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Dry Weather Water Quality Sampling and Modeling
Blackstone River Feasibility Study

Phase 1: Water Quality Evaluation and Modeling of the MA Blackstone River

Submitted to

U.S. Army Corps of Engineers

Submitted by

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1. Introduction

As part of the feasibility study to investigate impacts of potential projects in the Massachusetts (MA) section of the Blackstone River, this project provides a detailed accounting of the current state of the river's water quality and builds on the work completed in the 1991 Blackstone River Initiative (BRI) (Wright et al. 2001). The main purpose of the program is to develop field and laboratory data that expands the steady-state water quality model used in the BRI and provide further model calibration and validation. In addition, a comprehensive water quality monitoring and modeling effort is used to evaluate existing water quality in two impoundments along the Blackstone River. The data is used to further document water quality problems, which may be corrected by projects identified as part of this study. This program is sponsored by the U.S. Army Corps of Engineers and will be referred to as the BAC for the Blackstone Army Corps program.

1.1 Background

In order to provide perspective into the selection of the Blackstone River for this study, an understanding of the evolution of the work on the river and the receiving estuary system is necessary. In 1987, the U.S. Environmental Protection Agency (EPA) created the Narragansett Bay Project (NBP) through the National Estuary Program. The NBP produced a Comprehensive Conservation and Management Plan (CCMP), which detailed present and future long-term management actions, which are carried forward by governmental and local agencies and authorities. One of the primary recommendations of the CCMP was a coordinated assessment and sampling project for the Blackstone River in Massachusetts and Rhode Island. In order to support and implement some of the recommendations in the CCMP, the EPA established the BRI in 1991. To further support the work on the BRI, the Governors of Massachusetts and Rhode Island signed a Memorandum of Understanding in 1992. This MOU underscored the importance of the river system and estuary. In response to the MOU, the MADEP selected the Blackstone River as one of four rivers to target in the development of a total maximum daily load allocation (TMDL) management plan.

Additional national recognition of the importance of the river system was provided through the United States Congress designating the Blackstone River Valley a National Heritage corridor in 1993 for its historical importance. The BRI was a multi-phased, interagency, interstate project to conduct the sampling, assessment, and modeling work necessary for restoration of the river system and to prevent further deterioration of the resources of Narragansett Bay.

The BRI was reviewed by the EPA Science Advisory Board (SAB), Ecological Processes and Effects Committee (EPEC) in March 1998. The SAB review (EPA-SAB-EPEC-98-011) can be found at http://www.epa.gov/science1/fiscal98.htm with the follow-up response. The findings of this review were a major factor in the development of the current BAC study.

Review of the Relative Results of the BRI: To understand the BAC goal, a brief summary of the BRI conclusions, as they relate to the BAC study, are summarized below. A complete list of the BRI conclusions and recommendations can be found in Wright et al. (2001).

BRI Conclusions

- The loadings from the headwaters are small relative to other sources along the river.
- The flow in the river to which the Upper Blackstone Wastewater Pollution Abatement District (UBWPAD) discharges is very low in these upper reaches, offering little dilution. Therefore, the characteristics of the effluent often determines the characteristics of the river at this point.
- Based on a comparison of mass loadings between the UBWPAD and Blackstone River, the UBWPAD is clearly the dominant source in these upper reaches relative to nutrients.
- Large diurnal swings in dissolved oxygen (DO) and pH are evident in the

impoundments downstream of Fisherville Pond to the MA/RI state line. These result from an increased algal productivity (represented by increased chlorophyll *a* concentrations). Along the river the growth of algae is first stimulated by the increase of nutrients from the UBWPAD, especially phosphorus, and the low velocities in the impoundments.

- Even with large diurnal swings, violations of DO water quality standards was not evident.
- High primary productivity is a result of phosphorus from the municipal wastewater facilities on the river. The major source of phosphorus in MA is the UBWPAD.
- The impoundments along the river serve to reduce velocities and increase the time of travel in the river reaches directly behind the dams. These conditions compound the problems presented by high levels of phosphorus by providing the appropriate hydraulic conditions for the growth of algae.
- The impoundments along the river in Massachusetts are sediment traps. These sediments are a major sink of oxygen and often govern the oxygen profile. This is especially true in the upstream reaches where productivity and instream nitrification are relatively small compared with the lower reaches.
- Model application to the 7Q10 flow indicated minor violations in Massachusetts upstream of four impoundments from high sediment oxygen demands.
- Based on a comparison of data from the early 1980s and this study, it was clear that the advanced wastewater treatment implemented in the mid 1980s at UBWPAD has made a significant improvement to the dissolved oxygen concentrations in the river. The improvements involved a reduction in the facility's discharge of CBOD and ammonia.

<u>BRI Recommendations</u>

- Dams and their current and future role in the Blackstone watershed are a complicated issue. Dams are having a negative impact on the river oxygen profile, as related to the discussion above on productivity and sediment oxygen demand. A ranking of the dams, based on their importance to dissolved oxygen, may be made. Those considered to be significant may require a comprehensive study similar to Rice City Pond.

- The QUAL2E model may be used to provide insight into the potential impact of dam modification or dam elimination on river oxygen profiles. At a minimum the model may be used to evaluate the impact of rerouting the river around the dams, for instance by utilizing the Blackstone Canal system around Fisherville Dam (BLK06) and Rice City Pond (BLK08). The result would be a lowering of the detention times in these reaches, which would have a positive impact on oxygen by decreasing the impacts of SOD, while also reducing the time available for algae growth.
- The sampling program for the 1991 surveys did not include DO measurements upstream of impoundments. The 7Q10 simulations indicate that there is the potential for violations in a number of impoundments in MA and RI, caused mostly by the SOD values measured during this initiative. A validation of these SOD rates is possible through an inexpensive DO monitoring effort during low flows upstream and downstream of these dams.

1.2 BAC Goals and Objectives

The BAC has, in part, directed its field program design and analysis to respond to the BRI recommendations given above. There are two phases to this project. Phase 1 includes several comprehensive water quality surveys on the Blackstone River in Massachusetts, data analysis and interpretation and an expansion of the BRI steady-state water quality model. Phase 2 is a series of monthly river impoundment surveys for the Fisherville and Rice City Pond drainage areas, data analysis and interpretation and application of a water quality model to the systems.

The objectives of the BAC Phase 1 included:

1. The reduction of the distance between measurements through an increase of the number of sampling stations.
2. A refinement of the modeling by dividing the river into smaller reaches.
3. The monitoring of the river under flows significantly different than those observed in the 1991 BRI (Wright et al. 2001).
4. An analysis of several water quality parameters including dissolved oxygen, biochemical oxygen demand (BOD), nutrients, and chlorophyll *a* for use in calibrating and validating the revised river model.
5. An evaluation of dam reaeration, sediment oxygen demand and long-term BODs as they relate to the revised river model.

The objectives of the BAC Phase 2 included:

1. The water quality and quantity monitoring of selected impoundments through the growing season
2. A refinement of the modeling by dividing the river impoundment into smaller reaches.
3. The assessment of the dissolved oxygen/nutrient interactions in these river segments through light and dark bottle studies, continuous DO measurements, and monthly sampling.

Phase 1; Water Quality Evaluation and Modeling of the MA Blackstone River

2.0 Study Site and Field Program Description

The design of the BAC Phase 1 expands on the field, laboratory and modeling protocols followed in the BRI (Wright et al., 2001). Section 2.1 is a general overview of the sampling strategy first described in the BRI for the Blackstone and modified here to the goals and objectives of this study.

2.1 General Field Program Design Considerations

Watershed Description - The design of the BAC considered available information on the river and its watershed including:

- the location, discharge history and permit requirements of all point sources;
- the location and history of known or potential non-point sources of pollution;
- the physical characteristics of the watershed including land use, river miles and drainage areas;
- all political boundaries; and
- the river uses including water supply, diversions, hydro power, navigation, fisheries, waste assimilation.

Water Quality Station Selection - The selection of water quality sampling stations should provide a sufficient number of sampling sites to adequately describe the impact of point and non-point sources of pollution on the river under dry weather conditions. The key point and non-point sources of pollution should be highlighted with the monitoring program.

The strategy is based on the following considerations: (a) a single sample at a cross section will adequately characterize the water quality, based on the assumption that the river at these sites is completely mixed; (b) the major sources of point and non-point pollution have been identified; (c) the location of all hydraulic structures and/or changes are known; and (d) the sampling stations are safely accessible at all time.

The location of the water quality stations should consider:

- the upstream and downstream boundaries (headwaters and mouth of the river);
- the major tributaries as defined by size or anticipated water quality impact;
- other key locations such as political boundaries;
- the major pollutant point sources (if resources allow, all point sources should be sampled during the study);
- the boundaries of key modeling reaches;
- sites immediately above the major tributaries and pollutant point sources;
- sites sufficiently below the major tributaries and pollutant point sources to assure complete mixing under both high and low flow conditions;
- sites further downstream below the major tributaries and pollutant point sources to observe the fate of the pollutants and the response of the river;
- sites above and below expected sources of non-point pollutants; and
- locations where there is significant change in cross-section that will have an impact on water quality and modeling decisions.

Water Quality Sampling Frequency - The water quality sampling frequency should include a sufficient number of samples to adequately describe the river's water quality and provide sufficient information concerning the river and its pollutant sources. The strategy is based on the following considerations: (a) point sources can be represented by a daily, flow-weighted composite; and (b) a daily river concentration can be represented by the average of multiple grab samples distributed over 24 hours.

The design of the dry weather sampling frequency should consider the following:

- each river survey should be conducted over at least a 24 hour period;
- each station should be sampled a minimum of 4 times in this period at 6 hour intervals (start time should coincide with predawn hours to observe the lowest oxygen concentrations);
- point sources should be sampled for five days leading up to, and including, the day of river sampling (these samples should be 24 hour composites, with samples taken hourly, and weighted by flow); and
- at least three surveys should be completed with significantly different flows.

Sample Collection and Handling - In the collection of the water quality samples, a field laboratory should be set up if sample holding time will be exceeded during sampling. This lab should be central to the watershed. The chain of command for field and laboratory crews must be clear. The location for sample collection should be clearly defined and consistent and procedures for sample collection and preservation must be defined and understood by all field crews. Laboratory analysis procedures and schedules should be available and understood. Staffing levels need to be sufficient for expected sampling duration. Field equipment needs to be inspected with adequate back-up equipment.

Flow Monitoring - Developing an understanding of the watershed hydrology and hydraulics before the final design of the water quality surveys is essential to the success of any study, given the typical constraints of time and money. One should schedule the field program between high spring flows and low summer flows.

Sufficient flow measurements should be determined at key locations to permit the development of accurate flow profiles. Cost of data collection should be a consideration in the final design of the water quality surveys. The data should be adequate to allow for the calibration and validation of a transport model.

The strategy is based on the following assumptions: (a) ground water inflow is not directly measured but, instead is back calculated from direct measurements of river and point source flows, assuming it is generally proportional to the drainage area; and (b) a dry weather survey is under steady-state conditions, which means that flows do not change significantly over the period of the survey.

Some river systems will already have flow monitoring occurring. The obvious and most extensive source of historic and current flows will be the United States Geological Survey's (USGS) permanent gauging stations. In addition, several other sources including federal, state or municipal agencies (ie. fisheries) or industries (ie. hydro power) may have data. The quantity, quality and format of this data needs to be determined. In addition to river flows, the quantity, quality and format of all point source data will also need to be determined.

Water Quality Computer Modeling - Many watershed studies will include modeling of some factor(s) affecting the quality of the water. Models can vary greatly in their level of sophistication, ranging from simple spreadsheet calculations to complex computer models. Perceived problems and regulatory mandates are the usual considerations central to the decision to model. Cost, time, and data availability determine the magnitude of the study.

Occasionally, a problem that initiates a model application may be identified during the course of a study. A decision to apply a computer model usually is made during the study's design phase. It is crucial that the aspects of the study described in the hydrology and hydraulics section be designed to collect the data needed for the construction, calibration, and validation of a model(s). When computer models for quantity and/or quality are to be applied, it is important, from a time and cost standpoint, that all the data necessary to support the model application is collected from the start of the study. Additionally, the legitimacy of the calibrated and validated models can be undermined by the lack of supporting data directly acquired.

Many authors have addressed the issues involved in applying a computer model and the appropriate applications for particular models. A means for identifying the most appropriate model and its data requirements should be a significant part of a study design. Some general recommendations include the following;

- that an application of a computer model have an explicit goal;
- expertise in the model should be available (a large study should not be regarded as an opportunity to learn how to use a model); and
- the model should be selected with a clear understanding of the study's goal and the available data with the understanding that additional model complexity can always be added if conditions warrant it (typically the more complex a model the more data intensive).

<u>Model Data Collection</u> - As in the collection of hydrologic data, collection of data for quality modeling will include background research into many sources of historic and current data. Federal, state, and municipal agencies may have data. Public and private dischargers will have point source quality data.

The results of previous modeling efforts, whatever the degree of success, can be a valuable reference tool. A more sophisticated modeling effort could be added onto the results of previous efforts. Additionally, previous modeling attempts or preliminary sampling may have highlighted a certain stream reach that has a particular problem worthy of further study. A more sophisticated application of a model could focus on this reach, rather than the entire length of the system, sharply reducing the number of samples to be taken.

Just as in the hydrologic data collection, data for quality models should, at a minimum, be collected for a period of time that spans the expected range of water column chemistry. It is important to capture data to the extremes of expected ranges.

Some supporting data for particular model applications may not be available from local sources. For instance, solar radiation values applied to plant growth. Some data may be available from federal agencies or private vendors at a price, but recent data from the immediate region of the study may not be readily available. Models may have default settings recommended, but judicious application of values obtained from other studies or extrapolation from historical or spatially remote data may also be necessary.

Special considerations must also be made for other complex sampling procedures. In general, this would include anything outside the experience of the organization doing a study. The study plan would then include consideration of using other resources with individuals more familiar with the processes involved, or training of personnel and purchase or building of new equipment.

Some sampling and analysis is best left to specialty organizations. Costs associated with certain analyses, safety, or specialized sample collection procedures would dictate that this data collection be assigned to those best able to deal with these issues. Some data (ie. sediment oxygen demand [SOD]) has limitations whether the data is collected in-situ or via sample extraction and laboratory analysis. An understanding of the requirements and difficulties associated with data collection has to be considered in modeling.

Model Selection and Calibration - As data is collected and tabulated, the quality of the data needs to be evaluated. Values need to be compared to expected ranges and to the other data acquired. Appropriate statistical analysis should be applied, and outliers identified and evaluated. Deficiencies in the sampling plan can be corrected before needless expense is incurred. Modifications needed to correct some possible skew in the data, must be built into the study.

Model applications should be accomplished in stages, rather than all at once. Generally, a model should be proven hydraulically, then for conservative constituents, and finally for non-conservative constituents. Conditions influencing the calculation of decay rates must be understood.

In developing a water quality study to calibrate and validate a model, the following questions should be asked during the study's design.

- What is an acceptable level of accuracy?
- How is that accuracy to be defined?
- Is the model sufficiently accurate for the purpose of the study (ie. 7Q10, first flush) and over the full range of observed values?
- Will it support regulatory decisions, or worst case, a challenge?

Model Application Considerations - After a computer model is satisfactorily calibrated and validated, it may be applied for development of regulations, interpretation of current quality, and prediction of future modifications. The final step to development of computer models could be a critical review of the model. Does it answer the questions that were central to the initiation of the study? Does it raise new questions that justify more sophisticated modeling?

2.2 BAC Field Program Design

Water Quality Station Selection - This study is focused only on the Blackstone River in Massachusetts. The watershed map with water quality river stations and WWTFs are presented in Figure 1. The lists of river sampling stations and WWTFs are summarized in Table 1 and are listed by GPS in Appendix A. A primary station was monitored 4-6 times in a 24 hour period, while a secondary station was only monitored twice. All stations were coded with the prefix BAC (Blackstone Army Corps stations). There were 24 river stations including 18 along the Blackstone River and 6 on tributaries. Twenty of these stations are primary; BAC01-04, BAC06-07, BAC10-23, and 4 are secondary; BAC05, BAC08-09,

BAC24. There were 5 WWTFs that were monitored; UBWPAD, Millbury, Grafton, Northbridge, and Uxbridge.

The detail provided by the BAC is a major expansion from the 1991 BRI. The BRI study included 12 river stations, 8 along the Blackstone River and 3 on tributaries. Only one WWTF was sampled in Massachusetts, UBWPAD. The comparison of BAC and BRI (water quality station coded with the prefix BLK) stations are presented in Table 2.

River Water Quality Sampling Frequency – This study includes four water quality surveys, labeled as DWS1-4. DWS1 was completed on October 21-22, 2000 and DWS2 on June 8-9, 2001. DWS1 and DWS2 included 4 runs in 24 hours for the 20 primary water quality stations and 2 runs for 4 secondary water quality stations. Thunderstorms interrupted the sampling during DWS3 August 10, 2001. Rainfall started in the central part of the watershed after the second run was completed. The survey was discontinued with only 2 runs complete. The decision was made to conduct a fourth river survey (DWS3a) on October 12, 2001. This included only two runs on the river. It was not used for model calibration or validation. A fifth water quality survey, labeled as DWS4, was completed on July 10-11, 2003. Six runs were made in 24 hours for the primary stations. All tributaries were considered to be secondary. The bridge at BAC02 had been removed by the state, a result of new road patterns during the road reconstruction in that area. No water quality samples could be taken from this station.

River Water Quality Parameters: A recommendation of the SAB was the expansion of the parameter listing to include the " … addition of several important parameters to the field program and sample analyses … (in) future BRI efforts in order to improve the dry and wet season condition assessments and the model results: a) dissolved organic matter; b) total phosphorus; and c) long-term BOD." The constituents included in the BAC reflect this SAB recommendation: total suspended solids (TSS); volatile suspended solids (VSS); total phosphorus (TP); dissolved ammonia (NH_3-N); dissolved nitrate (NO_3-N); dissolved orthophosphate (PO_4-P); pH; conductivity; chloride (Cl); unfiltered BOD_5 ($UBOD_5$); filtered BOD_5 ($FBOD_5$); dissolved oxygen (DO); temperature (T); and chlorophyll *a*. The later surveys (DWS2, 3, 3a, and 4) included filtered nitrogen-inhibited BOD_5 ($FNBOD_5$); and turbidity. Long-term BOD analyses are discussed in Chapter 7.

Table 2. Comparison of Water Quality Stations for the BAC and BRI

BAC Station	BRI Station	River	**BAC Station**	BRI Station	River
BAC01	BLK00	Blackstone	**BAC13**		Blackstone
BAC02	BLK01	Blackstone	**BAC14**		Blackstone
BAC03	BLK02	Blackstone	**BAC15**	BLK07	Blackstone
BAC04		Blackstone	**BAC16**		Blackstone
BAC05		Singletary	**BAC17**	BLK08	Blackstone
BAC06	BLK03	Blackstone	**BAC18**	BLK09	Mumford
BAC07	BLK04	Blackstone	**BAC19**		Blackstone
BAC08		Cold Spring	**BAC20**	BLK10	West
BAC09		Cronin Brook	**BAC21**	BLK11	Blackstone
BAC10		Blackstone	**BAC22**	Near BLK12	Blackstone
BAC11	BLK05	Quinsigamond	**BAC23**		Blackstone
BAC12	BLK06	Blackstone	**BAC24**		Blackstone

Wastewater Treatment Facility Sampling Frequency – This study sampled 5 WWTFs that included UBWPAD, Millbury, Grafton, Northbridge and Uxbridge. They were monitored three times during the study. In general, the sampling was conducted every day for five days leading up to the start of the river survey for DWS1 (October 16-20, 2000), DWS2 (June 4-8, 2001) and DWS3 (August 6-10, 2001).

The UBWPAD personnel provided a 24-hour composite sample of the effluent. The UBWPAD discharges its effluent into a concrete channel which flows approximately ¼ mile to the confluence with the Blackstone River. Essentially, all the water in the channel is from the treatment facility (Wright et al., 2001). This channel is about 10 feet wide and flows at a depth of around 2.5 to 3.0 feet. Grab samples were taken in the channel right below the effluent's discharge (referred to as the outfall sample), in the discharge channel just before the confluence with the Blackstone River (referred to as the downstream sample); and in the channel above the UBWPAD discharge (DWS1 only) (referred to as the upstream sample). At the Millbury WWTF, a grab sample was taken of the effluent and Millbury personnel provided a 24-hour composite sample once each week. At the Grafton WWTF a 24-hour composite sample of the effluent was taken from a composite sampler. At Northbridge and Uxbridge WWTF personnel provided a 24-hour composite sample of the effluent.

The WWTFs were not sampled for DWS3a or DWS4. However, for DWS4 in place of the critical UBWPAD composite, samples were taken in the discharge channel just below the effluent's release for river runs 1, 2, 4, 5, and 6.

<u>WWTF Parameters</u>: TSS; VSS; TP; NH_3-N; NO_3-N; PO_4-P; pH; conductivity; CL; $UBOD_5$ with WWTF Seed; and $FBOD_5$ with WWTF Seed. The later surveys DWS2 and 3 included $FNBOD_5$ and unfiltered nitrogen-inhibited BOD_5 ($UNBOD_5$).

For comparison, the 1991 BRI study included 3 dry weather water quality surveys: July 10-11, 1991, August 14-15, 1991 and October 2-3, 1991. River samples for dissolved oxygen were taken every six hours over the 48 hour period, and river samples for water chemistry determinations were taken four times in the first 24 hour period at six hour intervals. Effluent analyses for the UBWPAD were conducted on 24-hour composite samples collected daily for five days prior to the water quality surveys. The water quality parameters analyzed included the same parameters as listed above for the this study with some notable exceptions. The BRI analyses included total and dissolved metals (cadmium, chromium, copper, lead, and nickel), hardness (calcium and magnesium); toxicity; fecal coliform and only $UBOD_5$.

Sample Collection and Handling – This study collected all samples in Teflon buckets and pre-cleaned plastic bottles. Samples were stored in ice for transport to the Civil and Environmental Engineering Department, University of Rhode Island, Kingston, RI. Maximum holding time between the first sample of every run and the start of analysis was 3 hours. Details of the handling and preservation of the samples may be found in Appendix B Methods and Materials.

Flow Monitoring - The details of the flow profile development are given in Chapter 5.

Water Quality Model Selection – The dry weather data collected during the 1991 BRI was used to calibrate and validate QUAL2E to the entire Blackstone River (Wright et al., 2001). The goal of the BAC modeling effort was to expand the detail provided in the BRI version of the model for the Blackstone River in MA, by providing considerably more data for its calibration and validation. Also, the BAC field and modeling changes were designed to address concerns raised in the SAB.

3. Water Quality Results for River Stations and WWTFs

The measured data for the wastewater treatment facilities (WWTFs) and the river for each survey are organized differently in this report. A complete dataset for DWS1 through DWS3a can be found in Appendix A. The dataset for DWS4 is in Appendix B. The measured values from the WWTFs define boundary conditions of the model. Since QUAL2E is a steady-state model, an average daily value for each water quality constituent was assumed reasonable.

Samples from the WWTFs were 24-hour composites (described in Section 2.1), weighted typically by flow. Composites were available for 5 days leading up to the field survey. Since the UBWPAD is large in comparison to the river flow and has a significant affect on the river, care should be taken with the actual average used. For instance, if variability is high, should fewer days be considered? The BRI modeling effort used the 5 day average, because the model included the entire Blackstone River to its end in Pawtucket. The BAC, because of its concern for only the Massachusetts section, did not use the 5 day average. The UBWPAD composites were averaged twice in Table 3; (1) for all days (1-5 for DWS 2 and 3 and 1-6 for DWS1) and (2) for only days 3-5 (or 3-6 for DWS1). In a few instances the variability between the two averages was significant and would have a dramatic impact in the concentration at BAC03, directly below the discharge.

Why 3 days? Flow and velocity relationships (reported in Chapter 5 of the BRI) were used to determine the time of travel between UBWPAD and the MA/RI state line: DWS1 = 2.85 days; DWS2 = 2.52 days; DWS3 = 3.65 days and DWS4 = 2.77 days. The average was approximately 3 days. The composite averages used in the BAC model included only the 3 days leading up to the field survey. For instance if the field survey was on Friday to Saturday, samples for the 3rd, 4th, and 5th days (Wednesday, Thursday and Friday) were used for the average model input. Although the problem may only be apparent in the reaches directly below the UBWPAD, which include BAC03 and BAC04, to be consistent, days 3-5

Table 3. Averages at UBWPAD for DWS1, DWS2, and DWS3

	DWS1		DWS2		DWS3	
	Average Composite		Average Composite		Average Composite	
Point load	Days 1-6	Days 3-6	Days 1-5	Days 3-5	Days 1-5	Days 3-5
$FNBOD_5$	2.58 ± 1.01	2.85 ± 1.13	8.27 ± 3.81	6.07 ± 3.02	5.65 ± 2.84	3.95 ± 0.43
NH_3-N	3.76 ± 1.60	3.02 ± 1.42	11.8 ± 1.0	12.6 ± 0.94	2.58 ± 0.78	3.08 ± 0.52
NO_3-N	4.79 ± 1.59	4.79 ± 2.00	0.57 ± 0.29	0.67 ± 0.29	4.48 ± 1.07	3.96 ± 0.88
PO_4-P	1.50 ± 0.89	1.37 ± 1.12	0.57 ± 0.28	0.40 ± 0.07	0.85 ± 0.38	0.92 ± 0.42
Cl	114 ± 3.61	114 ± 4.35	153 ± 15.5	163 ± 2.84	118 ± 10.9	125 ± 1.53

FNBOD - Filtered Nitrogen Inhibited BOD

were also used for the other 4 WWTFs. The impact of these facilities on the river stations directly below their discharge is minor by comparison. The daily variability of these facilities are reported in Appendix B.

Table 4 summarizes the results of the UBWPAD discharge channel sampling for DWS1-3. A comparison of the outfall and downstream grab samples would show any water quality changes that may be a result of nitrification, deoxygenation, or settling, or additional sources that might occur within the ¼ mile discharge channel before the confluence with the Blackstone River. There were essentially no changes.

The sampling in the UBWPAD discharge channel during DWS4 provides an opportunity to view the variability in the effluent over 24 hrs. The five samples taken were averaged and reported in Table 5, along with the minimum and maximum. (All values can be found in Appendix B). The greatest range was with $FNBOD_5$: ND (1/2 detection limit or 0.25 mg/L) to 3.73 mg/L or 15 to 1. The other constituents were in order PO_4-P at 6 to 1; NH_3-N at 2 to 1; NO_3-N at 1.6 to 1; and Cl at 1 to 1. This indicates that a failure to

accurately match the mass balance of the UBWPAD discharge and the Blackstone River at BAC03 and BAC04 may be a result of the inability of using a 24 hr composite to describe the WWTF discharge and an average of grab samples to describe the daily average at each river station. The assumption that WWTF and river averages represent similar variability's may be unreliable for certain days and for certain constituents.

Table 5. Daily Variability at UBWPAD Discharge Channel for DWS4

Point load	UBWPAD-Discharge Channel Ave River-Run 1-5	Min	Max
$FNBOD_5$	1.85	ND	3.73
NH_3-N	4.27	2.89	6.00
NO_3-N	2.82	2.31	3.88
PO_4-P	1.07	0.29	1.71
Cl	178	173	182

Were the water quality conditions in 1991 (BRI) similar to the current study (BAC)? Table 6 compares the two surveys for the UBWPAD discharge (modified for the BRI to include days 3-5 only). The overall BRI and BAC survey average is reported in Table 7. The most significant difference (supported later in this study) is with NH_3-N. Where in the BRI, the facility was providing complete nitrification, it is evident that during the surveys for the BAC they were not. It is interesting to note that the PO_4-P loads were the same in 1991 (BRI) as they were in 2000-3 (BAC).

Tables 8-14 are a summary of the BAC averages reported for each station during the 4 DWS surveys. Appendix A and B have a complete listing of all constituents for all river runs and all daily composite values for all the WWTFs. These concentrations were coupled with the flow profiles developed in Chapter 5 to provide the observed mass profiles.

4. Model Description and River System Representation

QUAL2E is a steady state stream water quality model and it is primarily used to simulate dissolved oxygen (DO) and water quality parameters that influence DO concentrations. It assumes that the major transport mechanism is velocity in the direction of flow. Input to the model includes wastewater discharges, tributary flows, incremental flows and withdrawals. A complete discussion of the model's capabilities is available in the QUAL2E documentation. The river representation in the model includes computational elements (CE). The water quality within each element is assumed to be completely mixed. The output is then linked with the next element downstream. River reaches are defined by 1 to 20 elements that have the same hydraulic characteristics, stream slope, channel cross section, and biological and chemical rate constants (Brown and Barnwell 1987).

The equations in QUAL2E allow the input of the hydraulic characteristics of the river reaches as empirical equations: $u = aQ^b$ and $D = cQ^d$ where, u = stream velocity (ft/sec); Q = stream flow (cfs); D = stream depth (ft); and a, b, c and d are empirical constants. The DO balance in a stream system is a function of the internal sources and sinks and is represented by the differential equation shown below:

$$\frac{dC}{dt} = K_2\left(C_s - C\right) + (\alpha_3\mu - \alpha_4\rho)A - K_d L - K_4 / D - \alpha_5\beta_1 N_1 - \alpha_6\beta_2 N_2 \qquad (4.1)$$

where, C = the concentration of dissolved oxygen (mg/L); C_s = the saturation concentration of dissolved oxygen at the local temperature and pressure (mg/L); α_3 = the rate of oxygen production per unit of algal photosynthesis (mg-C/mg-A); α_4 = the rate of oxygen uptake per unit of algae respired (mg-C/mg-A); α_5 = the rate of oxygen uptake per unit of ammonia nitrogen oxidation (mg-C/mg-N); α_6 = the rate of oxygen uptake per unit of nitrite nitrogen oxidation (mg-C/mg-N); μ = algal growth rate, temperature dependent (day^{-1}); ρ = algal respiration rate, temperature dependent (day^{-1}); A = algal biomass concentration (mg-A/L);

L = concentration of ultimate carbonaceous BOD (mg/L); K_d = carbonaceous BOD deoxygenation rate, temperature dependent (day^{-1}); K_2 = the reaeration rate, temperature dependent (day^{-1}); K_4 = sediment oxygen demand, temperature dependent (g-C/ft^2-day); β_1 = ammonia oxidation rate coefficient, temperature dependent (day^{-1}); β_2 = nitrite oxidation rate coefficient, temperature dependent (day^{-1}); N_1 = ammonia nitrogen concentration (mg-N/L); and N_2 = nitrite nitrogen concentration (mg-N/L).

The growth and decay kinetics of algae are complex and involve many parameters in the mathematical formulations. Chlorophyll *a*, a component of algae, is used as an indicator to simulate algal kinetics. The algal biomass is then estimated based on the ratio of chlorophyll *a* to algal biomass. The change of algal biomass is represented in the model by:

$$\frac{dA}{dt} = \mu A - \rho A - \frac{\sigma_1}{D} A \qquad (4.2)$$

where σ_1 = the local settling rate (ft/day). The algal settling rates can be input by reach. The algal growth rate, μ, is a function of light, nitrogen, and phosphorus. These are represented by Monod functions and each may limit growth. QUAL2E has three options available to model μ. The option used is $\mu = \mu_{max}$ (FL)(FN)(FP) where μ_{max}= maximum specific growth rate (day^{-1}); FL = algal growth limitation factor for light; FN = algal growth limitation factor for nitrogen; and FP = algal growth limitation factor for phosphorus. The algal respiration rate, ρ, is defined by a single parameter in the model and is constant.

The BAC QUAL2E model provides an expanded model for the 27.6 miles of Blackstone River in Massachusetts. The 20 reach BAC model representation of the model is supported by 18 main river stations, 6 tributaries and 5 WWTFs. This is in comparison to the 15 reach BRI model, which was supported by 9 main river stations, 3 tributaries and 1 WWTF. On average, the BAC model provides a reach for every 1.4 miles with 7 computational elements, and an average drainage area of 10.4 square miles. In comparison, the BRI reaches were an average 1.8 miles with 9 computational elements, and an average drainage area of 14 square miles. In the BAC model, most reaches now start and end with a water quality station A detailed listing of the reaches, computational elements (CEs), drainage area and mile points can be found in Tables 16 and 17.

Table 15. Comparison between BAC and BRI Model Reaches

BAC	BRI	Reach Boundaries	BAC MP	Cumulative MP
1	1	BAC01 to BAC02	0.87	0.87
2	1	BAC02 to UBWPAD	1.42	2.29
3	2	UBWPAD to BAC04	1.66	3.95
4	2	BAC04 to BAC06	1.26	5.21
5	3	BAC06 to MWWTF	0.63	5.84
6	4	MWWTF to BAC07	0.87	6.71
7	5	BAC07 to BAC10	1.58	8.29
8	6	BAC10 to BAC12	1.26	9.55
9	7	BAC12 to BAC13	0.76	10.31
10	8	BAC13 to BAC14	2.05	12.36
11	8	BAC14 to BAC15	1.55	13.91
12	9	BAC15 to BAC16	0.87	14.78
13	9	BAC16 to NWWTF	1.74	16.52
14	10	NWWTF to BAC17	1.26	17.78
15	11	BAC17 to BAC19	1.45	19.23
16	12	BAC19 to Mumford River	0.31	19.54
17	13	Mumford River to BAC21	2.19	21.73
18	14	BAC21 to BAC22	4.11	25.84
19	15	BAC22 to BAC24	1.10	26.94
20	15	BAC24 to BAC23	0.63	27.57

MP - Mile Points

5. Flow Profiles

The flow profiles for all dry weather surveys were developed based upon flows from the USGS gauging stations situated in the Blackstone River drainage area (Quinsigamond River, Millbury, Northbridge, Branch River, and Woonsocket) and the wastewater treatment facilities (WWTFs). Table 18 summarizes the daily average flows. The WWTF flows were received from personnel at each facility.

Table 18. Daily Average Flows (cfs) for BAC Surveys

Location	DWS1	DWS2	DWS3	DWS4
Quinsigamond River *	11.0	35.0	5.90	16.0
Millbury*	NI	NI	NI	80.0
Northbridge*	125	163	79.0	NA
Branch River*	116	103	26.0	60.9
Woonsocket*	327	478	174	344
UBWPAD	44.6	51.0	45.4	48.6
Millbury WWTF	1.69	2.05	1.96	1.69
Grafton WWTF	1.86	2.27	2.00	1.86
Northbridge WWTF	1.39	1.70	1.08	1.39
Uxbridge WWTF	0.85	1.01	0.90	0.85

* = USGS gauging station; NI = Not Installed, NA = Not Active

For each dry weather survey, two different incremental river reach flows (q) were computed in order to take advantage of the Northbridge and Woonsocket gauging stations. The first river segment ranges from the headwater (BAC01) to Northbridge (to Millbury for DWS4). The second river segment was from Northbridge (or Millbury) to Woonsocket. The following two formulas were used to calculate the incremental river reach flow q (cfs/mi^2):

$$q_1 = \frac{Q_{N/M} - (Q_Q + Q_{WWTF1})}{DA_{N/M} - DA_Q} \tag{5.1}$$

in which $Q_{N/M}$ = flow at the Northbridge or Millbury gauge; Q_Q = flow at the Quinsigamond gauge; Q_{WWTF1} = flows from the WWTFs above the Northbridge or Millbury gauge; $DA_{N/M}$ = drainage area at the Northbridge or Millbury gauge; and DA_Q = drainage area at the Quinsigamond gauge.

$$q_2 = \frac{Q_W - (Q_{N/M} + Q_Q + Q_B + Q_{WWTF2})}{DA_W - (DA_{N/M} + DA_Q + DA_B)} \tag{5.2}$$

in which Q_W = flow at the Woonsocket gauge; Q_Q = flow at the Quinsigamond gauge; Q_B = flow at the Branch gauge; Q_{WWF2} = flows from the WWTFs between the Northbridge or Millbury gauge and the Woonsocket gauge; DA_w = drainage area at the Woonsocket gauge; and DA_B = drainage area at the Branch gauge.

An example of the application of these equations is given in Table 19 for DWS1. The daily average flows and drainage areas are given for the Northbridge, Quinsigamond, Branch and Woonsocket stations from data reported by the USGS. The wastewater flows, along with the river flows, are entered into equations 5.1 and 5.2, resulting in incremental inflows of 0.56 cfs/mi^2 for the area above the Northbridge gauge, and 0.46 cfs/mi^2 for the area between the Northbridge and Woonsocket gauges.

The incremental inflows are then used to develop the flows in the river by computational element. Table 20 shows the flow profile for the example divided into reaches. Column 3 is the flow reported for this day at the Quinsigamond River USGS gauge. The area below this gauge, 8.63 mi^2, is multiplied by the appropriate incremental inflow to obtain the flow from this tributary between the gauge and the confluence with the Blackstone

Table 19. Example Determination of the Incremental River Reach Flows for DWS1

Incremental flow (q_1) from the Headwater to USGS Station Northbridge

$q_1 = Q_N-(Q_Q+Q_{WWTF1})/(DA_N-DA_Q) =$	0.56cfs/mi^2

Flows (Q)	cfs
Northbridge USGS Station (Q_N)	125
Quinsigamond River USGS Station (Q_Q)	11.0
All WWTF above USGS Station Northbridge (Q_{WWTF1})	48.2

WWTF	cfs
UBWPAD	44.6
Millbury	1.69
Grafton	1.86
Q_{WWTF1}	48.2

Drainage area (DA)	mi^2
USGS Northbridge (DA_N)	143
USGS Quinsigamond (DA_Q)	25.6

Incremental flow (q_2) from USGS Station Northbridge to USGS Station Woonsocket

$q_2 = Q_W-(Q_N+Q_B+Q_{WWTF2})/(DA_W-(DA_N+DA_B) =$	0.46cfs/mi^2

Flows (Q)	cfs
Woonsocket USGS Station (Q_W)	327
Northbridge USGS Station (Q_N)	125
Branch River USGS Station (Q_B)	116
All WWTF below USGS Station Northbridge and above	2.24

WWTF	cfs
Northbridge	1.39
Uxbridge	0.85
Q_{WWTF2}	2.24

Drainage area (DA)	mi^2
USGS Woonsocket (DA_W)	416
USGS Northbridge (DA_N)	143
USGS Branch River (DA_B)	91.2

River. The entire drainage area of all the ungauged tributaries was handled in a similar manner. These are reported in column 4. The final tributary flows are reported in column 5. The flow attributed to direct drainage from areas that are not included in the tributaries is determined in column 6. The headwater flows are also reported here. The contribution from the WWTFs is given in column 7. The summation of all the contributions gives the flow profile in column 8. Table 21 summarizes the incremental flow rate by survey, and Table 22 is a summary of the flow at each station for each survey. These flows coupled with the river concentrations provide the mass profiles for each constituent.

Flows for both this study and for the 1991 BRI are plotted in Figure 2. Since the BRI station BLK12 and this study's BAC22 are within 0.2 miles, it is easiest to compare the flows from 1991 and this project with these stations. In 1991, flows for the three dry weather surveys were 102, 112 and 460 cfs (Wright et al, 2001). In 2000-3, flows were 126, 184, 235, and 307 cfs. The 7Q10 flow is approximately 115 cfs. The range of flows provides an excellent test of the model.

A significant amount of the river flow at BAC03 is from the discharge at the UBWPAD; about 54% DWS1, 55% DWS2, 76% DWS3, and 64% DWS4. The dilution ratio of river flow to wastewater flow (Q_R:Q_{UBWPAD}) is less than 1 (Q_R) to 1 (Q_{UBWPAD}); approximately 0.9:1 for DWS1, 0.8:1 for DWS2, 0.3:1 for DWS3, and 0.6 to 1 for DWS4.

Before the flow profiles are accepted, they are tested with chloride, a conservative tracer. Chloride is both chemically and biologically stable and indifferent to adsorptive processes and, therefore, settling and resuspension. These properties make chloride a valuable tool in evaluating flows, since dilution is the factor controlling stream concentration. The BAC and BRI chloride concentration profiles are similar, and are shown in Figure 3. Figure 4 is a composite of the BAC and BRI studies representing chloride mass (lbs/day). Again, the range covered by the 7 surveys is excellent, and will provide a reasonable test of the model.

Before modeling the system for chloride, a concentration of chloride had to be assigned to the incremental inflow into each computational element. Since the Blackstone River and its major tributaries are effluent streams, it has been assumed that tributaries without significant point and non-point sources will provide an estimate into groundwater contributions. With 6 tributaries monitored in the study, the decision to apply them to the closest upstream drainage areas results in the following: Singletary Brook is applied to Reaches 1 to 6; the average of Spring Brook, Cronin Brook and Quinsigamond River, weighted by drainage area, is applied to Reaches 7 to 11; and the average of Mumford and West Rivers, weighted by drainage area, is applied to Reaches 12 to 20. This procedure is followed for every constituent that is modeled in this report.

There was an undefined source of chloride between BAC01 and BAC02. This was observed for DWS1, 2, 3 and 3a. This would not have been a problem except the bridge at BAC02 was removed before DWS4 and a sample could not be taken. As a result, either a prediction needed to be made of the amount of chloride to be added to BAC01 for DWS4, or the modeling for chloride would need to start at BAC03. Both options were tried. First, the mass loadings from the two stations (BAC01 and BAC02) were evaluated for the 4 surveys using least-squared regression. The result was a statistically significant relationship (R^2 = 0.99): BAC02 = 1.6145 (BAC01) – 1261. With the DWS4 value at BAC01, the equation was used to predict the loadings at BAC02.

The results of the chloride modeling (Figures 5 and 6) are excellent for DWS 1, 2 and 3 and support the procedure used to develop the flow profiles (statistical results are given in Chapter 10). For DWS4, the predicted profile parallels the observed profile, but underpredicts it. The underprediction could be a result of the value estimated by the equation. Since the 4 surveys used in the regression were taken in 2000-1, the source may have increased by 2003 (DWS4). If the simulation for DWS4 was to begin at BAC03, the model prediction to observation would be acceptable.

6. River and Dam Reaeration

Reaeration occurs in the Blackstone River under natural conditions and under conditions facilitated by dams. Eight dams are situated on the Blackstone River between BAC04 and BAC19. Table 23 lists the characteristics of the dams, which were included in the model. The information for the dams was obtained from the BRI and from the Army Corps report in 1994 (USACE 1994). The first two dams, New England Power and Singing Dams (Figure 7), are located in a critical river segment (BAC03 to BAC07) that has impaired water quality. This will be proven in a later chapter.

Table 23. Summary of the Stream Dam Characteristics

#	Dam	Reach	Elements	ADAM	BDAM	HDAM
1	New England Power Co.	5	3	1.60	0.70	15*
2	Singing	7	1	1.60	0.70	10*
3	Wilkinson	7	4	1.60	0.70	4
4	Saundersville	7	7	1.60	0.70	4
5	Fisherville	8	6	1.60	0.70	10*
6	Farnumsville	9	4	1.60	0.70	4
7	Riverdale	12	1	1.60	1.05	14*
8	Rice City Pond	15	1	1.60	0.70	21*

Dam reaeration is computed with following equation from QUAL2E:

$$D_a - D_b = \left| 1 - \frac{1}{1 + 0.116 * a * b * H(1 - 0.034 * H) * (1 + 0.046 * T)} \right| * D_a \qquad (6.1)$$

where, D_a = the oxygen deficit above dam (mg/L); D_b = the oxygen deficit below dam (mg/L); T = the temperature (ºC); H = HDAM, the height through which water falls (ft);

a = ADAM, the empirical water quality factor (1.8 in clean water, 1.6 in slightly polluted water, 1.0 in moderately polluted water, and 0.65 in grossly polluted water); b = BDAM, the empirical dam aeration coefficient (0.7 to 0.9 for flat broad crested weir, 1.05 for sharp crested weir with straight slope face, 0.8 for sharp crested weir with vertical face, and 0.05 for sluice gates with submerged discharge).

Since direct measurements for computation of the reaeration rate was not conducted, commonly used formulas for computation of the reaeration in natural streams were utilized. The Blackstone River can be mostly considered as a slow moving stream with somewhat isotropic conditions. The equation by O'Connor and Dobbins was selected for application in the 1991 modeling effort. Modelers have used this equation to represent moderate to deep streams, with moderate to low velocities (Chapra 1997; Brown and Barnwell 1987):

$$K_2^{20} = \frac{(D_m \overline{u})^{0.5}}{d^{1.25}} * 2.31 \qquad (6.1)$$

in which, D_m = molecular diffusion coefficient, ft^2/sec; $\overline{u}$ = mean velocity, ft/sec; d = mean depth, ft.

This equation was used for the reaches 1 through 3 and 5 through 20. It was not used for Reach 4 between BAC04 and BAC06. Fast moving riffles were observed in this reach with depths of approximately 1-2 feet. The photograph in Figure 8 is typical for this reach. In order to take into account the reaches characteristics, the reaeration equation by Owen et al. (1964) was selected. This equation was developed based upon data from Churchill et al. (1962) and is often used for shallow, fast moving streams (Brown and Barnwell 1987):

$$K_2^{20} = 9.4 * \frac{\overline{u}^{0.67}}{d^{1.85}} * 2.31 \qquad (6.2)$$

7. BOD Simulations

Total BOD or unfiltered BOD (UBOD) will include particulate and dissolved BOD and nitrification. This is misleading if the intent is to account for impacts of carbonaceous BOD (CBOD) on stream DO. The loss of particulate BOD to settling is not an immediate oxygen demand in the water column. Instead, it becomes part of the bottom sediments and is incorporated in the sediment oxygen demand. Nitrification should be handled separately.

Filtered BOD (FBOD) will eliminate the particulate BOD, but not nitrification which will be included in the loss of oxygen. This leads to an increase in the BOD value and a higher prediction of the ultimate demand associated with CBOD. Nitrogen-inhibited filtered BOD (FNBOD) provides the best representation of CBOD. The difference between the filtered and nitrogen-inhibited filtered BOD is illustrated in Figure 9.

The nitrogen inhibited BODs were determined by adding 3 ml of POLYSEED®NX-CBOD5 Seed Ioculum (InterBio®). The poly seed has a chemical additive for the inhibition of ammonia nitrogen in water samples. It is formulated as a seed population for the $CBOD_5$ test as conducted according to Standard Methods for the Examination of Water and Wastewater. A detailed procedure is given in the Appendix B. The various types of BOD analyzed for river and WWTF samples are given in Table 24.

7.1 Long Term BOD

In QUAL2E, the global rate converting $CBOD_5$ to ultimate CBOD ($CBOD_u$) is k_1. In the BRI, this was defaulted to the value suggested in the model of 0.25 day^{-1} base e at 20ºC. A major concern of the SAB was the use of this rate and the failure to measure it in the river. Although no values were suggested by the SAB during the review, the SAB did present work that indicated a reaction value of 0.07 day^{-1} base e at 20ºC. A comprehensive sensitivity analysis was completed in the BRI, for values ranging from 0.1 to 0.5 day^{-1} base e at 20ºC.

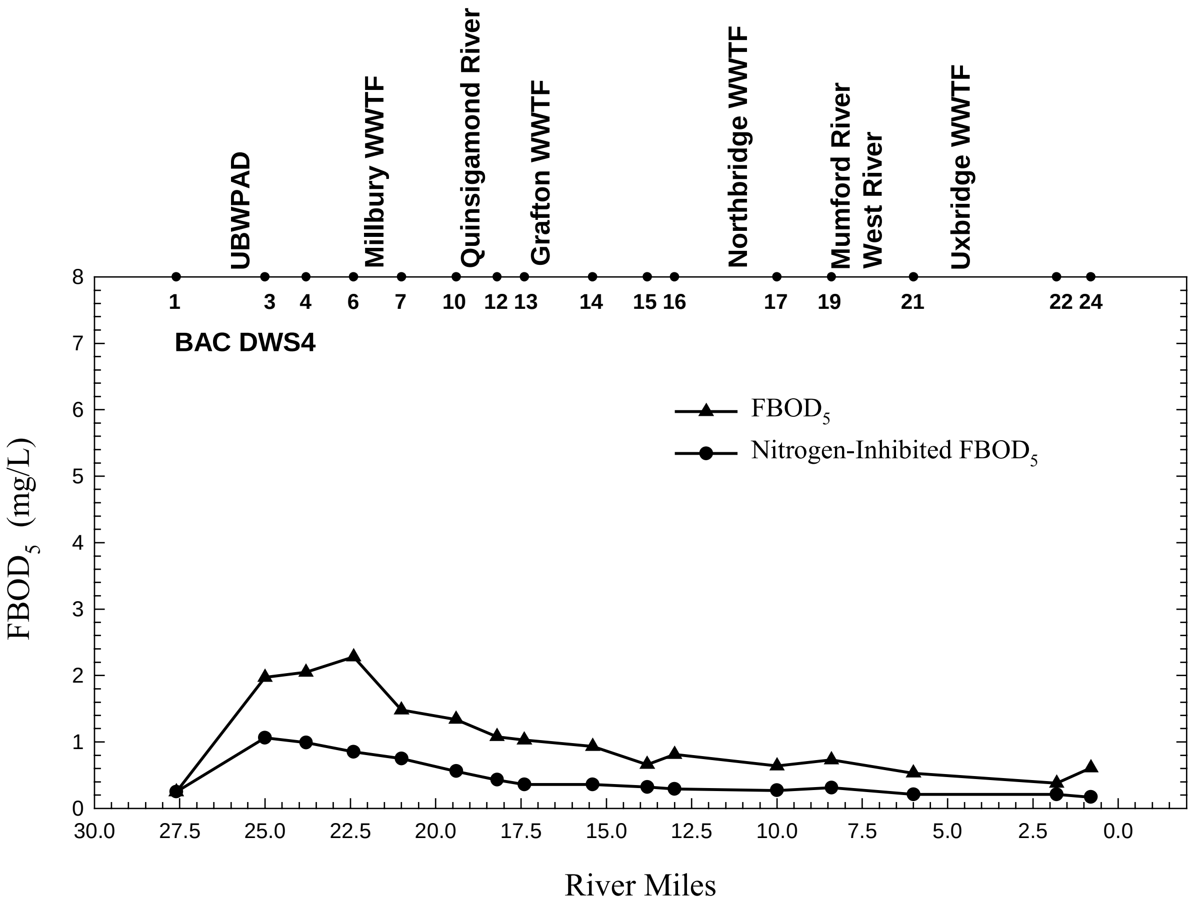

Figure 9. Comparison Between Filtered and Nitrogen-Inhibited Filtered BOD_5 for DWS4.

The conclusion was that the impact on the Blackstone River's DO was insignificant for this range of k_1's and for the typical BOD values reported in 1991, which were less than 2.0 mg/L UBOD. The difference in DO was less than 0.1 mg/L for the three 1991 surveys.

The purpose for determining long-term biochemical oxygen demand in this study was to remove this as a modeling concern. Therefore, it was the goal to generate the first order reaction rate (k_1) at several locations in the river to allow conversion of the $CBOD_5$ to $CBOD_u$. Six stations were selected to cover the Blackstone River in Massachusetts

Table 24. Summary of BOD_5 Analyzed for each DWS

River	$FBOD_5$	$UBOD_5$	$FNBOD_5$	$UNBOD_5$
DWS1	✓	✓	NA	NA
DWS2	✓	✓	NA	NA
DWS3	✓	✓	✓	NA
DWS4	✓	NA	✓	NA

WWTF	$FBOD_5$	$UBOD_5$	$FNBOD_5$	$UNBOD_5$
DWS1	✓	✓	NA	NA
DWS2	✓*[1]	✓*[1]	✓	✓
DWS3	✓*[2]	✓*[2]	✓*[3]	✓
DWS4	✓*[5]	NA	✓*4	NA

*[1] Day 4 Only; *[2] Days 3 and 4 Only; *[3] Days 1 and 2 Only;
*[4] UBWPAD Channel Only; NA = Not analyzed;
F = Filtered; U = Unfiltered; FN = Filtered Nitrogen Inhibited;
UN = Unfiltered Nitrogen Inhibited

from its headwaters to the MA/RI state line. They were:

1. BAC02 Above the UBWPAD wastewater discharge
2. BAC06 Between the UBWPAD and Millbury WWTFs
3. BAC12 Below Fisherville Pond
4. BAC15 In Riverdale Pond
5. BAC17 At the end of Rice City Pond
6. BAC21 After the confluence with the West and Mumford Rivers

Samples were taken on three days: August 10, 2001; July 26, 2002; and May 30, 2002 between 0800 and 1200. The laboratory analysis included UBOD, FBOD, FNBOD (with 3 ml of polyseed), with dilutions of 100 and 150 ml samples. Long-term analysis included the determination of BOD at 5, 7, 10, 15 and 30 days. The results for the FNBOD with 150 ml dilution are reported in Table 25.

The daily difference method was used to determine the first order reaction rate of CBOD (k_1) and the ultimate carbonaceous BOD_u (Tsviglou 1958; Gaudy, Jr, et al. 1967; Bowie et al. 1985). An example of this method is presented in Table 26 and Figure 10. A complete set of all graphs, tables and regression analyses are presented in Appendix B. The final results are presented in Table 27.

The overall average for the three sampling periods was 0.0848 day^{-1} base e at 20ºC for k_1 and 6.04 mg/L for $CBOD_u$. The reaction rate ranged from a low of 0.0484 to a high of 0.1215 day^{-1} base e at 20ºC. The $CBOD_u$ ranged from 4.44 to 7.87 mg/L.

7.2 BOD_5 – Modeling

Since the BRI only measured $UBOD_5$, this form of BOD is the only form used for comparison with the BAC values in Figures 11 and 12. The BOD_5 values measured during DWS2 were discarded due to equipment malfunction. The BOD_5 values measured in DWS3 were only sampled twice over 24 hours and average daily values at river stations close to the UBWPAD discharge are not well represented by a 2 sample average because of the high daily variability in the UBWPAD discharge (Tables 4 and 5). DWS4 is the only data set available to have acceptable $FNBOD_5$ data. DWS1 has $FBOD_5$ data but no $FNBOD_5$. A conversion of the $FBOD_5$ observations for DWS1 into $FNBOD_5$ values was made.

The decay of carbonaceous BOD_5 is usually governed by the initial concentration of carbonaceous BOD_5, the time of travel, and two rates. The deoxygenation rate (K_d) controls

the decay of carbonaceous BOD_5 and is directly related to heterotrophic bacteria, which decay organic matter. The settling removal rate (K_s) controls the loss of carbonaceous BOD_5 due to settling and depends upon the velocity and depth of the stream. Since the samples for the CBOD modeling were filtered, the settling removal rate was not defined in the model. Given similar deoxygenation rates, high concentrations of carbonaceous BOD_5 and long (slow) times of travel usually result in high dissolved oxygen losses, and, in contrast, very low concentrations of carbonaceous BOD_5 and a very short (fast) time of travel usually result in low dissolved oxygen losses.

Like most biological rates, the deoxygenation rate strongly depends upon temperature. Therefore, the actual measured temperature during the survey has to be entered in initial conditions of the QUAL2E input file.

The impact of temperature is directly related to the magnitude of heterotrophic bacteria (living either in river water or in benthos,) activity and efficacy. The impact by bacteria attached to the benthos can be the most governing effect for decay of carbonaceous BOD_5 in shallow streams and, therefore, may exceed the impact of bacteria in the water column (Chapra 1997). Between BAC01 and BAC17, the Blackstone River is essentially a shallow river, except for areas directly upstream of dams. In this part of the river, the depth varies between 1 and 3 feet under low-flow, dry weather conditions. Mesotrophic bacteria, which heterotrophic bacteria also belong to, are active in the range of 5 to 35°C. The maximal activity of these bacteria will be reached at approximately 37°C, which cannot be encountered in the Blackstone. The survey temperature was determined in the field for each sample taken and the average daily value was input into the model. Since the input for all reaction rates for QUAL2E are adjusted to 20°C, the choice of the temperature correction coefficient, θ, has to be carefully selected.

The evaluation of temperature coefficient was presented in Figure 13 for a typical deoxygenation rate (2.0 day^{-1} base e at 20°C), and three commonly used temperature

correction coefficients. The range of temperature, 5 through 30°C, was used. Figure 13 suggests two distinct characteristics, and confirms the significance of choosing θ values: (1) At temperatures above 20°C, the higher the θ value, the higher the value of K_d; (2) The higher the θ value, the greater the slope of the line, and, therefore, the greater the net change in K_d per °C; and (3) At temperatures below 20°C, the higher the θ value, the lower the K_d at each temperature. A temperature correction value of 1.047 was chosen from the literature (Bowie et al., 1985; Bedford et al. 1983; and Harleman et al. 1977).

The difference between $FBOD_5$ and $FNBOD_5$ for each river sample (DWS3, 3a, and 4) and all WWTF samples were determined. These were averaged and the results were used to convert $FBOD_5$ to $FNBOD_5$ (Table 28).

The development of the deoxygenation rate is based upon the statistical computation of a trendline and its slope between natural logarithmic concentration of $FNBOD_5$ measured during the river survey, versus cumulative travel time of the river. Shallow streams can be influenced by riverbed effects and exhibit a higher deoxygenation rates (Wright and McDonnell 1979). Rates may typically vary between 0.15 and 4 day^{-1} in water depths between 0.5 and 1 meter (Bowie et al. 1985). The Blackstone River is shallow, typically not exceeding 1 meter between BAC03 and BAC12. The values determined from the analysis of DWS1 and 4, are within the range reported above, and are reasonable (Table 29). No discernible BOD decay could be observed in the stations above UBWPAD (BAC01 – 02) and below BAC17.

All wastewater inputs were developed from survey data as discussed in Chapter 3 of this report. All tributary inputs were also described in the results section. A complete listing of this data is included in both Appendix A and B. Figure 14 shows the comparison between model predictions and average $FNBOD_5$ observations with 95 % confidence limits. The results are excellent. Statistical analysis of these simulations are given in Chapter 10. There was no $FNBOD_5$ data from 1991 BRI.

8. Nutrient Considerations

8.1 Nutrient Distribution

Figures 15 and 16 (NH_3-N) provide the first illustrated account of the difference between the ammonia concentration and mass loading profiles during the BRI and the BAC surveys. Table 30 helps to reduce the information on these two figures, and highlights the distribution of nitrogen at the station just below UBWPAD (BAC03 and BLK02) and near the MA/RI state line (BAC23 and BLK12). During the BRI, about 90% of the nitrogen in the river at BAC03 was in the form of NO_3-N. By the state line, there was very little change with NO_3-N at 93%. It was evident that the UBWPAD was providing significant nitrification and there was little nitrification in the river. In contrast, during the BAC, a significantly higher percent of the nitrogen was in the form of NH_3-N (56%) at BAC03, but by the state line (BAC23) the conversion of NH_3-N to NO_3-N was complete and approximate 92% of the nitrogen was in the form of NO_3-N. It was evident that the UBWPAD was not providing complete nitrification and as a result there was significant nitrification in the river. To support these observations, a comparison of the UBWPAD NH_3-N and NO_3-N loads for the BRI and BAC surveys is given in Table 31.

In general, for the BAC surveys, the highest ammonia levels occur just below the UBWPAD discharge at BAC03. For 3 out of 4 surveys (DWS1, 3 and 4) there is a sharp reduction in the concentrations to BAC07, about 4.4 miles downstream. From here, concentrations remain near detection for the river downstream of BAC10. The one exception to this was DWS2. The rate of ammonia decrease is more gradual and continues to occur well downstream of BAC07 to the state line

The nitrate profiles for concentration (Figure 17) and for mass (Figure 18) support the observations above. Nitrate levels were typically at their highest at BLK02 during the BRI, with no increase in downstream reaches (supporting the fact that the UBWPAD was

essentially discharging no ammonia and therefore the instream nitrification was limited). In contrast, during the BAC, the nitrate levels steadily increased in the reaches immediately below the UBWPAD, usually reaching the highest values by BAC07. This supports the observed reduction in ammonia in the same reaches. The following mass balance will provide proof of this nitrification.

Is ammonia disappearance in the reaches below the UBWPAD discharge due to instream nitrification, aquatic plant uptake or both? To investigate this, a nitrogen balance is completed for DWS 2, 3 and 4 (Tables 32, 33, and 34, respectively). In these tables, the river is divided into two segments: BAC03-BAC07 and BAC07-BAC24. Important observations are indicated immediately after each segment. A confirmation of instream nitrification would be the equivalent appearance of NO_3-N to a disappearance of NH_3-N. For these three surveys the following was determined:

- DWS2 - 547 lbs/day ammonia reduction vs 584 lbs/day nitrate production.
- DWS3 - 750 lbs/day ammonia reduction vs 932 lbs/day nitrate production.
- DWS4 – 825 lbs/day ammonia reduction vs 888 lbs/day nitrate production.

This confirms that the loss of ammonia in this reach is due to instream nitrification. This will be significant in dealing with the process for ammonia disappearance to be modeled, as well as the impact this process will have on the dissolved oxygen simulation.

A second nitrogen balance, in the reach between BAC07 and BAC24, is not as definitive. Ammonia loss does not equal nitrate production. It is evident that there is a loss of nitrogen to the water column for all three surveys:

- DWS2 – loss of 687 lbs/day of nitrogen
- DWS3 – loss of 949 lbs/day of nitrogen
- DWS4 – loss of 606 lbs/day of nitrogen

In the literature, the ratio of nitrogen uptake to phosphorus uptake by aquatic plants has been found to be about 7 to 1. The phosphorus loss (described by PO_4-P) in the river from BAC07 to BAC24 is the following, along with the N/P ratios:

- DWS2 – loss of 110 lbs/day of phosphorus: N/P ratio = 6.3
- DWS3 – loss of 151 lbs/day of phosphorus: N/P ratio = 6.3
- DWS4 – loss of 122 lbs/day of phosphorus: N/P ratio = 5.0

Strong evidence that the loss of nutrients in this reach suggests an increase in the aquatic plant biomass. This will be discussed in Chapter 10.

The combined nitrogen comparison is given in Figure 19. This figure provides us with an opportunity to view nitrogen independent of nitrification. Nitrogen values, directly below the UBWPAD (BAC03 or BLK02), are typically the highest values reported on this river. Values downstream of these stations are relatively constant with a few exceptions. Nitrogen values are similar for the two studies.

8.2 NH_3-N Modeling

The modeling of ammonia nitrogen consists of two sinks: conversion of ammonia to nitrite, and ammonia uptake by aquatic plants (ie. algae, macrophytes and/or periphyton) and two sources: the conversion of organic nitrogen to ammonia and the flux of ammonia from bottom sediments.

In the preceding section, a case was made for instream nitrification as the dominant process controlling ammonia concentrations in the upper reaches. The calculation of the reaction rate converting ammonia to nitrite will be the most important factor leading to a successful model of NH_3-N.

Plants are able to use nitrogen in the form of NH_3 ions, NO_2 ions, NO_3 ions and various organic compounds (Round 1981). Ruane and Krenkel (1978) comment that some workers found that aquatic plants grow better with nitrate than with ammonia, whereas others report the opposite. Both comment further that the process is sensitive to many environmental factors. In the QUAL2E BRI application, an algae preference factor of zero was used for ammonia nitrogen. This was not challenged by the SAB during their review, and, as a result, it is considered a reasonable assumption and will continue to be used in the BAC application. The influence of algae in the ammonia balance is, therefore, neglected but is important when simulating nitrate.

Chiaro and Burke (1980) found NH_3 release rates from the sediment between 3.72 and 46.93 mg/ft^2-day (44-505 mg/m^2-day) from a polluted river (Saginaw River in East Central Michigan). Fillos and Swanson (1975), who did experiments based on simulated sludge, found that high release of NH_3 from the sediment such as 25 mg/ft^2-day is caused by high gradients of NH_3 between benthos and surface water and dissolved oxygen concentration lower than 2 mg/L. Concentrations of DO for the Blackstone River never fell below 4 mg/L. The flux of ammonia from sediment was a constant rate for each reach and was defined as 5 mg/ft^2-day for the BRI model application. This was not challenged by the SAB during their review, and, as a result, it is considered a reasonable assumption and will continue to be used in the BAC application.

Organic nitrogen was not determined for this study. During the BRI study, organic nitrogen was calculated from Total Kjeldhal Nitrogen (TKN) and NH_3-N measurements for the UBWPAD. The overall average for the study was 1.05 mg/L organic nitrogen as N. This is reasonable based on Viessman and Hammer (1985) which indicated a conventional biological treatment plant, such as UBWPAD, with substantial nitrification should be about 2.0 mg/L. Viessman and Hammer also indicated conventional biological treatment plants without nitrification would have a typical concentration of 4.0 mg/L. This concentration was used for the other 4 WWTFs . Other values were set to the BRI averages; Headwaters = 0.29

mg/L, Quinsigamond River = 0.40 mg/L, Mumford River = 0.40 mg/L and West River = 0.40 mg/L. No organic nitrogen was input for the incremental inflow or other small tributaries. The reaction rate converting organic nitrogen to NH_3-N was set to 0.02 day^{-1} base e at 20°C (Bowie et al. 1985).

To test the model for organic nitrogen only a limited amount of data was available from the USGS station at Millville, MA, located near BAC22 (Reach 19, CE 1). There were a total of six measurements (April, June (2), August (2), November) published in the Water-Data Report MA-RI for 2001 and 2002. These ranged from 0.45 to 0.74 mg/L with an average of 0.57 mg/L.

The model results for the 4 dry weather surveys at BAC22 were 0.49 mg/L for DWS1, 0.46 mg/L for DWS2, 0.81 mg/L for DWS3 and 0.49 mg/L for DWS4. The average for the 4 surveys was 0.56 mg/L. Both the range and average for the observations and model predictions were similar. The organic nitrogen simulations were successful.

The process of nitrification usually starts in the second stage of BOD decay after approximately five days. The process can be stimulated or retarded by the amount of BOD discharged from the WWTFs and will depend on the degree of treatment (Bansal 1976). In secondary treatments plants, in which carbonaceous BOD is reduced, the growth of nitrifying bacteria just below the WWTFs discharge is accelerated and may exceed the impact of carbonaceous BOD on the dissolved oxygen regime (Gowda 1983).

The oxidation rate of NH_3-N to NO_2-N is described by the first order reaction rate, β_1. The magnitude of β_1 depends upon the observed loadings of NH_3-N, the stream temperature, the travel time, and the amount of nitrifying bacteria. The first step of nitrification is due to the chemoautrophic bacteria, called nitrosomonas. Nitrifying bacteria can be encountered both in river water (planktonic nitrifiers) and on the surface of the riverbed, attached on the benthos (Dunnette and Avedovech 1983). Similar to the heterotrophic bacteria for the decay of BOD, the shallower the river the more important the nitrifying bacteria attached on the wetted perimeter to nitrification (Williams and Lewis 1986; Cooper 1986). Therefore, the rate of nitrification in shallow streams can be greater than in deeper streams.

Typical values for β_1 range between 0.1 to 0.5 base e day^{-1} at 20 ºC for deep rivers with large bodies of water. In contrast, shallow and small rivers show values of β_1 greater than 1 day^{-1} base e at 20 ºC, which is not unusual (Thomann and Mueller 1987). Stratton and McCarty (1967) and Bowie et al. (1985) have published β_1 values ranging from 1.68 to 1.84 and 3.1 to 6.2 day^{-1} base e at 20 ºC for shallow streams, respectively.

Many researchers do not distinguish between the two steps of nitrification, since the rate of conversion from nitrite to nitrate is very fast and nitrite concentrations in rivers are very low. It is evident in many reports that researchers are determining the rate of nitrification from ammonia disappearance, similar to this study, and, as a result, are defining β_1 as described here. Table 35 summarizes the other nitrification rates from the literature.

Table 35. Summary of β_1 Values for Some Rivers

Reference	Study Area	β_1 (day^{-1})*
Bansal (1976)	Flint River (Michigan)	0.1-1.32
Cooper (1986)	Weiohewa (New Zealand)	5.4-6.4
Deb and Klafter-Snyder (1983)	Leatherwood Creek (Arkansas)	1.1-7.1
Dunnette and Avedovech (1983)	Willamate River (Oregon)	0.7-4.6 (18 and 21 ºC)
Gowda (1983)	Speed River (Ontario)	0.2-4.41
Koltz (1982)	Iowa and Cedar River (Iowa)	0.5-9.0

* If not otherwise identified, all nitrification rates are at 20 ºC

For the BAC, β_1 values were determined from linear regression of the logarithmic concentration of NH_3-N and the cumulative travel time in the river. The slopes of the linear regression equation equal the β_1 values. Table 36 is a list of the average β_1 values determined for DWS 1 and 4.

The impact of the actual temperature in the river is also crucial and ranges between 10 to 30ºC. The optimum range for nitrifying bacteria lies between 25 and 28ºC (Courchaine 1968). All rates in QUAL2E are based on a temperature at 20ºC. The conversion of the rates to actual temperature in the river is computed by means of the Arrhenius-formula. The Arrhenius-formula is governed by the difference between actual river temperature and the reference temperature at 20ºC and the temperature correction coefficient, θ. The temperature correction coefficient is an important factor in modeling and must be selected carefully. In Figure 20, two different θ values commonly used in water quality models were chosen in order to show the impact of θ on the oxidation rate, β_1, over a temperature range of 5 through 30 ºC. The values on the figure are the calibrated BAC model β_1 values (0.01 through 3.25), and the lines for each rate are the result of the two temperature correction values. These curves for each β_1 value cross each other at 20ºC, called the cross over point on the figure. The default value for θ in QUAL2E is 1.083, which was selected by Bierman et al. (1980) and O'Connor et al. (1981) for modeling water quality in a lake and in a bay, respectively. The θ value of 1.02 was chosen by Chen and Orlob (1972, 1975), Baca and Arnett (1976), Smith (1978), and Brandes (1976), who modeled water quality in rivers, lakes and bays. The higher the reaction rate the greater the impact associated with temperature change (Figure 20).

At a rate of 3.25 (at 20 ºC), the largest effect on the change of the oxidation rate is for DWS1 and 3. Table 37 provides an example of the impact of the temperature coefficient, 1.02 and 1.083, on the maximum β_1 of 3.25 day^{-1} for an NH_3-N concentration of 3.0 mg/L and a constant time of travel of 0.08 hour. The average temperature for each survey is given in column 3. The temperature adjusted β_1 values are given in columns 4 and 5. The change

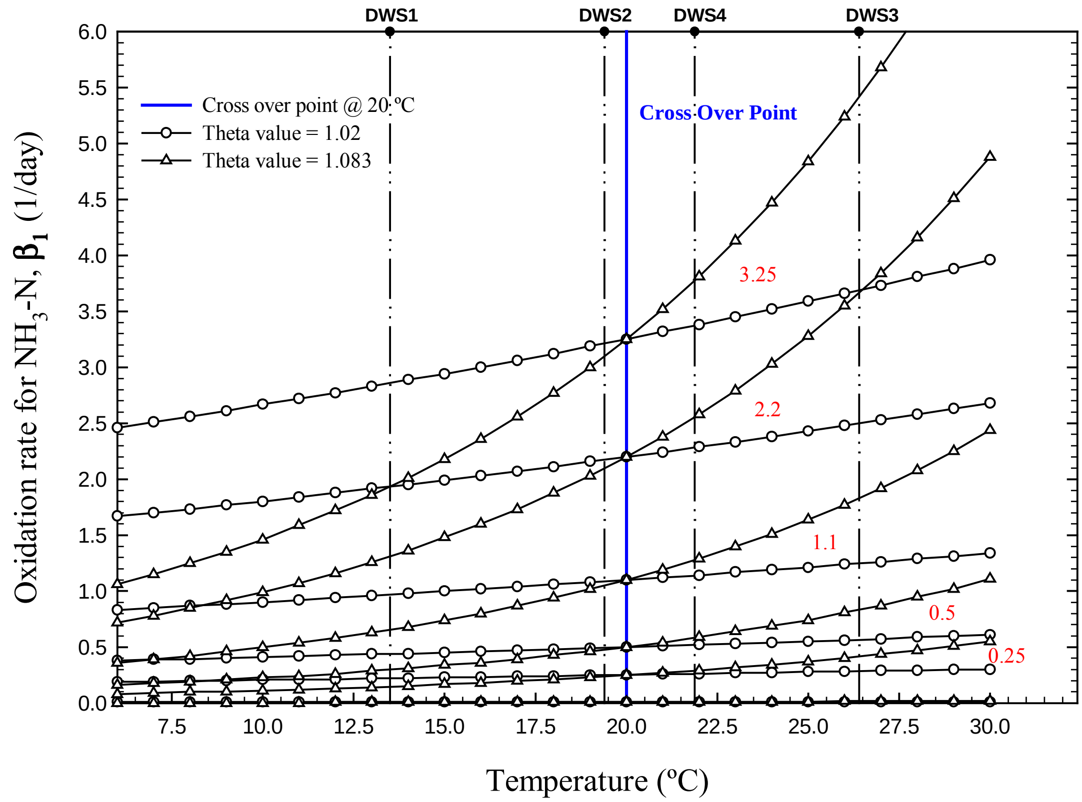

Figure 20. Impact of temperature correction coefficients on β_1 rates for each DWS.

Table 37. Impact of temperature correction coefficients (θ) on NH_3-N concentrations

1	2	3	4	5	6	7	8
			β_1		Predicted NH_3-N		
	NH_3-N	T	θ = 1.02	θ = 1.083	θ = 1.02	θ = 1.083	Δ
	mg/L	ºC	day^{-1}	day^{-1}	mg/L	mg/L	%
DWS1	3.00	13.5	2.86	1.93	2.39	2.57	7.2
DWS2	3.00	19.4	3.21	3.12	2.32	2.34	0.7
DWS3	3.00	26.4	3.69	5.46	2.23	1.94	13.2
DWS4	3.00	21.9	3.38	3.81	2.29	2.21	3.4

β_1 = 3.25 day^{-1} base e @ 20 ºC; NH_3-N concentration = 3.0 mg/L; Time of Travel = 0.08 hrs.

of NH_3-N concentration is given in columns 6 and 7. The greatest impact of the temperature correction coefficients was for DWS1 and DWS3, 7.2% and 13.2%, respectively. This is not surprising, since the temperature of both dry weather surveys is significantly below or above 20 ºC. The greatest difference in concentration for NH_3-N (13.2%) can be found for DWS3, which had the warmest survey water temperature. The selection of the temperature correction coefficient has a significant impact on the success or failure of a model to predict ammonia profiles, when the data is collected at stream temperatures significantly below or above 20 ºC. For the NH_3-N modeling, a temperature correction coefficient of 1.02 was selected.

All wastewater inputs were developed from survey data (Chapter 3). All tributary inputs were also described in the results section. A complete listing of this data is included in both Appendix A and B. The NH_3-N modeling involves four dry weather surveys. The average rates reported in Table 36 were an average from DWS4 and DWS1 and were used for model calibration (Figure 21). For model validation, the NH_3-N DWS3 profile used the rates of Table 36 (Figure 22). The simulation was successful. Statistical analyses of these simulations are given in Chapter 10.

When the stream rates and the mass loadings from the UBWPAD were used for DWS2, the simulations were not successful. First of all the UBWPAD measured NH_3-N concentration was too high compared to the concentrations in the river downstream of the discharge (BAC03-07). The concentrations from the UBWPAD were unusually high in comparison with the other surveys: 12.6 mg/L for DWS2 and 3.02 mg/L; 3.08 mg/L; and 4.27 mg/L for DWS1, 3 and 4 (channel), respectively. There was no obvious reason for this difference and it was not seen with any other constituent. From the river stations and the UBWPAD flow, a concentration of 8.4 mg/L would be necessary. Either way the amount of ammonia in the discharge is high for DWS2 and the comparable nitrate levels are low. Very little nitrification was going on in the facility during DWS2. Nitrification begins in the river below the UBWPAD discharge and continues well beyond BAC07 to BAC19 at a slower rate

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than what was observed when UBWPAD was providing nitrification (DWS1, 3 and 4). Possibly, this slower rate is a better representation of the river when the UBWPAD is not nitrifying and all nitrifying bacteria must develop in the reaches downstream of BAC03. While on the other hand the other three surveys were completed in months when considerable nitrification at the UBWPAD had been on-going before and during the surveys. Possibly the population of nitrifying bacteria was sufficient to provide the higher rates of nitrification and the rapid conversion of ammonia to nitrate.

The rates and simulation for the river without UBWPAD nitrification (DWS2) also are reported in Table 36 and the model simulation with both the higher and lower values at UBWPAD are given in Figure 23. The slower rate of nitrification continues throughout the Blackstone River in MA to the state line.

9. Sediment Oxygen Demand

The SOD measurements were conducted by means of an in situ benthic chamber, which was designed and constructed by Kugler (1997). The in situ benthic chamber has been used on the Saugatucket River, RI, and the measurements have been incorporated into a successful water quality modeling study. Six sites were selected for measurement of the Sediment Oxygen Demand (SOD). These stations along with those monitored in the BRI are given in Figure 24.

The chamber was submerged and sealed in the riverbed and dissolved oxygen measurements were conducted every 5 minutes for an approximately 1.5 hours. The chamber was then covered in order to prevent oxygen production by algae or periphyton. The first adjustment to the chamber's DO signal was made for the water column by having light and dark bottles in the river along side the chamber for the same period of time. An analysis of the change in the bottles gave an estimate of the water column DO demand and production. The second adjustment comes from the information provided by the covered and uncovered chamber. An analysis of this data allows the elimination of the DO production from the sediment (periphyton). The remaining DO signal is a result of sediment DO demand only. The adjusted data when plotted versus time and evaluated by simple linear regression provides an estimate of the SOD rate of DO change. The computation of the SOD rate was calculated with the following formula after Davis and Herdendorf (1985):

$$SOD = 24 * \frac{S * V_c}{1000 * A} \left| \frac{gO_2}{ft^2 day} \right| \tag{9.1}$$

in which: SOD is the rate at the ambient temperature during the measurement; S is the slope of the regression line of DO concentration (mg/L) vs. time (min.); V_c is the volume of the benthic chamber (L); here, $V_c = 17.5$ L; and A is the bottom area of the benthic chamber (ft^2); here, $A = 1.4\ ft^2$.

The computed SOD rates were adjusted to 20ºC by using the Arrhenius relationship. For the temperature coefficient, θ, a value of 1.048 was used (Duke and Masch 1973; Roesner and Evenson 1977; JRB 1983; Tetra Tech 1980; and Porcella et al., 1983). Table 38 shows a comparison among SOD values measured in the BRI 1991 by EPA (Wright et al., 2001) and in this study.

The BAC SOD stations were selected to complement the EPA BRI stations. It was intended that the results of the two studies would reduce the need to apply a rate far beyond the reach of measurement. This was accomplished. The rates are in Table 38.

In the BRI, SOD was measured at BAC07 (Singing Dam) (0.55 g-O_2/ft^2-day) and was applied in the reaches upstream to the UBWPAD discharge. In this study measurements were made midway between the discharge and Singing Dam. The average of the three measurements was 0.20 g-O_2/ft^2-day. This rate replaced the higher BRI value in Reaches 3-5. The higher BRI rate was still used for the reach containing Singing Dam (Reach 6).

The BAC values monitored at BAC10 (0.17 g-O_2/ft^2-day) and BAC13 (0.19 g-O_2/ft^2-day) were applied in Reaches 7-8 and 9-10, respectively, replacing the higher BRI rate monitored at BAC14 (Reach 11) (0.38 g-O_2/ft^2-day). The higher value was retained in Reach 11, since it was confirmed by the BAC measurement in Reach 12 of 0.35 g-O_2/ft^2-day.

SOD measurements from both the BRI and BAC for Rice City Pond (Reaches 13-14) and the reaches below the pond to the state line were consistent and the rates essentially did not.

Measurements in Reach 17 confirmed the use of the low values of SOD. This study measured the SOD in the Reach 12 The BAC SOD rate of 0.35 g-O_2/ft^2-day monitored in this study in the reach leading to the Riverdale Dam. The value 0.35 g-O_2/ft^2-day was similar to the value measured in the Reach 11 by EPA.

Table 38. Measured SOD rates on the Blackstone River in Massachusetts

Reach	Station	SOD - BRI* g-O_2/ft^2-day	SOD - BAC* g-O_2/ft^2-day	Model 1991** g-O_2/ft^2-day	Model 2003** g-O_2/ft^2-day
1	BAC01	0.15		0.15	0.15
2	BAC02			0.55	0.15
3	BAC03			0.55	0.20
4	BAC04		0.27/0.21/0.13	0.55	0.20
5	BAC06			0.55	0.20
6		0.55		0.55	0.55
7	BAC07			0.37	0.17
8	BAC10		0.23/0.17/0.10	0.37	0.17
9	BAC12			0.37	0.19
10	BAC13		0.23/0.17/0.17	0.37	0.19
11	BAC14	0.22/0.53		0.37	0.37
12	BAC15		0.35	0.23	0.35
13	BAC16			0.23	0.24
14		0.16/0.30	0.22/0.25	0.23	0.24
15	BAC17			0.15	0.13
16	BAC19			0.15	0.13
17			0.18/0.13/0.09	0.15	0.13
18	BAC21			0.15	0.13
19	BAC22			0.15	0.13
20	BAC24	0.15/0.14		0.15	0.15

* All SOD rates have been adjusted to 20 degrees C using a temperature coefficient of 1.048.
** Rates are reported that were used in the application of the DO Model.

10. Dissolved Oxygen

The following chapter is divided into several areas. A procedure for development of solar radiation values is described and the model input data is presented. The productivity of the river is discussed with a comparison between the 1991 BRI and the current BAC surveys. The model is used to predict NO_3-N, PO_4-P, chlorophyll *a*, and dissolved oxygen and a comparison is made with observed averages and 95% confidence limits. The BAC model is then applied to the BRI 1991 survey data. This is followed by all supporting statistics for all the modeling efforts. Finally, changes of dissolved oxygen are broken down into individual reach gains and losses.

10.1 Solar Radiation

The solar radiation data were obtained from the National Oceanic and Atmospheric Administration (NOAA) National Data Center. The majority of the measured solar radiation data were collected by NOAA's National Weather Service (NWS) and supplied to the National Renewable Energy Laboratory (NREL). The foundation for the National Solar Radiation Data Base (NSRDB) is the hourly measured solar radiation data collected by the NWS over the past several decades. The estimates of solar radiation were a combination of modeled data (using the METSTAT model) and meteorological data. Altogether, the data covers a time frame of 30 years from 1960 through 1990. Fortunately, two solar radiation stations were situated in the drainage area of the Blackstone River: Worcester and Providence. In Figure 25 a comparison is made between the two stations for data averaged by month for 30 consecutive years. The differences between the stations were minor.

The data from Worcester was used in the BAC QUAL2E model. The solar radiation input in QUAL2E were filtered in order to obtain an average of 30 consecutive years (1960-1990) for each hour. The data was then modified in two steps. First, a conversion of the

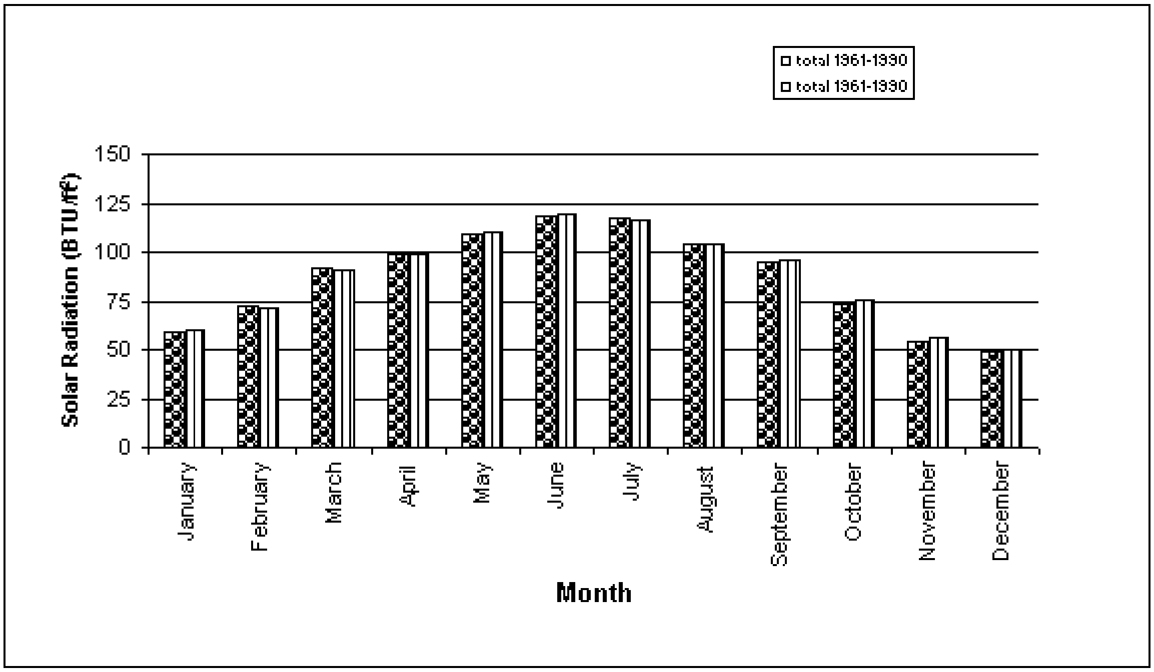

Figure 25. Comparison between Monthly Average Solar Radiation from Worcester and Providence NWS Stations.

total solar radiation into photosynthetically active radiation (PAR or PhAR) was completed followed by an adjustment for light-attenuation by organic and inorganic solids. Both steps are prerequisites for modeling algae in QUAL2E.

In developing PAR data, it is necessary to understand that plants (in QUAL2E algae) are only able to use a certain spectrum of the solar radiation for photosynthesis. Thus, the total/global solar radiation cannot be applied for the modeling of algae. By definition, PAR includes the wavelengths from 400 to 700 nm (0.4-0.7 µm) and covers approximately the range of the visible light (Field and Effler 1983; Kozlowski et al. 1991; Pearcy et al. 1991; Stefan et al. 1983; and Yocum et al. 1964). Larcher (1980) suggests the use of a spectra of wavelengths from 380-710 nm. According to Larcher, only 45 to 50 % (average is 48 %) of the total incident short wave length radiation (total solar radiation) is considered as PAR.

Yocum (1964) published a similar value of 47 %, which he obtained by comparison between a single measurement of incident radiation with a grating photometer and the results from an Eppley pyrheliometer. Stefan et al. (1983) published a value of 45.5 %, which he obtained by correlation of measured total solar radiation and PAR. The manual in QUAL2E recommends values between 43 and 45 %, but mentions that field measurements are preferable. Chapra (1997) mentions that values about 40 to 50 % are typical and also quotes Stefan et al. (1983). Since field measurements for PAR were not conducted during the dry weather surveys, the literature value by Stefan et al. (45.5%) was chosen to develop PAR for total solar radiation. The ratio is deemed steady being independent of time (Yocum 1964).

The 30-year record was accessed for each day of the year and for each hour of record. For example, 30 values were organized for January 1st for every hour of solar radiation. Since our model expects the solar radiation data to be in 3 hour intervals, the 30 values per hour were then grouped every three hours to arrive at a single value which would represent the 30 yr average solar radiation that occurred on January 1st at 6 AM, 9 AM, 12 PM, 3 PM, and 6 PM. This was completed for every day in a year.

The data input to the model requires the solar radiation to cover the time span of twice the estimated travel time. Therefore, for each DWS the following days were considered leading up to the survey: DWS1 October 15-22; DWS2 June 3-9; DWS3 August 1-10; and DWS4 July 3-10. The data was adjusted to PAR. Individual years are represented in Appendix B along with the 30-year average that was used in the model. Figures 26 and 27 are a summary of these tables and show the variability (95% confidence limits) of the PAR data for each of the surveys. The largest confidence limit is in the DWS2 (June) with approximately ± 10 for an average of about 100 BTU/ft^2. It was concluded that since direct measurement was not made, and collection of solar radiation data at the Worcester NWS station was discontinued in 1990, this 30-yr record and the procedure outlined here are reasonable for establishing the average PAR data for the days leading up to the field survey.

The second step considers the attenuation of solar radiation by the water body beneath the water surface including attenuation by living and nonliving suspended solids. QUAL2E takes into account the attenuation in the water body by a nonlinear equation, which requires the input of the non-algal portion of the light extinction coefficient. The nonalgal portion of the light extinction coefficient was computed with following equation (Chapra 1997; Di Toro 1978):

$$\lambda_0 = k_{ew} + 0.052 * N + 0.174 * D \tag{10.1}$$

in which, λ_0 = non-algal portion of the light extinction coefficient, 1/ft; k_{ew} = light extinction due to particle-free water and color, 1/ft = 0.01; N = nonvolatile suspended solids, mg/L; D = detritus (nonliving organic suspended solids), mg/L. The equation above is based upon regression equations among measured concentrations of nonvolatile suspended solids, algal chlorophyll *a*, non-algal volatile suspended solids, and turbidity.

The following table, Table 39, shows the non-algal portion of the light extinction coefficients computed by equation 10.1 for each reach and for each dry weather survey. The computations are based on measured values of TSS, VSS, and chlorophyll *a* from each dry weather survey. For DWS4, TSS and VSS were not measured so that the average of the non-algal portion of the light extinction coefficient for DWS2 and DWS3 was taken for DWS4.

10.2 Productivity in the Blackstone River

In addition to the nitrogen balance completed earlier, phosphorus, chlorophyll *a*, and dissolved oxygen profiles may be an indication of the river's productivity. It will be shown below that there are several indicators that show the productivity observed in 1991 during the BRI is different than the situation that the BAC DWS described. The analysis starts with a comparison of surveys for chlorophyll *a* and phosphorus.

For chlorophyll *a*, the concentration profiles in the river are similar for the two studies (Figures 28 and 29). Concentrations are very low in the river leading up to BAC07 (BLK04). Below this station concentrations and mass loadings rise reaching their maximum values in the reaches between Rice City Pond (BAC17; BLK06) and the MA/RI state line (BAC24).

For phosphorus, in the 1991 BRI survey, in the reach from the UBWPAD confluence to Singing Dam (BLK02-04; BAC03-07), there is little change in the phosphorus concentrations (Figures 30 and 31). The major decrease does not begin until the Fisherville Pond station at BLK06 (BAC12). In contrast, for the BAC surveys, phosphorus concentrations begin to decrease, around BAC04, with at least a 50% reduction in concentration by Singing Dam (BAC07; BLK04).

What is causing the reduction in phosphorus in the upstream reaches during the BAC? In Tables 32-34, the nutrient mass balances for DWS2, 3 and 4 included PO_4-P. It was established in Chapter 8 that the ammonia disappearance and nitrate appearance balanced, and nitrification was the major factor controlling the concentration of ammonia and nitrate in the reach from the UBWPAD discharge to Singing Dam (BAC03-BAC07). Although there was no loss of nitrogen in this reach, there was a loss of PO_4-P of 25.2 lbs/day in DWS2, 132 lbs/day in DWS3, and 120 lbs/day in DWS4. The loss of PO_4-P could not have been from settling, because the PO_4-P was filtered before analysis. The alternative is uptake by aquatic plants in the water column (ie. algae) or attached to the wetted perimeter (ie. periphyton and/or macrophytes). If it was algae, the chlorophyll *a* profiles would support this. They do not (Figures 28-29). If it is rooted aquatic plants, no chemical analyses will provide direct evidence, although large dissolved oxygen daily variations will provide indirect evidence of high productivity, and visual confirmation of plant growth may be possible. If productivity is confirmed in this reach, the nitrogen mass balance indicates that the nitrogen uptake must be from another source other than the water column.

In the rest of the MA Blackstone River, below BAC07 to the state line (BAC24), phosphorus loss is in proportion to the nitrogen loss (see Chapter 8 and Tables 32-34). The rate is approximately 6:1. This is very close to the ratio of 7:1 (N:P) reported in the literature (Chapra 1997) for reaches with high productivity. This suggests that the two nutrients are being lost to the water column due to plant uptake. The chlorophyll *a* profiles support this (Figure 28-29). There is a consistent rise of algae in this section. If rooted aquatic plants are also important, it may be possible to visibly confirm this.

Three sets of figures have been developed to evaluate the system productivity. These include the overlay of: (1) PO_4-P and chlorophyll *a* (Figures 32-33); (2) chlorophyll *a* and dissolved oxygen (Figures 34-35); and (3) PO_4-P and dissolved oxygen (Figures 36-37). Included in the comparison are the summer surveys for the BAC July 2003 DWS4 and for the BRI July and August 1991.

During DWS4 (Figure 32), the PO_4-P decrease from BAC03 to BAC07 did not result in a growth of algae (indicated by no increase in chlorophyll *a*). PO_4-P approaches its minimum value by Fisherville Pond (BAC10 to BAC12). In contrast, the surveys in July and August 1991 (Figure 33) show PO_4-P starts to decrease between Singing Dam and Fisherville Pond and algae (chlorophyll *a*) starts to increase. This trend continues to the river segments below Rice City Pond where PO_4-P reaches its minimum and chlorophyll *a* reaches its maximum.

Figures 34 and 35 compare chlorophyll *a* and dissolved oxygen. For DWS4, major dissolved oxygen swings only appear at the upper reaches (BAC03 through BAC07) and do not correspond with the trend of chlorophyll *a*, whereas, for the dry weather surveys in July and August 1991, major dissolved oxygen swings only occur at the lower reaches, which do correspond with the trend of chlorophyll *a*.

Figures 36 and 37 compare PO_4-P and dissolved oxygen. In the recent dry weather survey (DWS4), it is striking that the large dissolved oxygen swings occur between the river stations BAC03 and BAC07. Higher dissolved oxygen swings can be encountered in the downstream reaches, where PO_4-P decreases.

Therefore, areas of high productivity and phosphorus loss in the upstream reaches (BAC03-07) occur in the BAC, but do not occur in the BRI. Is this associated with rooted aquatic plants? Photographs were taken at every station during the survey in July 2003. Eight of these photographs are presented in Figures 38 and 39 (A complete set of photographs is presented in Appendix B). Figure 38 shows the river at 4 locations between BAC03 and BAC07 and Figure 39 for 4 locations below BAC10 to BAC22. These photographs provide dramatic evidence that macrophytes dominated the early reaches and are not evident in the downstream reaches. Why? Certainly phosphorus concentrations are near detection in the lower reaches and depths increase, but another important factor is the ability of light to penetrate the water column to the channel bottom and stimulate plant growth. The turbidity concentration profile for DWS 4 provides the link between macrophyte growth and light penetration (Figure 40). Turbidity increases significantly at BAC12 and continues increasing until BAC17. From the photographs of Figure 39, the water has grown turbid and no macrophytes are evident.

If the phosphorus loadings from UBWPAD had not changed between 1991 and 2000-3, why were the water quality conditions in the reach between BAC03-07 (BLK02-04) so different in the two studies? One possibility is the impact of dechlorination at the UBWPAD on the river in this reach. In 1991, the facility was not dechlorinating and sufficient chlorine residual was being discharged to create an instream toxicity (Wright et al. 2001) that could have inhibited the growth of plants. In the mid-1990s the facility began to dechlorinate. This would have eliminated the stream toxicity and stimulated the growth of plants. The system is shallow in this reach with a relatively high velocity that would not encourage the growth of algae but would not hinder the growth of rooted aquatic plants.

Net productivity and plant respiration values should reflect the activity of the plant communities throughout the river. The average daily net productivity (P_{av}) can be computed with the delta method presented by Chapra and Di Toro (1991). Although this measure has deficiencies (insensitivity for low and high reaeration rates and photosynthesis is represented as a half-sinusoid), the delta method is deemed an easy and quick procedure to obtain an independent check, if diurnal curves of dissolved oxygen are available.

The delta method is based on the following mass balance, where plants are uniformly distributed for a sufficiently long distance, and the deficit does not vary spatially:

$$dD/dt = R_c - P(t) - K_2 D \qquad (10.2)$$

in which, D = oxygen deficit (mg/L), t = time (d), K_2 = reaeration rate (day^{-1}), R_c = community respiration (mg/L d^{-1}), and P = primary production (mg/L d^{-1})

An analytical solution of equation 10.2 is extensive. Chapra and Di Toro provide graphical and simplified analytical expressions to calculate the ratio Δ/P_{av} (difference of max and min dissolved oxygen over average productivity), which depends on the reaeration and the photoperiod. In this report, only the simplified analytical expressions were utilized in order to compute Δ/P_{av}. The graphical expression has the disadvantage that it is insensitive for $k_a<1.0\ d^{-1}$ and for highly aerated systems ($k_a>10\ d^{-1}$) while the analytical expressions provides an equation for low reaeration rates ($k_a<1.0\ d^{-1}$). In addition, the reading of the graphs always includes an uncertainty. Two equations were used to compute Δ/P_{av} depending on the reaeration rate. Both equations are widely applied and recommended (Thomann and Mueller 1987; Bowie et al. 1985).

For $k_a<1.0\ d^{-1}$:

$$\frac{\Delta}{P_{av}} = T - f \tag{10.3}$$

in which, Δ = diurnal dissolved oxygen range (mg/L), P_{av} = average daily primary productivity rate (mg/L d^{-1}), T = period (d), and f = photoperiod (d).

For k_a>1.0 d^{-1}:

$$\frac{\Delta}{P_{av}} = \frac{T}{k_a f} * \frac{(1 - e^{-k_a f}) * (1 - e^{k_a (T-f)})}{1 - e^{k_a T}} \tag{10.4}$$

Altogether, equation 10.3 was only applied for stations BAC12 and BAC24, which have reaeration rates smaller than 1. Only station BAC06 exceeded the reaeration rate of 10 day^{-1}.

Community respiration, R_c, is calculated from equation 10.2. Plant respiration, R_p, is then estimated by subtracting the oxygen demand estimated for BOD, nitrification and SOD. An estimate of P_{av}/R_p is the photosynthetic quotient. When P_{av}/R_p is greater than 1.0 the system is autotrophic where the plants are able to use CO_2 as a sole source of carbon and plant biomass will increase. This should occur in the early summer. When P_{av}/R_p is less thant 1.0 the system is heterotrophic and the plants will obtain carbon from organic compounds and the plant biomass will decrease. This should occur in the fall. When P_{av}/R_p is equal to 1.0 plant photosynthesis just equals plant respiration and the biomass remains relatively constant.

First, DWS4 was evaluated for the average daily productivity (P_{av}), plant respiration (R_p), and its ratio (P_{av}/R_p) (Table 40). It is striking that the highest P_{av} and R_p is situated between stations BAC03 and 07, which is not surprising, since these sections belong to the most productive and active sections of the Blackstone River, in which macrophytes thrive. It is interesting that the P_{av}/R_p ratios are around 1 in this section, indicating daily photosynthetic oxygen production = daily plant respiration. Consequently, if the model is simulating daily average DO, like QUAL2E, the results would not be impacted by the failure to address the macrophyte issue. This certainly would not be the case if the model were used to provide dynamic simulations of DO that result in large DO swings at various times of the day, or if the model were applied in periods when the P_{av}/R_p ratios were not equal to 1.0. At BAC12, where algae play an increasingly important role, the growth rate rises gradually with some interruptions and reaches maximum as the river approaches the state line. The last four stations indicate clearly that more oxygen is produced than expended.

In contrast, the high P_{av} and R_p values in the upper reaches in DWS4 were not occurring in July 1991, another confirmation that macrophytes were not present in 1991. In 1991, the highest productivity occurred in the reaches just before the state line.

Table 40. Average Productivity (P_{av}) and Plant Respiration (R_p) for DWS4 and July 1991

	BAC				BRI		
Station	P_{av}	R_p	P_{av}/R_p	Station	P_{av}	R_p	P_{av}/R_p
	mg/L d^{-1}	mg/L d^{-1}			mg/L d^{-1}	mg/L d^{-1}	
BAC03-BAC04	18.4	17.7	1.04	BAC03-BAC06	6.82	6.64	1.03
BAC04-BAC06	23.5	23.7	0.99				
BAC06-BAC07	19.0	17.7	1.07	BAC06-BAC07	2.47	2.21	1.12
BAC07-BAC10	5.52	8.82	0.63	BAC07-BAC12	2.55	2.29	1.11
BAC10-BAC12	2.27	2.37	0.96				
BAC12-BAC13	1.79	2.37	0.76	BAC12-BAC15	10.4	10.4	1.00
BAC13-BAC14	2.74	2.54	1.08				
BAC14-BAC15	4.42	4.12	1.07				
BAC16-BAC17	3.55	-	-	BAC15-BAC17	18.1	3.74	4.84
BAC17-BAC19	2.63	2.33	1.13	BAC17-BAC21	17.7	-	-
BAC19-BAC21	2.12	1.02	2.08				
BAC21-BAC22	3.15	0.65	4.85	BAC21-BAC22	16.1	2.03	7.94
BAC22-BAC24	4.90	3.60	1.36				

To compare the two studies, mass loads for P_{av} and R_p were determined and the BAC survey was grouped in the same manner as the BRI (Table 41). Values of productivity seen in the later reaches in 1991 were similar to those seen in the early reaches in the BAC. Typically, the BRI survey was an order of magnitude lower in the early reaches compared to the BAC.

Table 41. Comparison of the River Productivity for the BRI (DWS4) and BAC (July 1991)

	BRI			BAC		
Station	P_{av}	R_p	P_{av}/R_p	P_{av}	R_p	P_{av}/R_p
	lbs/day	lbs/day		lbs/day	lbs/day	
BAC03-BAC06	1987	1934	1.03	17887	17679	1.01
BAC06-BAC07	744	665	1.12	8562	7946	1.08
BAC07-BAC12	925	836	1.11	6042	7911	0.76
BAC12-BAC15	4001	3997	1.00	6336	6118	1.04
BAC15-BAC17	7197	1496	4.81	5858		
BAC17-BAC21	8083			4054	2732	1.48
BAC21-BAC22	8605	1086	7.92	3952	837	4.72

P_{av}/R_p values of 2 and greater are not unusual in high productive rivers. Erdmann (1979) applied a similar approach for obtaining P_{av}/R_p values, which range from 0.2 through 5.1. Wright and McDonnell (1986) provide P_{av}/R_p values, which range from 0.79 through 2.53.

The macrophytes were identified by species in the reach between BAC03 and BAC07 (Table 42). These macrophytes may be grouped in two formations. Both formations have in common that the macrophytes grow in dense, thick, and long masses covering most parts of the riverbed.

Table 42. Macrophyte Species between UBWPAD and BAC07

Species	Water Depth 1- 4 ft	Water Depth 4- 6 ft
Myriophyllum heterophyllum	✔	
Elodia canadensis	✔	
Potamogeton crispus		✔
Potamogeton pusillus		✔
Vallisnaria americana	✔	✔
Callitriche heterophylla	✔	✔

The first formation appears in shallower water of 1-3 ft and consists of <u>Myriophyllum heterophyllum</u> (Watermilfoil) and <u>Elodea canadensis</u> (Common waterweed), which are distributed relatively equally. <u>Myriophyllum heterophyllum</u> is a perennial and submerged except for the flowering spikes that are emerged. <u>Elodea canadensis</u> (Common waterweed) is a perennial and lives entirely submerged, except for small white flowers, which bloom at the water-surface.

The second formation appears in deeper water of 3-6 ft and consists of <u>Potamogeton crispus</u> (Curly Pondweed) and <u>Potamogeton pusillus</u> (Small Pondweed). Both can be encountered growing in very close communities of dense, thick, and long masses. <u>Potamogeton crispus</u> is a perennial with sections submerged and emerged. <u>Potamogeton pusillus</u> is a perennial and submerged.

In both formations, 2 other species can be found, which grow either in clusters (<u>Vallisneria americana</u> -Water Celery) or solitarily (<u>Callitriche heterophylla</u>). <u>Vallisneria ammeriana</u> is a perennial and submerged. The 4 plants that dominated the reaches are all indicators of eutrophic freshwater when they appear in dense clusters (Cox 1985).

In addition to the macrophytes, the river-section between BAC06 and BAC07 mats of Spirogyra spp., a filamentous green algae, were also observed growing at the side of the channel on macrophytes or fallen branches of trees. Spirogyra can be found in enriched streams with low velocities (Stevenson et al. 1996).

10.3 Dissolved Oxygen Modeling for DWS1, DWS2, and DWS4

Similar to the constituents that have been modeled above (Cl, $FNBOD_5$ and NH_3-N), the incremental inflow concentrations for NO_3-N, PO_4-P, DO, and temperature were determined from the tributaries, and for nitrite, chlorophyll *a*, organic N and organic P they were set to zero. All wastewater and tributary inputs were developed from survey data unless noted. A complete listing of all data is included in both Appendix A and B.

In the QUAL2E BRI application, an algae preference factor of one was used for NO_3-N. This was not challenged by the SAB review, and, as a result, it is considered a reasonable assumption and will continue to be used in the BAC application. The influence of algae in the nitrate balance is important; where in the simulation of ammonia it was not considered significant. Both the nitrate simulations and the chlorophyll *a* simulations will be evaluated together.

The coefficient to convert nitrite to nitrate, $\beta_{2,}$ has been set sufficiently high to keep nitrite near zero. The major factors impacting nitrate predictions are the sources of nitrate from direct and indirect discharges, instream nitrification and the loss of nitrate due to algae uptake. Since the main factors impacting nitrate (including β_1 and $\beta_{2,}$ and the algal nitrate preference of 1.0) undergo no adjustment in this phase of the modeling, the nitrate simulations for all surveys are a validation of the model parameter selection, especially for ammonia oxidiation. The results of the nitrate modeling are excellent for DWS 1, 3 and 4 (Figures 41 and 42) and slightly underpredicted for DWS2. The supporting statistics are

given later in this Chapter.

A small source of chlorophyll *a* was observed in the data between BAC02 and BAC03. This loading was consistent in all four surveys. The possible source may be the aqueduct connection, which is approximately 400 yards downstream from the UBWPAD discharge channel. Although the aqueduct does not provide any significant flow into the Blackstone River under dry weather conditions, it will continue to flow because of groundwater additions. These low flows result in low velocities. During wet weather, flows from Worcester are bypassed through the canal and have the potential for containing a nutrient loading from the city's storm water and combined sewer system. The combination of low velocities and nutrients appeared to stimulate the growth of aquatic plants. The loading from the aqueduct of about 0.3-0.4 lbs/day was added in the reach just before BAC03.

A second source of chlorophyll *a* was identified from the data between river stations BAC10 and BAC12. Between these two stations is the remnants of Fisherville Pond. The Quinsigamond River meets the Blackstone River in this reach. BAC11 was sampled during the DWSs, but the station is about 5 miles upstream of the river's confluence with the Blackstone River. The impoundment survey of Phase 2 completed between late June to December 2001, sampled the Quinsigamond River from BAC11 to the Blackstone River. A significant growth of algae was found for most of the months sampled (Table 43). The impoundment stations FP06 and FP05 are situated on the Blackstone River between BAC10 and the confluence with the Quinsigamond River and the impoundment station FP04 is the last station monitored on the Quinsigamond River before its confluence with the Blackstone River. It is clear that the low velocities of the Quinsigamond River in the impoundment below BAC11 and above FP04 provide an ideal environment for algae growth. Therefore, FP04 was used to represent the Quinsigamond River's contribution to the Blackstone River for the two late summer surveys, DWS3 and DWS4. Table 44 summarizes the input values of the tributaries for each study.

Table 43. Measured Chlorophyll *a* During the 2001 Impoundment Study for Fisherville Pond in μg/L

	28-Jun	10-Jul	15-Aug	16-Aug	23-Sep	24-Sep	12-Oct	21-Oct	9-Nov	26-Nov	11-Dec
BAC10	NS	NS	1.9	2.06	1.36	1.63	2.69	2.12	NS	NS	2.23
FP06	NS	3.79	2.22	2.56	1.71	2.16	2.93	2.26	NS	NS	2.37
FP05	NS	3.17	2.57	2.35	1.35	2.1	7.57	2.1	NS	NS	2.96
BAC12	NS	4.84	5.42	5.78	5.31	5.56	5.15	2.37	NS	NS	2.6
FP04	9.66	5.86	10.9	20.1	13.9	13.9	7.73	47.4	13.1	10.5	NS

NS = Not Sampled

Table 44. Tributary Chlorophyll *a* Input in μg/L

Point sources	DWS1	DWS2	DWS3	DWS4
Singletary Brook	0.17	0.18	ND	0.65
Spring Brook	0.06	0.25	ND	0.67
Cronin Brook	0.11	0.20	ND	1.00
Quinsigamond River	0.08	0.71	15.50	5.86
Mumford River	0.10	0.30	0.81	0.82
West River	0.15	0.69	ND	0.78

ND = Not Detected

The chlorophyll *a* modeling included DWS1 and DWS4 for calibration and DWS2 and DWS3 for validation. Almost all the parameters and rates for modeling chlorophyll *a* were taken from the 1991 BRI application of the model. One minor change involved the selection of the algae settling rate. In the BRI this rate was defined as zero for all reaches up to the station BLK08 (BAC17 – the end of Rice City Pond (RCP)) and 1.0 between this station and the state line. In the BAC model the settling rate of 1.0 was extended upstream to include RCP (up to BAC16). This seemed reasonable because of the higher chlorophyll *a* values in RCP and the lower travel times.

Figures 43 and 44 are the results of the simulations for chlorophyll *a*. The test of the model requires a prediction of chlorophyll *a* concentrations that range from 1.0 to over 20 μg/L. The modeling of chlorophyll *a* was successful. The supporting statistics are given later in this Chapter.

In 1991, the DO concentrations did not vary a great deal (Figure 45). Concentrations ranged between 7 and 9 mg/L, even though the flows in July and August 1991 were near the 7Q10 flow. The UBWPAD was providing advanced waste treatment resulting in low BOD and ammonia and high nitrate. Oxygen demanding factors such as deoxygenation and nitrification were being handled in the facility. The major sink of oxygen in the reaches below UBWPAD was SOD (Wright et al. 2001).

In the BAC, there was considerable more variation in the DO profiles (Figure 46). Immediately below the UBWPAD, DO values decreased reaching values below 5.0 mg/L at station BAC04 during the lower flow surveys of DWS 1 and 4. Three factors were impacting DO during the BAC surveys that were not happening in the BRI surveys between BAC03 and BAC07: 1. Higher BOD concentrations from the UBWPAD were causing a measurable oxygen demand; 2. Higher ammonia concentrations from the UBWPAD were causing instream nitrification and a substantial oxygen demand; and 3. High productivity due to significant mats of rooted aquatic plants (Figure 38) were causing large diurnal swings of DO. SODs continue to be a sink of oxygen, especially in the reach leading up to Rice City Pond.

DO simulations were limited to DWS1, DWS2, and DWS4. DWS3 and DWS3A were not used, since only two runs in the morning were taken and would not represent the average DO over the day. The results of the DO modeling are given in Figures 46 and 47. The simulations are excellent.

One area of weakness is in the reach between BAC03 and BAC04. The model over predicted the DO for two of the three surveys. The sharp increases of DO between BAC06 and BAC07 and at BAC07 are due to the reaeration at the Electric Millbury Dam and Singing Dam, respectively.

A striking difference between Figure 46 (DWS1 and 4) and Figure 47 is the lower values reported in the Rice City Pond reach (BAC16-17) for the June survey (DWS2). This is opposite to the expectations that surveys in the summer/fall with lower flows, longer time of travels and warmer temperatures would have lower DO values. In the inventory of sources and sinks that follow, it is clear that the high ammonia concentrations from the UBWPAD are still resulting in high oxygen demands from nitrification in the central reaches of the MA Blackstone River.

The PO_4-P simulations are presented in Figures 48-49. The modeling of dissolved phosphorus was successful for the late fall survey in October (DWS1). Although the DWS2 application of the model over predicted the stream observations, these observations were suspiciously low, at or just above detection. The model prediction was dominated by the UBWPAD composite, with the greatest changes occurring from dilution, not plant uptake. The model and observation trends are similar. Why did the PO_4-P simulations for DWS3 and 4 fail? QUAL2E does not include the modeling of rooted aquatic macrophytes. This was not of concern in the 1991 surveys, but is an issue in the BAC surveys. The inability of the modeling effort to reduce PO_4-P in the reach between BAC04 and BAC07 from macrophyte uptake, leads to an over prediction in later reaches. A model that handles macrophyte production is needed.

10.4 Model Simulation of the 1991 BRI Data

The BAC version of QUAL2E was used to simulate NH_3-N, NO_3-N, PO_4-P chlorophyll *a*, and DO for the BRI 1991 inputs. This version represents the reaction rates as

they have been defined and developed in this report. These profiles are included with average observations and the 95% confidence limits, standard deviations or maximum and minimums on Figures 50-54.

For comparison, the predicted profiles from the BRI version of QUAL2E are also on these figures. These represent the reaction rates used in the model calibration and validation defined in the BRI (Wright et al. 2001). All supporting statistics for these model runs are also presented in the next section.

10.5 Statistical Evaluation of the Model Simulations

Throughout this report, success or failure of the modeling effort was based on a visual comparison between model prediction and observation with 95% confidence limits. Statistical tests have been suggested to provide a quantitative evaluation of the model's performance (Thomann and Barnwell 1980; Thomann 1982). These tests do provide a means to compare model applications between surveys. However, care must be taken to avoid misinterpreting the results.

Three different statistical measures for the verification of the predicted results were selected. The measures provide a better quantitative understanding of the credibility of the model results. All three measures were conducted based on suggestions published by R. V. Thomann and include all results of the dry weather surveys for each water quality constituent.

The first statistical measure determines the Root Mean Square Error (RMS) by the following formula:

$$E_{RMS} = \sqrt{\frac{\left(x_i - c_i\right)^2}{N}} \qquad (10.5)$$

in which x are the observed values, c the predicted values, and N the amount of the values determined. RMS was calculated for each water quality constituent for the BAC model application for this study (2000-3) and the BRI data set (1991) and the BRI modeling effort (Table 45).

The RMS is reflective of the magnitude of the concentration range and cannot be compared between constituents, for instance: Chloride simulations provided the best match between predictions and observations, yet it had the highest RMS of 8.54. It had the highest concentrations ranging from approximately 75 to 125 mg/L. The closest constituent to this was DO at about 7 to 9 mg/L. In contrast, the PO_4-P concentrations were the worst match between predictions and observations, yet it had one of the lowest RMS values of 0.30.

The following are general observations for this statistical test:

- The NO_3-N, chlorophyll *a*, and DO RMS values for this study are significantly better than those reported in the BRI.

- For NH_3-N, although the RMS values were slightly lower in 1991 (0.18 vs 0.24) the level of ammonia in the two studies is important. Success in 1991 meant the model was used to accurately simulate the low levels of NH_3-N (0.1 to 0.5 mg/L) for the most part through dilution. Therefore, NH_3-N, NO_3-N, and DO predictions were not sensitive to the selection of β_1. The NH_3-N concentrations (0.1 to 5.0 mg/L) in the BAC study were an order of magnitude higher below the UBWPAD. As a result, the calibration and validation of ammonia and β_1 were important and the successful simulation of NH_3-N, NO_3-N, and DO were tied into the selection of this rate.

The second measure determines the median relative error (MRE, e), which has been used in comparing DO modeling efforts (Thomann and Barnwell 1980; Thomann 1982):

$$e = \frac{|\bar{x} - \bar{c}|}{\bar{x}} \tag{10.6}$$

in which $\bar{x}$ are the median of the observed values, $\bar{c}$ the median of the predicted values. This test is often times misinterpreted when dealing with very low concentrations, for instance: NH_3-N simulations for 1991 showed a low MRE of 7%. Concentrations were relatively constant ranging from 0.5 to 0.1 mg/L. For the BAC survey, the MRE value was 47%, yet from the figures the model did an excellent job simulating the high values of ammonia 2-3 mg/L for DWS1, 3 and 4 (Figures 21 and 22). The simulation reached levels near background of 0.1 mg/L by BAC13. The concentrations of the river stations downstream of BAC13 were typically near detection or less than 0.05 mg/L. Therefore, in this area for observations of 0.05 mg/L and model predictions of 0.10 mg/L, the MRE would essentially be 100%.

The following are general observations for this statistical test:

- The MRE for DO is 2% for the current study. This is more impressive than the 2% reported in the BRI for the 1991 data, because there is considerably more range of DO values in the BAC surveys. Thomann (1982) suggested the current state of DO modeling in 1982 had reached a level of 10% MRE, which was based upon 15 small and larger streams in the US.
- The NO_3-N, chlorophyll *a*, and chloride MRE values for this study are significantly better than those reported in the BRI.
- The large MRE for PO_4-P of 244% for DWS1-4 is evidence that the model is not capable of handling the uptake from the rooted aquatic plants in the reach between BAC03-07, and the lower MRE in 1991 of 49% supports the comment above that macrophytes were not a concern in this study.

The third measure is a regression analysis, which determines the coefficient of determination, the slope, and the intercept. This measure provides an additional level of insight into the comparison between the model prediction and the observation (Thomann and Barnwell 1980; Thomann 1982). The regression analyses were performed for each constituent for the BAC study. With the exception of PO_4-P, the regression analyses support the success of the model simulations (Figures 55 and 56).

To test whether a correlation coefficient could be 0, one must refer to a table of the critical values (which, naturally, depend upon the sample size n). Such a table is printed in many statistic textbooks. It can be calculated by the following equation:

$$R/(1-R^2)^{0.5} * (n-2)^{0.5} \tag{10.7}$$

and interpreted as a t statistic with n-2 degrees of freedom. The results of the 7 regression analyses indicate that the null hypothesis of zero correlation is rejected at the 95% confidence level and a relationship exists between the predicted and observed data.

The regression line should approximate a 45 degree line where the slope is equal to 1.0 and the intercept is 0.0. The slope and intercept of the regression equation can be tested with a t statistic and n-2 degrees of freedom and a desired level of significance. By testing the slope of the regression line, one can determine if the slope could be 0 and, thereby, conclude that there is no linear relationship between the predicted and observed data. Furthermore, unless we can see some particular non-linear relationship when inspecting the standard residual plot, one may conclude there is no relationship at all. A similar test may be run on the intercept. The results of the 7 regression analyses indicate that the null hypothesis of the intercept equal to 0 is rejected at the 95% confidence level and the regression line approaches a one to one relationship indicated by a 45-degree line.

10.6 DO Source or Sink Inventory

Tables 46 and 47 are an inventory of the sources and sinks of DO for BAC DWS4 and BRI July 1991, respectively. The accounting of DO loss or gain is completed between stations. Figure 57 represents this as an overlay for the river.

- 2000-3 DWS4 Nitrification: Nitrification was the most important process in the reaches below the UBWPAD discharge accounting for approximately 60-70 % of the DO loss. It was the major reason why an oxygen sag was observed in the reach from BAC03 to BAC07. It decreased steadily to the state line where its importance to the DO balance had diminished to less than 5% of the DO lost.
- 1991 BRI Nitrification: Nitrification was insignificant throughout the Blackstone in MA. It was typically less than 10% of the DO loss.
- 2000-3 DWS4 SOD: As nitrification ended, SOD became the greatest loss steadily increasing in importance until it was 90% of the DO loss by the state line. SOD was very important in the oxygen sag observed in Rice City Pond (BAC16-17).
- 1991 BRI SOD: With minor loss from BOD and nitrification, the SOD demand was the controlling factor the DO losses.
- 2000-3 DWS4 BOD: Deoxygenation from BOD was important in the early reaches typically reaching 15-20%. By BAC 14 it was less than 10%.
- 1991 BRI BOD: It was insignificant typically less than 5% of the DO loss

Figures 58 and 59 support the observations made above concerning DO loss. The center graph in each figure illustrates these observations: for DWS4 the change over from nitrification to SOD by the midpoint in the river; and for the BRI July 1991, the importance of SOD throughout the river.

The bottom graphs are an interesting look at the two sources of DO. Clearly the relative importance of reaeration over algae photosynthesis is evident in the DWS4 surveys.

Algae are certainly more important to DO in 1991, clearly providing much of the DO by the state line.

Since travel time also plays a role in source/sinks of dissolved oxygen, the travel time between each station is also exhibited in these figures. If the travel time is higher, there is a greater opportunity for larger gains and losses of DO in a given reach.

Table 48 provides a ranking of the dams relative to their importance to DO increases for DWS4. The equation for dam reaeration was presented earlier in Chapter 6. The amount of oxygen gain is directly proportional to the amount of the deficit above the dam and the height through which the water falls. In this there are three values of DO gain, the first is from the model using equation 6.1 for the DO predicted above the dam and the input of temperature and dam height. The second and third values are from the actual observation using equation 6.1 and the daily average and minimum DO deficit and temperature from field data, respectively.

For the model prediction Singing Dam and the New England Power Co. dam were ranked first and second. Both are in the river segment between BAC03 and BAC07 where the highest demands from the oxidation of CBOD and ammonia are occurring and the lowest values of DO on the river are reported. Because the DO deficit is the highest in these reaches, the DO gain from the dams is the greatest. If the dams were further downstream where the system has recovered from the UBWPAD CBOD and ammonia discharges and DO concentrations are higher, the DO gain would be less. From the DO and temperature field data, Singing Dam is still, by far, the most important source of DO reaeration. In fact, the gain of DO for the minimum DO conditions was over 4.0 mg/L.

11. Phase 1: Summary and Conclusions

11.1 Summary

- Four dry weather surveys were completed including DWS1 (October 2000), DWS2 (June 2001), DWS3 (August 2001) and DWS4 (July 2003).
- The range of flows between this study and the 1991 BRI provide an excellent test of the model. Near station BAC22 (BRI station BLK12), the flows for the three BRI surveys were 102, 112 and 460 cfs, and the flows for the four BAC surveys were 126, 184, 235, and 307 cfs. The 7Q10 flow is approximately 115 cfs.
- The BAC model was represented by 20 reaches, which were supported by 18 main river stations, 6 tributaries and 5 WWTFs. This is in comparison to the 15 reach 1991 BRI model, which was supported by 9 main river stations, 3 tributaries and 1 WWTF.
- This study sampled 5 WWTFs, that included UBWPAD, Millbury, Grafton, Northbridge and Uxbridge, three times during the study. In general, the daily composite sampling was completed every day for five days before the start of the river survey for DWS1, DWS2 and DWS3. Model input included the three-day average leading up to the survey to coincide with the time of travel of the Blackstone River in MA. During the 1991 BRI study, only 1 WWTF in MA was sampled, UBWPAD.
- For the river surveys, three forms of BOD_5 were analyzed including unfiltered BOD_5, filtered BOD_5 and nitrogen-inhibited filtered BOD_5. The nitrogen-inhibited filtered BOD_5 values were used for the simulation of $CBOD_5$.
- Long-term BODs were developed for August 10, 2001, May 30, 2002 and July 26, 2002 at 6 locations for unfiltered BOD, filtered BOD and nitrogen-inhibited filtered BOD. The BOD_u and K_1 values from the nitrogen-inhibited filtered BOD are used in this model. The study average for K_1 was 0.0848 day^{-1} base e at 20°C and 6.04 mg/L for $CBOD_u$. The reaction rate ranged from a low of 0.04840 to a high of 0.1215 day^{-1} base e at 20°C. The $CBOD_u$ ranged from 4.44 to 7.87 mg/L.

- Prior to calibration, the deoxygenation coefficient was obtained by a semilog linear regression analysis between measured nitrogen-inhibited filtered BOD_5 loads and travel time. The deoxygenation coefficient, K_d, was the highest directly below the UBWPAD: 2.05 to 1.40 day^{-1} base e at 20ºC.
- All temperature correction coefficients were carefully reviewed in the QUAL2E model from 1991 and only the temperature correction coefficients for NH_3-N and NO_3-N were changed to 1.02 from the 1991 values of 1.083 and 1.047, respectively.
- Values of β_1 were developed from a semilog linear regression analysis between NH_3-N loads and travel time. For DWS1, 3, and 4, the calculated nitrification rate, β_1, ranged from 3.25 to 2.20 day^{-1} base e at 20ºC starting directly below the UBWPAD and ending at BAC10. This was followed by another active nitrifying section with values from 1.1 to 0.5 day^{-1} base e at 20ºC ending at BAC16.
- The lower nitrification rates developed for DWS2 represent a situation when the UBWPAD is not providing complete nitrification.
- Sediment Oxygen Measurements (SOD) were measured at six sites with an insitu SOD chamber. These sites were located in different reaches from the sites monitored in the 1991 BRI. The values range from 0.13 (BAC21) to 0.35 (BAC16) g-O_2/ft^2-day at 20ºC. Most SOD values are based upon an average of three chamber placements at each station.
- Two additional chlorophyll *a* sources were determined from the river chlorophyll *a* data. These included the aqueduct just below the UBWPAD discharge, and the impounded area of the Quinsigamond River just before the confluence with the Blackstone River.
- For steady state and dynamic algae simulations, input files of photosynthetically active radiation (PAR) were prepared for each dry weather survey. The PAR data is based on solar radiation data (global solar radiation data) from the National Oceanic and Atmospheric Administration.
- DO values for DWS4 fall below 5.0 mg/L in the reaches below the UBWPAD through Singing Dam. DO values approached 5.0 mg/L for DWS2 (BAC04 and

BAC07) and DWS1 (BAC04). This did not occur in the 1991 BRI, in which DO does not fall below 6 mg/L.

- The non-algal portion of the light extinction coefficient was computed based on volatile suspended solids, total suspended solids, and algae concentration.
- All rates for modeling chlorophyll *a* were the same as the 1991 BRI QUAL2E model except for the settling rate of 1.0 day^{-1}, which was extended through Rice City Pond Dam to BAC16.
- All modeling was evaluated for all constituents by three statistical tests including root mean square error, median relative error and the statistical analysis of observation versus prediction.

11.2 Conclusions

- For chloride, DWS1, 2, 3 and 4 were modeled successfully and provide additional support for the procedure used in developing the river flow profile.
- For $FNBOD_5$, ($CBOD_5$ in QUAL2E) DWS1 and 4 were modeled successfully.
- NH_3-N and NO_3-N loads from UBWPAD were seasonally dependent. Whether or not UBWPAD was providing nitrification had an impact on the instream nitrification rate.
- For NH_3-N, DWS1, 3 and 4 were modeled successfully. During these surveys the UBWPAD was providing a higher level of nitrification than for DWS2.
- For NO_3-N, DWS1, 2, 3 and 4 were modeled successfully. In the reach between BAC03 and BAC07, NO_3-N appearance is a direct result of NH_3-N disappearance.
- The highest SOD demands were found before Singing Dam, Riverdale Dam, and between BAC15 and Rice City Pond.
- Nitrification is the greatest contributor to the DO deficit, followed by $CBOD_5$ and SOD.
- The usual reaeration method by O'Connor and Dobbins was not sufficient for the reach between BAC04 and BAC06 (rapids). The reaeration method by Owen was

used instead.

- Dams play an important role for the recovery of the river. In particular, Singing Dam provides the greatest contribution of dam reaeration.
- In the BRI 1991 surveys, there was no significant productivity in the early reaches of the Blackstone River (BAC03 to BAC07). In contrast, high productivity was evident in the data of the BAC surveys.
- The high productivity in the reach between BAC03 and BAC07 was due to rooted aquatic plants and not algae. In 1991, the UBWPAD, just above BAC03 (BLK02), was discharging levels of residual chlorine that caused measurable toxicity in the reaches directly downstream. De-chlorination started in the mid-1990s. The high productivity from rooted aquatic plants below the UBWPAD outfall was probably a direct result of the elimination of the residual chlorine in the effluent.
- The dense stands of rooted aquatic plants ended by BAC07 and were not evident in the Blackstone River from BAC10 to BAC23. This was attributed to an increase in river depth and the increase in turbidity, which prevented the penetration of light to the channel bottom. Instead, the growth of algae, defined by the increase of chlorophyll *a*, occurred below BAC10. The amount of algae biomass generated was similar in both the BRI and the BAC.
- The presence of macrophytes in the early reaches did not present a problem with respect to nitrogen. Since analysis of the data indicate all ammonia loss resulted in nitrate appearance in these reaches, nitrogen uptake by macrophytes had to be through the root structures not from the water column.
- QUAL2E does not include the simulation of macrophytes. Therefore, the PO_4-P simulations fail to reflect the loss associated with the macrophyte uptake in the reaches between BAC03 and BAC07. Conversely, for DWS1 in October at the end of the growing season, the removal of PO_4-P is not influenced by macrophytes. DWS1 was modeled successfully to BAC17.
- For chlorophyll *a*, DWS1, DWS2, DWS3 and DWS4 were modeled successfully. The modeling relied on solar radiation data (global solar radiation) and the correct

computation of the non-algal portion of the light extinction coefficient.

- For DO, DWS1, DWS2, and DWS4 were modeled successfully. In addition, the BAC version of QUAL2E was successfully applied to the 1991 data sets.
- If the nitrification within the wastewater treatment facility at UBWPAD had progressed more, the daily dissolved oxygen sag would not have been as large downstream. DO sags would only have occurred in the early morning, caused by plant respiration from macrophytes.
- The median relative error for the DO simulations for the BAC and BRI surveys were in the range of 2-3%.

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