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APPENDIX C

DOWNSTREAM WATER QUALITY IMPACTS OF DIVERSIONS FROM SUDBURY RESERVOIR

Downstream Water Quality Impacts of Diversions from Sudbury Reservoir - Data Analysis and Model Calibration

prepared for

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Summary

A mathematical model is developed and tested for the purpose of predicting the potential downstream water quality impacts of diversions from Sudbury Reservoir. The model simulates spatial variations in important water quality components, including nutrients, algae, organic matter, dissolved oxygen, and coliform bacteria, from Framingham to Billerica. Model simulations and data analysis indicate that river water quality conditions are controlled primarily by the relatively low hydraulic gradient and interactions between the river and adjacent wetlands, especially during summer flooding events.

Water Quality Modeling

This section describes the technical bases for the assessment of the potential impacts of diversion from the Upper Sudbury on downstream water quality with respect to dissolved oxygen, algae, and fecal coliform bacteria, and temperature. The assessment is based upon analysis of historical monitoring data and upon a mathematical model which has been calibrated and applied to predict downstream water quality profiles for various base flows, temperatures, and diversion strategies. In accordance with the hydraulic analysis, the impact assessment is focused on the 51-kilometer portion of the Sudbury and Concord Rivers between MDC Dam Number 1 at Winter Street, Framingham, and Talbot Dam in Billerica. This section is organized according the following topics:

Data Compilation
Data Analysis
Model Description
Model Calibration

Results of model applications to assess potential diversion impacts are discussed in the Sudbury River Environmental Impact Report and summarized in Tables 7, 8, and 9 of this Appendix.

Data Compilation

An extensive effort was undertaken at the beginning of the study to compile and computerize pertinent hydrologic and water quality data from the study area, including:

(1) Monthly Hydrologic File: monthly mean discharges at three key gauging stations in the basin (Sudbury River at MDC Dam #1 in Framingham, Assabet River at Maynard, Concord River at Lowell), water years 1959 through 1981;

(2) Daily Hydrologic File: mean discharges at the above locations on and for two weeks preceeding the days of water quality sampling by the DEQE;

(3) Elevation Record File: measured elevations at Sherman Bridge and Stone Bridge, Wayland, and corresponding daily flows at the above three flow gauging stations;

(4) Concord DNR Water Quality File: monthly measurements at three

locations on the Assabet, Sudbury, and Concord Rivers conducted by the Concord Department of Natural Resources between 1973 and 1978;

(5) DEQE Water Quality File: data from intensive surveys conducted by the Massachusetts Division of Water Pollution Control, Department of Environmental Quality Engineering, during 1965, 1973, 1977, and 1979;

A station coding system based upon river kilometer has been designed to permit sorting and merging of DEQE water quality data from different survey periods. The station codes and river kilometer indices in Table 1 provide a frame of reference for interpretation of the plots discussed below. Climatologic data pertinent to the modeling effort have also been acquired and used in the analysis, but not computerized.

Data Analysis

The water quality data bases described above have been subjected to a variety of statistical and graphical analyses in order to develop some perspective on historical water quality conditions, spatial and temporal variations, and controlling factors. Key results are summarized below with respect to the variables of interest (oxygen, algae, and coliform bacteria).

Table 2 summarizes relevant hydrologic and climatologic data during DEQE monitoring periods. The most reliable water quality information comes from the intensive DEQE surveys in July 1973, August 1973, July 1979, and August 1979. These included three-day, diel sampling for dissolved oxygen and duplicate sampling for the other water quality variables. The remaining surveys employed single, grab-sampling only. The August 1979 data are not useful for modeling purposes because of wet weather (3.7-inch antecedent rainfall) and rising flow conditions described below.

Figure 1 depicts the relationship between USFWS elevation measurements at Sherman Bridge (RKM 36.1) and USGS flow measurements at Lowell (RKM 1.6). The correlation between these measurements is attributed to the backwater effects of Talbot Dam, an important hydrologic feature of the basin also indicated by HEC-II simulations. The plot and regression equation are based upon daily measurements during 1981 and 1982. Additional testing indicates that Sherman Bridge elevations are less strongly correlated with upstream flows (Saxonville or MDC#1) and that the residuals from the regression equation in Figure 1 are independent of the ratio of flow at MDC#1 to the flow at Lowell.

Hydrographs for the periods preceeding and during sampling are depicted in Figure 2. To provide some perspective on flow ranges, Figure 3 depicts seasonal variations in mean monthly flows measured by the USGS at Lowell between 1959 and 1981. The August 1973 survey was representative of summer, low-flow conditions, since the average discharge at Lowell was 171 cfs, compared with the median August flow of 115 cfs and the U.S. Fish and Wildlife Service summer criterion of 185 cfs (.5 cfs/mi2). The other surveys were conducted during relatively high flow (673 - 760 cfs) periods in relation to "normal" summer flow regimes.

Despite the fact that average flows were similar, hydrologic conditions varied considerably among the three high-flow surveys. As shown in Figure 2, the July 1973 and June 1979 surveys were conducted

Table 1
Stations and River Kilometer Indices - Sudbury/Concord Rivers

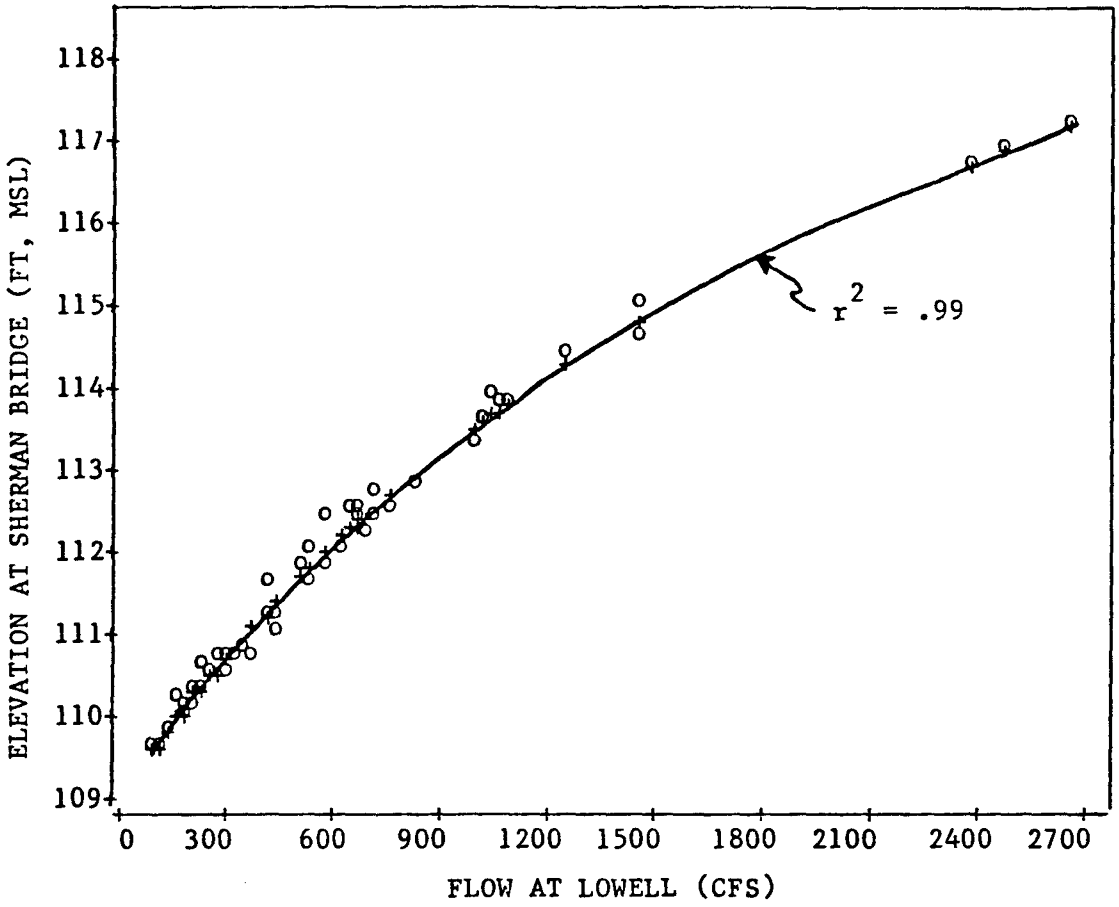
CODE	Location	Town	Mile	Kilometer
S01	Fruit St	Hopkinton	44.7	72.0
S02	Cordaville Road	Ashland	40.7	65.5
S03	Chestnut Street	Ashland	39.3	63.3
S04	Route 135	Ashland	38.9	62.6
S05	Winter Street, above Dam #1	Framingham	36.1	58.1
	MDC Dam Number 1	Framingham	36.1	58.1
S06	Central Street, first bridge	Framingham	34.2	55.1
S07	Central Street, above dam	Framingham	31.8	51.2
	Colonna Dam, Saxonville	Framingham	31.8	51.2
S08	Central Street, below dam	Framingham	31.8	51.2
S09	Elm Steet	Framingham	31.1	50.1
S10	Danforth Street Bridge	Framingham	30.1	48.5
S11	Stone Bridge Road	Fram./Wayland	30.0	48.3
S12	Pelham Island Road	Wayland	26.6	42.8
	Confluence Hop Brook	Wayland	26.2	42.2
S13	Route 20	Wayland	26.1	42.0
S14	Route 27	Wayland	25.1	40.4
S15	Lincoln Road/Sherman Bridge	Lincoln/Sudb.	22.4	36.1
S16	Route 117	Lincoln/Conc.	20.0	32.2
S17	Sudbury Road	Concord	17.7	28.5
S18	Nashawtuc Road	Concord	15.7	25.3
	Confluence Assabet	Concord	15.2	24.5
S19	Lowell Road	Concord	15.4	24.8
S20	Route 225	Carlysle/Bedfd	11.0	17.7
S21	River Street	Billerica	7.1	11.4
S22	Route 129	Billerica	6.0	9.7
S23	Pollard Street	Billerica	5.2	8.4
S24	Above Talbot Dam	Billerica	4.5	7.2
S25	Route 495	Lowell	2.5	4.0
S26	Lawrence Street	Lowell	1.6	2.6
S27	Rogers Street	Lowell	1.0	1.6
S28	Route 133	Lowell	0.6	1.0
S29	Route 38/110	Lowell	0.2	0.3
	Confluence Merrimack River	Lowell	0.0	0.0

Table 2
Summary of Hydrologic and Climatologic Data for
DEQE Intensive Survey Periods

Survey	1	2	3	4
Year	73	73	79	79
Month	7	8	6	8
Days	10-12	28-30	10-13	13-15
Mean Flows (cfs)				
Concord R. @ Lowell	760	171	673	690
Assabet R. @ Maynard	183	58	84	300
Sudbury R. @ MDC#1	102	24	156	145
5-Day Antecedent Precipitation (in)	.19	.10	.12	3.70
Water Temp (Deg. C)	24-25	24-25	19-21	17-18
Air Temp (Deg. C)	19-24	25-27	12-18	13-17
Cloud Cover (tenths)	.4-1.0	.1-.8	.2-.6	.6-.9

Figure 1

Elevation at Sherman Bridge vs. Flow at Lowell



Regression Equation based upon 1981-82 measurements:

$$E = 112.55 + 3.845E{-}03\ X - 1.216E{-}06\ X^2 + 2.482E{-}10\ X^3$$

$$X = Q - 700$$

$R^2 = .992$, $SE^2 = .031$, $n = 55$

where,

E = Elevation at Sherman Bridge (ft, msl)

Q = Flow at Lowell (cfs)

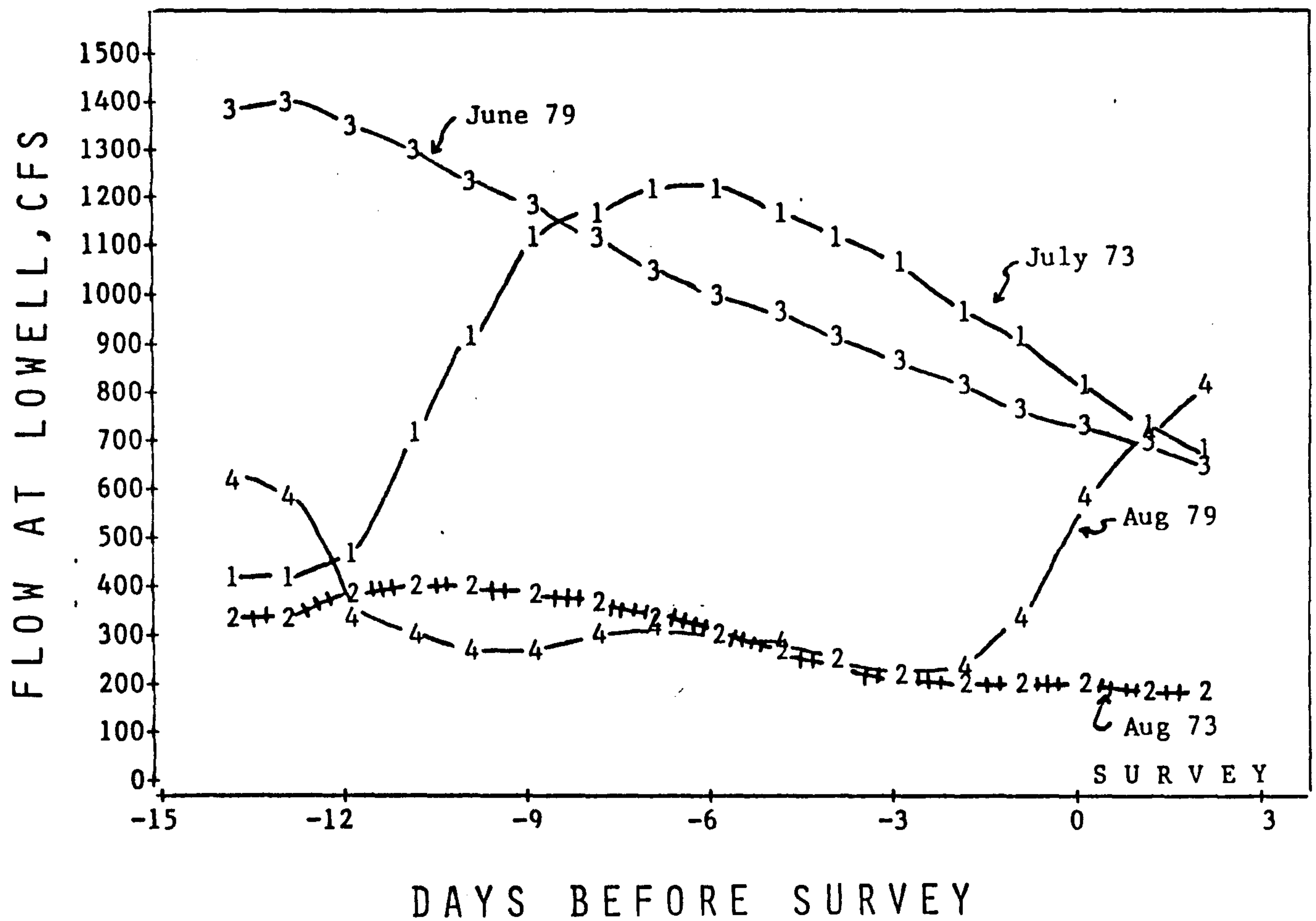
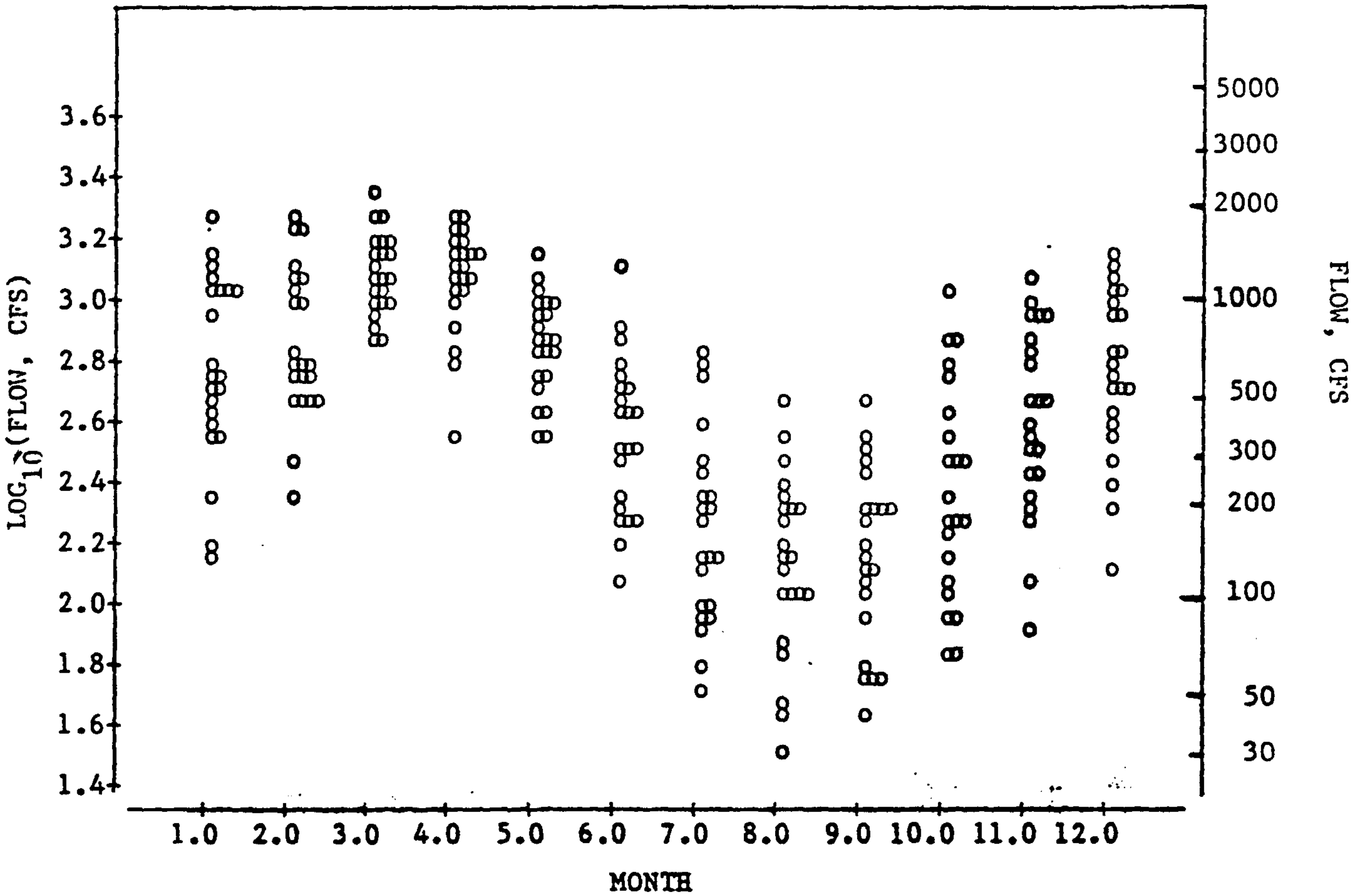


Figure 2

River Hydrographs for DEQE Intensive Monitoring Periods

Figure 3

Distribution of Mean Monthly Flows Measured by the USGS at Lowell, 1959-1981



during periods of falling discharge, and the August 1979 survey, during a period of rising discharge, attributed to a 3.7-inch rainstorm which began 3 days prior and ended on the second day of sampling. The major hydrologic difference between the July 1973 and June 1979 surveys is that the former followed a major summer storm, while flows were decreasing seasonally during the latter. The peak daily flow of 1200 cfs prior to the July 1973 survey was exceeded only three other times during July between 1937 and 1981. These hydrologic differences have important effects on water quality conditions because of the wetland interactions discussed below.

Spatial variations in daily mean and daily minimum dissolved oxygen concentrations are plotted in Figure 4 and 5, respectively. Violations of the 5 mg/liter Class B criterion are indicated for three out of the four DEQE surveys based upon mean dissolved oxygen. All surveys indicated violations between Pelham Island Road (RKM 42.8) and Nashawtuc Road (RKM 25.3), just above the confluence of the Assabet, based upon daily minimum concentration. This section of river is characterized by a low elevation gradient and large areas of adjacent wetlands.

Dissolved oxygen levels were clearly lower during the July 1973 sampling. Diel oxygen fluctuations below Stone Bridge Road (RKM 48.3) were also low compared with those measured during the other surveys, indicating suppression of photosynthesis. These severe conditions can be attributed primarily to death and decay of flooded wetland vegetation and subsequent loading of organic materials (oxygen demand) during the falling limb of the storm hydrograph. Correspondence on file with the Massachusetts Division of Fisheries and Wildlife contains evidence of even more severe water quality conditions occuring during 1938, following a larger July flood (3710 cfs). While the July 1973 and June 1979 flows were similar, the latter occurred relatively early in the growing season, when the surrounding wetlands would tend to be dominated by flood-tolerant vegetation and accumulated biomass would be relatively low.

Additional perspectives on oxygen variations in the Sudbury are provided by Figures 6, 7, and 8. These plots are based upon oxygen and temperature data taken at the Route 117 bridge (RKM 32.2) by the Concord DNR, between 1973 and 1978, and by the Mass. DEQE, between 1973 and 1979. Figure 6 shows dissolved oxygen concentrations at this location as a function of Concord River flow at Lowell. As described above, the elevation and morphometry of the Sudbury below Stone Bridge Road (RKM 48.3) are controlled more by the flow at Talbot dam (6 kilometers above the USGS gauging station at Lowell) than by the inflow at MDC#1 because of backwater effects. During summer months, oxygen levels are negatively correlated with flow. This probably reflects increasing wetland impacts during summer high flows. Increased algal and/or aquatic plant photosynthesis during low flows is probably another contributing factor, since the diel oxygen fluctuation measured during the DEQE low-flow survey of August 1973 was relatively large (4 to 13 mg/liter). A similar flow/concentration relationship is not apparent for October-May samples, which are consistently above the 5 mg/liter criterion. This presumably reflects lower water temperatures and less biological activity during the fall, winter, and spring months.

Corresponding plots of dissolved oxygen deficit vs. flow are given in Figure 7. The oxygen deficit is computed as the saturation concentration of oxygen at the river temperature minus the observed oxygen concentration. The oxygen deficit reflects the balance between oxygen inputs (attributed to aeration and photosynthesis) and outputs

Figure 4

Spatial Variations in Daily Mean Oxygen Concentrations
During DEQE Intensive Monitoring Periods

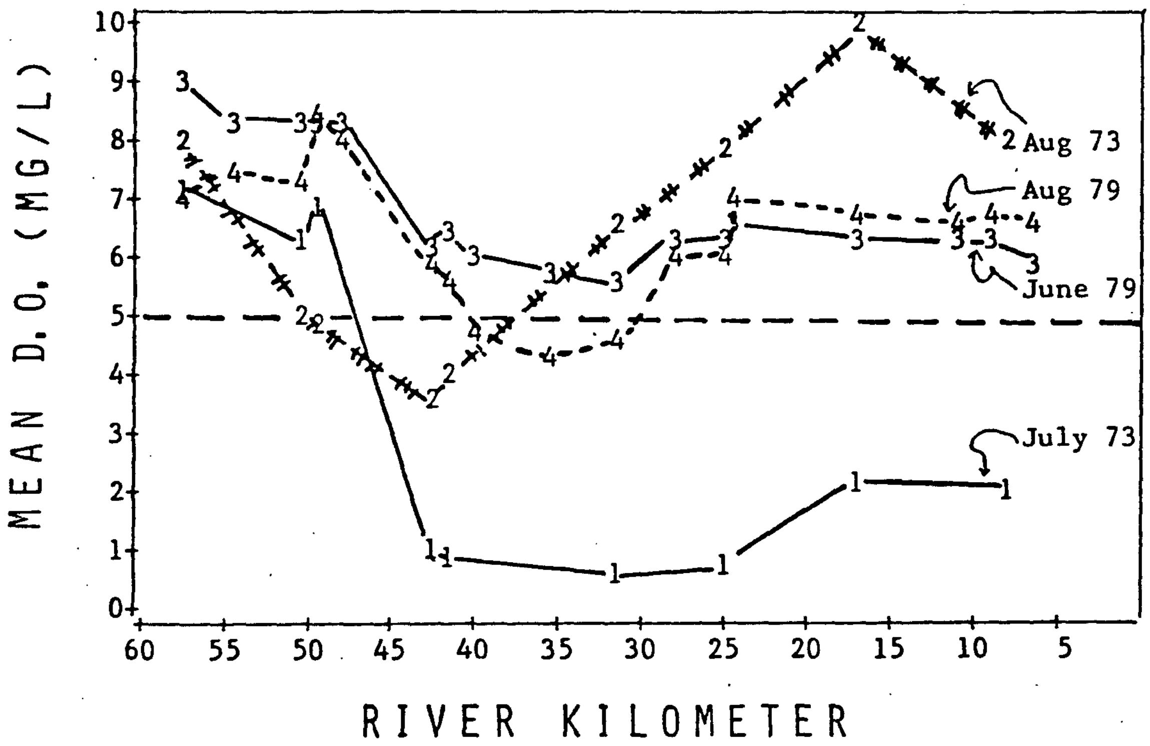


Figure 5

Spatial Variations in Daily Minimum Oxygen Concentrations
During DEQE Intensive Monitoring Periods

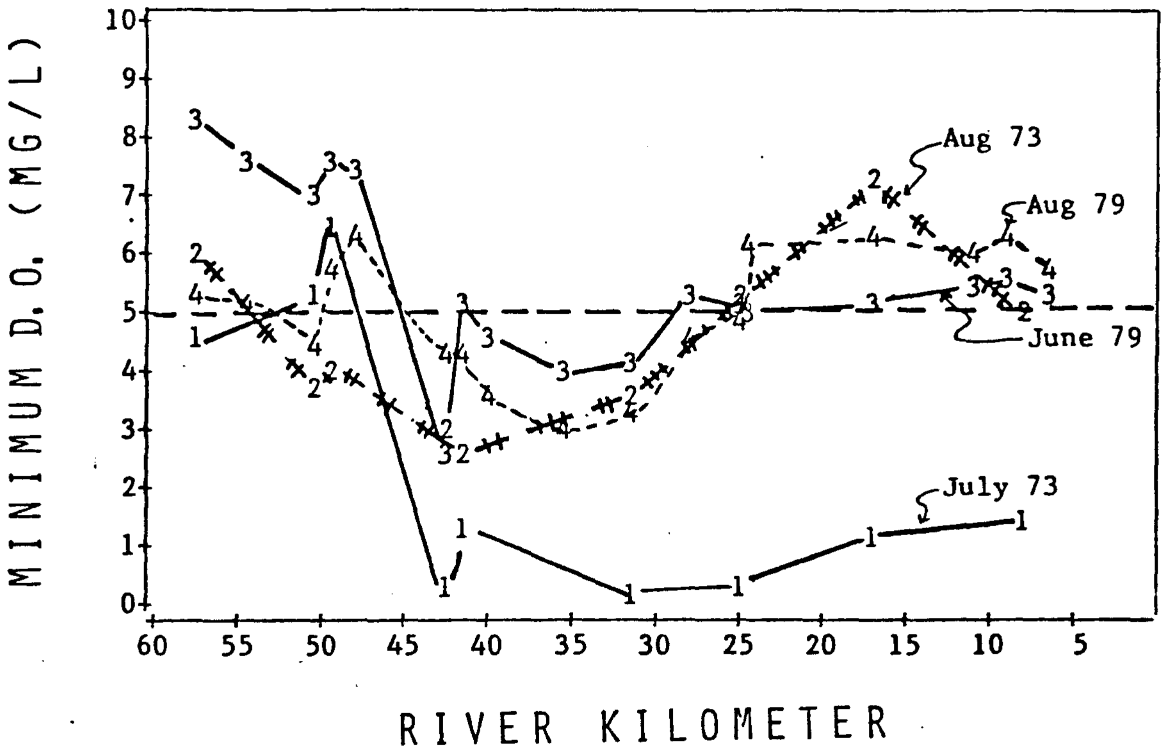
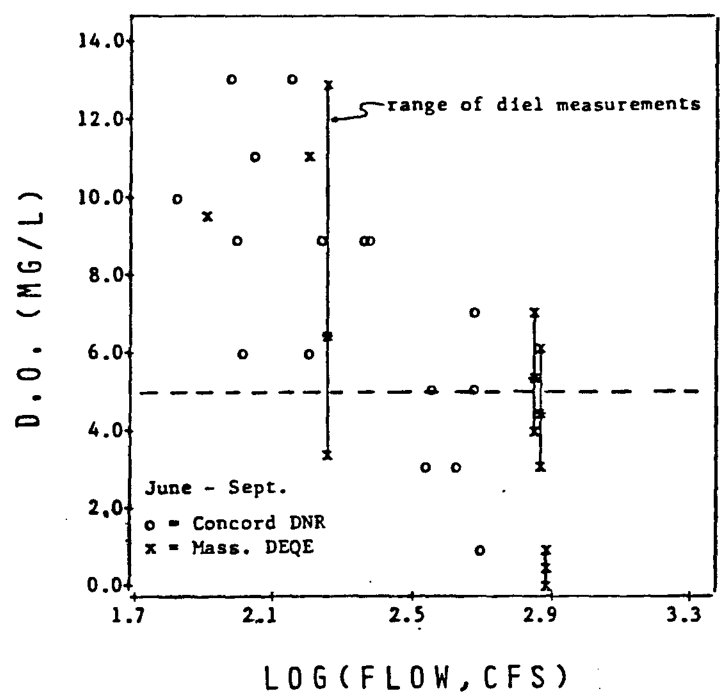


Figure 6

Relationship between Flow at Lowell and Dissolved Oxygen
at Route 117 Bridge by Season



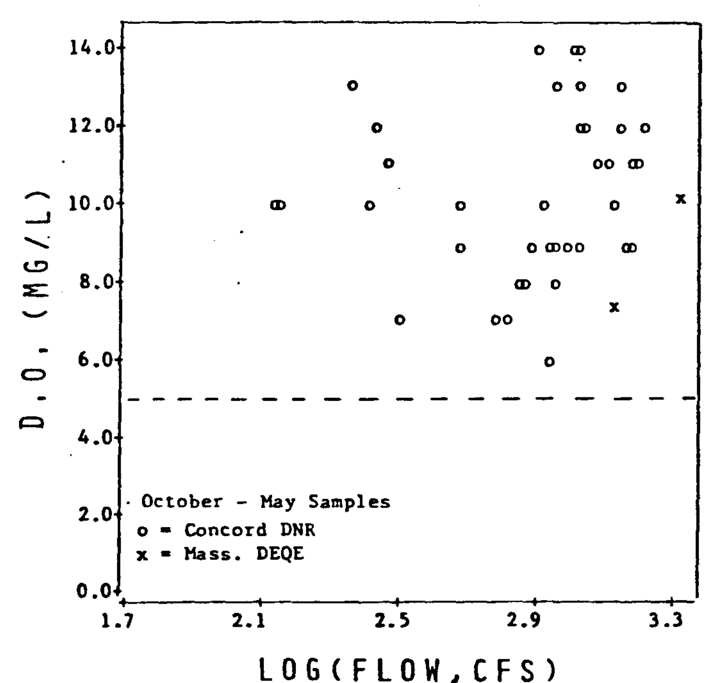
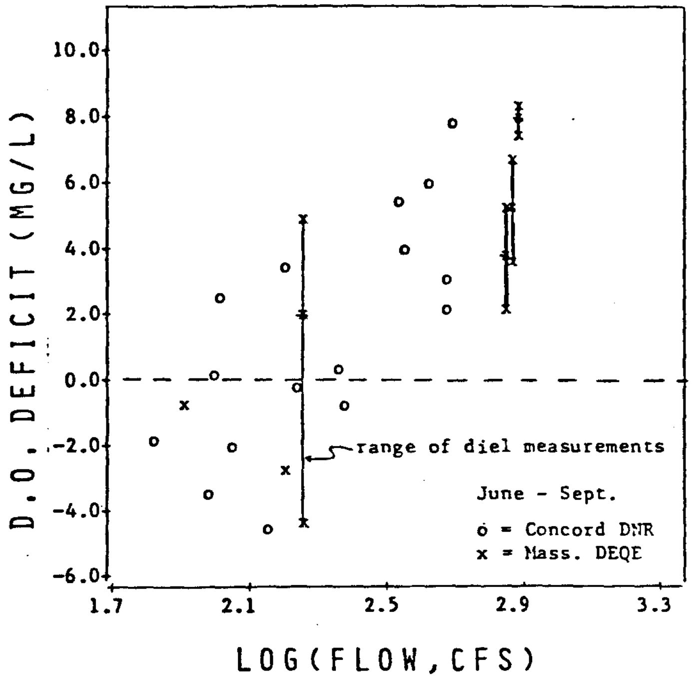


Figure 7

Relationship between Flow at Lowell and Dissolved Oxygen Deficit at Route 117 Bridge by Season



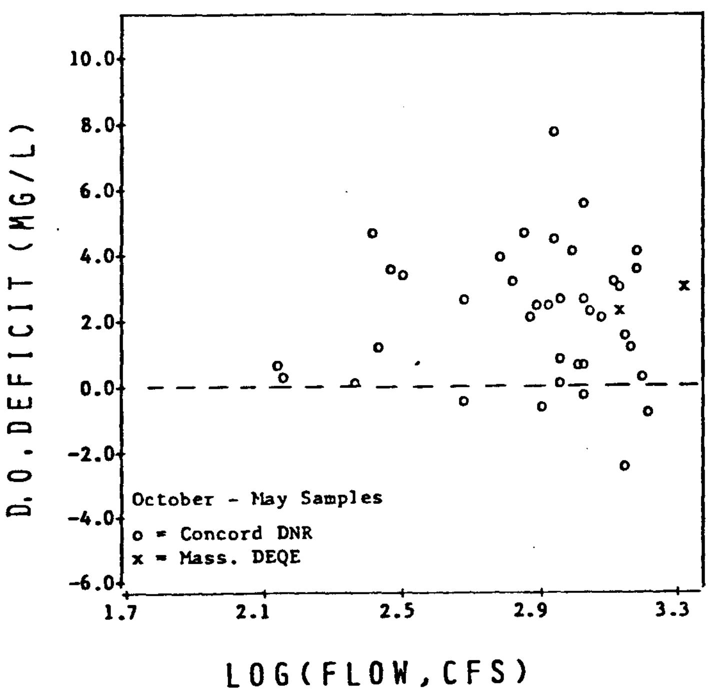
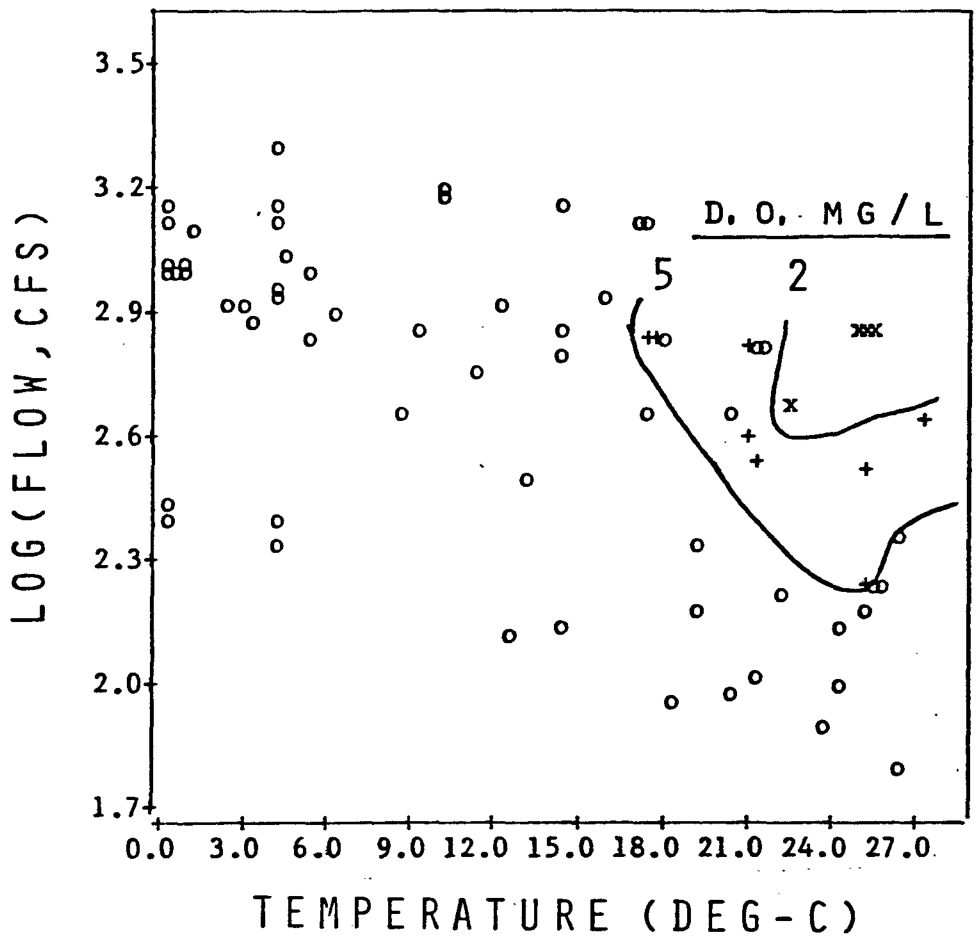


Figure 8

Oxygen Violations Observed at the Route 117 Bridge as a Function of Flow at Lowell and River Temperature



(attributed to microbial respiration and decay of organic materials). A value of zero indicates that oxygen concentrations are at equilibrium with the atmosphere. Dominance of respiration or photosynthesis is indicated by positive or negative values, respectively. The summer plot shows that the balance is shifted toward oxygen sinks during high flows and toward oxygen sources during low flows. The importance of photosynthesis as an oxygen source is indicated by the negative deficits measured on several occasions during low-flow periods.

Figure 8 provides an alternative representation of the Route 117 data. Approximate contours of 5 and 2 mg/liter oxygen measurements are indicated on a plot of flow against river temperature. This display clearly shows that violations of these criteria are typical of warm, wet periods. In a "normal" river system, oxygen concentrations would generally be lowest during warm, dry periods.

The steadily increasing oxygen concentrations between river kilometers 42 and 20 during the low-flow survey of August 1973 can be partially attributed to increasing algal concentrations, as shown in Figure 9. The peak algal population of 30 mg Chlorophyll-a/m3 was roughly three times that observed during the high-flow survey of June 1979. The increased algal growth in July of 1973 can be attributed to higher temperatures (24 vs. 20 degrees C) and longer hydraulic residence time (approximately 21 days vs. 7 days). The lack of ortho-phosphorus data precludes direct assessment of growth-limiting nutrients, although total phosphorus data and model simulations (described below) indicate that phosphorus was probably not limiting during any of the survey periods. Nitrate and ammonia profiles indicate that nitrogen limitation may have developed at peak chlorophyll-a levels during the low-flow survey.

Total and fecal coliform data derived from surveys by Concord DNR (1973-1978), Mass. DEQE (1973-1979), and IEP (1982) are displayed in Figures 10-12. In interpreting the spatial displays (Figures 11 and 12), note that data from 1973 and 1979 are based upon replicate sampling and are therefore more reliable than data from other years.

Figure 10 shows time series of Concord DNR total coliform data at three locations in the basin between 1973 and 1978. Coliform counts are displayed in relation to 1000 organisms/100 ml, formerly the state criterion for Class B waters. Decreasing trends in the measurements are indicated at all three stations. The trend is most apparent at the Assabet station and probably reflects upstream point source abatements.

Spatial variations in total coliform levels measured during four different years are shown in Figure 11. Consistent with the trends noted in the Concord DNR data, the 1973 spatial profiles were generally higher than those measured in subsequent years. Concentrations tend to increase moving through the Framingham area (RKM 58 to 48) and probably reflect urban runoff and the effects of a leaking sewage pumping station between Saxonville Dam and Elm Street, which was repaired prior to the 1979 surveys, according to DEQE sources. During the June 1979 survey, coliform levels declined below Framingham and increased at the mouth of the Assabet, while levels showed less spatial variation during other periods. Fecal coliform variations for 1977 and 1979 are shown in Figure 12 in relation the Class B criterion of 200 organisms/100 ml (monthly log-mean).

Interpretation of the total and fecal coliform profiles is complicated by changes in source conditions leading to trends in the data, by inconsistencies in the sampling freqencies and locations from one year the next, by possible seasonal effects on coliform loadings,

Figure 9

Spatial Variations in Chlorophyll-a Concentrations During DEQE Intensive Monitoring Periods

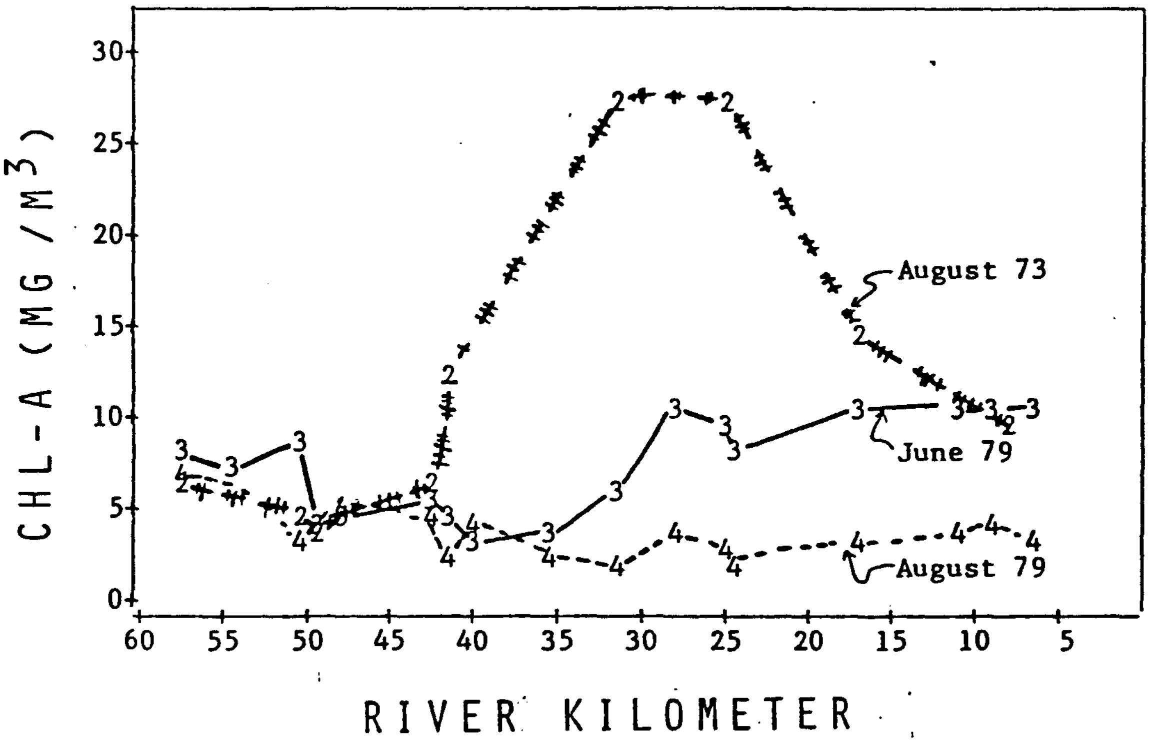
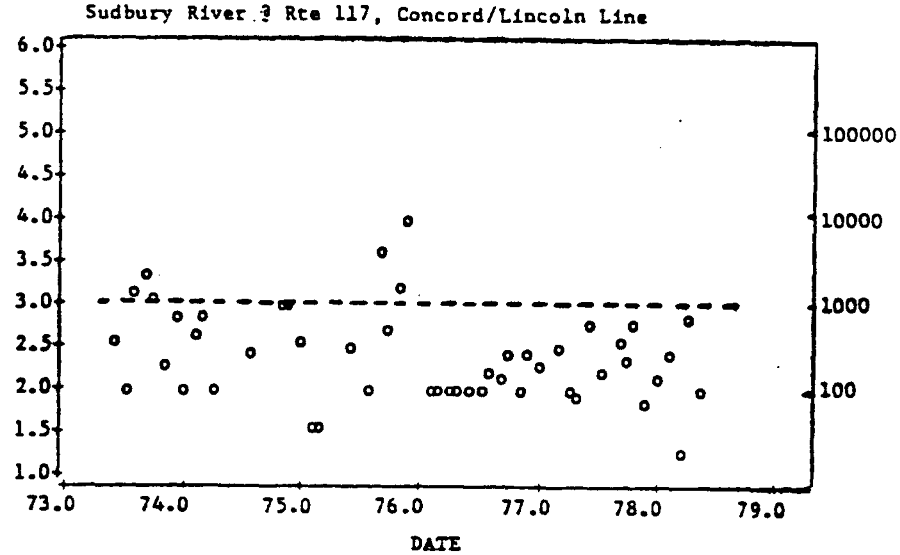


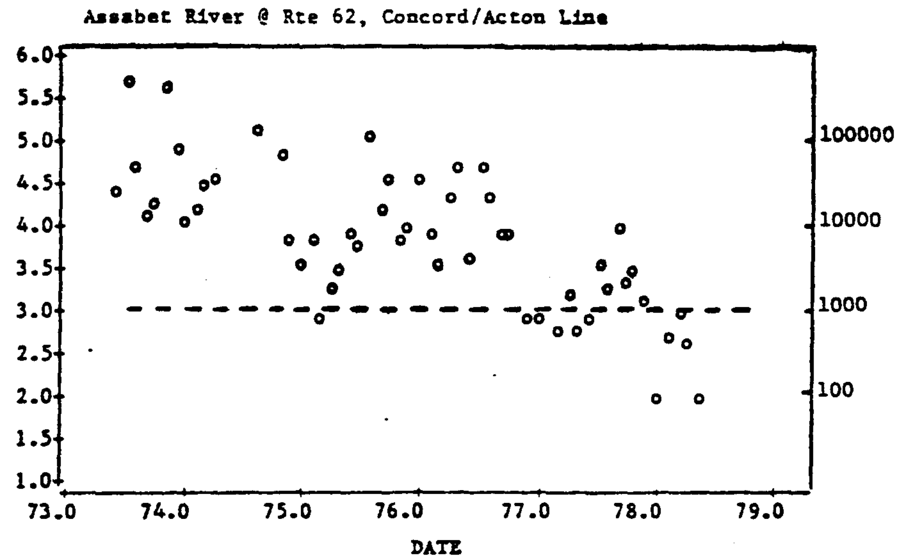
Figure 10

Time Series of Total Coliform Measurements at Three Locations in the SUASCO Basin Derived from Concord DNR Data

Units: Organisms / 100 ml

Log Scales — Linear Scales





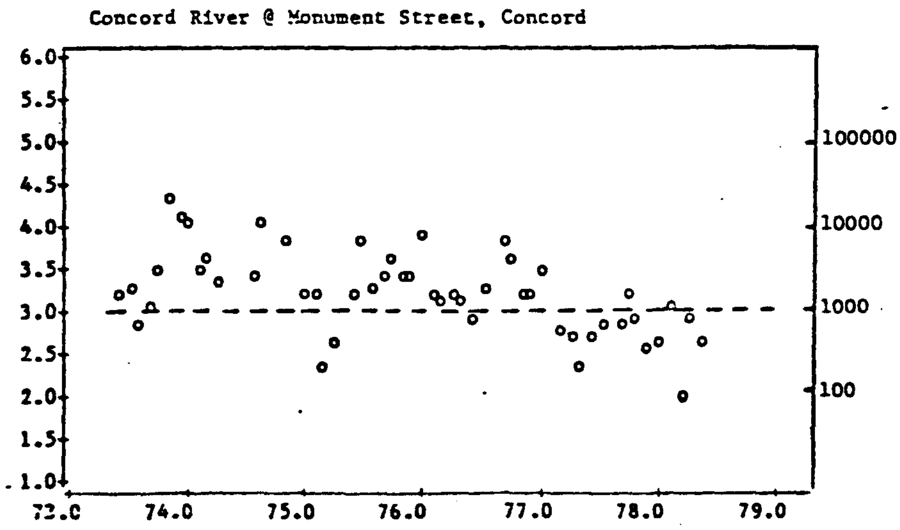


Figure 11

Spatial Variations in Total Coliform Measurements (#/100 ml)
During Different Survey Periods

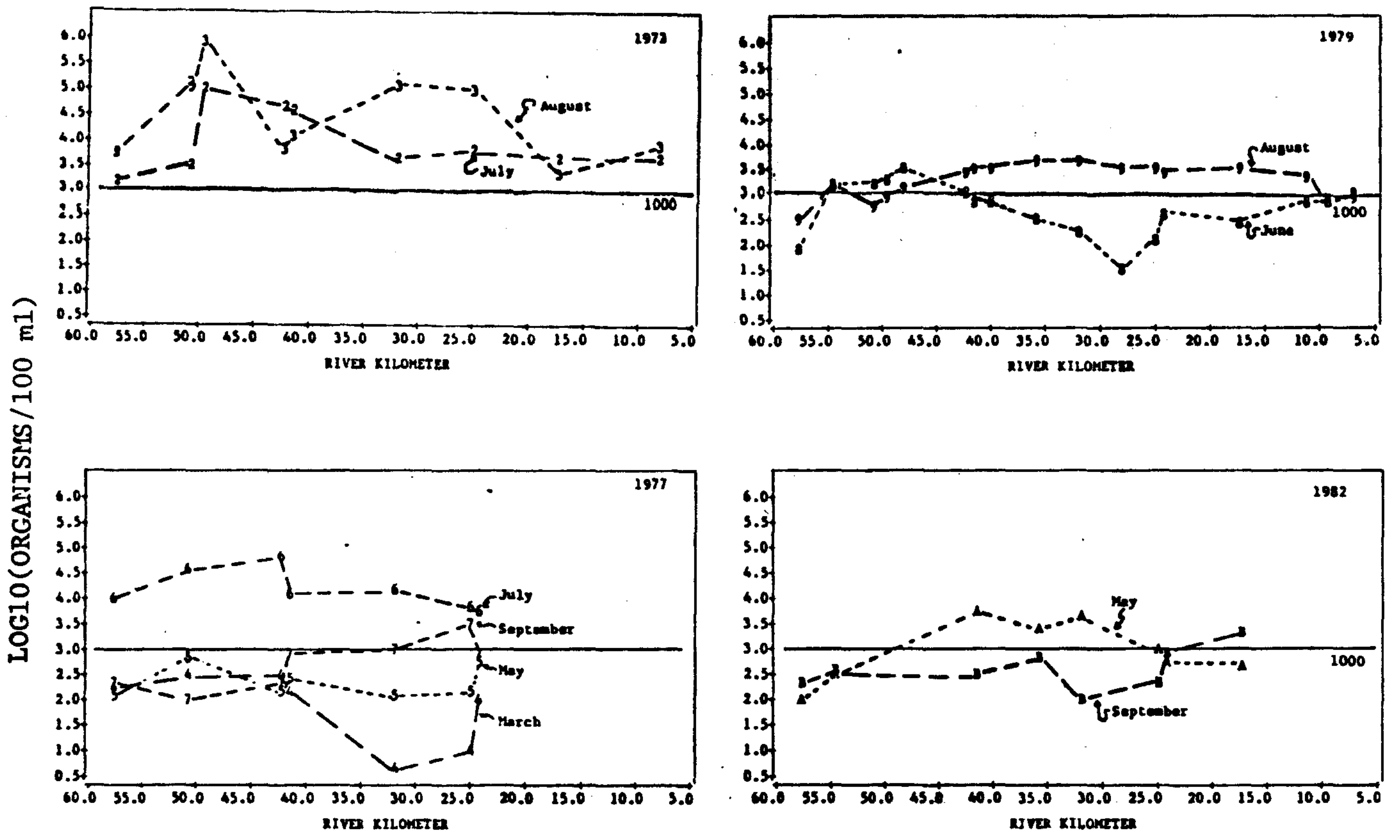
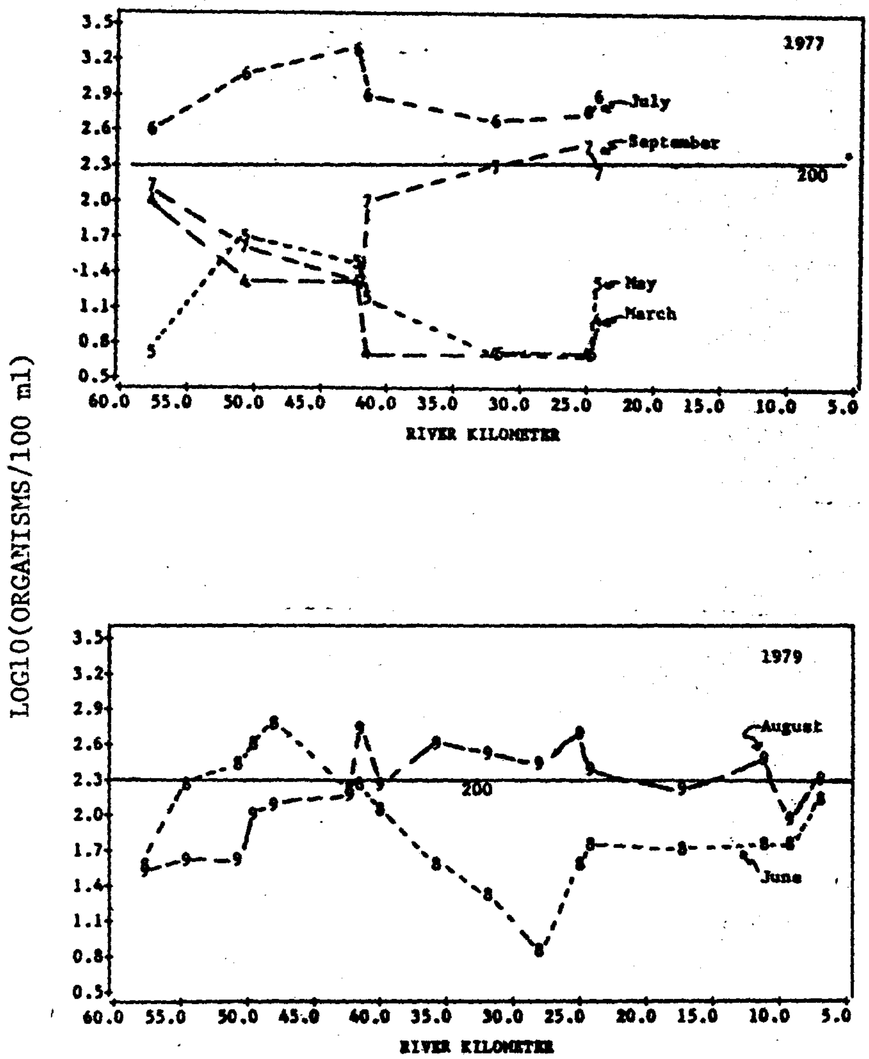


Figure 12

Spatial Variations in Fecal Coliform Measurements (#/100 ml)
During Different Survey Periods



and by the high variability which is inherent in these types of measurements. Figure 13 shows, however, that total and fecal coliform measurements are highly correlated. Generally, summer coliform profiles tend to be higher than those observed for other seasons. Higher summer counts may reflect (a) less dilution of coliform sources because of lower flows; (b) lower coliform decay rates attributed to higher algal concentrations and less light penetration; (c) possible growth of organisms in the river during periods of high temperature and organic matter concentration; (d) increased non-point loadings during the summer attributed to wildlife and/or other natural sources. Existing data do not permit sorting or ranking of these factors.

The measurements themselves are relative indicators and violations of the criteria do not necessarily suggest public health problems related to pathogenic bacteria. The significance of coliform levels and variations should be interpreted in relation to present and projected uses of the water for bathing and drinking. Interest and use for bathing is low; there are no "beaches" and the appeal is low because of natural water color and organic substrate. Billerica's drinking water supply is protected by routine filtration and chlorination procedures.

Regression analyses of DEQE and Concord DNR river temperature data from the Route 117 bridge have been performed in order to assess relationships with flow and climatologic variables. Air temperature is a major generally a major determinant of surface water temperatures (Linseley et al., 1968). Variations in flow may influence water temperatures because of changes in mean depth, velocity, surface area, and residence time. The spillage of water over dams (MDC#1 and Saxonville) before entering the Lower Sudbury would tend to promote equilibration of water temperatures with ambient climatologic conditions and reduce temperature sensitivity to flow. Water temperatures between May and October are generally of primary concern with respect to dissolved oxygen, other water quality aspects, and fisheries.

The following regression equations are based upon 27 water temperature measurements made at the Route 117 bridge between May and October:

$$Tw = 5.88 - .016\ Qs + .84\ Ta \qquad (R^2 = .84,\ Se^2 = 3.6)$$

$$Tw = 5.80 - .0033\ Ql + .85\ Ta \qquad (R^2 = .84,\ Se^2 = 3.7)$$

where,

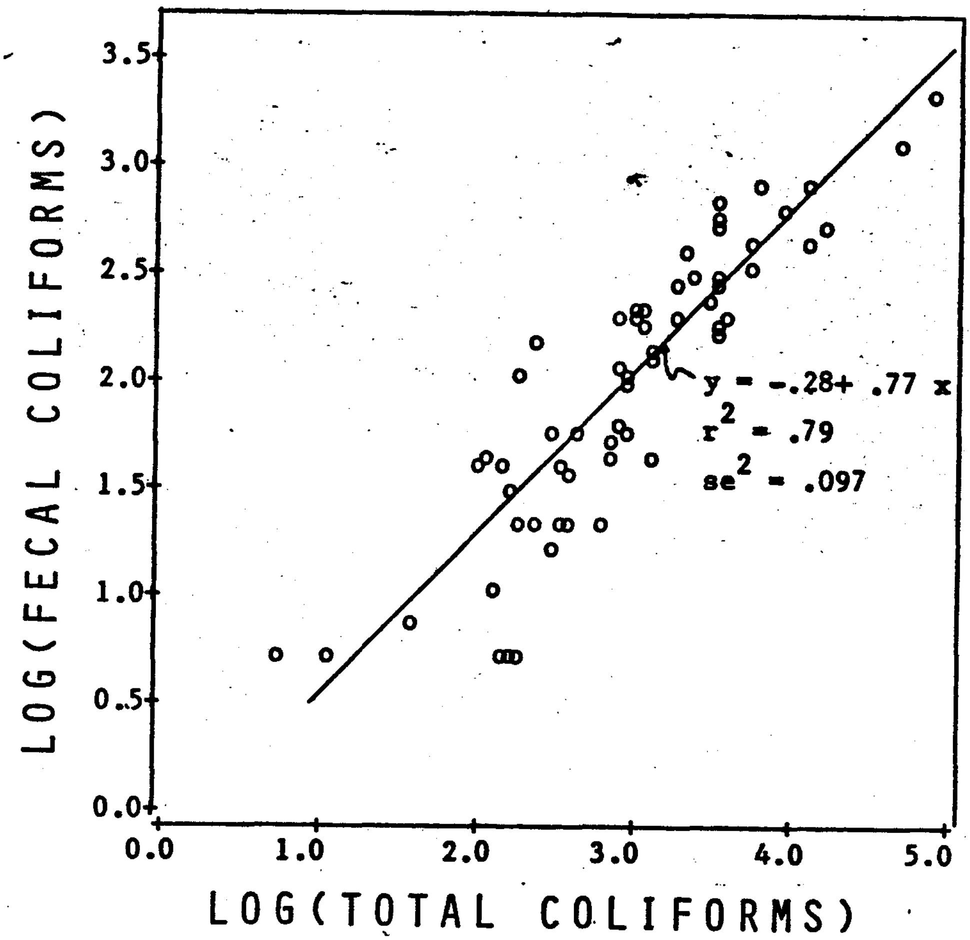
Tw = water temperature at route 117 (deg C)
Ta = monthly-mean air temperature (deg C)
Qs = flow at MDC#1 (cfs)
Ql = flow at Lowell (cfs)

The regressions indicate that air temperature is the major controlling factor, although the flow terms are also statistically significant ($p < .05$) in both cases. Logarithmic transformations for the flow terms have also been tested, but found to explain less variance than the linear models. Using the regression with the highest flow coefficient (.016 vs. Qs), a diversion of 40 mgd (64 cfs) would be expected to result in a temperature increase of 1.06 degrees C at the Route 117 bridge. The

Figure 13

Correlation Between Total and Fecal Coliform Measurements

Units: Organisms / 100 ml



maximum feasible diversion under a median August flow regime, 12 mgd, (based upon HEC-II simulations) would result in a .32 deg C increase. Temperature changes of this scale would not be expected to have significant impacts on aquatic life or water quality.

Model Description

A version of the QUAL-2 water quality simulation model has been used to assess diversion options. QUAL-2 is a steady-state model which was originally developed for use in wasteload allocation studies (Water Resources Engineers, 1972). Many subsequent versions of the model have followed (Roesner et al., 1981). The basic algorithm used here was developed and used in wasteload allocation studies of the Lower Winooski River in Vermont (Meta Systems, 1979, Vermont Department of Water Resources, 1982, VanBenschoten and Walker, 1982). Improvements over earlier versions especially important for this application include:

(1) addition of detrital organic nitrogen and organic phosphorus compartments;

(2) updated algal growth kinetics (self-shading, algal ammonia uptake, alternative (vs. multiplicative) nutrient limitation by nitrogen or phosphorus);

(3) provision for simulating diel variations in oxygen attributed to photosynthesis and respiration by algae and aquatic plants;

(4) improved simulation of longitudinal dispersion (Fischer et al., 1979);

(5) provision for nonlinear hydraulic geometries;

(6) modification of the numeric solution algorithm to permit application to systems with relatively long residence times;

The program code has been adapted for use on microcomputers and tested against the original code using input files for the Lower Winooski.

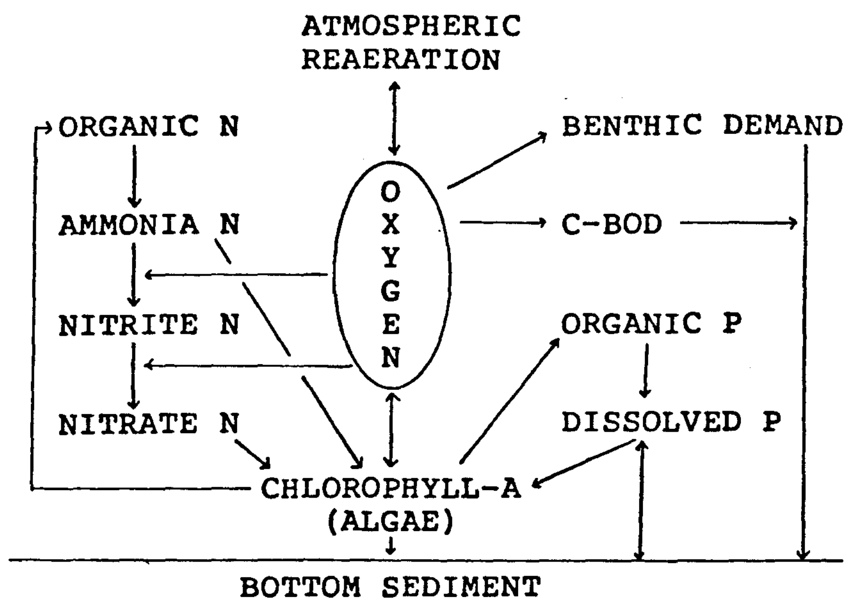
Control pathways in the model are shown in Figure 14. The model simulates the transport and transformations of water quality components in a one-dimensional system under steady-state hydraulic conditions. Boundary conditions are specified in terms of source and tributary flows and concentrations, day length, solar radiation, benthic sources and sinks, channel hydraulic geometry, and water temperatures. The model divides the river in to a series of computational elements and performs water and mass balances on each element, while taking into account sources, sinks, and transformations. All rate processes are temperature-dependent. The output is a longitudinal profile (concentration vs. river kilometer) for each water quality component at equilibrium with the specified boundary conditions.

Model Calibration

The uniqueness of each system and limitations in the state-of-the-art require that the model be calibrated and tested in each application. Calibration involves selecting an appropriate set of parameters so that model predictions are in "reasonable" agreement with measured water

Figure 14

Control Pathways in QUAL-II



quality conditions. The parameters characterize various physical, chemical, and biological processes which influence water quality (e.g., organic matter oxidation rate, algal growth rate, etc.). Parameter estimates for the Winooski River and other systems (Zison et al., 1978) provide reasonable starting points for calibration of the model to the Sudbury. Testing involves demonstrating that the calibrated model can simulate water quality conditions during more than one survey period.

The first step in applying a model of this type is the definition of reaches with relatively uniform morphometric and hydraulic characteristics. Output from the HEC-2 hydraulic simulation model has been used for this purpose. The HEC-2 output consists of estimated hydraulic characteristics (flow cross-section, top width, velocity, etc.) at each of 278 locations along the length of the river between Talbot Dam and MDC Dam #1, under each of nine different hydrologic regimes which represent various seasons and diversion strategies.

Reach boundaries and morphometric characteristics have been assessed by integrating the cross-sectional areas and top widths with respect to distance moving upstream from Talbot Dam. The integrated values, representing cumulative volume and surface area, respectively, have been plotted against river kilometer for each hydrologic regime. Reach boundaries have been specified at river kilometers corresponding to changes in the average slopes of these curves. This procedure averages over high-frequency variations in cross-section (attributed to bridges, for example). A total of 11 reaches have been defined in this way, as identified in Table 3. Tributary and local drainage areas for each reach are also listed in Table 3. The total wetland area draining into each model reach has also been estimated from aerial photographs of the 1968 flood and topographic maps. A schematic reach map is given in Figure 15. In performing balance calculations, the model further divides each reach into a series of "computational elements" (nominal length .4 km), so that the final simulation represents the river as a series of about 170 linked segments.

Table 4 summarizes the morphometric and hydraulic characteristics of each reach based upon integration of the HEC-2 profiles for each hydrologic regime. The relatively large fluctuations in width in reaches 4-6 reflect flooding of adjacent wetlands under high flow conditions. Hydraulic geometries are specified in QUAL-2 using functions of the following form:

$$F = \log_{10}(Qi/Qri)$$

$$W = Wr\ 10^{W1\ F + W2\ F^2}$$

$$A = Ar\ 10^{A1\ F + A2\ F^2}$$

where,

F = relative flow (dimensionless)
Qi = outflow from reach i (m3/sec = cms)
Qri = reference flow for reach i (cms)
W = top width
Wr = top width at flow Qri (m)

Figure 15

QUAL2 Reach Map for Sudbury/Concord Rivers

RKM	SOURCES	RKM	REACHES
58.60	1 Upper Sudbury	58.60	1 MDC Dam #1
		53.30	2 Mass Turnpike
		51.30	3 Saxonville Dam
		48.30	4 Stone Bridge Road
42.20	2 Raytheon	42.80	5 Pelham Island Road
42.20	3 Hop Brook		
		36.00	6 Sherman Bridge
		31.30	7 Above Fairhaven
		30.50	8 Below Fairhaven
25.10	4 Assabet River	25.20	9 Above Assabet
23.50	5 Concord STP		
14.50	6 Billerica HOC	15.30	10 Bedford/Carlylse Line
		7.60	11 Talbot School
		7.10	Talbot Dam

A = cross-section (m^2)
Ar = cross-section at flow Qri (m^2)
A1,A2,W1,W2 = empirical parameters for each reach

The model permits referencing of width and cross-section in given reach to the flow in any other reach. In a "normal", free-flowing stream, morphometry is referenced to discharge within each reach. To represent the backwater effects in this system, the morphometries of segments 4 to 11 (below Stone Bridge Road) are referenced to the discharge at Talbot Dam (Reach 11). Regression analyses of HEC-2 output confirm that morphometric properties in these reaches are more strongly correlated to flows at Talbot Dam than to flows within the respective segments.

Regression analyses have been done to estimate the parameters A1,A2,W1, and W2 for each reach based upon the data in Table 4. Because the model is not applied under extreme high-flow conditions, data from the spring flood simulation (Run 9) have not been used in these regressions. This improves the quality of the fit for the lower flow regimes. In all cases, the regressions explain more than 98% of the variance in the simulated widths and cross-sections, with a maximum standard error of 5% and median standard error of about 1%. The above relationships essentially permit interpolation of the HEC-2 output.

Final parameter estimates for the morphometric relationships are included in the model output listings which have been submitted as an addendum to this report and which are available for review by interested parties at the offices of Interdisciplinary Environmental Planning, Inc.. Observed stage-discharge relationships at Sherman Bridge and the Billerica Water Treatment Plant intake have been used in combination with the HEC-2 output to estimate the morphometric parameters of each model reach. Reference width and cross-section have been adjusted in Reach 7 (Fairhaven Bay) to reflect the single HEC-2 cross-section included in this area. The model requires estimates of Manning's n values for calculation of longitudinal dispersion rates within each reach; these values have been derived from the HEC-2 input file.

DEQE surveys in July 1973, August 1973, and June 1979 have been used for calibration and testing of the parameters used in simulating water quality transformation processes. The 1973 surveys did not include monitoring of point sources (Raytheon-Wayland, Concord STP, and Billerica House of Correction). Flows and concentrations from the 1979 survey have been used in these cases. Based upon information supplied by the Concord STP operator, discharge during the August 1973 survey was about 600,000 gallons/day, or about half that measured during the 1979 survey. Model simulations are relatively insensitive to point source loadings. Average concentrations measured at MDC Dam #1 during each survey have been used to characterize upstream inflow and local, ungauged inflows to each model reach. Measured concentrations have also been used for the two major tributaries, Hop Brook and Assabet River.

The water balance during each survey has been derived from average measured flows of the Sudbury at MDC Dam #1, Assabet at Maynard, and Concord at Lowell. Ungauged inflows have been estimated by difference from the above measured flows and distributed on a drainage area basis. Assabet flows measured at Maynard have been adjusted based upon drainage area to reflect flows at the confluence with the Sudbury. Hop Brook flows have been estimated using the average ungauged runoff rates for the basin, estimated by difference according to the above scheme. Details on the relative drainage areas are summarized in Table 3.

Table 3
Summary - Reach Definitions for QUAL2
Sudbury River MDC Dam #1 to Talbot Dam

Rch	Approximate Upstream Boundary	River Index km	Drainage Areas Total km2	Local km2	Wetland km2
	MDC Dam #1	58.6	193.9		
01				48.4	.43
	Mass Turnpike	53.3	242.3		
02				10.0	.20
	Saxonville Dam	51.3	252.3		
03				24.3	.05
	Stone Bridge	48.3	276.6		
04				14.5	4.54
	Pelham Island Road	42.8	291.1		
	Hop Brook --->	42.2		58.3	
05				32.4	16.70
	Sherman Bridge	36.0	381.8		
06				17.9	7.63
	Above Fairhaven Bay	31.3	399.7		
07				2.8	0.28
	Below Fairhaven Bay	30.5	402.5		
08				17.1	2.05
	Above Assabet	25.2	419.6		
	Assabet --->	25.1		458.8	
09				53.6	10.94
	Bedford/Carlysle	15.3	932.0		
10				52.1	1.80
	Talbot School	7.6	984.1		
11				4.7	0.12
	Talbot Dam	7.1	988.8		

Table 4
Morphometric and Hydraulic Properties of QUAL-2 Model
Reaches Derived from HEC Output

RCH	RUN	ELEV	QOUT	XSEC	WIDTH	ZM	VEL
1	1	149.29	3.4	48.2	51.6	0.93	0.070
1	2	149.77	22.4	78.4	60.3	1.30	0.286
1	3	150.16	41.4	94.0	57.2	1.64	0.440
1	4	150.41	62.4	123.3	66.0	1.87	0.506
1	5	150.74	93.3	151.0	71.0	2.13	0.618
1	6	150.64	82.9	141.9	69.8	2.03	0.584
1	7	151.19	144.1	197.0	84.9	2.32	0.732
1	8	151.63	206.0	242.6	97.4	2.49	0.849
1	9	152.34	331.5	291.9	110.2	2.65	1.136
2	1	144.48	3.5	760.1	229.4	3.31	0.005
2	2	145.00	22.5	1143.6	260.8	4.39	0.020
2	3	145.25	42.0	1160.2	261.5	4.44	0.036
2	4	145.48	62.9	1174.5	262.1	4.48	0.054
2	5	145.78	93.8	1193.1	262.9	4.54	0.079
2	6	145.69	83.0	1186.9	262.7	4.52	0.070
2	7	146.22	145.6	1220.9	264.1	4.62	0.119
2	8	146.67	207.5	1249.9	265.4	4.71	0.166
2	9	147.59	334.0	1303.4	267.6	4.87	0.256
3	1	143.32	5.8	50.4	44.4	1.14	0.115
3	2	144.89	24.8	70.1	51.3	1.37	0.354
3	3	144.95	54.0	88.8	55.7	1.59	0.608
3	4	145.01	87.0	108.8	58.2	1.87	0.799
3	5	145.07	118.0	125.1	60.1	2.08	0.943
3	6	145.05	108.0	119.8	59.5	2.01	0.902
3	7	145.17	204.7	165.7	63.6	2.61	1.235
3	8	145.28	266.6	194.2	65.7	2.96	1.373
3	9	145.46	430.0	272.6	93.2	2.92	1.578
4	1	110.63	6.7	135.0	67.0	2.02	0.050
4	2	111.05	25.7	160.1	78.1	2.05	0.161
4	3	112.03	58.0	212.0	87.8	2.41	0.274
4	4	112.38	96.3	292.9	117.7	2.49	0.329
4	5	112.68	127.3	332.6	134.2	2.48	0.383
4	6	112.59	117.0	321.7	129.7	2.48	0.364
4	7	113.89	227.5	776.5	387.5	2.00	0.293
4	8	114.31	289.4	923.5	469.8	1.97	0.313
4	9	115.59	468.0	1828.1	788.0	2.32	0.256
5	1	109.42	10.3	249.5	83.0	3.01	0.041
5	2	109.60	29.3	261.5	85.2	3.07	0.112
5	3	110.25	77.0	310.6	97.9	3.17	0.248
5	4	111.10	133.1	417.4	162.3	2.57	0.319
5	5	111.38	164.1	457.7	182.1	2.51	0.359
5	6	111.32	154.0	449.1	178.1	2.52	0.343
5	7	113.13	317.8	1036.7	643.8	1.61	0.307
5	8	113.45	379.7	1235.6	754.1	1.64	0.307
5	9	114.89	615.0	2606.4	1150.8	2.26	0.236

(continued)

Table 4 (continued)

RCH	RUN	ELEV	QOUT	XSEC	WIDTH	ZM	VEL
6	1	109.41	11.4	479.2	125.1	3.83	0.024
6	2	109.53	30.4	493.5	127.1	3.88	0.062
6	3	109.98	83.0	553.1	155.3	3.56	0.150
6	4	110.74	145.3	705.9	259.4	2.72	0.206
6	5	110.93	176.3	753.4	282.2	2.67	0.234
6	6	110.90	166.0	747.5	279.5	2.67	0.222
6	7	112.73	347.8	1513.8	553.2	2.74	0.230
6	8	112.99	409.7	1654.9	571.5	2.90	0.248
6	9	114.47	664.0	2575.8	677.7	3.80	0.258
7	1	109.41	11.4	3739.0	622.8	6.00	0.003
7	2	109.52	30.4	3808.3	626.3	6.08	0.008
7	3	109.93	83.0	4068.3	640.9	6.35	0.020
7	4	110.65	145.3	4544.0	678.1	6.70	0.032
7	5	110.81	176.3	4653.7	687.8	6.77	0.038
7	6	110.80	166.0	4642.9	686.6	6.76	0.036
7	7	112.59	347.8	6122.1	976.3	6.27	0.057
7	8	112.82	409.7	6354.7	989.5	6.42	0.064
7	9	114.28	664.0	7855.5	1075.6	7.30	0.085
8	1	109.41	11.4	809.8	194.5	4.16	0.014
8	2	109.52	30.4	831.1	196.5	4.23	0.037
8	3	109.93	83.0	911.5	204.6	4.46	0.091
8	4	110.65	145.3	1069.1	229.4	4.66	0.136
8	5	110.81	176.3	1102.4	237.2	4.65	0.160
8	6	110.80	166.0	1101.7	236.9	4.65	0.151
8	7	112.59	347.8	1603.7	355.0	4.52	0.217
8	8	112.82	409.7	1683.0	367.1	4.58	0.243
8	9	114.27	664.0	2258.9	427.4	5.28	0.294
9	1	109.41	90.0	1819.4	405.6	4.49	0.049
9	2	109.51	109.0	1856.1	412.5	4.50	0.059
9	3	109.88	182.0	1986.3	441.3	4.50	0.092
9	4	110.55	332.0	2258.5	527.6	4.28	0.147
9	5	110.67	363.0	2314.6	546.1	4.24	0.157
9	6	110.67	363.0	2232.9	518.4	4.31	0.163
9	7	112.38	855.1	3275.0	802.2	4.08	0.261
9	8	112.57	917.0	3416.1	882.5	3.87	0.268
9	9	113.95	1452.0	4790.2	1080.3	4.43	0.303
10	1	109.37	96.0	886.7	193.6	4.58	0.108
10	2	109.45	115.0	902.2	195.4	4.62	0.127
10	3	109.74	185.0	953.1	201.8	4.72	0.194
10	4	110.28	338.1	1047.6	212.9	4.92	0.323
10	5	110.37	369.0	1064.7	214.9	4.96	0.347
10	6	110.38	370.0	1065.1	214.9	4.96	0.347
10	7	111.76	873.1	1344.3	281.1	4.78	0.649
10	8	111.92	935.0	1381.4	283.0	4.88	0.677
10	9	113.14	1480.0	1669.5	304.8	5.48	0.886

(continued)

Table 4 (continued)

RCH	RUN	ELEV	QOUT	XSEC	WIDTH	ZM	VEL
11	1	109.32	96.0	617.9	185.7	3.33	0.155
11	2	109.40	115.0	629.3	188.0	3.35	0.183
11	3	109.62	185.0	670.7	194.5	3.45	0.276
11	4	109.99	338.1	744.5	203.5	3.66	0.454
11	5	110.05	369.0	756.0	204.7	3.69	0.488
11	6	110.05	370.0	756.1	204.8	3.69	0.489
11	7	110.80	873.1	908.0	220.1	4.13	0.962
11	8	110.89	935.0	929.0	221.0	4.20	1.006
11	9	111.47	1480.0	1056.7	225.3	4.69	1.401

RCH = QUAL2 Model Reach (see Table 3)
RUN = HEC Run ID Number:

Flows (cfs)

Run	MDC#1	Talbot	Diversion	
1	3.4	96.0	19.0	Summer
2	22.4	115.0	0.0	"
3	41.4	185.0	0.0	"
4	62.4	338.0	31.0	Fall
5	93.3	369.0	0.0	"
6	82.9	370.0	0.0	"
7	144.1	873.0	62.0	Spring
8	206.0	935.0	0.0	"
9	331.5	1480.0	0.0	"

QOUT = segment outflow (cfs)
XSEC = mean cross-section (ft2)
WIDTH = mean top width (ft)
ZM = mean depth = XSEC/WIDTH (ft)
VELOCITY = mean velocity = QOUT/XSEC (ft/sec)

Monitoring data from the August 1973 (low-flow) survey indicated significant increases in ammonia nitrogen, total phosphorus, and total coliform bacteria in the short river reach between Saxonville Dam (RKM 51.2) and Elm Street (RKM 50.1) in Framingham. Discussions with DEQE staff indicate that significant sewage loadings were probably entering this section of river during that period from a leaking pumping station. Since these loadings were not monitored, they have been estimated in the model calibration procedure and held constant for the July and August 1973 surveys. The situation was reportedly corrected prior to the June 1979 survey.

The estimation of parameters in a model of this complexity is a subjective exercise, aided by the following:

(1) literature studies which provide indications of typical values and feasible ranges for the parameters which describe various water quality transformations at the process level (e.g., algal growth rate, organic matter decay rate, etc.), based upon laboratory experiments, field experiments, and/or other modeling efforts;

(2) published empirical relationships which permit estimation of certain parameters as a function of other system characteristics (e.g., reaeration rate as a function of depth, velocity, and temperature);

(3) direct measurement of certain parameters in the system being studied (e.g., light extinction coefficients based upon Secchi depths);

(4) inference of certain parameters by empirical adjustment to optimize the fit between observed and predicted profiles;

Key parameter estimates and sources for the water quality simulation are summarized in Tables 5 and 6. Parameters are placed in two categories. The values in Table 5 are site-specific characteristics which have been estimated primarily by calibrating the model to observed water quality profiles; these are generally within the ranges reported in modeling studies of other systems (Zison et al., 1978). The values in Table 6 are generalized parameters which describe various water quality transformations; these have been estimated primarily from literature data and empirical functions. Because the parameters in Table 6 describe more or less fundamental (process-level) reactions they tend to be more constant from one river basin to another than those in Table 5.

Model calibration has focused initially on nutrient and chlorophyll-a data from the August 1973 and June 1979 surveys. Observed and predicted chlorophyll-a profiles are compared in Figure 16. The maximum algal growth rate (2.3 1/day), respiration rate (.12 1/day), and settling velocity (.75 m/day) parameters used in simulating the chlorophyll-a profiles are identical to those used in the Lower Winooski model (Van Benschoten and Walker, 1982). The model simulates the peak observed chlorophyll-a concentrations of 30 and 10 mg/m3 for the two surveys. For the low-flow survey, chlorophyll-a concentrations are over-predicted below the Assabet. This may indicate violations of the steady-state assumption in the lower portions of the river during the August 1973 survey, since the total time of travel under these conditions (approximately 21 days) was long in relation to the survey

Table 5
Parameter Values Derived Primarily from Calibration of Model to Observed Water Quality Profiles and Other Site-Specific Characteristics

Parameters	Value/Comments
Benthic Photosynthesis	1.5 - 10 g/m2-day channel
	4 g/m2-day overbank
Benthic Plant Respiration	assumed equal to benthic photosynthesis
Benthic Oxygen Demand	1 - 3 g/m2-day channel + wetland impact
	3 g/m2-day overbank
Benthic Sources/Sinks (g/m2-day) (negative values are sinks)	
BOD-U	4 (reach 6) , 0 other reaches
Dissolved P	.005 impounded reaches, 0 other reaches (1,3)
Ammonia N	.025 impounded reaches, 0 other reaches (1,3)
	-.05 overbank
Nitrate N	-.10 overbank
Wetland Export Concentrations (g/m3)	
D.O.	4
BOD-U	5
Organic N	3.5
Organic P	.50
Organics	120 (expressed as benthic oxygen demand)
Non-Algal Light Extinction Coef. (1/m)	
	1.4 reaches 1-9
	2.6 reaches 10-11

note: all biological rates and benthic fluxes input at 20 degrees C

Table 6

Generalized Parameter Estimates Derived Primarily from the Literature

Parameters	Value/Comments
Reaeration Rate	O'Connor & Dobbins (1958) equation constrained to K2 > 1 / mean depth (Manhattan College, 1968, Banks, 1973)
Longitudinal Dispersion Rates	Fischer et al. (1979) equation
Decay Rates	
BOD-U	.2 1/day
Ammonia N	.6 1/day (calibrated)
Nitrite N	3.0 1/day
Organic N	.1 1/day
Organic P	.1 1/day
Fecal Coliforms	1.6 1/day (calibrated)
Algal Parameters	
Maximum Growth Rate	2.3 1/day
Respiration Rate	.12 1/day
Settling Velocity	.75 m/day
Chlorophyll Content	.010 mg Chl-a / mg Algae
P Content	.011 mg P / mg Algae
N Content	.080 mg N / mg Algae
Light Extinction	43.2 m2/g Chl-a
Ammonia Preference Factor	.9
Photo. Oxygen Equiv.	1.6 mg O2 / mg algae
Resp. Oxygen Equiv.	2.0 mg O2 / mg algae
Half-Saturation Constants	
Algal Phosphorus Uptake	.005 g/m3
Algal Nitrogen Uptake	.03 g/m3
Algal Growth vs. Light	1.5 calories/cm2-hr
Benthic O2 Demand	.5 g O2 / m3
Other Benthic Sinks	.03 g / m3
Other Oxidation Rates	1.0 g O2 / m3
Temperature Sensitivity Coefficients (THETA)	
Ammonia N Oxidation	1.080
Benthic Oxygen Demand	1.072
Other Benthic Sinks/Sources	1.047
Reaeration Rate	1.022
Other Biological Rates	1.047
Longitudinal Dispersion Rates	Fischer et al. (1979) equation
Solar Radiation	Estimated from latitude, month, and cloud cover (Mc Gaughey, 1968)

$$\frac{\text{Rate at T}}{\text{Rate at 20}} = \text{THETA}^{(T-20)}$$

references: Zison et al. , 1978
Manhattan College, 1968
VanBenschoten and Walker, 1982

all biological rates input at 20 degrees C period; i.e., algal populations measured in the lower reaches may have been influenced by higher flow periods previous to the monitoring period. Alternative explanations would include zooplankton grazing, increased turbidities, or effects of floating duckweed (leading to increased light limitation) in the lower river. Non-algal turbidities have been increased in the last two river reaches to limit peak biomass levels.

Preliminary simulations of the high-flow surveys indicated relatively large over-predictions of observed nitrate concentrations below Pelham Island Road. The role of wetlands as nitrate sinks has been well-documented (Kadlec and Kadlec, 1978) and can be attributed to combined effects of nutrient uptake by plants and denitrification supported by organic substrates. Reasonable simulation of the observed nitrate profiles (Figure 17) has been achieved by specifying a nitrate loss of .1 g/m2-day in overbank areas (defined below). A corresponding overbank loss of .05 g/m2-day has been specified for ammonia nitrogen. These nutrient transformations during high-flow periods provide a more complete description of the system but are of little consequence to water quality with respect to algal populations or dissolved oxygen because algal populations are limited only by light and residence time during high-flow periods.

Calibration of the daily mean oxygen profiles was initially achieved by adjusting the benthic oxygen demand rates in each reach and using the O'Connor-Dobbins (1958) formulation to estimate reaeration rates. Benthic demands estimated in this procedure ranged from 1 to 8 g/m2-day and were found to correlate with the wetland areas tributary to each model segment and to be higher during the high-flow (June 1979) survey. Possible mechanisms for wetland impacts on benthic demands include:

(1) export of particulate organics from the wetland areas and subsequent settling and decay on the river bottom;

(2) percolation of water through organic swamp deposits, transport of dissolved organics with seepage into the river bed, and subsequent oxidation;

(3) increased benthic demand in overbank areas during flooded periods attributed to the decay of accumulated organic materials;

(4) reduced reaeration rates in overbank areas attributed to stagnation of water by aquatic vegetation;

The first two mechanisms would be flow-dependent and are reflected in the export model described below. The third mechanism is simulated by specifying a higher effective benthic demand (3 g/m2-day) in flooded areas adjacent to the channel. The effective channel width is estimated for each reach to correspond to USFWS summer flow criterion of 185 cfs, based upon review of HEC-2 model output which indicates minimal overbank flow under these conditions. Inflection points in the width and cross-section vs. discharge curves are also indicated in reaches with adjacent wetlands at approximately this flow value. This definition does not necessarily correspond to that used in the HEC-2 simulations. To some extent, the effects of vegetation on reaeration rate are implied in the hydraulic simulation, because overbank Manning's n values are higher and

Figure 16

Observed and Predicted Chlorophyll-a Profiles

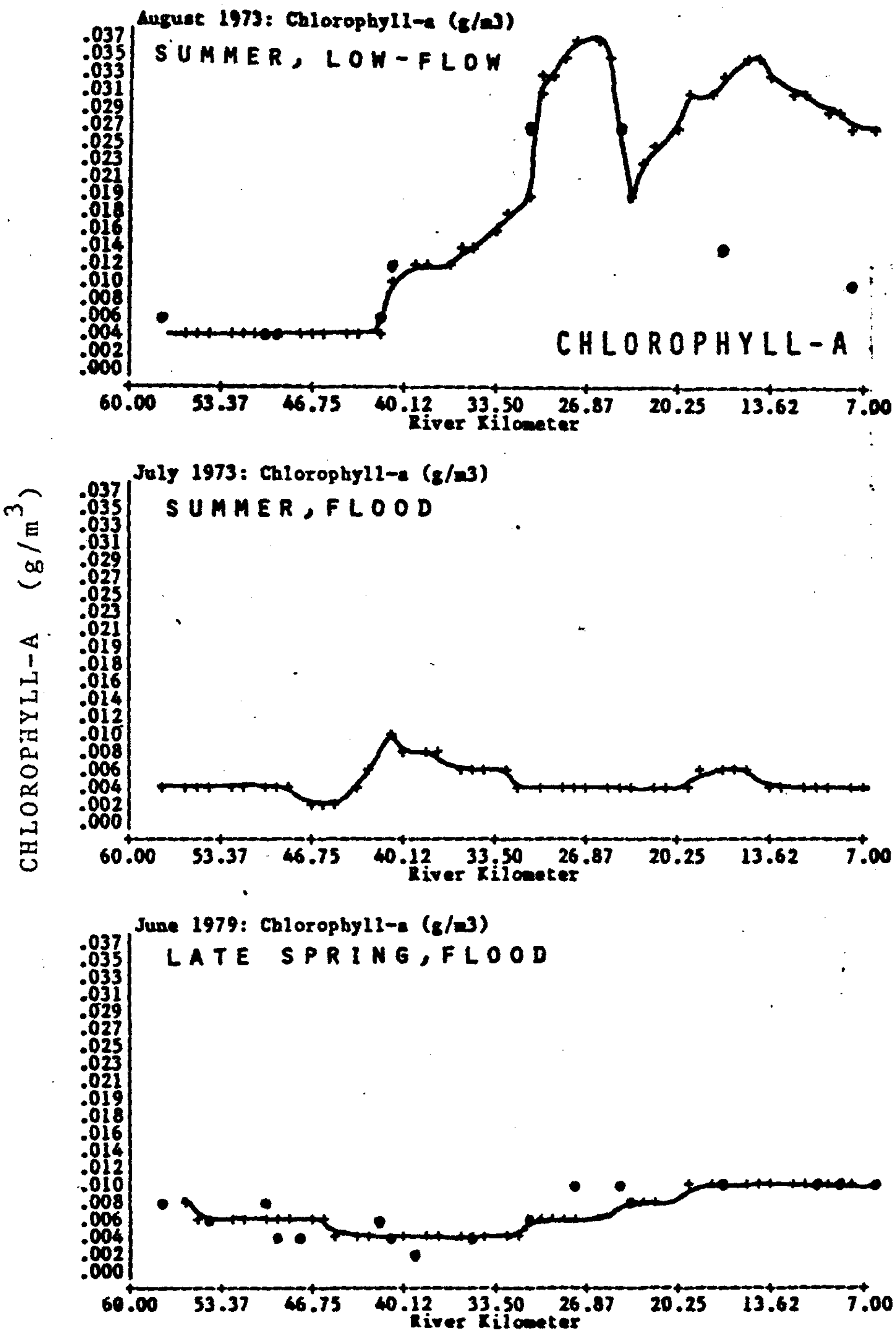
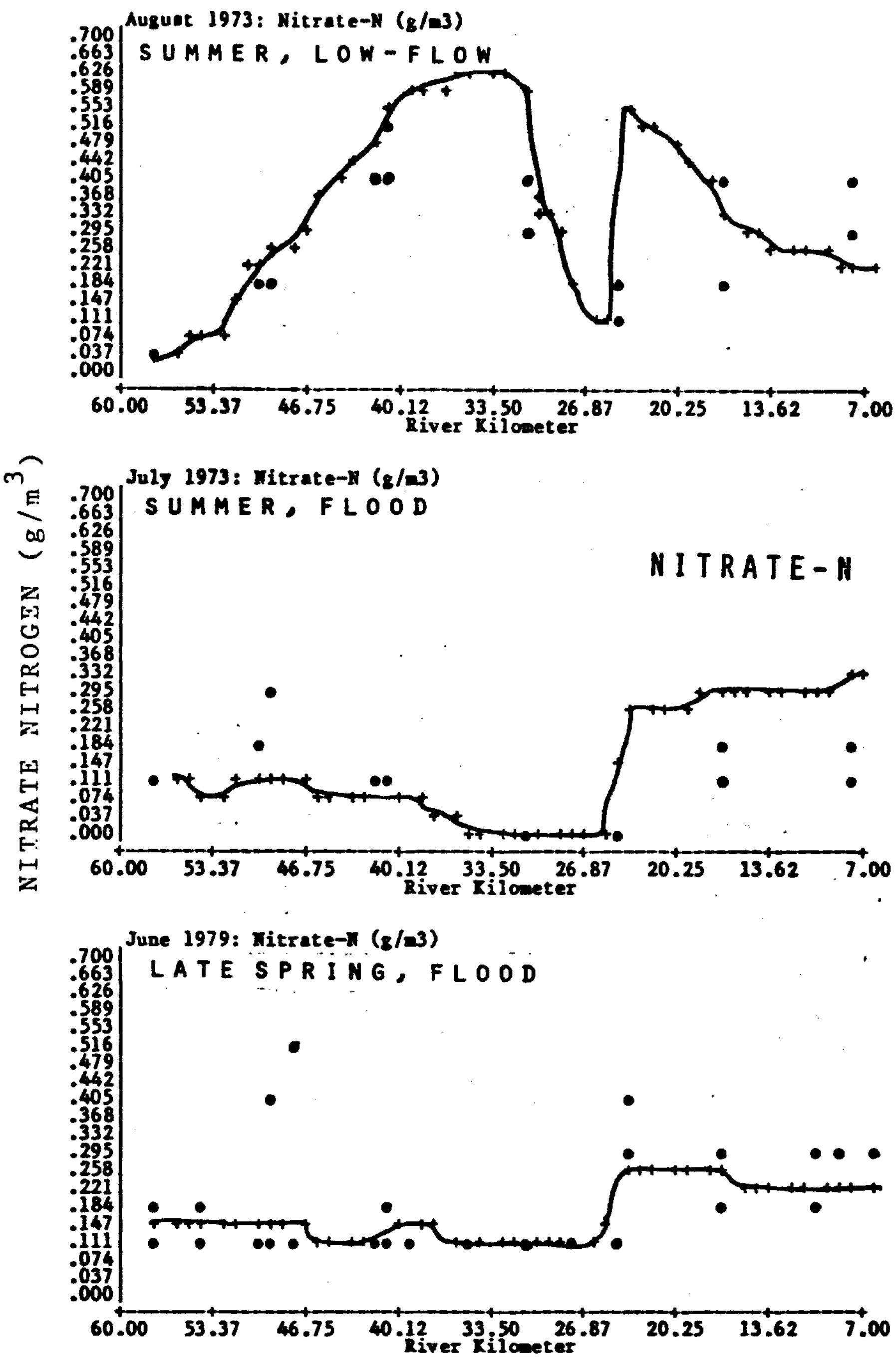


Figure 17

Observed and Predicted Nitrate-Nitrogen Profiles



lead to greater flow resistence, greater depths, and lower velocities; the last two factors would, in turn, reduce reaeration rates calculated using O'Connor-Dobbins (1958) formula.

In general, alternative combinations of wetland export oxygen demand concentration, overbank benthic demand, and overbank reaeration rate reduction may give approximately the same oxygen profile simulations for the various surveys and are thus indistinguishable based upon existing data. The selected parameter combination emphasizes the export component and is conservative for the simulation of diversion impacts because the export component of the organic loading to each model reach is dependent upon basin runoff and wetland area and is independent of upstream diversions (i.e., the organic matter from the wetlands reaches the river and is oxidized, regardless of upstream diversion or river elevation), whereas the overbank loading and reaeration mechanisms would be somewhat sensitive to diversions and elevations. Additional field studies and intensive monitoring would be required to develop an adequate data base for detailed discrimination among potential wetland impact mechanisms. Existing data support a conservative analysis based primarily upon export relationships of the type routinely used in modeling other types of land use/water quality relationships (Omernik, 1977, Meta Systems, 1982).

The wetland export model is based primarily upon a mass balance which relates the benthic oxygen demand and sources of organic nitrogen and organic phosphorus in each model reach to the tributary wetland drainage areas using a model of the following form:

$$Lij = R \; Awj \; Cwi$$

where,

Lij = loading of component i into reach j (g/sec)
R = basin unit discharge (cms/km2)
Awj = wetland area tributary to reach j (km2)
Cwi = concentration of component i in wetland drainage (g/m3)

The model is consistent with higher wetland loadings during periods of higher unit discharge (R) and in segments with larger areas of tributary wetlands. Since the average export concentrations (Cwi) are calibrated, they implicity include any bias attributed to differences in unit discharge (R) between wetland and upland drainage areas in the basin. Impacts on benthic demands are estimated by dividing the wetland loadings by the water surface area within each reach.

The estimation of wetland export concentrations has been guided, in part, based upon reasonable values for the nutrient contents of plant detritus and measured values of productivity and detritus export in other wetland systems. The total oxygen demand export of 125 mg/liter (120 mg/liter expressed as as benthic demand and 5 mg/liter as suspended BOD-U) corresponds roughly to 117 mg/liter of organic matter (assuming that the organic matter has the oxidation state of carbohydrate or CH2O). The export organic nitrogen and organic phosphorus concentrations of 3.5 mg/l and .5 mg/l correspond to detritus compositions of 3% nitrogen and .4% phosphorus, respectively, which are within the ranges of measurements for aquatic plants (Mackenthun, 1968).

At the average annual unit discharge of .016 cms/km2, the oxygen demand export concentration 125 mg/liter corresponds to an average annual export of 2 g/km2-sec, .17 g/m2-day, or .63 metric tons/hectare-year. In contrast, estimates of annual net primary productivity for freshwater macrophytes on fertile sites in temperature regions range from 30 to 45 tons/hectare-year for emergent species and from 1 to 7 metric tons/hectare-year for submersed species (Wetzel, 1975). The above oxygen demand export corresponds to only 1.3-2.1 percent of the productivity range for emergent species. The net productivity numbers reflect the potential biomass generated within adjacent wetlands; only a fraction of this biomass would be exported and exert an oxygen demand on the receiving water body; the remainder would (a) accumulate in place; (b) be decomposed in place; (c) accumulate in the receiving water as undecomposed organic sediment; or (d) be flushed downstream without decomposition (De la Cruz, 1978). Direct measurements of actual organic matter export from freshwater wetland systems are not readily available in the literature. Detritus exports of 3.4 tons/hectare-year were reported by De la Cruz (1965) for a Georgia salt marsh and 3.6 tons/hectare-year were reported by Heald (1969) for a mangrove estuary. Indirect export estimates from other wetland systems range from 0 to 50% of the annual net, above-ground primary productivity (De la Cruz, 1978). Thus, the calibrated export oxygen demand concentration of 125 mg/liter or equivalent annual export of .63 tons/ha-year is feasible in relation to literature values of wetland organic matter production and export.

A channel benthic BOD source of 4 g/m2-day has been included in Reach 6 (Sherman Bridge to Route 117) to account for increases in BOD concentrations between these locations, particularly during the low-flow, August 1973 survey. This increase may be attributed to unidentified point sources, sloughing of organics from benthic plants, or to wetland interactions not considered in the above export model.

The model includes a provision for simulating the effects of dam reaeration on oxygen levels. This option has been used for MDC#1 (RKM 58.5) for a small dam near the Mass Pike (53.3). Comparisons of data from above Saxonville Dam (RKM 51.2) and Elm Street (RKM 50.1) during the August 1973 and June 1979 indicated no increases in average oxygen concentrations, despite the considerable elevation drop over this short section (approximately 7 meters). Effects of intervening unmonitored sources, channelization for flood control, and/or diversions around the dam spillway may account for the apparent lack of reaeration in this section. Reasonable calibration to the oxygen profiles in this section was achieved without accounting for dam reaeration, although this has little effect on oxygen simulations below Pelham Island Road (RKM 42.3).

After calibration of the mean oxygen profile, benthic photosynthesis and respiration rates have been adjusted within reasonable ranges (Zison et al., 1978, Wetzel, 1975) to fit the observed daily minimum oxygen profiles. These rates reflect productivity by rooted aquatic plants, floating aquatic plants, periphyton, and aufwuchs communities. For lack of a better assumption, photosynthesis and respiration by these communities are assumed to be in balance under normal conditions; thus, the calibrated rates influence only the diel fluctuations of oxygen and not the daily mean values. Any impacts of the aquatic plant communities on the mean oxygen concentrations are implicit in the calibration of the net benthic demands discussed above. Calibrated photosynthesis and respiration rates range from 1.5 g/m2-day to 10 g/m2-day in the various reaches; the highest value is Fairhaven Bay, where aquatic plants are relatively abundant. A rate of 4 g/m2-day has been used for overbank areas to reflect productivity in the wetlands and shallow marginal waters during flooded periods.

Observed and predicted mean and minimum oxygen concentrations for each of the three surveys are compared in Figures 18 and 19, respectively. The model calibration is based upon data from the summer low-flow survey of August 1973 and the late-spring flood survey of June 1979. Simulations of the summer flood (July 1973) employ the same set of model parameters, with the exception that all benthic photosynthesis is inhibited; this is consistent with our "working understanding" of the response of the plant community to extreme summer floods, as described in the previous section.

Replicate data from one survey (June 1979) are available for calibration of the fecal coliform sub-model. This has involved adjustment of the effective decay rate and source concentrations in the Framingham area to match observed and predicted profiles, as shown in Figure 20. The calibrated decay rate (1.6/day) is within the range of literature values cited by Zison et al. (1978). The calibrated local drainage concentrations in the Framingham area (1000 - 5000 organisms/100 ml) probably reflect urban non-point sources. As indicated in Figure 20, observed fecal coliform levels are highly variable in the Billerica area (below river kilometer 13) and may also reflect urban impacts.

The calibrated model reproduces observed profiles with with reasonable accuracy and thus represents a useful tool for assessing impacts of diversions on downstream water quality. One limitation is that data from only one-low flow survey (August 1973) are available for model calibration and testing. It is possible that the wetland flooding event experienced in July of that year and/or effects of poorly quantified sewage loadings from a leaking pumping station in Framingham could have had residual effects on oxygen profiles measured during August. Summer oxygen profiles during low-flow periods which are not preceded by wetland flooding events may show higher concentrations and fewer violations of the 5 mg/liter criterion. Because of these considerations, the calibrated model may provide a conservative assessment of baseline conditions and diversion impacts with respect to dissolved oxygen. Additional surveys would be required to test this possibility. Another limitation is the lack of low-flow data for testing the fecal coliform model. Impact assessments indicate, however, that diversion strategies are more likely to be limited by dissolved oxygen impacts than by fecal coliform or chlorophyll-a impacts. Results of the simulations are summarized in Tables 7,8, and 9 and discussed in the main body of this report.

Figure 18

Observed and Predicted Daily-Mean Dissolved Oxygen Profiles

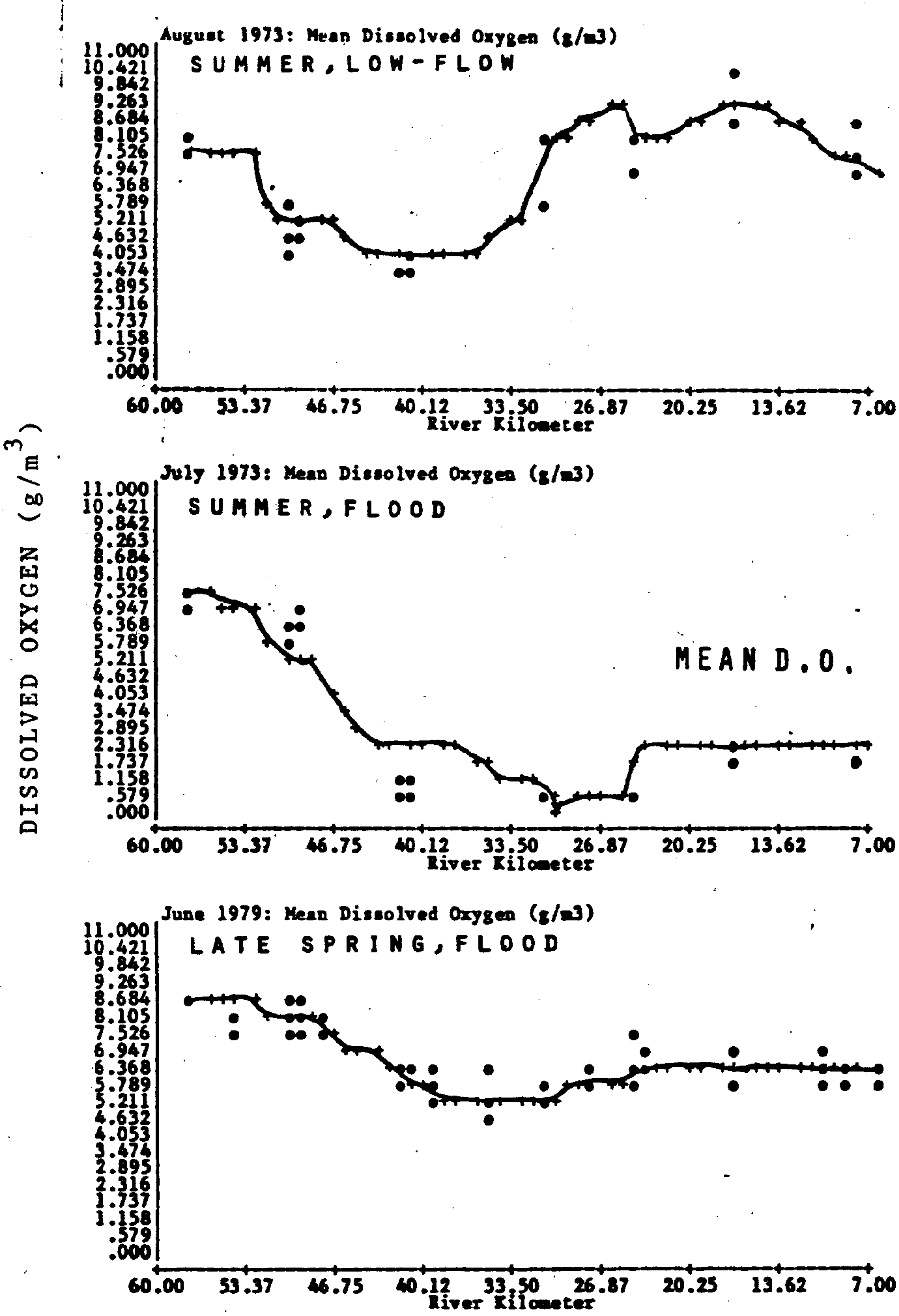


Figure 19

Observed and Predicted Daily-Minimum Dissolved Oxygen Profiles

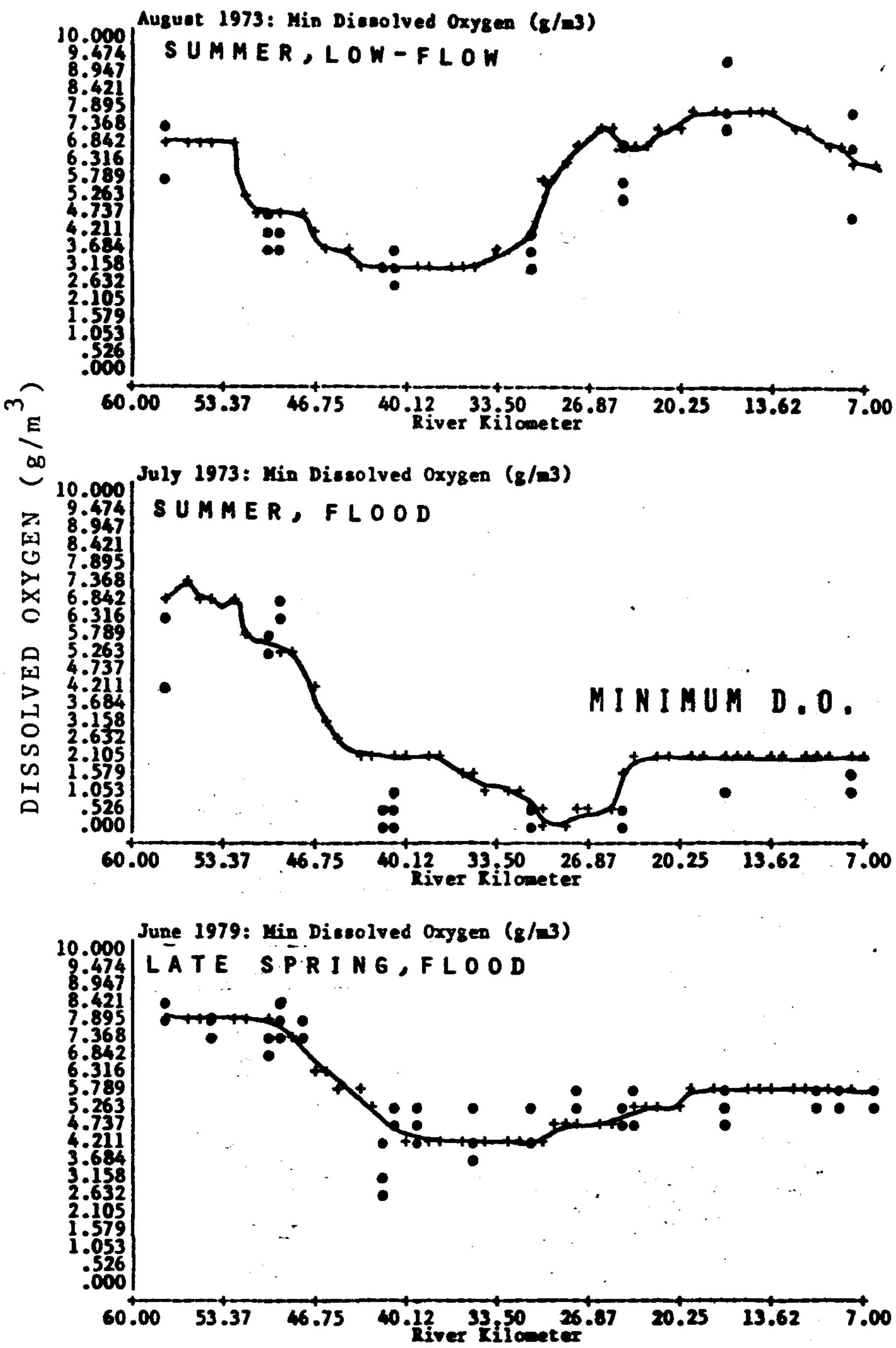


Figure 20

Observed and Predicted Fecal Coliform Profile

June 1979

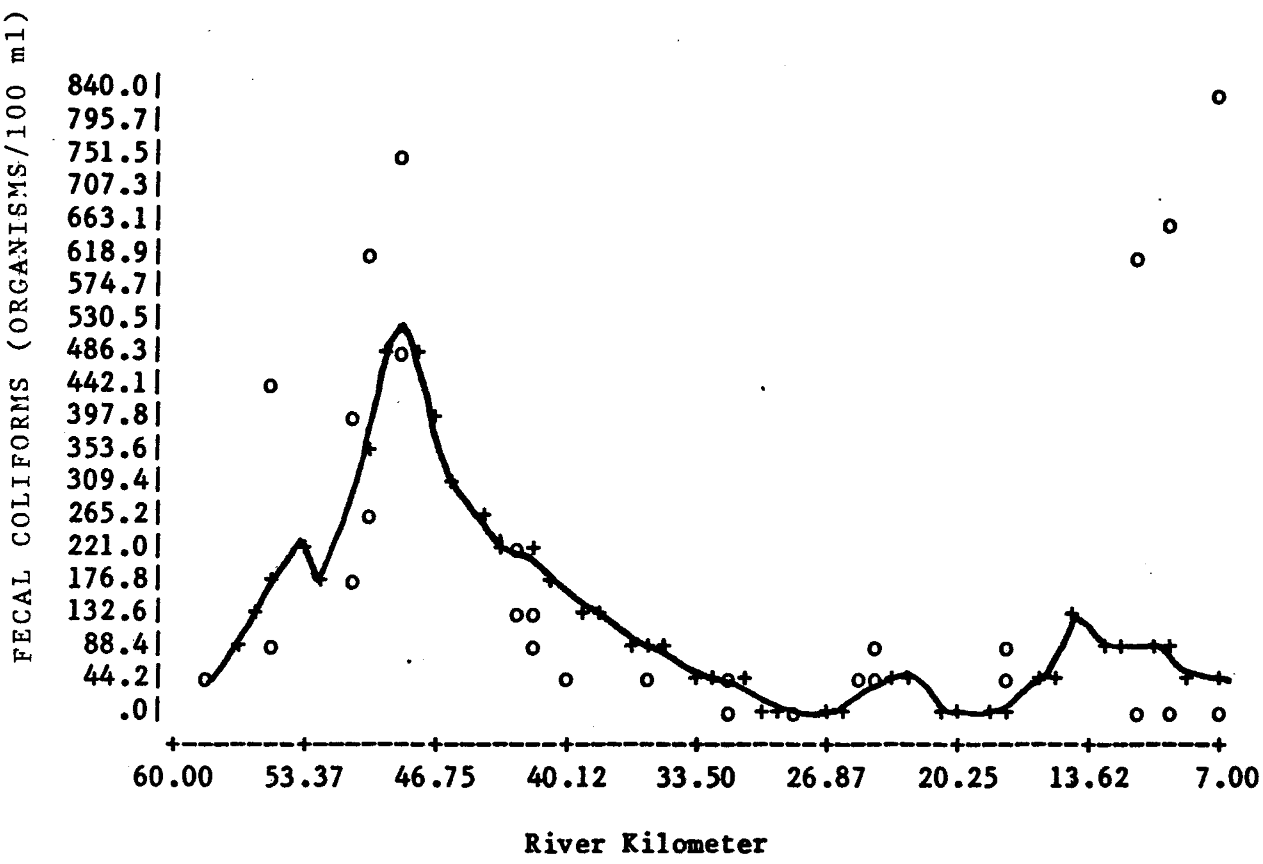


Table 7
Results of Water Quality Impact Simulations

Base Flow cms/km2	Diver. mgd	Total Flows Dam#1 cms	Total Flows Talbot cms	DISSOLVED OXYGEN Daily Mean Min	DISSOLVED OXYGEN Daily Mean RKM<5	DISSOLVED OXYGEN Daily Mean RKM<2	DISSOLVED OXYGEN Daily Minimum Min	DISSOLVED OXYGEN Daily Minimum RKM<5	DISSOLVED OXYGEN Daily Minimum RKM<2	Max Chl-a mg/m3	Max Fecal Coliforms (#/100ml)
Temperature = 26 degrees C											
.0033	0.	.64	3.34	4.3	10.7	0.	3.4	18.9	0.	37.2	262
.0033	6.	.37	3.07	3.9	11.9	0.	3.0	20.1	0.	38.9	352
.0033	12.	.10	2.80	3.5	12.7	0.	2.7	28.8	0.	40.3	477
.0106	0.	2.06	10.54	3.5	13.9	0.	2.2	20.0	0.	30.1	377
.0106	12.	1.52	10.00	3.2	14.6	0.	1.9	19.7	2.4	31.6	437
.0106	20.	1.16	9.64	2.9	15.0	0.	1.6	19.3	5.2	32.2	493
.0200	0.	3.88	19.84	3.6	19.2	0.	2.3	33.0	0.	16.9	429
.0200	12.	3.34	19.30	3.5	19.6	0.	2.1	33.0	0.	17.6	463
.0200	20.	2.98	18.94	3.4	20.0	0.	2.0	33.0	0.4	18.1	490
.0200	40.	2.08	18.04	3.1	21.2	0.	1.7	27.9	8.7	19.5	576
Temperature = 20 degrees C											
.0033	0.	.64	3.34	5.5	0.	0.	5.0	0.8	0.	28.4	291
.0033	6.	.37	3.07	5.4	0.	0.	4.9	6.4	0.	29.1	398
.0033	12.	.10	2.80	5.3	0.	0.	4.8	9.6	0.	30.2	557
.0106	0.	2.06	10.54	5.2	0.	0.	4.3	10.7	0.	14.3	414
.0106	12.	1.52	10.00	4.9	2.4	0.	4.0	11.2	0.	15.6	481
.0106	20.	1.16	9.64	4.6	6.4	0.	3.8	12.4	0.	16.4	542
.0200	0.	3.88	19.84	5.5	0.	0.	4.5	12.3	0.	6.9	468
.0200	12.	3.34	19.30	5.4	0.	0.	4.3	12.7	0.	7.2	506
.0200	20.	2.98	18.94	5.3	0.	0.	4.2	13.5	0.	7.4	536
.0200	40.	2.08	18.04	5.0	0.	0.	3.9	15.8	0.	8.0	630
Temperature = 17 degrees C											
.0033	0.	.64	3.34	6.3	0.	0.	5.8	0.	0.	19.7	305
.0033	6.	.37	3.07	6.1	0.	0.	5.7	0.	0.	20.6	420
.0033	12.	.10	2.80	6.0	0.	0.	5.6	0.	0.	20.7	596
.0106	0.	2.06	10.54	6.2	0.	0.	5.4	0.	0.	7.7	432
.0106	12.	1.52	10.00	5.9	0.	0.	5.1	0.	0.	8.2	503
.0106	20.	1.16	9.64	5.6	0.	0.	4.9	3.6	0.	8.6	567
.0200	0.	3.88	19.84	6.5	0.	0.	5.5	0.	0.	4.6	487
.0200	12.	3.34	19.30	6.3	0.	0.	5.4	0.	0.	4.7	527
.0200	20.	2.98	18.94	6.2	0.	0.	5.3	0.	0.	4.8	558
.0200	40.	2.08	18.04	6.0	0.	0.	5.0	0.	0.	5.0	658

(Continued)

Table 7
Results of Water Quality Impact Simulations (Continued)

Base Flow cms/km2	Diver. mgd	Total Flows Dam#1 cms	Total Flows Talbot cms	DISSOLVED OXYGEN Daily Mean Min	Daily Mean RKM<5	Daily Mean RKM<2	Daily Minimum Min	Daily Minimum RKM<5	Daily Minimum RKM<2	Max Chl-a mg/m3	Max Fecal Coliforms (#/100ml)
Temperature = 14 degrees C											
.0106	0.	2.06	10.54	7.1	0.	0.	6.5	0.	0.	4.4	451
.0106	20.	1.16	9.64	6.6	0.	0.	6.0	0.	0.	4.7	592
.0106	40.	0.26	8.74	5.8	0.	0.	5.2	0.	0.	5.3	852
.0200	0.	3.88	19.84	7.4	0.	0.	6.6	0.	0.	3.5	505
.0200	20.	2.98	18.94	7.2	0.	0.	6.4	0.	0.	3.6	580
.0200	40.	2.08	18.04	7.0	0.	0.	6.2	0.	0.	3.7	685
Temperature = 26 degrees C, No Plant Photosynthesis *											
.0200	0.	3.88	19.84	0.4	47.9	13.6	0.2	47.9	14.8	18.1	429
.0200	12.	3.34	19.30	0.3	47.9	14.8	0.1	47.9	16.4	18.9	463
.0200	20.	2.98	18.94	0.3	47.9	16.0	0.1	47.9	18.0	19.6	497
.0200	40.	2.08	18.04	0.3	47.9	19.2	0.0	48.7	20.0	21.0	576

Notes:

All simulations assume plant photosynthesis/respiration = June 79/August 73 = "normal" conditions, except *, which assumes July 1973 conditions (summer flood)

Base Flows:
.0033 cms/km2 = median august flow (HEC simulation)
.0106 cms/km2 = median fall flow (HEC simulation)
.0200 cms/km2 = late spring/early summer flow
(approx. equal to June 79 calibration)

RKM<5 = total river length violating oxygen criterion of 5 g/m3
RKM<2 = total river length violating oxygen criterion of 2 g/m3

Minimum dissolved oxygen concentrations generally located:
near Rte 20 for .0033 cms/km2 simulations
near Sherman Bridge for .0106 cms/km2 simulations
upstream of Rte 117 for .020 cms/km2 simulations

Maximum chlorophyll-a generally located below Assabet River
(see Table 3 for additional details)

Maximum Fecal Coliform levels located at Stone Bridge Road
(see Table 4 for additional details)

Table 8

Summary of Simulated Chlorophyll-a Concentrations

River Kilometer:		48.3	36.0	30.5	9.5		
Location:		Stone	Sherman	Below	Billerica	Chl-a	Maximum
Base Flow (cms/km2)	Diver. (mgd)	Bridge	Bridge	Fairhaven	Water Intake	Conc.	Location*
.0033	0	12.2	13.7	20.7	23.1	37.2	18.8
.0033	6	12.0	13.2	20.4	21.5	38.9	18.9
.0033	12	7.4	16.0	19.6	20.0	40.3	20.1
.0106	0	8.2	8.5	19.0	28.4	30.7	14.9
.0106	12	8.3	9.2	20.6	28.9	31.6	15.3
.0106	20	8.5	9.5	21.7	29.5	32.2	15.3
.0200	0	7.3	3.7	5.1	16.7	16.9	7.1
.0200	12	7.2	3.8	5.3	17.3	17.6	7.1
.0200	20	7.1	3.8	5.5	18.0	18.1	7.1
.0200	40	6.8	3.9	6.0	19.4	19.5	7.1

based upon 26 deg C simulations
concentrations in mg/m3
* river kilometer of maximum chlorophyll concentration

Table 9

Summary of Simulated Fecal Coliform Levels

River Kilometer:		48.3	42.8	36.0	30.5	9.5	Maximum	
Location:		Stone	Pelham	Sherman	Below	Billerica	Fecal	Count
Base Flow	Diver.	Bridge	Island	Bridge	Fairhaven	Water	Count	Loc.*
(cms/km2)	(mgd)	Road	Road	Road	Bay	Intake		
.0033	0	250	37	6	<1	36	262	48.7
.0033	6	352	23	4	<1	34	352	48.3
.0033	12	477	9	1	<1	31	477	48.3
.0106	0	377	84	21	1	55	377	48.3
.0106	12	437	79	19	1	57	437	48.3
.0106	20	493	73	16	1	55	493	48.3
.0200	0	429	97	28	3	50	429	48.3
.0200	12	463	97	27	3	52	463	48.3
.0200	20	490	96	23	3	52	490	48.3
.0200	40	576	93	22	3	52	576	48.3

based upon 26 deg C simulations
fecal coliform counts in organisms/100 ml
* river kilometer of maximum fecal coliform concentration

References

Banks, R.B., "Some Features of Wind Action on Shallow Lakes", Journal of the Environmental Engineering Division, ASCE, No. EE5, pp. 813-827, October 1975.

Concord Department of Natural Resources, "Concord River and Stream Water Quality Monitoring Project", Annual Reports, 1973-78.

De La Cruz, A.A., "A Study of Particulate Organic Detritus in a Georgia Salt Marsh-Estuarine Ecosystem", Ph.D. Dissertation, University of Georgia, Athens, Georgia, 1965.

De La Cruz, A.A., "Production and Transport of Detritus in Wetlands", in Wetland Functions and Values: The State of Our Understanding, Americal Water Resources Association, pp. 162-174, November 1978.

Fischer, H.B., E.J. List, R.C.Y. Koh, J. Imberger, Mixing in Inland and Coastal Waters, Academic Press, New York, 1979.

Heald, E.J., "The Production of Organic Detritus in a South Florida Estuary", Ph.D. Dissertation, University of Miami, Florida, 1969.

Heald, E.J., "The Production of Organic Detritus in a South Florida Estuary", Ph.D. Dissertation, University of Miami, Florida, 1969.

Kadlec, R.H. and J.A. Kadlec, "Wetlands and Water Quality", Wetland Functions and Values: The State of Our Understanding, Americal Water Resources Association, pp. 436-456, November 1978.

Linsley, R.K., M.A. Kohler, J.L.H. Paulhus, Hydrology for Engineers, McGraw-Hill, New York, Second Edition, 1975.

Mackenthun, K.M., The Practice of Water Pollution Biology, U.S. Department of Interior, FWPCA, 1969.

Manhattan College, "Basic Models of Natural Water Systems", prepared for U.S. Environmental Protection Agency, Contract No. 68-01-5918, 1968.

Massachusetts Department of Environmental Quality Engineering, "The Concord and Sudbury Rivers - 1973 Part A" Division of Water Pollution Control, Westborough, May 1974.

Massachusetts Department of Environmental Quality Engineering, "Water Quality Monitoring Program of the Metropolitan Area Planning Council's 208 Management Area 1977 Data Report" Water Quality and Research Section, Division of Water Pollution Control, Westborough, May 1980.

Massachusetts Department of Environmental Quality Engineering, "The Concord and Sudbury Rivers - 1979 Water Quality Data" Division of Water Pollution Control, Westborough, May 1980.

Massachusetts Department of Environmental Quality Engineering, "The Concord and Sudbury Rivers - 1979 Water Quality Data" Division of Water Pollution Control, Westborough, May 1980.

McGauhey, P.H., Engineering Management of Water Quality, McGraw-Hill, New York, 1968.

Meta Systems, Inc., "Documentation for the Meta Systems Version of the Qual-II Water Quality Simulation Model", prepared for U.S. Environmental Protection Agency, Water Planning Division, 1979.

Meta Systems, Inc., "Land Use/Water Quality Relationships in the West River System, Connecticut", final report, prepared for New Haven Water Company, Connecticut, 1982.

O'Connor, D.J. and W.E. Dobbins, "Mechanism of Reaeration in Natural Streams", Trans. ASCE, Vol. 123, 1958.

Omernik, J.M., "Nonpoint Source - Stream Nutrient Level Relationships: A Nationwide Study", EPA Ecological Research Series, Corvallis Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, Oregon, EPA-600/3-77-105, 1977.

Roesner, L.A., et al., "Computer Program Documentation for Stream Quality Model (QUAL-II)", U.S. Environmental Protection Agency, Center for Water Quality Modeling, Athens Environmental Research Laboratory, Georgia, EPA-600/9-81-014, 1981.

SAS Institute, "SAS Users Guide", Raleigh, NC, 1979.

Van Benschoten, J. and W. W. Walker, "Calibration and Application of QUAL-II to the Lower Winooski River", draft manuscript submitted to Water Resources Bulletin, American Water Resources Association, 1982.

Vermont Agency of Environmental Conservation, "Lower Winooski River Wasteload Allocation Study - Part B: Mathematical Modeling Report", Department of Water Resources and Environmental Engineering, Montpelier, January 1982.

Water Resources Engineers, Inc.; "Progress Report on Contract No. 68-01-0713, Upper Mississippi River Basin Model Project", prepared for U.S. Environmental Protection Agency, September 1972.

Wetzel, R.G., Limnology, W.B. Sunders Company Philadelphia, 1975.

Zison, S.W., W.B. Mills, D. Deimer, C.W. Chen, "Rates, Constants, and Kinetics in Surface Water Quality Modeling", U.S. Environmental Protection Agency, Athens Environmental Research Laboratory, Georgia, EPA-600/3-78-105, December 1978.