1977

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

Location and station: Auburn wastewater-treatment facility (18-16)			Location and station: Cedar Creek at U.S. Route 27 (18-25)		
Day	CBOD (mg/L)	Percent remaining	Day	CBOD (mg/L)	Percent remaining
2	1.8	81	2	2.4	56
4	3.3	65	4	3.2	42
5	3.5	62	5	3.4	38
8	4.5	52	8	4.2	24
11	5.3	43	11	4.8	13
15	6.3	32	15	5.2	5
18	7.0	25	18	5.4	2
20	7.3	22	20	5.4	2
Ultimate CBOD = 9.3 mg/L			Ultimate CBOD = 5.5 mg/L		
K_d (base e) = 0.07 day^{-1}			K_d (base e) = 0.14 day^{-1}		

A plot of the percentages of long-term CBOD remaining against time provided an estimate of the deoxygenation rate for CBOD, K_d (Thomas 1950). (See figs. 14 and 15.) K_d values of 0.11 and 0.14 day^{-1} at 20°C were used in the August 1976 and July 1977 model calibrations, respectively.

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

$$K_r = \ln \left[\frac{C_d}{C_u}\right] t^{-1} \qquad (10)$$

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 the August 1976 survey ranged from 0.1 day^{-1}, in reaches that did not receive significant waste loads, to 7.9 day^{-1} downstream from the Tenth Street ditch at RM 21.30 to the Auburn wastewater-treatment facility at RM 19.90. Values of K_r for the July 1977 survey ranged from 0.6 day^{-1} in the upper reaches to 0.1 day^{-1} downstream. These values of K_r are high and suggest that most of the CBOD is removed through settling or some other process.

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

$$(K)_T \quad \text{Carbonaceous} = (K)_{20°C}(1.047^{T-20°C}) \qquad (11)$$

where

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

T the temperature, in degrees Celsius,

and

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

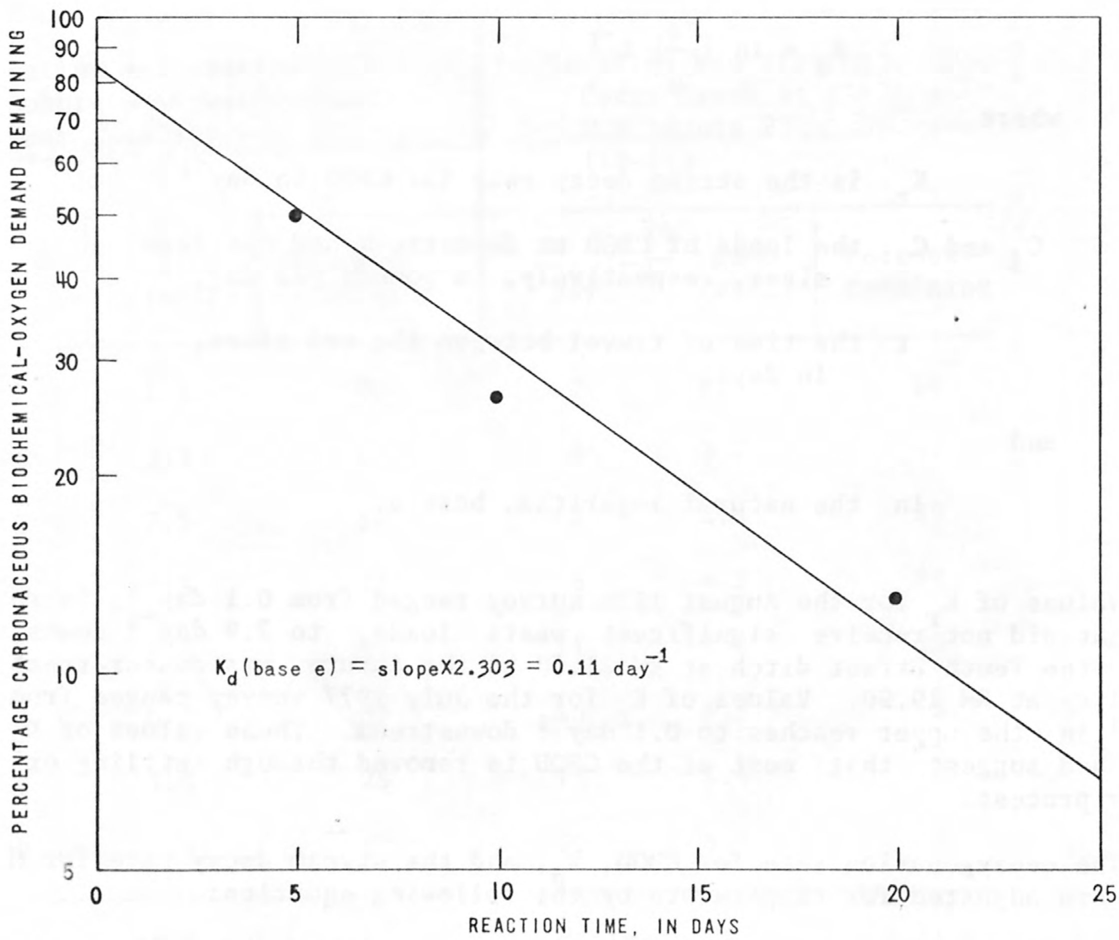

Figure 14. -- Relation of percentage of remaining carbonaceous biochemical-oxygen demand to time at river mile 13.64, Cedar Creek, August 11, 1976.

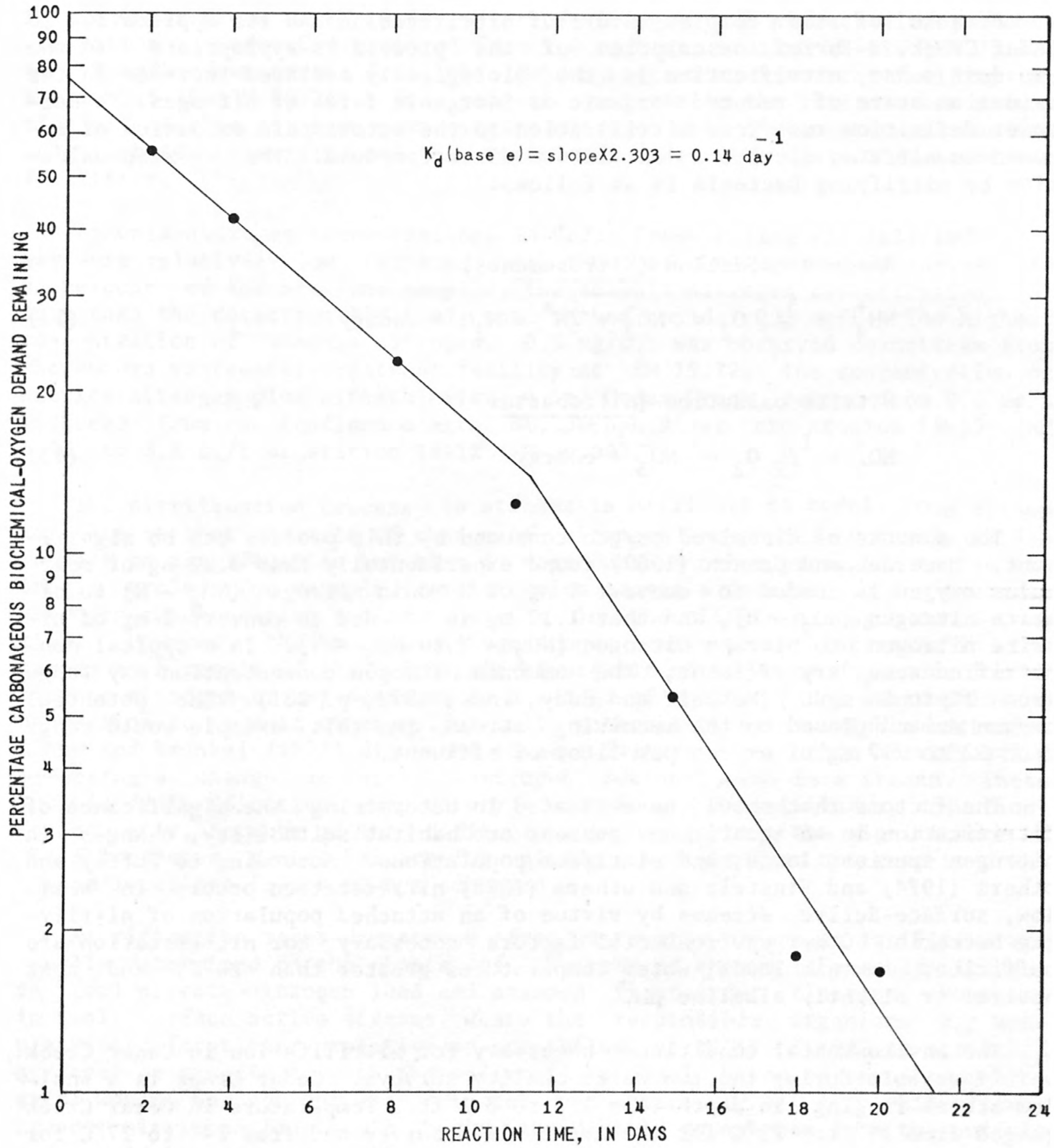

Figure 15.-- Relation of percentage of remaining carbonaceous biochemical-oxygen demand to time at river mile 10.78, Cedar Creek, July 20, 1977.

Nitrogenous Biochemical-Oxygen Demand

Because of the complex nature of nitrification and its application to Cedar Creek, 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 (Nitrosomonas)

$$NH_4^+ + {}^3/_2\, O_2 = NO_2^- + 2H^+ + H_2O + \text{energy} \qquad (12)$$

Nitrite oxidation (Nitrobacter)

$$NO_2^- + {}^1/_2\, O_2 = NO_3^- + \text{energy} \qquad (13)$$

The amounts of dissolved oxygen consumed by this process 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, Inc., 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 nitrogen species loads, and nitrifier populations. According to Tuffey and others (1974) and Finstein and others (1978) nitrification occurs in shallow, surface-active streams by virtue of an attached population of nitrifying bacteria. Other environmental factors necessary for nitrification are sufficient ammonia loads, water temperatures greater than 20°C, and near neutral or slightly alkaline pH.

The environmental conditions necessary for nitrification in Cedar Creek were favorable during the two water-quality surveys. Cedar Creek is a shallow stream ranging in depth from 1.3 to 3.1 ft. Temperature in Cedar Creek ranged from 18° to 22°C for the August 1976 survey and from 24° to 27°C for the July 1977 survey. Values of pH were at or near neutrality for both surveys. Ammonia-nitrogen concentrations at the wastewater-treatment facilities ranged from 0.2 to 12.0 mg/L for the August 1976 survey. The ammonia-nitrogen concentration of the Tenth Street ditch (18-11) was 15 mg/L at the same time. For the July 1977 survey, the ammonia-nitrogen concentration at the wastewater-treatment facilities ranged from 1.8 to 8.2 mg/L. Enumerations of nitrifying bacteria were not made.

Ammonia-nitrogen concentration in Cedar Creek during the August 1976 survey ranged from 0.2 mg/L, downstream from the Waterloo wastewater-treatment facility at RM 26.42, to 1.8 mg/L upstream from the confluence with Little Cedar Creek at RM 10.78. Nitrite-nitrogen plus nitrate-nitrogen concentration in Cedar Creek ranged from 1.0 mg/L at station 18-01 (RM 27.02) to 7.8 mg/L downstream from the Auburn wastewater-treatment facility at station 18-15 (RM 20.21). The general increase in the instream concentration of nitrite nitrogen plus nitrate nitrogen with distance downstream can be attributed primarily to the wasteloads from the wastewater-treatment facilities.

Ammonia-nitrogen concentrations in Cedar Creek during the July 1977 survey were relatively low compared with those of the August 1976 survey. In 59 percent of the stations sampled, the ammonia-nitrogen concentration was less than the detection limit of the method used, 0.1 mg/L. The highest concentration of ammonia nitrogen, 0.5 mg/L, was observed downstream from the Auburn wastewater-treatment facility at RM 19.22. The concentration of nitrite nitrogen plus nitrate nitrogen in Cedar Creek ranged from 0.6 mg/L upstream from the confluence with St. Joseph River at station 18-32 (RM 0.83) to 3.8 mg/L at station 18-12 (RM 20.90).

The nitrification process in streams is difficult to model. One reason for this is that the rate of change of nitrogen compounds not only is dependent on nitrification but also is dependent on other processes 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. These include 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.

Nitrification rates downstream from wastewater-treatment facilities are usually determined on the basis of the rate of change in ammonia-nitrogen and (or) nitrate-nitrogen load and assumed first-order kinetics. However, in small surface-active streams, where the responsible organisms are nonplanktonic (stationary relative to streamflow, as cells fixed to streambed surfaces or associated with the sediment), a model based on zero-order kinetics would be appropriate (Finstein and others, 1978). In this circumstance, nitrification would begin immediately downstream from the outfall and would be independent of the ammonia-nitrogen concentration in the water column. If the responsible organisms were planktonic, nitrification would be delayed until the nitrifying population had expanded in response to the ammonia enrichment. Under zero-order kinetics, nitrification would result from the action of pregrown cells and would be independent of growth, except in the historical sense. In some circumstances, both planktonic and non-planktonic cells could contribute to the nitrification. However, Strom and others (1976) determined that the number of nitrifiers released from wastewater effluents was insufficient to exert an appreciable oxygen demand.

The decision to include nitrification in the model calibration was based on the analysis of data from stations 18-16 to 18-22. In this reach, the calculated dissolved-oxygen deficit due to CBOD could not account for the observed dissolved-oxygen sag downstream from the Auburn wastewater-treatment facility. Instream ammonia-nitrogen concentrations, though low, would not limit the nitrification process if the responsible organisms were non-planktonic. The reach from stations 18-16 to 18-22 was the only reach where the reduction in ammonia-nitrogen load equaled the increase in nitrate-nitrogen load.

For the model calibration, the deoxygenation rates for NBOD (K_n) downstream from the wastewater-treatment facilities were determined on the basis of the change in ammonia-nitrogen load and assumed zero-order kinetics.

$$K_{n,\ zero} = \frac{C_u - C_d}{t} \tag{14}$$

where

$K_{n,\ zero}$ is the zero-order deoxygenation rate for NBOD, in pounds per day per day,

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

and

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

The upstream ammonia-nitrogen load was determined by mass-balancing the ammonia-nitrogen load from stations 18-15, 18-16, 18-17, and 18-19.

$$C_u = \frac{(23\ ft^3/s)(0.1\ mg/L)+(2.6\ ft^3/s)(7.1\ mg/L)+(15\ ft^3/s)(0.1\ mg/L)+(1.8\ ft^3/s)(0.6\ mg/L)}{42.4\ ft^3/s}$$

$$C_u = 0.55\ mg/L = 118.2\ lb/d$$

The downstream ammonia-nitrogen concentration, C_d, was 0.1 mg/L (21.88 lb/d) at station 18-22.

The removal rate of ammonia nitrogen for the July 1977 survey in the reach from station 18-16 to station 18-22 was estimated to be 300 (lb/d)/d at 20°C.

$$K_{n,\ zero} = \frac{118.2\ \text{lb/d} - 21.88\ \text{lb/d}}{0.208\ \text{d}}$$

$$K_{n,\ zero} = 463.1\ (\text{lb/d})/\text{d} \qquad \text{at } 25°\text{C}$$

$$K_{n,\ zero} = 300\ (\text{lb/d})/\text{d} \qquad \text{at } 20°\text{C}$$

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

$$(K)_T \ \text{nitrogenous} = K_{20°C}(1.09^{T-20°C}) \tag{15}$$

where

(K) is the reaction constant, in day^{-1} or pounds per day per day,

T the temperature, in degrees Celsius,

and

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

An average first-order K_n of 0.3 day^{-1} was assumed in reaches downstream from the influence of wastewater-treatment facilities (Thomann, 1972, p. 97).

The same nitrification rates were used for both sets of water-quality data except at reach 3, downstream from the Waterloo wastewater-treatment facility at RM 26.75. For the July 1977 model calibration, the instream removal rate of ammonia nitrogen for this reach was calculated to be 33 (lb/d)/d at 20°C.

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 removal by biochemical oxidation. As a result, a benthic-oxygen demand would be expected to be significant, particularly during the August 1977 survey where values of K_r were highest.

In large rivers, benthic-oxygen demand can often be omitted as a significant oxygen demand. However, in shallow rivers and streams, benthic-oxygen demand can become one of the most significant factors affecting the dissolved-oxygen dynamics. For example, a benthic-oxygen demand of 4 $(g/m^2)/d$ (Thomann, 1972, p. 104), in a river 10 ft deep uses 1.3 (mg/L)/d dissolved oxygen. The same benthic-oxygen demand in a shallow 1-ft deep stream uses 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 during which settling may occur. 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 the level of accumulation and the rate of utilization. For short periods of time after a scouring event, sufficient to remove previous deposits, the level of accumulation would be low.

If the rate of settling is greater than the rate of utilization, CBOD will accumulate much faster than it is utilized. In addition, only the upper half inch of the benthic deposit is assumed to be active in the stream deoxygenation process. CBOD underlying this upper layer is utilized by anaerobic decomposition (Velz, 1970). Consequently, CBOD can be removed from the water without exerting an oxygen demand until it is resuspended during a scouring event.

Examination of daily discharge records for the U.S. Geological Survey gage at Cedarville showed that the August 11, 1976, survey was done only 3 days after a sizable storm event had subsided and that the July 20, 1977, survey was done during the receding stage of a small hydrograph. The calculated mean stream velocity exceeded 0.6 ft/s in all reaches during these events and possibly caused bed scour. Consequently, even though CBOD was being deposited on the stream bottom during both surveys, the oxygen demand from these deposits may have been reduced because of the short period of accumulation.

Benthic-oxygen demand was not measured during the two water-quality surveys.

For the model calibrations a benthic-oxygen demand of 0.3 $(g/m^2)/d$ was assumed for most of the modeled reaches. This value represents the lower limit of observed benthic-oxygen demand values reported by Butts and Evans

(1978) for streams in Illinois. Benthic-oxygen demand values used in the model calibrations that were greater than 0.3 $(g/m^2)/d$ represent the residual dissolved-oxygen deficit not accounted for by other processes affecting the dissolved-oxygen dynamics.

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 assume that gaseous exchange varies directly with stream velocity and inversely with stream depth and that reaeration increases with decreasing flow (Churchill and others, 1962; O'Connor and Dobbins, 1958; Owens and others, 1964; Thackston and Krenkel, 1969). 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, even at low flows. In addition, several investigators using the radioactive-tracer technique to determine reaeration have reported a strong correlation between reaeration and channel slope (Tsivoglou and Neal, 1976; Foree, 1976). Foree (1976), in a study of small streams in Kentucky, observed a general tendency for reaeration to decrease with decreasing specific discharge (discharge per unit area).

The equation used to predict reaeration in this study is the energy-dissipation model developed by Tsivoglou and Neal (1976, p. 2686).

$$K_a = 0.110\ SV, \quad \text{when } 1 \le Q \le 10\ ft^3/s \tag{16}$$

$$K_a = 0.054\ SV, \quad \text{when } 25 \le Q \le 3{,}000\ ft^3/s \tag{17}$$

where

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

0.110 and 0.054 the gaseous-escape coefficients for equations 16 and 17, respectively, in $feet^{-1}$,

S the stream channel slope, in feet per mile,

V the stream velocity, in miles per day,

and

Q the stream discharge, in cubic feet per second.

Grant (1976a), using the radioactive-tracer technique to determine reaeration in small streams in Wisconsin with flows less than 37 ft^3/s, determined a gaseous-escape coefficient of 0.081 $ft^{-1} \pm 0.014$ ft^{-1} at 20°C. This coefficient is in fair agreement with the gaseous-escape coefficient determined by Tsivoglou and Neal (1976) for small streams.

Wilson and Macleod (1974) concluded from a statistical evaluation of 16 predictive reaeration equations that equations containing slope give better results than equations based on velocity and depth. In a similar study, Rathbun (1977) compared 19 reaeration equations with reaeration rates determined directly and concluded that the energy-dissipation model of Tsivoglou and Neal (1976, p. 2686) is the best overall equation.

Reaeration rates were adjusted for temperature by the following equation:

$$(K)_T \text{ reaeration} = (K)_{20°C} \, (1.021^{T-20°C}), \qquad (18)$$

where

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

T the temperature, in degrees Celsius,

and

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

Photosynthesis

Net-photosynthetic oxygen production was evaluated by the upstream-downstream approach of Odum (1956) as modified by Stephens and Jennings (1976). The upstream-downstream method generates a relationship describing the rate of change in dissolved-oxygen concentration between two stream stations at discreet sampling intervals. The model assumes that oxygen is produced only during daylight hours and that any change in dissolved-oxygen concentration during this period, after correcting for diffusion, is due to photosynthetic-oxygen production. Any change in dissolved-oxygen concentration during hours of darkness, after correcting for diffusion, is attributed to respiration.

Average diel fluctuations of dissolved oxygen for main-stem sites were 1.1 and 1.6 mg/L for the August 1976 and July 1977 surveys, respectively. These fluctuations are significant because of the high rates of oxygen diffusion in shallow streams.

Calculated net-photosynthetic oxygen-production values for the two surveys are presented in table 9. Sufficient data were not available to evaluate all the stream reaches. However, several generalizations regarding the effect of photosynthesis on the oxygen dynamics of Cedar Creek may be made. The calculated values are similar to values reported by Hoskin (1959) for small streams in North Carolina.

Table 9.--Calculated net-photosynthetic oxygen production values for modeled stream reaches on Cedar Creek, Dekalb and Allen Counties, Ind.

Reach (river miles)	Calculated net-photosynthetic oxygen production [$(g/m^2)/d$]	
	August 11, 1976	July 20, 1977
24.81 - 24.04	------	-0.436
17.42 - 16.02	------	-1.842
16.02 - 13.64	------	-.647
14.36 - 10.78	-0.896	------
13.64 - 10.78	------	-.398
7.82 - 5.79	------	-.366
7.82 - 0.90	-.117	------
5.79 - 0.90	------	-.136

All the net-photosynthetic oxygen-production values determined for the two surveys were less than zero. Therefore, algal respiration must also be considered as one of the factors contributing to the oxygen deficit in Cedar Creek. O'Connell and Thomas (1965) indicated that negative net-photosynthetic oxygen-production values are common occurrence when the algal populations are nonplanktonic. The rise in respiration that seems to occur downstream from Auburn at RM 17.42 may be due to the increased organic matter and increased populations of respiratory organisms. This agrees with the theory that the respiratory metabolism of a polluted stream is proportional to the organic matter content at a rate determined by the deoxygenation constant (Streeter, 1935; Velz, 1939; and Phelps, 1944).

Net-photosynthesis was not included in all modeled reaches for the model calibrations. Net-photosynthesis, as in benthic-oxygen demand, was included in several reaches to show that it may be responsible for the dissolved-oxygen deficit not accounted for by carbonaceous and nitrogenous biochemical-oxygen demands.

Model-Calibration Results

The calculated and observed CBOD, NBOD, and dissolved-oxygen concentrations, and flows are illustrated in figures 16-23. Several observations of these results follow.

The calculated and observed CBOD concentrations for the July 1977 survey agree more closely than those for the August 1976 survey. A modified form of the normalized-mean error as defined by Wilson and MacLeod (1974) was used as the criterion for comparison. The equation is as follows:

$$\text{Normalized-mean error} = \left[\sum_{i=1}^{N} \frac{|A_{calc} - A_{obs}|}{A_{obs} N} \right] 100 \text{ percent} \qquad (19)$$

where

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

and

N the number of observations.

The absolute value of the error term was included to avoid situations where large positive and negative errors cancel and result in a small error term. The normalized-mean error was 4.9 percent for the July 1977 survey and 11.7 percent for the August 1976 survey. The large discrepancy between the observed and calculated CBOD at RM 0.77 for the August 1976 survey was probably due to algal respiration.

The normalized-mean error for NBOD was 7.7 percent for the July 1977 survey and 16.3 percent for the August 1976 survey.

The normalized-mean error for dissolved oxygen was 4.9 percent for the July 1977 survey and 2.5 percent for the August 1976 survey. The maximum calculated dissolved-oxygen deficit occurred downstream from Auburn. The calculated deficits are primarily due to the dissolved-oxygen concentration of the Auburn wastewater effluent, nitrification, and algal respiration. The current instream dissolved-oxygen standard was not violated during the two water-quality surveys.

Several factors may be responsible for the differences between calculated and observed values. Single samples collected in mid-channel may not have been representative of the average stream condition for the mixing zone downstream from wastewater-treatment facilities. For these reaches, samples should have been collected in equal-width increments across the stream transect and should have been composited so that the volume from each increment was proportional to the discharge. Other potential sources of error

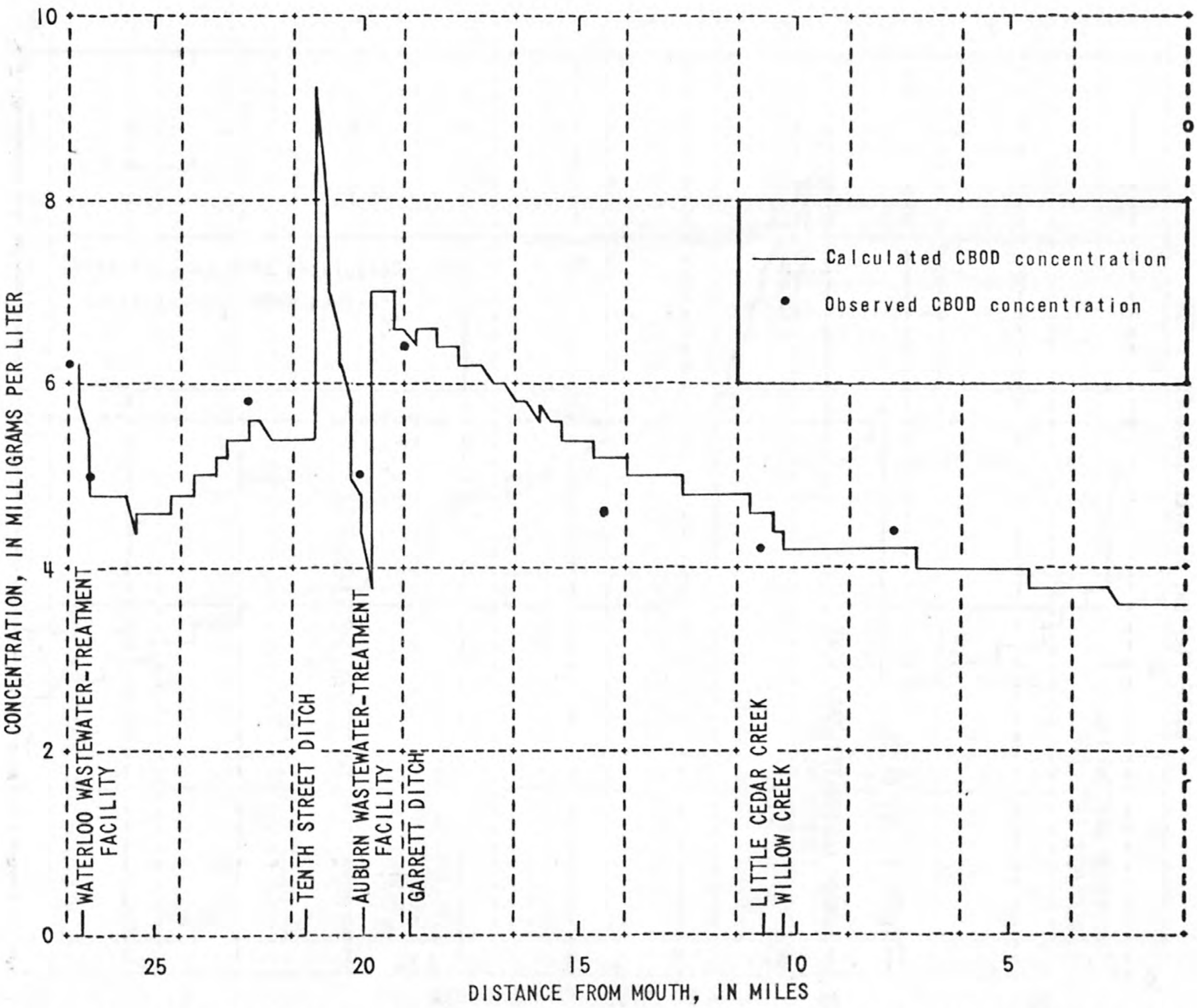

Figure 16.-- Relation of calculated and observed carbonaceous biochemical-oxygen demand concentrations to distance, Cedar Creek from Waterloo to mouth, August 11, 1976.

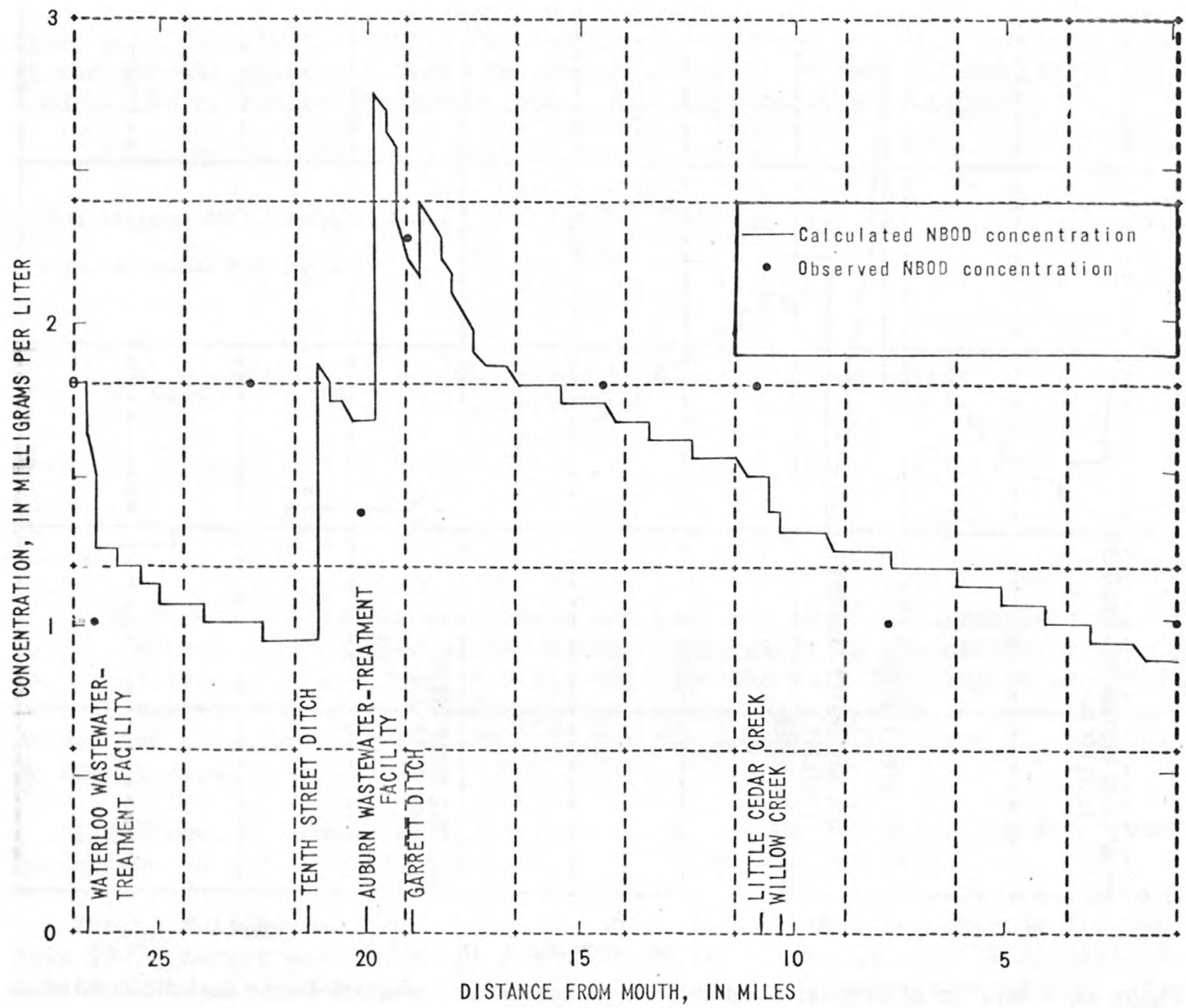

Figure 17.-- Relation of calculated and observed nitrogenous biochemical-oxygen demand concentrations to distance, Cedar Creek from Waterloo to mouth, August 11, 1976.

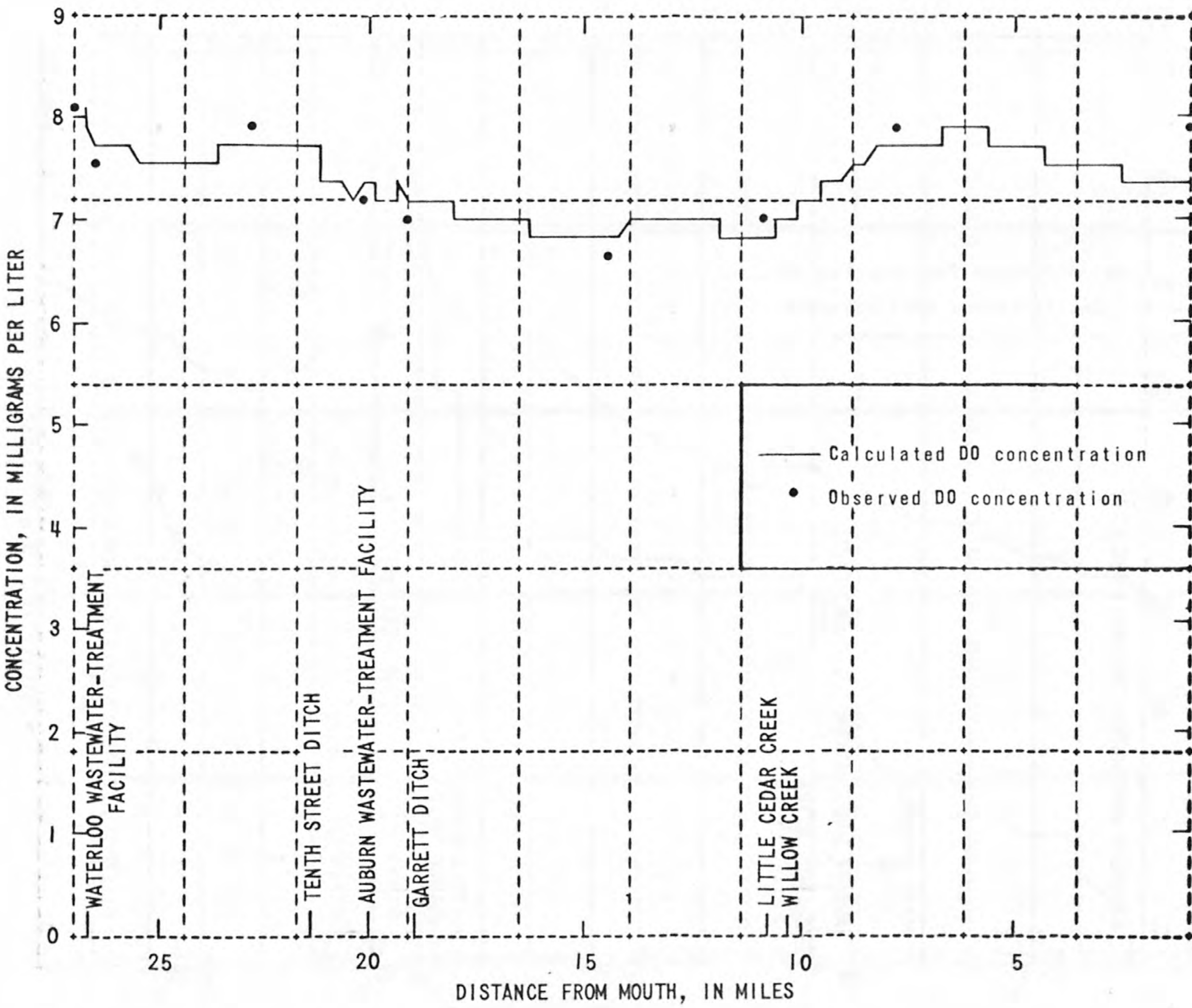

Figure 18.-- Relation of calculated and observed dissolved-oxygen demand concentrations to distance, Cedar Creek from Waterloo to mouth, August 11, 1976.

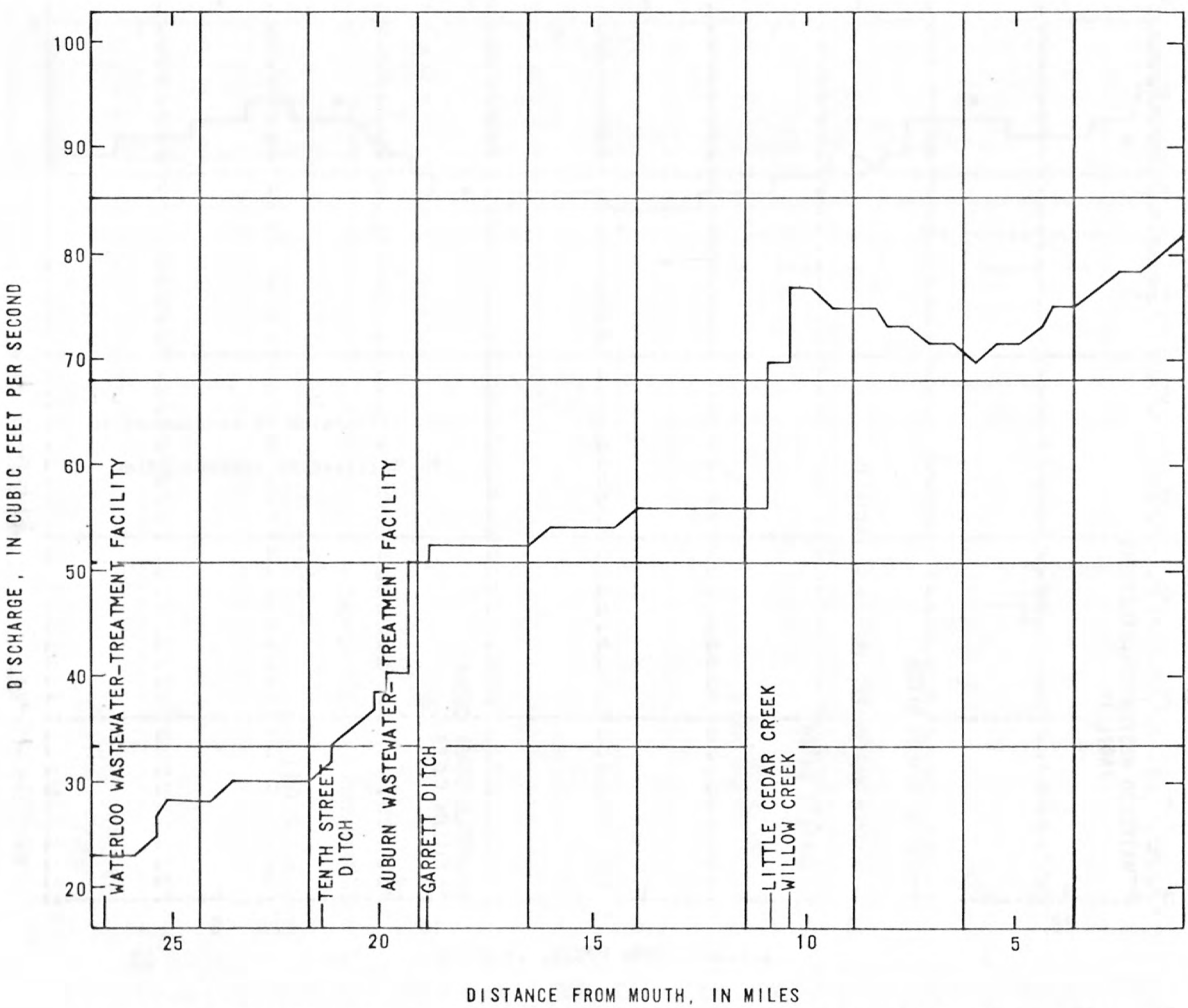

Figure 19.-- Relation of discharge to distance, Cedar Creek from Waterloo to mouth, August 11, 1976.

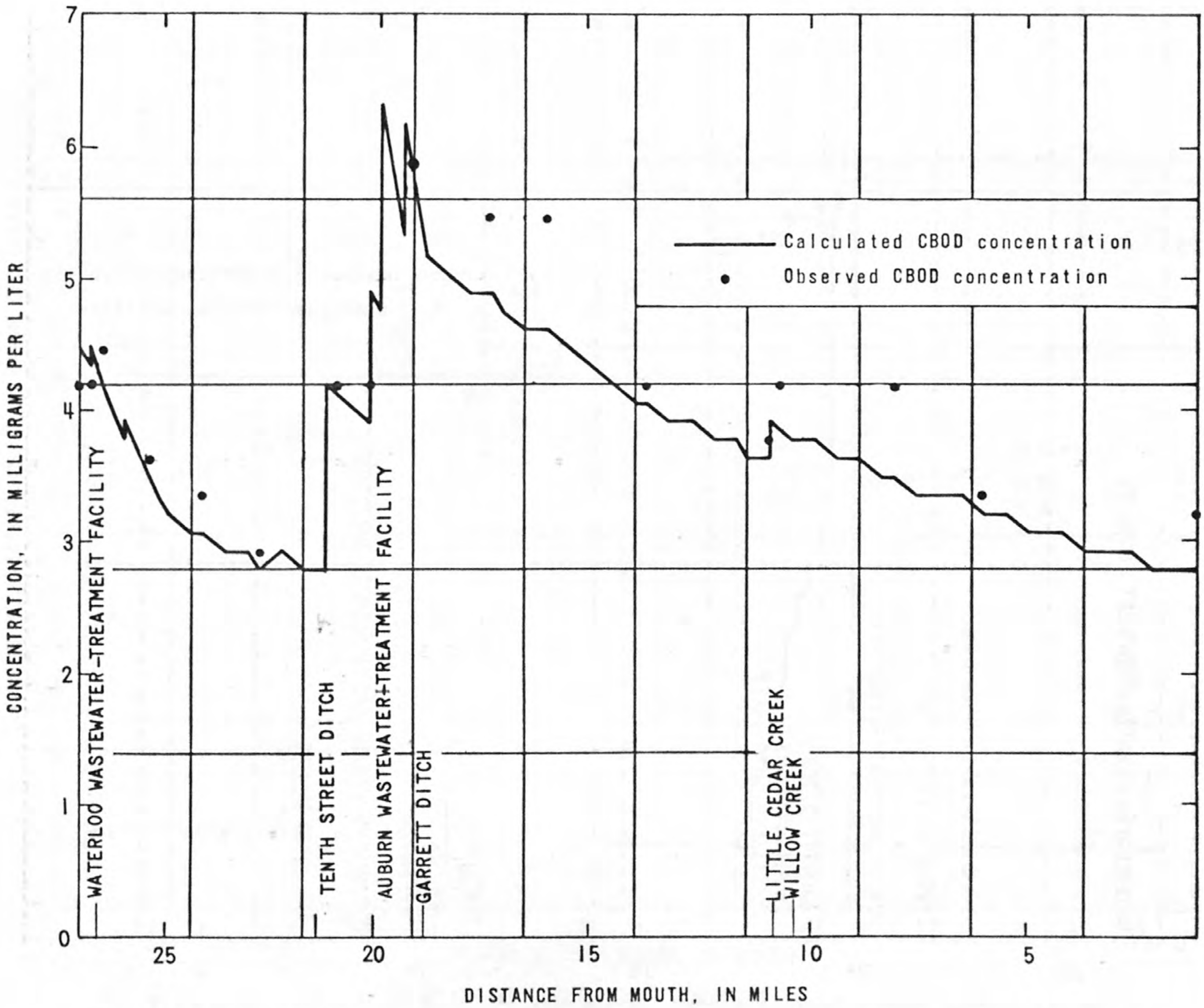

Figure 20. -- Relation of calculated and observed carbonaceous biochemical-oxygen-demand concentrations to distance, Cedar Creek from Waterloo to mouth, July 20, 1977.

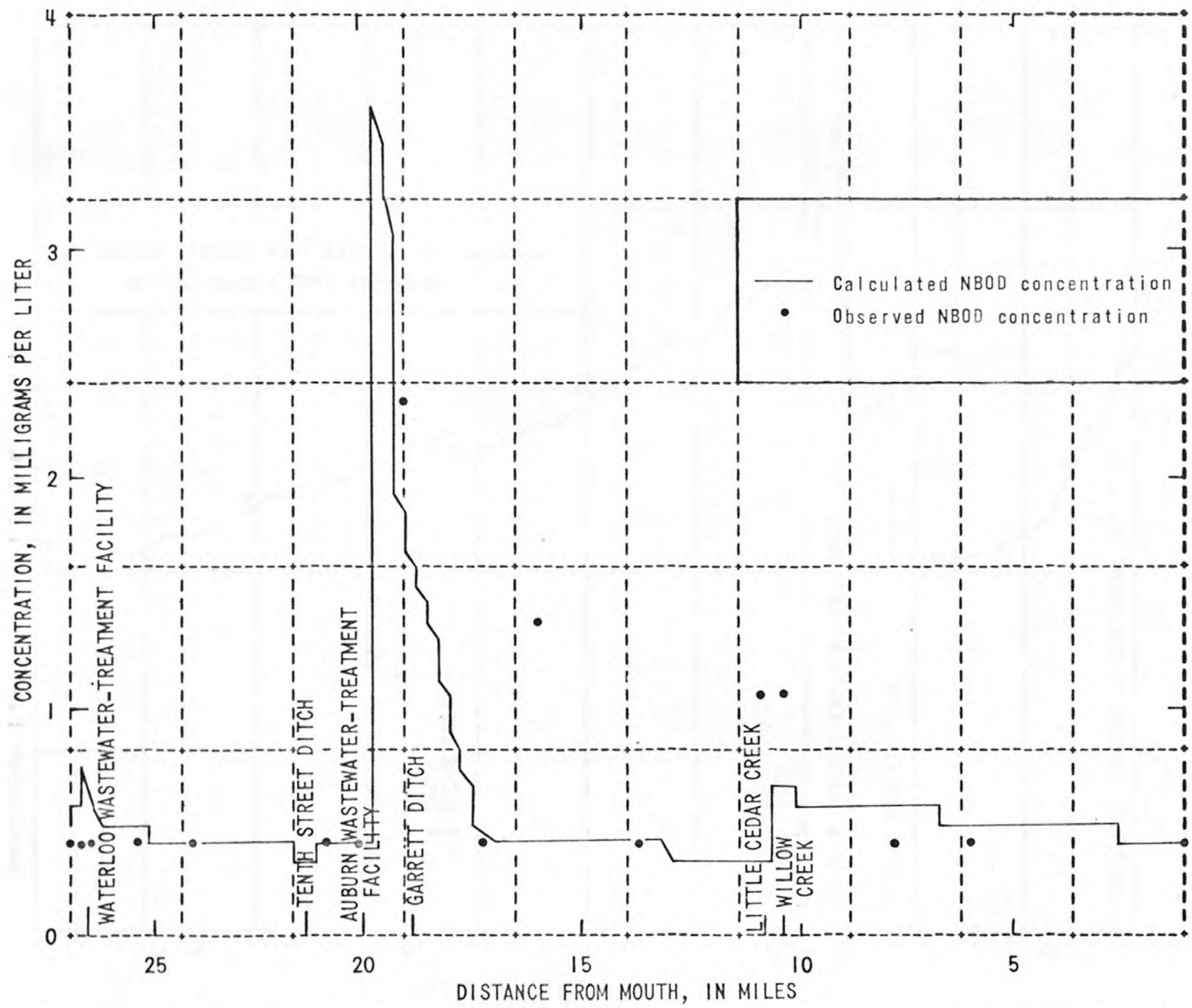

Figure 21.-- Relation of calculated and observed nitrogenous biochemical-oxygen demand concentrations to distance, Cedar Creek from Waterloo to mouth, July 20, 1977.

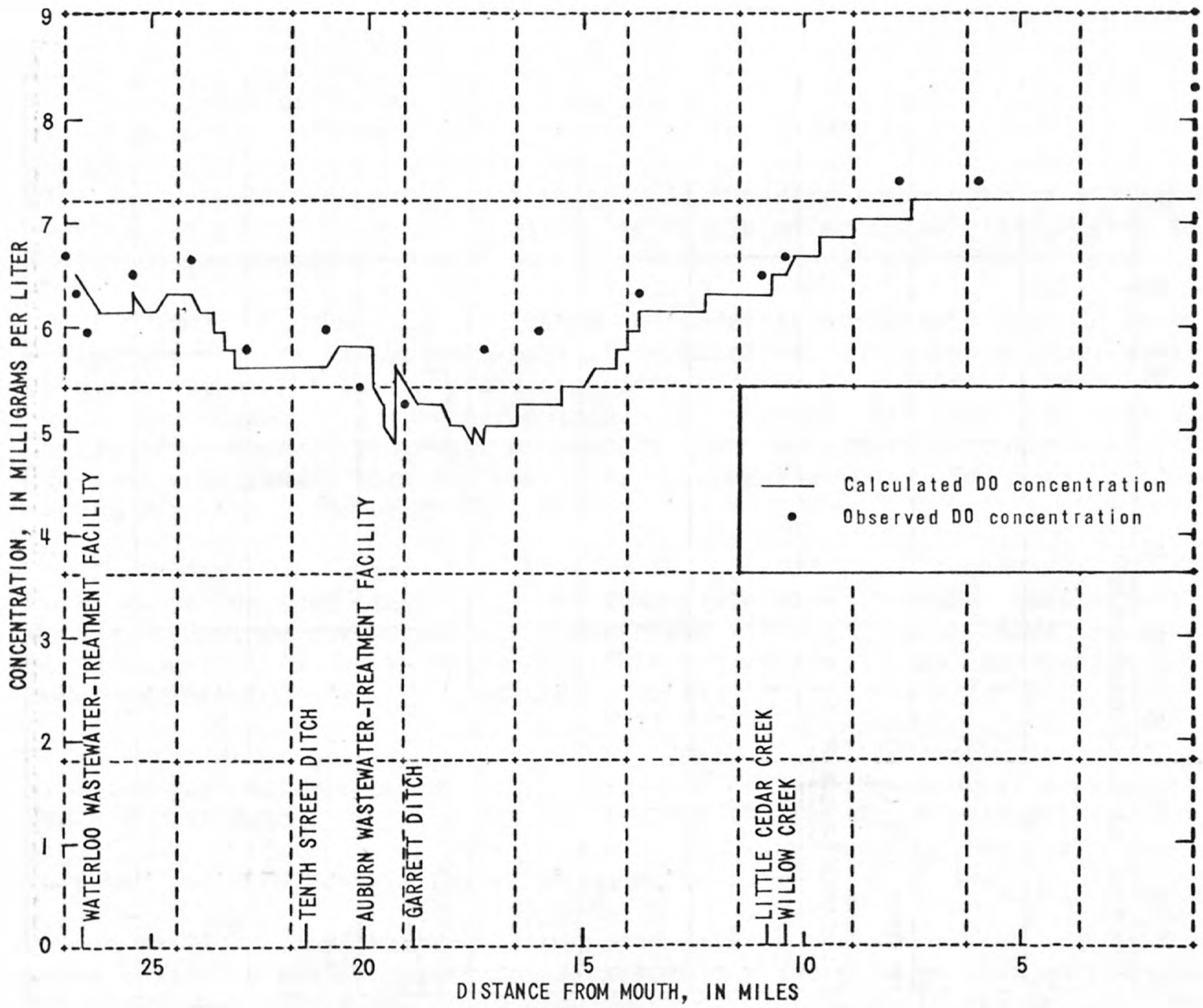

Figure 22.-- Relation of calculated and observed dissolved-oxygen concentrations to distance, Cedar Creek from Waterloo to mouth, July 20, 1977.

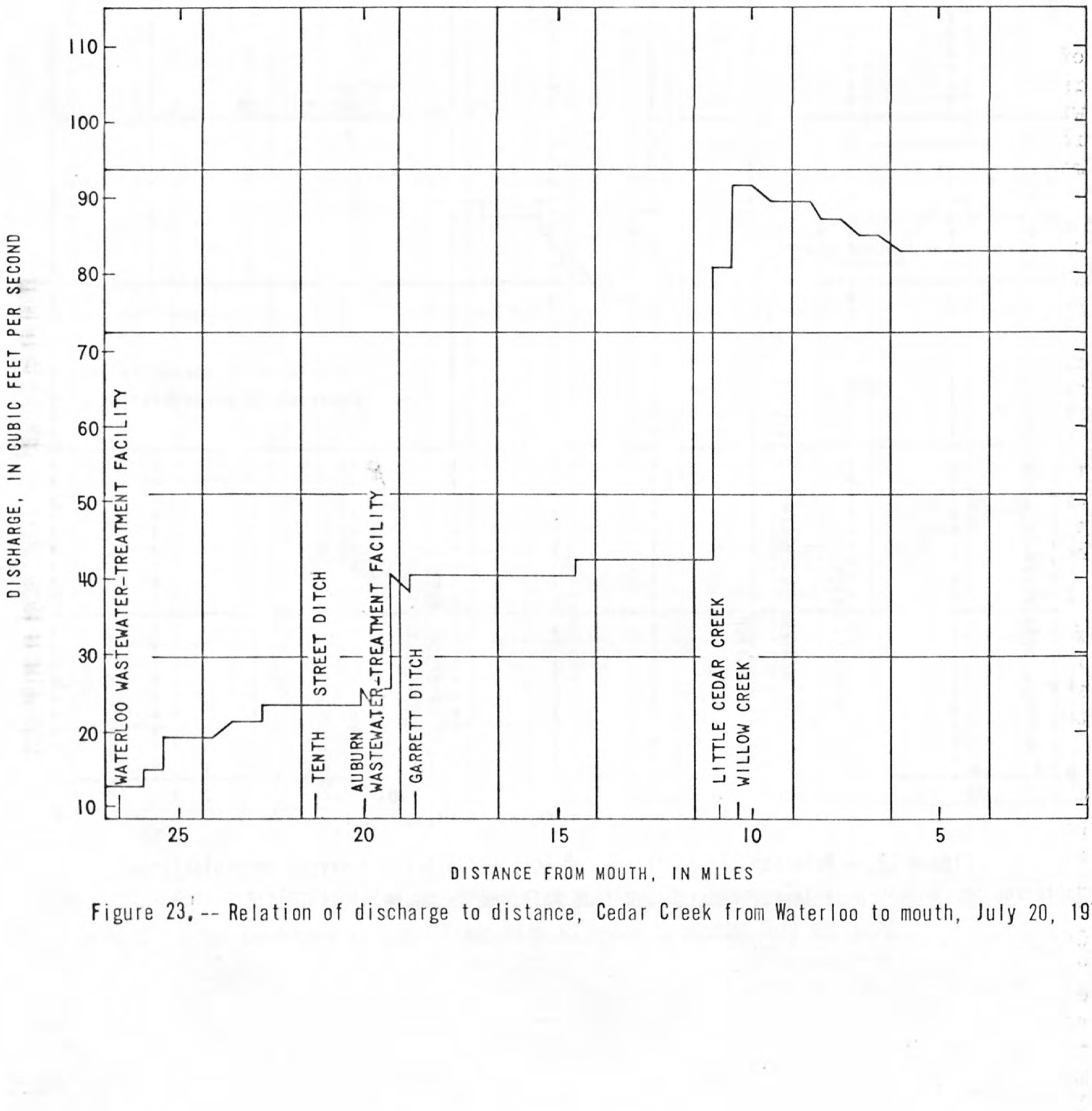

Figure 23.-- Relation of discharge to distance, Cedar Creek from Waterloo to mouth, July 20, 1977.

include failing to dechlorinate stream samples collected downstream from wastewater-treatment facilities before CBOD analysis and determining ultimate CBOD by different dilution ratios (Grant, 1976b, p. 6; Colston, 1975, p. 195).

MODEL VERIFICATION

The purpose of the model-verification process is to measure the degree of acceptability 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 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 Cedar Creek 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 budgeting constraints for this study, prohibited adequate data collection for model calibration during the critical low-flow period.

The two water-quality surveys were done with different waste loadings from the wastewater-treatment facilities and other point sources such as the Tenth Street ditch. For the Auburn wastewater-treatment facility there are significant differences between the waste loads reported by the ISBH Laboratory and the ISBH Monthly Report of Operations.

In small streams, where a large percentage of the streamflow is wastewater effluent, small changes in waste load may cause large changes in the processes that affect the dissolved-oxygen concentration in the stream. For example, the change in waste load from the Tenth Street ditch and the Auburn wastewater-treatment facility during the two surveys significantly affected the calculated values of K_r and K_d. Consequently, small streams such as Cedar Creek may need significantly more data to characterize the relationship between streamflow, temperature, wastewater discharge and the model coefficients than large rivers. The predictive capability necessary to describe the stream processes in Cedar Creek for a wide range of flow conditions has not been achieved. Consequently, the coefficients used in the model calibrations may not represent stream processes during the critical low-flow condition. The extrapolation of the calibration parameters to the low-flow $Q_{7,10}$ condition is an approximation of undetermined accuracy.

The Cedar Creek model cannot be considered verified at the low-flow $Q_{7,10}$ condition. However, lack of verification does not totally disqualify the Cedar Creek model for use in the waste-load assimilation analysis (in a qualitative sense). Because there is significant variability in several of the model parameters, conclusions from the waste-load assimilation studies were based on a sensitivity analysis of several of the critical parameters, particularly nitrification. The purpose of this analysis was to determine the significance of variations in these parameters on the predictions of the model.

WASTE-LOAD ASSIMILATION

Waste-load assimilation studies were made for both the summer (June-August) and winter (November-March) low flows to determine the combination of waste loadings that would meet the current Indiana water-quality standards for the years 1978, 1980, 1983, 1985, 1990, 1995, and 2000 (Indiana State Board of Health, 1977).

Procedures

Procedures used in establishing the waste-load assimilation capacity were furnished by the Indiana State Board of Health (1977).

In the NPDES permits (Indiana State Board of Health, 1978), daily maximum and daily average discharge loadings are given for industries and weekly maximum and monthly averages are given for the municipalities. (See table 1.)

To determine the waste-load assimilative capacity of the stream, as defined by the water-quality standards, the authors initially used maximum daily CBOD loadings (twice the monthly average) for the largest municipal discharger and average weekly loadings for the remainder of the municipal dischargers. For industries, the maximum daily CBOD loadings were used (Indiana State Board of Health, 1977). Where no CBOD permit had been issued, the combined data from the Monthly Report of Operations for June-August and data from the ISBH Water-Quality Surveillance Section were used to determine an appropriate value.

Effluent limits for ammonia nitrogen and dissolved oxygen have not been established for the municipalities and industries in the Cedar Creek basin. The concentration of ammonia nitrogen in wastewater effluents, observed during the two water-quality surveys, ranged from 0.2 to 12 mg/L. According to Metcalf and Eddy, Inc. (1972, p. 234), the ammonia-nitrogen concentration of untreated domestic sewage ranges from 12 to 50 mg/L. The amount of ammonia removal in conventional secondary treatment is usually small. Nitrification in the activated sludge process is usually insignificant because the detention times required for nitrifiers to proliferate are usually 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 was needed to ensure nitrification. The low effluent ammonia-nitrogen concentrations observed during the two surveys may have been due to dilution by ground water. As a result, the Indiana State Board of Health requested that 15 mg/L be used as the effluent ammonia-nitrogen concentration (65 mg/L NBOD) for the waste-load assimilation study. This value was assumed to be more representative of the effluent ammonia-nitrogen concentration during low flows than the ammonia-nitrogen concentration observed during the two water-quality surveys (Aolad Hossain, Indiana State Board of Health, oral commun., 1978). The determination of a representative ammonia-nitrogen concentration for each wastewater-treatment facility is critical to the waste-load assimilation study of Cedar Creek and will be needed in the future.

The observed dissolved-oxygen concentrations at the Auburn wastewater-treatment facility were low during the two water-quality surveys and averaged 30 percent of saturation. If the dissolved-oxygen concentrations of the wastewater effluents had been assumed to be this low for the waste-load 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 prior to discharge would be inexpensive compared to additional waste treatment. Consequently, the initial dissolved-oxygen concentration of the wastewater effluents was assumed to be 80 percent of saturation (Aolad Hossain, Indiana State Board of Health, oral commun., 1978).

Where the water-quality standards were violated, the CBOD and NBOD loads for the wastewater-treatment facilities were reduced until the appropriate standards were met. The determination of the combination of CBOD and NBOD loads that would just meet the Indiana water-quality standards defined the assimilative capacity of the stream.

Guidelines for the critical temperature and streamflow conditions used in the waste-load assimilation study were provided by the Indiana State Board of Health (1977). Summer $Q_{7,10}$ flows for stations on Cedar Creek at Auburn and Cedarville were used in the summer waste-load assimilation study and annual $Q_{7,10}$ flows (Rohne, 1972, p. 288) were used in the winter waste-load assimilation study. These flows are listed in the table on page 60.

A temperature of 22°C was used in the summer waste-load assimilation study. This temperature was based on the mean daily water temperature, which is exceeded 10-20 percent of the time for the months June through August. A temperature of 9°C was used in the winter waste-load assimilation study. This temperature was the average of the daily mean water temperatures for November, the month with the lowest flow.

Sampling station	Summer $Q_{7,10}$ discharge (ft^3/s)	Annual $Q_{7,10}$ discharge (ft^3/s)
USGS gage at Auburn (18-11)[1]	2.8	1.8
USGS gage at Cedarville (18-31)[1]	21.0	19.0

[1]Indiana State Board of Health sampling station number.

Temperatures for the Cedar Creek waste-load assimilation study were estimated by the following equation:

$$T = M + A\,[\sin\,(0.0172\,\underline{d} + C)] \quad \text{(Shampine, 1977, p. 13)} \qquad (20)$$

where

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

M is 11.48, the mean annual stream temperature, in degrees Celsius,

A is 10.78, the stream temperature amplitude, in degrees Celsius,

$\underline{d}$ is the Julian date,

and

C is 4.32, the angle-phase coefficient, in radians.

Concentrations of CBOD, NBOD, and dissolved oxygen for the headwaters and minor tributaries were based on data collected in July 1977. The initial upstream dissolved-oxygen concentration was assumed to be 7.4 mg/L, the ultimate CBOD was assumed to be 4.0 mg/L, and the initial NBOD was assumed to be 0.9 mg/L.

Reaction-rate coefficients for CBOD used in the waste-load assimilation study were the same as those used in the July 1977 model calibration analysis.

Sufficient data were not available to determine the relationship between streamflow and the deoxygenation rate for NBOD (K_n) downstream from wastewater-treatment facilities. Consequently, the waste-load assimilative capacity of Cedar Creek during summer low flows was estimated for four different rates of nitrification in the reaches downstream from wastewater-treatment facilities.

The first estimate of the stream waste-load assimilative capacity was based on the assumption that the nitrification process downstream from wastewater-treatment facilities follows zero-order kinetics. This assumption was also used in the model calibration. The removal rates of ammonia-nitrogen used in the calibration and the waste-load assimilation study were 300 (lb/d)/d at 20°C, downstream from the Auburn wastewater-treatment facility, and 300 and 33 (lb/d)/d at 20°C, downstream from the Waterloo wastewater-treatment facility. Because additional water-quality data, as well as nitrifier enumerations, are needed before the zero-order assumption can be verified, the waste-load assimilation capacity was also determined for first-order kinetics downstream from the wastewater-treatment facilities.

The second estimate of the stream waste-load assimilative capacity was based on a first-order reaction coefficient of 6.0 day^{-1} at 20°C, downstream from wastewater-treatment facilities. This value was based on data from the July 1977 survey.

For comparison, a third estimate of the waste-load assimilative capacity was determined by assuming a first-order reaction rate for nitrification of 0.6 day^{-1} at 20°C, downstream from wastewater-treatment facilities. This limit of the "normal range" for large rivers. The value was not based on data from the Cedar Creek surveys and may not be representative of nitrification rates for small streams.

An average first-order reaction rate for nitrification, 0.3 day^{-1} at 20°C, was assumed for reaches downstream from the influence of wastewater-treatment facilities (Thomann, 1972, p. 97).

During winter, nitrification was assumed to be negligible, owing to the inhibition of nitrifying organisms at low stream temperatures (Thomann and others, 1971). Consequently, for the winter waste-load assimilation study, ammonia nitrogen was treated as a nonbiodegradable constituent, and K_n was set equal to zero.

Reaeration rates were computed by the energy-dissipation equation of Tsivoglou and Neal (1976, p.2686) as discussed in the section "Reaeration."

All rates were corrected for temperature as discussed in the appropriate sections of "Parameter Estimation."

Benthic and algal effects were not included in the waste-load assimilation study.

Sufficient data were not collected to determine the waste-load assimilation capacity of Garrett ditch and Willow Creek. However, several assumptions had to be made in order to determine appropriate values for these inputs for the waste-load assimilation study of Cedar Creek.

For Garrett ditch (18-19) the following assumptions were made:

1) Natural background flow for Garrett ditch, upstream from the Garrett wastewater-treatment facility, was zero for both the summer and winter low flows.

2) Discharge from the Garrett wastewater-treatment facility was 1.2 ft^3/s (Indiana State Board of Health, 1978).

3) Time of travel from the Garrett wastewater-treatment facility to the confluence with Cedar Creek was 1.4 days.

4) Concentrations of ultimate CBOD and NBOD in the Garrett wastewater-treatment facility effluent were 20 and 65 mg/L, respectively.

5) K_r was 0.14 day^{-1}.

6) Reaction rates for nitrification were the same as those estimated for the reaches downstream from the Auburn wastewater-treatment facility.

On the basis of these assumptions, the concentrations of ultimate CBOD for Garrett ditch (18-19) were 16.2 and 9.4 mg/L for summer and winter low flows, respectively. The NBOD concentrations during summer low flows were 28.5 mg/L, for K_n = 0.6 day^{-1}; 0.4 mg/L, for K_n = 6.0 day^{-1}; and 0.4 mg/L, for zero-order kinetics. The NBOD concentration during winter low flows was 65 mg/L.

For Willow Creek and the Huntertown wastewater-treatment facility (18-28) the following assumptions were made:

1) Natural background flow for Willow Creek, upstream from the Huntertown wastewater-treatment facility, was 0.98 ft^3/s for both summer and winter low flows.

2) Discharge from the Huntertown wastewater-treatment facility was 0.22 ft^3/s (Indiana State Board of Health, 1978).

3) Time of travel from the Huntertown wastewater-treatment facility to the confluence with Cedar Creek was 5.2 days.

4) Concentrations of ultimate CBOD and NBOD in the Huntertown wastewater-treatment facility effluent were 60 and 65 mg/L, respectively.

5) Concentrations of ultimate CBOD and NBOD in the headwaters for Willow Creek were 4 and 0.5 mg/L, respectively.

6) K_r was 0.14 day^{-1}.

7) Reaction rates for nitrification were the same as those estimated for the reaches downstream from the Auburn wastewater-treatment facility.

On the basis of these assumptions, the concentrations of ultimate CBOD for Willow Creek at the confluence with Cedar Creek (18-28) were 6.6 and 9.4 mg/L for summer and winter low flows, respectively. The NBOD concentrations during summer low flows were 0.5 mg/L, for K_n = 0.6 day^{-1}; 0.0 mg/L, for K_n = 6.0 day^{-1}; and 0.0 mg/L, for zero-order kinetics. The NBOD concentration during winter low flows was 12.4 mg/L.

The effect of the Garrett wastewater-treatment facility on the dissolved-oxygen concentration in Cedar Creek during summer low flows was tested for a wide range of CBOD and NBOD loadings. Combinations of ultimate CBOD, as much as 60 mg/L, and NBOD, as much as 65 mg/L, were assumed for the effluent at the Garrett wastewater-treatment facility. A dissolved-oxygen sag in Cedar Creek did not result from these waste loadings.

As a result of these findings, the waste-load assimilative capacity downstream from Auburn could be determined independently of waste loads from Waterloo and Garrett.

Results and Discussion

According to the Indiana State Board of Health (1977), a part of the assimilative capacity of a stream reach should be reserved for future growth and development. Two modeling techniques are acceptable in this regard.

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 an interval of 5 years to the end of the planning period (year 2000) and used as input into the calibrated waste-load allocation model.

For this study, the second modeling technique was used. The ISBH projected waste loads and flows for the existing discharges to the year 2000 (table 10). These projections were used in the calibrated model, and the necessary concentrations were determined for the discharges to meet current water-quality standards. (See summary of computations in figs. 24 through 30 and table 11.) The goals were an average dissolved-oxygen concentration of 5.0 mg/L and maximum ammonia-nitrogen concentrations of 2.5 mg/L and 4.0 mg/L for the summer and winter low flows, respectively.

Several observations regarding the different estimates of the stream waste-load assimilative capacity should be made.

The model indicates that the dissolved-oxygen concentration of the Auburn wastewater effluent and nitrification are the most significant factors affecting the dissolved-oxygen concentration in Cedar Creek during summer low flows. The observed dissolved-oxygen concentration for the Auburn wastewater effluent was low and averaged 30 percent of saturation. If the dissolved-oxygen concentration of the wastewater effluent is this low during summer low flows, Cedar Creek will have virtually no additional waste-load assimilative capacity.

All four estimates of the stream waste-load assimilative capacity indicate that the projected nitrogenous biochemical-oxygen demand waste loads, from the Indiana State Board of Health, for wastewater-treatment facilities at Auburn and Waterloo will violate the State's instream dissolved-oxygen standard even if the effluent dissolved-oxygen concentration is raised to 80 percent saturation. This observation is based on the assumption that the ammonia-nitrogen concentration of the wastewater effluents is 15 mg/L, a value considerably higher than the ammonia-nitrogen concentration observed during the two water-quality surveys.

Carbonaceous biochemical-oxygen demand is less significant than nitrification in the dissolved-oxygen dynamics because most of the CBOD was estimated to have been removed through settling or some other process.

The following suggestions for future water-quality studies on Cedar Creek are intended to help verify the model and to clarify the extent to which various factors affect the dissolved-oxygen dynamics.

Additional water-quality data are needed to determine an appropriate ammonia-nitrogen concentration of wastewater effluents to be used in waste-load allocation.

Additional water-quality surveys should be made at steady-state, low-flow conditions on Cedar Creek, where collecting a greater number of water samples immediately downstream from wastewater-treatment facilities is emphasized. These samples should be collected in equal-width increments across the stream transect and should be composited so that the volume from each increment is proportional to the discharge. This type of sampling is preferable to grab-samples in situations, such as a mixing zone, where a sample collected in mid-channel may not be representative of the average stream condition. In particular, these surveys should address the fate of ammonia nitrogen and its effect on dissolved oxygen.

Table 10.--Population and flow projections through the year 2000 for municipalities in the Cedar Creek basin, Dekalb and Allen Counties, Ind.

[Vince Sommers, Indiana State Board of Health, written commun., 1978]

Input	Year						
	1978	1980	1983	1985	1990	1995	2000
Waterloo wastewater-treatment facility							
Population	1,776	1,977	2,025	2,116	2,280	2,452	2,640
Flow (Mgal/d)	0.175	0.198	0.202	0.212	0.228	0.245	0.260
Auburn wastewater-treatment facility							
Population	7,800	8,335	8,600	8,782	9,324	9,987	10,714
Flow (Mgal/d)	2.00	2.16	2.24	2.29	2.46	2.66	2.90
Garrett wastewater-treatment facility							
Population	4,812	4,864	4,925	5,001	5,148	5,304	5,474
Flow (Mgal/d)	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Huntertown wastewater-treatment facility							
Population	884	1,180	1,300	1,400	1,640	1,800	1,900
Flow (Mgal/d)	0.144	0.160	0.167	0.172	0.186	0.194	0.200

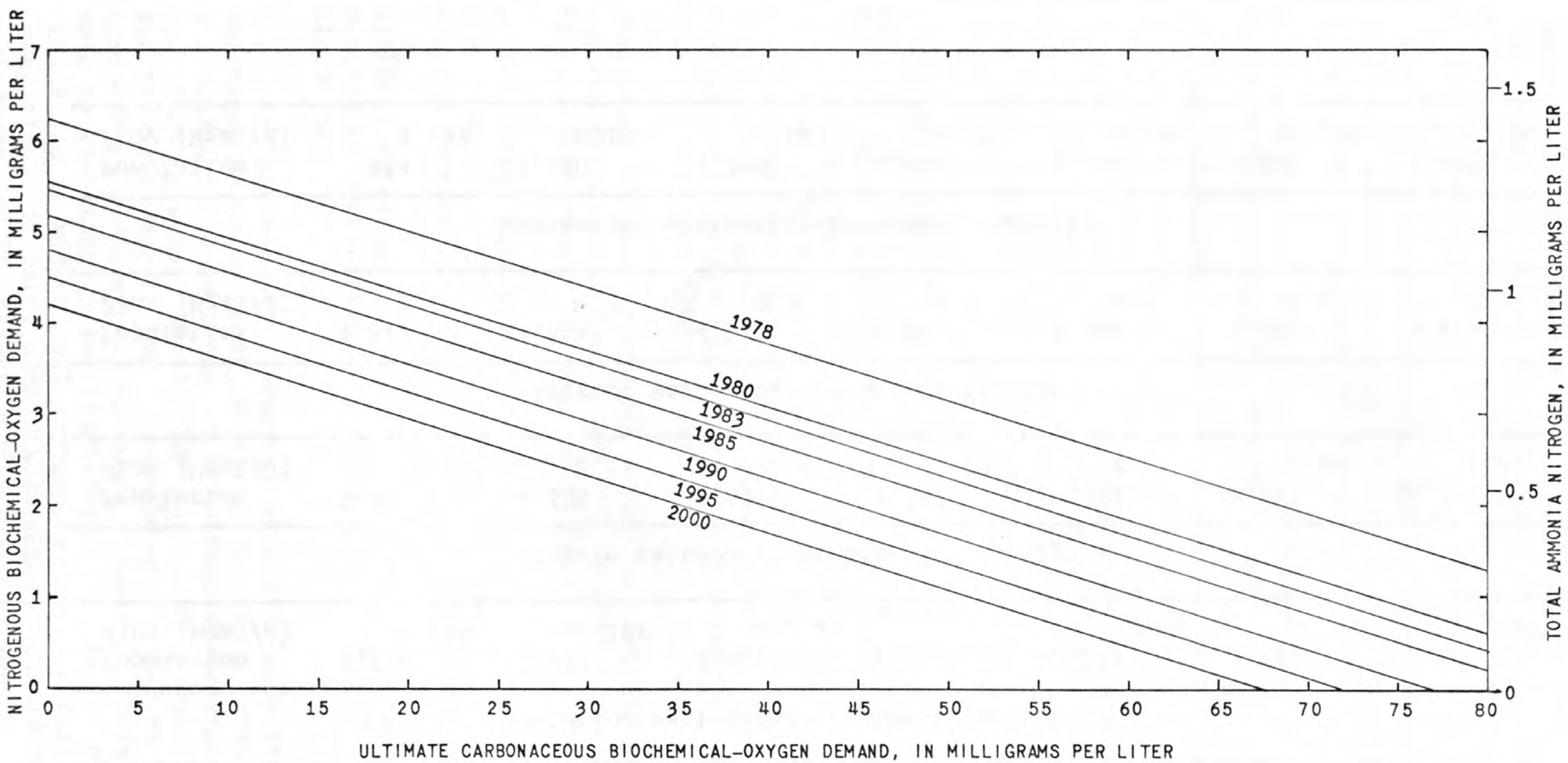

Figure 24.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Waterloo wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where the instream removal rate of ammonia nitrogen is 300 lb/day/day.

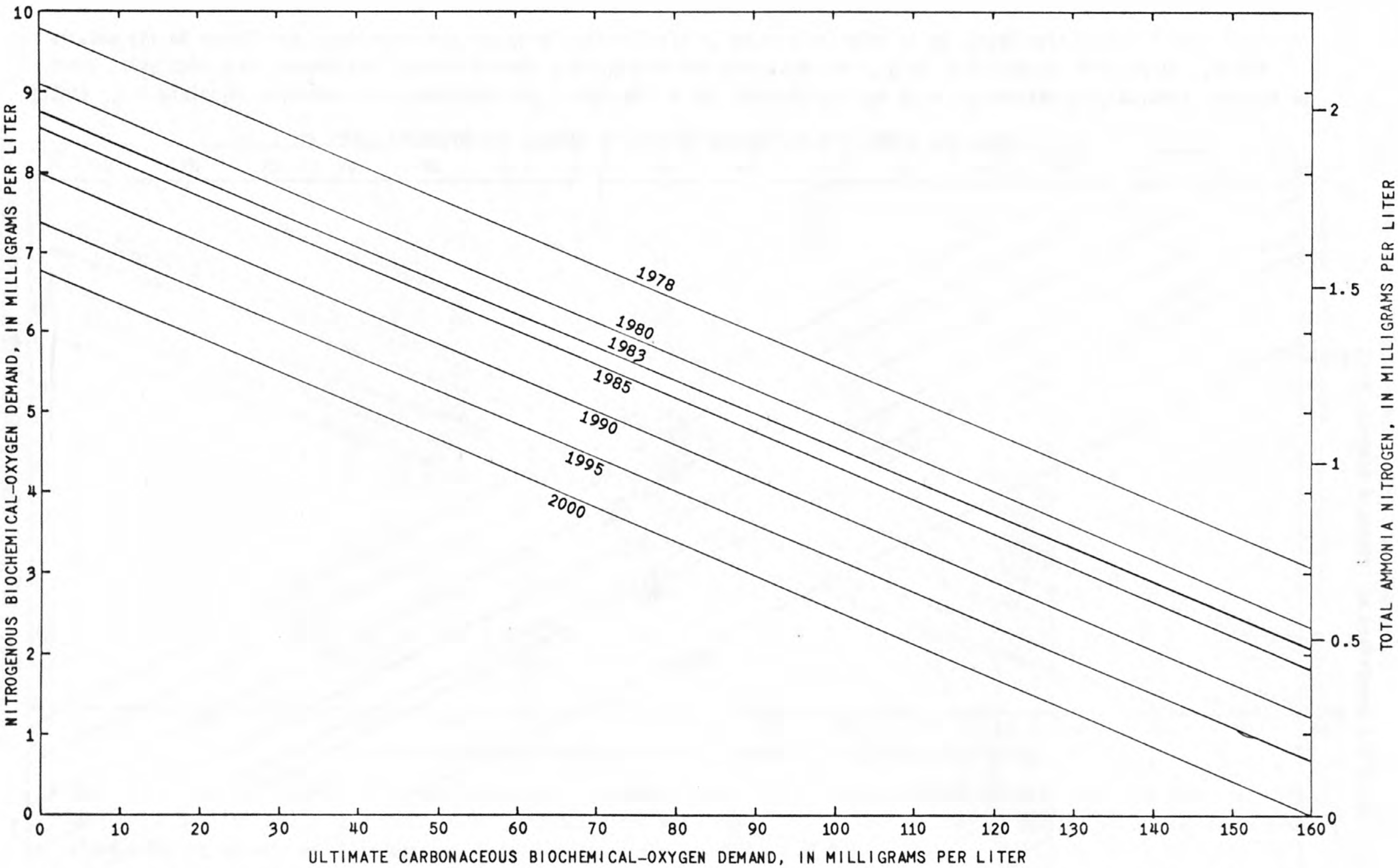

Figure 25.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Auburn wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where the instream removal rate of ammonia nitrogen is 300 lb/day/day.

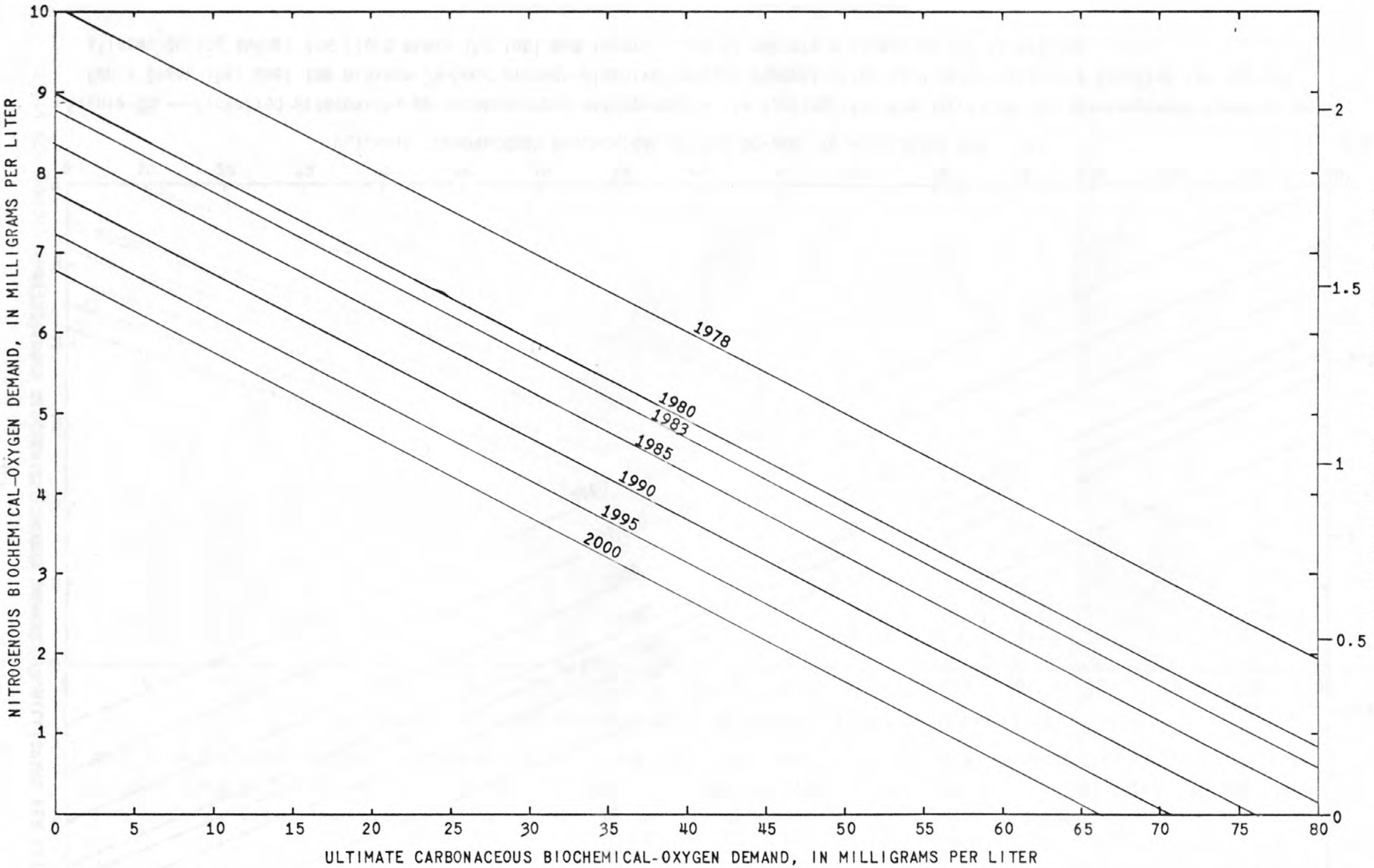

Figure 26.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Waterloo wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where the instream removal rate of ammonia nitrogen is 33 lb/day/day.

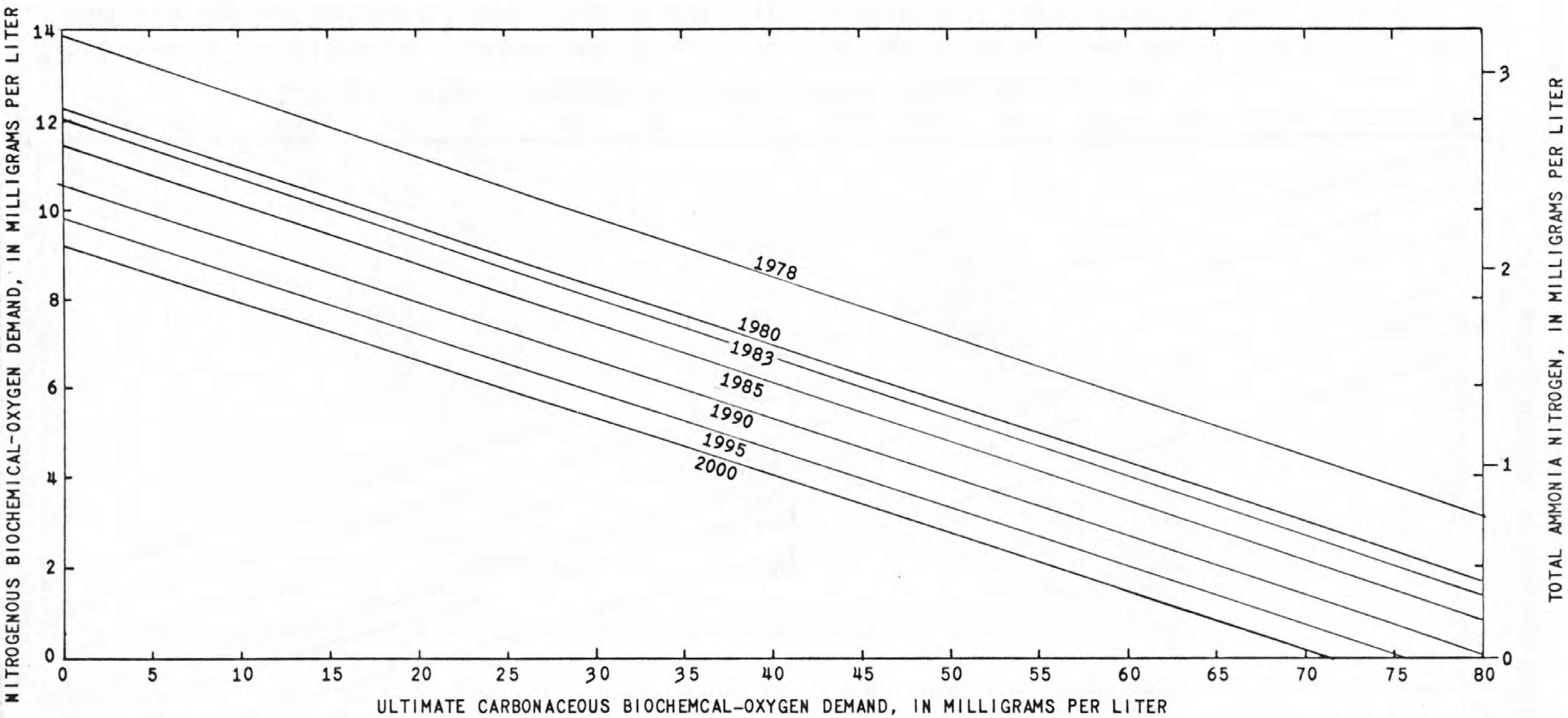

Figure 27.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Waterloo wasterwater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where K_n (deoxygenation rate for NBOD) is 6.0 day^{-1}.

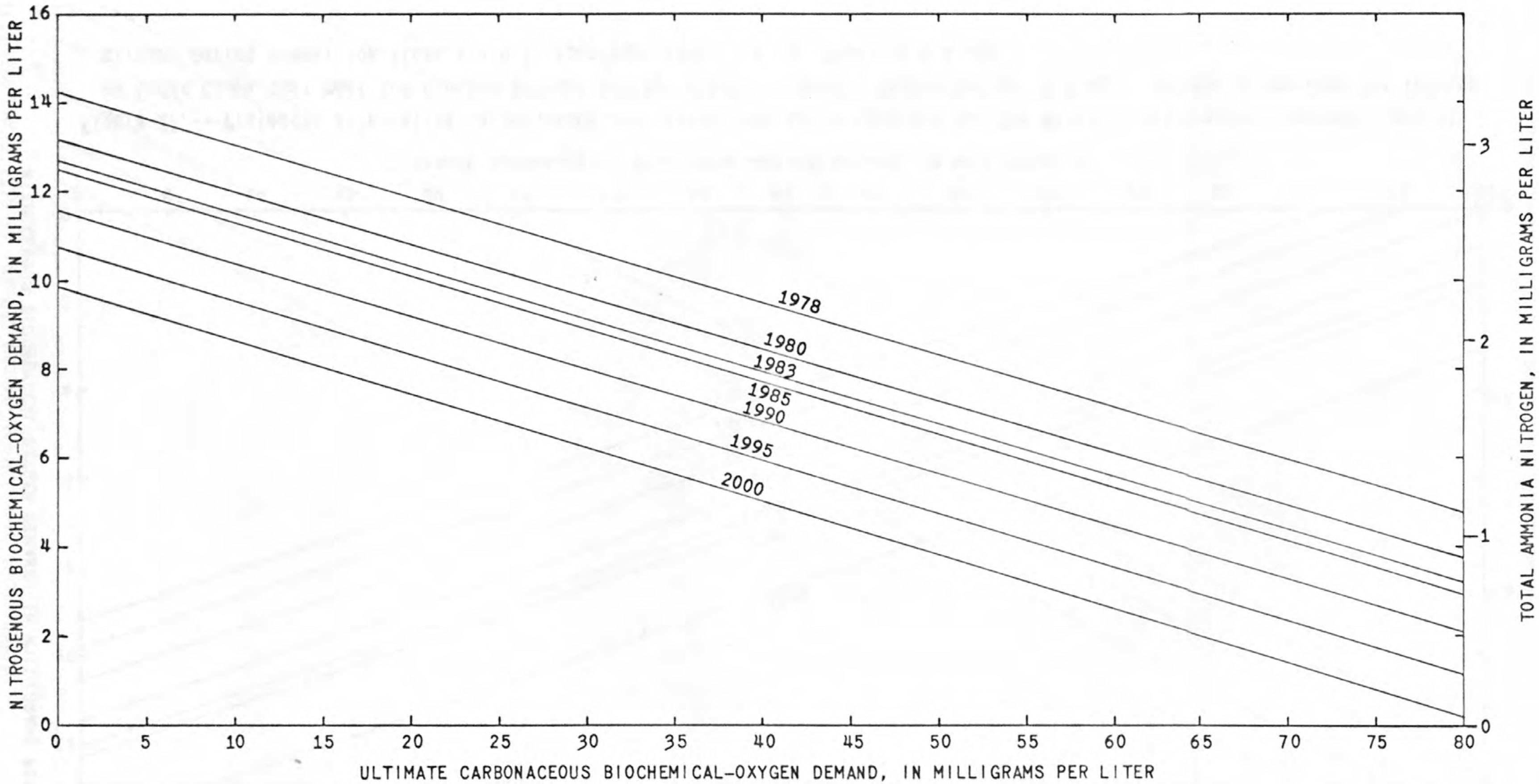

Figure 28.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Auburn wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where K_n (deoxygenation rate for NBOD) is 6.0 day^{-1}.

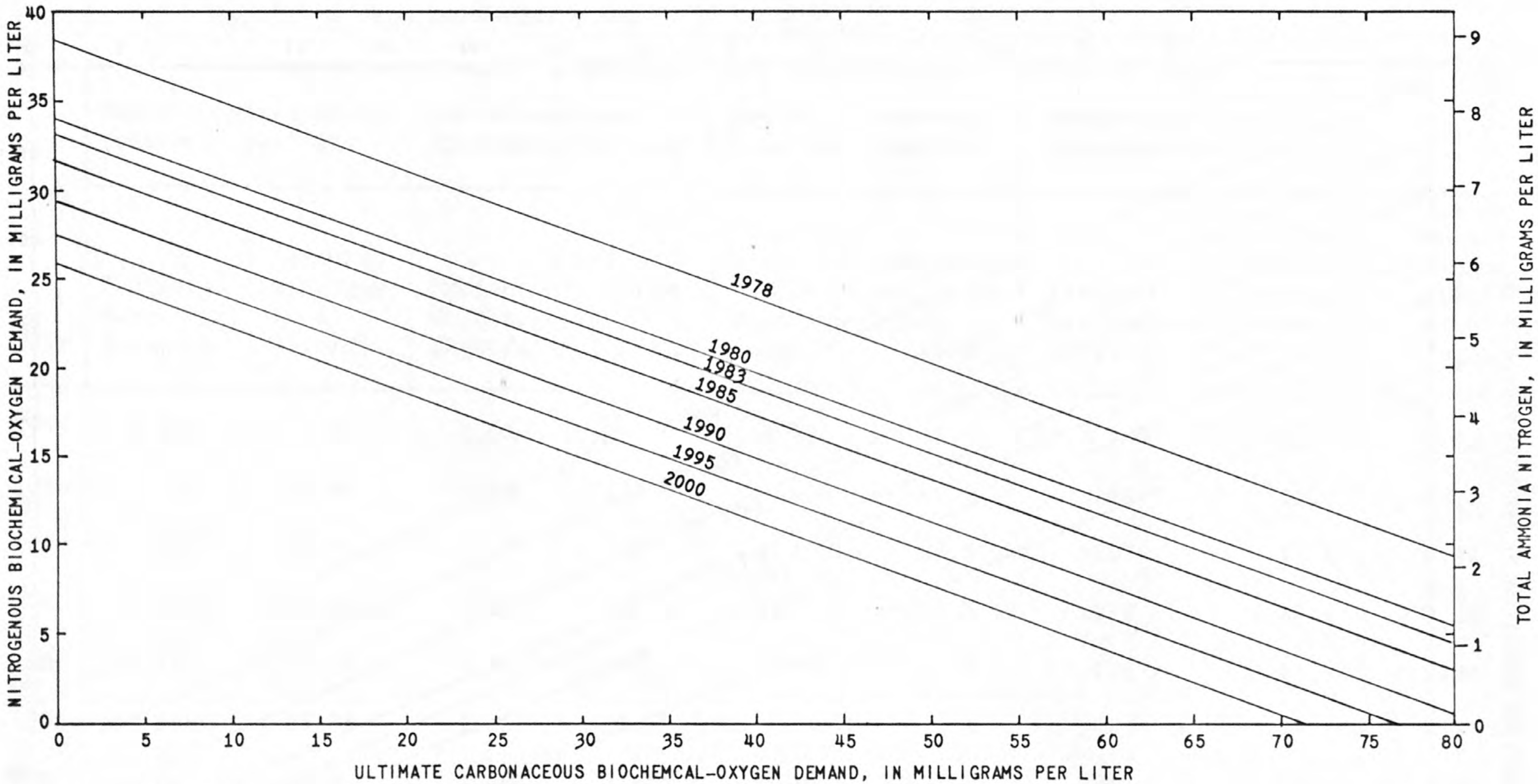

Figure 29.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Waterloo wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where K_n (deoxygenation rate for NBOD) is 0.6 day^{-1}.

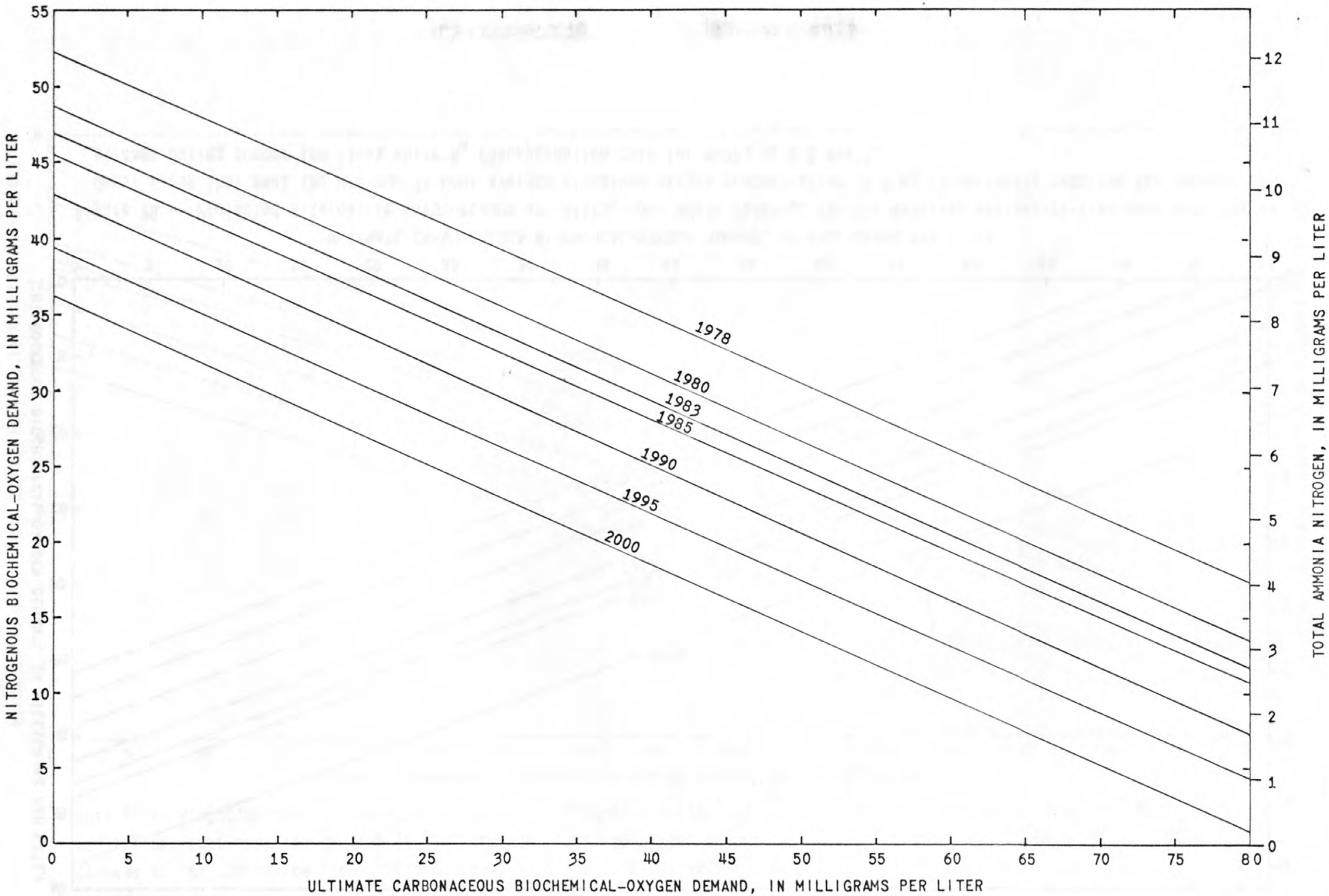

Figure 30.-- Projected alternative carbonaceous and nitrogenous waste loadings for the Auburn wastewater-treatment facility on Cedar Creek that meet the minimum 24-hour average dissolved-oxygen concentration (5.0 mg/L) currently required for Indiana streams during summer low flows, where K_n (deoxygenation rate for NBOD) is 0.6 day^{-1}.

Table 11.--Total ammonia-nitrogen loads that will result in an instream ammonia-nitrogen concentration of 4 mg/L in Cedar Creek during winter low flows at the Waterloo, Auburn, Garrett, and Huntertown wastewater-treatment facilities

	Waterloo wastewater-treatment facility		Auburn wastewater-treatment facility		Garrett wastewater-treatment facility		Huntertown wastewater-treatment facility		
Year	Projected discharge (Mgal/d)	Ammonia-nitrogen load (lb/d)	Projected discharge (Mgal/d)	Ammonia-nitrogen load (lb/d)	Projected discharge (Mgal/d)	Ammonia-nitrogen load (lb/d)	Projected discharge (Mgal/d)	Ammonia-nitrogen load (lb/d)	Ammonia-nitrogen concentration[1] (mg/L)
1978	0.175	14.49	2.00	165	0.800	66.3	0.144	11.9	9.93
1980	.198	15.80	2.16	172	.800	63.9	.160	12.8	9.57
1983	.202	15.90	2.24	176	.800	63.0	.167	13.1	9.44
1985	.212	16.48	2.29	178	.800	62.2	.172	13.4	9.32
1990	.228	17.19	2.46	185	.800	60.3	.186	14.0	9.04
1995	.245	17.88	2.66	194	.800	58.4	.194	14.1	8.75
2000	.260	18.32	2.90	204	.800	56.4	.200	14.1	8.45

[1] Ammonia-nitrogen concentration that each wastewater-treatment facility may discharge and that will produce a maximum instream ammonia-nitrogen concentration of 4.0 mg/L.

Finally, nitrifier enumerations should be made of both the bottom materials and the water column downstream from wastewater-treatment facilities to determine if sufficient populations are available to exert a significant oxygen demand.

SUMMARY AND CONCLUSIONS

A one-dimensional, steady-state, dissolved-oxygen model has been calibrated, and alternatives for waste-load allocation have been developed for Cedar Creek in Dekalb and Allen Counties, Indiana. The model indicates that the dissolved-oxygen concentration of the Auburn wastewater effluent and nitrification are the most significant factors affecting the dissolved-oxygen concentration in Cedar Creek during summer low flows. The observed dissolved-oxygen concentration of the Auburn wastewater effluent was low and averaged 30 percent of saturation. Whether or not the effluent is aerated before discharge will ultimately define the waste-load assimilative capacity of Cedar Creek. Projected nitrogenous biochemical-oxygen demand loads, from the Indiana State Board of Health, for the Auburn and Waterloo wastewater-treatment facilities will result in violations of the current instream dissolved-oxygen standard (5 milligrams per liter), even with an effluent dissolved-oxygen concentration of 80 percent saturation.

Ammonia toxicity is not the limiting water-quality criterion during summer low flows.

The time of travel for Garrett ditch and Willow Creek during summer and winter low flows is sufficient for these streams to assimilate waste loads from both the Garrett and Huntertown wastewater-treatment facilities.

The natural stream discharge upstream from the confluence of Willow and Little Cedar Creeks is small compared to the wastewater discharge, so benefits of dilution are minimal.

The stream reaeration capacity alone is not sufficient to maintain an average dissolved-oxygen concentration of 5 mg/L in Cedar Creek.

During winter low flows, ammonia toxicity rather than dissolved oxygen becomes the limiting water-quality criterion in the reach of Cedar Creek downstream from the Auburn wastewater-treatment facility and the confluence of Garrett ditch. The current CBOD limits for wastewater-treatment facilities do not exceed the instream dissolved-oxygen standard.

Critical ammonia-nitrogen concentrations were not predicted downstream from the Waterloo wastewater-treatment facility from 1978 through 2000.

The model indicates that future wastes in the Cedar Creek basin will probably have to be discharged into the reach downstream from RM 18.9. However, Cedar Creek downstream from County Road 68 (RM 13.64) to the confluence with the St. Joseph River is protected under Indiana's "Natural, Scenic and Recreational Water System," which mandates that the present water quality be maintained. Any future wastes discharged to Cedar Creek would be restricted to those that could be assimilated by RM 13.64.

The effect of storm-water discharge on water quality was not investigated. This information, as well as additional information on the process of nitrification, reaeration, and the effect of changing flow on these parameters, should be incorporated into future studies of the reach.

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