FINAL REPORT ON THE SOUTH BURLINGTON, VERMONT

ADVANCED ECOLOGICALLY ENGINEERED SYSTEM (AEES)

FOR WASTEWATER TREATMENT

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1. INTRODUCTION

The Vermont Advanced Ecologically Engineered System (AEES) treats 80,000 gallons per day (300 m^3/d) of raw domestic wastewater to advanced tertiary treatment standards. The facility is located in Chittenden County, Vermont, USA, adjacent to the City of South Burlington, Bartlett Bay Wastewater Treatment Facility.

The Vermont AEES is the final of four AEES demonstration projects funded under a United States Environmental Protection Agency grant to the Massachusetts Foundation for Excellence in Marine and Polymer Science (CX 8220496-01-0). Living Technologies, Inc. is a subcontractor to the Massachusetts Foundation responsible for the design, construction, and operation of the Vermont AEES facility. Other projects funded under the grant have been reported elsewhere.

The purpose of the Vermont AEES demonstration project is comprised of three elements: (1) To demonstrate the treatment performance of an AEES under the cold climate conditions prevailing in Vermont: (2) To test the ability of the system and individual components within the system to treat wastewater to advanced tertiary standards; and, (3) To determine the costs of operating the system under steady state conditions over specified time periods. All three elements are addressed in this report.

1.1. Background

Ecologically engineered technology for wastewater treatment encompasses a diverse array of technologies. Generally, ecologically engineered wastewater treatment systems tend to employ vascular plants as a significant element in biological treatment processes. Some natural wastewater treatment technologies, such as wetlands and water hyacinth lagoons, share this design element with ecologically engineered technologies. Ecologically engineered technologies explicitly incorporate other design elements from the ecological sciences not commonly used in the environmental engineering field, such as incorporating specific wildlife habitats into treatment wetlands.

The Vermont AEES incorporated vascular plants, higher invertebrates, and fish into a modified activated-sludge, extended-aeration treatment process. The AESS design concept was to use both the microbial community attached to plant roots, and suspended, flocculating bacteria to effect nutrient removal in aerated, complete-mix reactors prior to the clarifier. At the clarifier and in post-clarifier filters, higher invertebrates, such as snails, micro-crustacea, and fish were incorporated into the design to consume residual biosolids.

A greenhouse to protect the plants used in wastewater treatment encloses the Vermont AEES. The area occupied by the greenhouse, 730 m^3 (7,800 ft^2), was designed to accommodate school and community tours, which were an integral part of the educational

mission of the project[1], and to provide space for a smaller test treatment train. Approximately 60% of the greenhouse footprint is occupied by treatment components, appurtenances, and working space for the operator.

Within the greenhouse, the plants have also provided a uniquely beautiful aesthetic experience for the public and operators. Aesthetics were not a design parameter for the Vermont AEES, but the appeal of the plants has inspired enthusiasm for wastewater treatment in hundreds of people who have toured the facility.

Design of the Vermont AEES began in 1994. Construction was completed in 1995. Biological ramp-up began in December 1995 and was completed in April 1996. Steady state operations were reached in May 1996. The project closed at the end of 1999.

1.2. Report Scope and Purpose

This report describes and analyses wastewater treatment at the Vermont AEES during the steady state period from May 1996 through August 1999. Operations in the closing months of 1999 were devoted to obtaining supplemental data for this report.

The purpose of this report is to present an assessment of the AEES technology as configured at the South Burlington, Vermont site. Care has been taken to include both strengths and shortcomings of the Vermont AEES in the analyses that follow.

1.3. Report Organization

The principal topics covered in this report are the overall treatment performance of the two main treatment trains, the performance of individual components in the treatment system, and results of experiments conducted on a smaller test train. Section 2 covers the design basis, physical description, testing, and overall treatment performance of the Vermont AEES. Section 3 covers the performance of Vermont AEES system components. Section 4 covers studies conducted on a small test train located in the greenhouse. Section 5 covers costs of operations. There are appendices that provide further explanation of topics covered in the report, and the raw data for the analyses conducted in this report.

1.4. Invention Disclosure

All information presented in this report is in the public domain. No intellectual property rights or proprietary inventions are claimed for knowledge or products developed under funding from this project.

[1] Education is addressed in a separate report.

2. DESIGN BASIS AND TREATMENT PERFORMANCE ANALYSIS

2.1. INTRODUCTION

The purpose of this chapter is to describe the design basis, physical characteristics, process diagram, testing protocol, and treatment performance of the Vermont AEES. The principal focus of this chapter is on overall treatment performance. Detailed analysis of the treatment performance of individual elements of the system is reserved for the following section.

The Vermont AEES has met or bettered design treatment performance for five day carbonaceous biochemical oxygen demand ($CBOD_5$), total suspended solids (TSS), total Kjeldahl nitrogen (TKN), ammonia (NH_3), nitrate (NO_3), total nitrogen (TN), fecal coliform bacteria, and total phosphorous (TP). Treatment performance for total phosphorous (TP), however, would not meet the common advanced tertiary treatment discharge standard of 1 mg/L.

Efforts to optimize denitrification performance enter into most of the performance analyses of this chapter. Post-clarifier denitrification began in the early months of the project, using methanol as a carbon source in an anoxic EFB. Denitrification with methanol was adequate, but methanol added BOD, COD, and, indirectly, TSS to the effluent. Methanol had overall adverse impacts on treatment performance. Elimination of methanol, in favor of a single-stage, denitrification process prior to the clarifier, significantly improved performance of parameters had previously been adversely affected by methanol. It is therefore very important to account for the affects of methanol on treatment performance. Many of the analyses that follow differentiate treatment performance between the period when methanol was used and two separate periods when no methanol was used.

2.2. Design Treatment Performance

The Vermont AEES is designed to treat 300 m^3/d (80,000 gpd) of raw domestic wastewater to tertiary standards. The design parameters are $CBOD_5$, TSS, TKN, NH_3, NO_3, TN, TP and fecal coliform (Table 2-1). Chemical oxygen demand (COD) was not a design parameter, but was included for treatment performance evaluation.

The design reaction rates for $CBOD_5$ removal and nitrification were conservative. Treatment of $CBOD_5$ to 10 mg/L was to have taken place just before the clarifier. Nitrification was designed to be completed in the EFB immediately downstream of the clarifier. Please refer to Section 3.1 for a discussion of reaction rates and model equations.

2.3. Testing Procedures - Quality Assurance and Control.

Except where noted, treatment performance data are from samples processed by a certified contract laboratory. Influent and effluent samples are 24-hour composites. An in-house laboratory provided process testing twice per week and on an as needed

Table 2-1. Design parameters and standards.

Parameter	Influent mg/L	Effluent mg/L	Total influent kg/d	Total effluent kg/d	Influent per train, kg/d	Effluent per train, kg/d
$CBOD_5$	350	10	106	3	53	1.5
TSS	190	10	57	3	29	1.5
TKN	40	5	12	1.5	6.1	0.8
NH_3	30	1	9	0.3	4.5	0.2
NO_3	0	5	0	1.5	0.0	0.8
TN	40	10	12	3	6.1	1.5
TP	10	3	3.0	0.9	1.5	0.5
FC*	7.5×10^6	2000	***cfu/100 ml**			

basis. Parameters used for process testing were COD, TSS, NH_3, and NO_3. Weekly samples sent to the contract laboratory were split and tested in the in-house process laboratory. All in-house process laboratory testing was conducted according to Standard Methods.

Splitting of samples was the foundation of the quality assurance program to verify accuracy of in-house process laboratory results. The in-house process laboratory tended to get slightly higher results for NH_3, NO_3, and COD, but slightly lower results for TSS than did the contract laboratory (Table 2-2).

Additional quality assurance and control measures involved critical review of results from both the contract and process laboratories. Unusual results from the contract laboratory were re-tested within sample holding times permitted by Standard Methods. Suspected testing or sample labeling errors were flagged as unacceptable in data entry spreadsheets.

The in-house laboratory was used for testing of effluent parameters at intermediate points in the treatment system. Except were noted, intermediate position results are from grab samples.

2.4. Process diagram

In reviewing treatment performance of the Vermont AEES, it is important to consider the changes in the process diagram that have been made since the completion of ramp-up at the end of April 1996. These changes are documented in the process diagram (Figure 2-1).

Table 2-2. Comparison of contract and in-house process laboratory split samples.

		Contract Laboratory			In-house Process Laboratory		
Parameter mg/L	**Statistic**	**Influent**	**A-train effluent**	**B-train effluent**	**Influent**	**A-train effluent**	**B-train effluent**
NH_3	**mean**	**13.9**	**0.4**	**0.4**	**14.5**	**0.4**	**0.4**
	s	4.9	0.5	0.9	3.8	1.5	0.9
	n	77	54	55	70	67	68
NO_3	**mean**	**NA**	**4.7**	**5.5**	**NA**	**5.3**	**6.6**
	s		3.2	3.1		3.6	3.7
	n		71	66		71	66
TSS	**mean**	**192**	**7.8**	**3.6**	**158**	**3.4**	**2.7**
	s	108	8.3	3.8	476	2.4	2.6
	n	82	82	81	35	35	34
COD	**mean**	**488**	**40**	**28**	**557**	**39**	**28**
	s	191	21	13	172	16	10
	n	80	76	78	73	69	72

2.4.1. Headworks

Description: The headworks consist of a submersible pump located at the end of the degritting channel in the City of South Burlington, Vermont Bartlett Bay Sewage Treatment Plant. The pump is located inside a screen for rag protection.

Flows: The influent pump delivers 300 m^3/d (80,000 gpd) of raw influent that is split under pressure between two equally configured treatment trains, A and B. The average influent pump rate is 5.7 L/s (90 gpm). Influent pumping is cycled by a pin timer in 15-minute increments to approximate diurnal flow fluctuations (Figure 2-2).

2.4.2. Aerated reactors

Description: There are five aerated reactors in each train. Raw influent enters the first reactor. Effluent from the fifth reactor flows to the clarifier. The working volume of each reactor is 57 m^3 (14,900 gallons). The hydraulic residence time (HRT) in all five reactors was 45 hours at the design influent flow of 152 m^3/d.

Function: The aerated reactors are designed for nutrient removal. The nutrient removal design basis changes during the project are outlined below (
Table 2-3). In August 1996 automated return activated sludge (RAS) began in both trains. Other process changes, described in detail later in this report, modified the treatment trains to denitrify recycled nitrified effluent to a front end, single stage anoxic reactor.

2.4.3. Clarifier and solids holding tank

Description: Volume of the clarifier is approximately 25 m^3 (6,600 gallons) with an HRT of four hours. Biosolids pumped by airlift from the clarifier are deposited in the

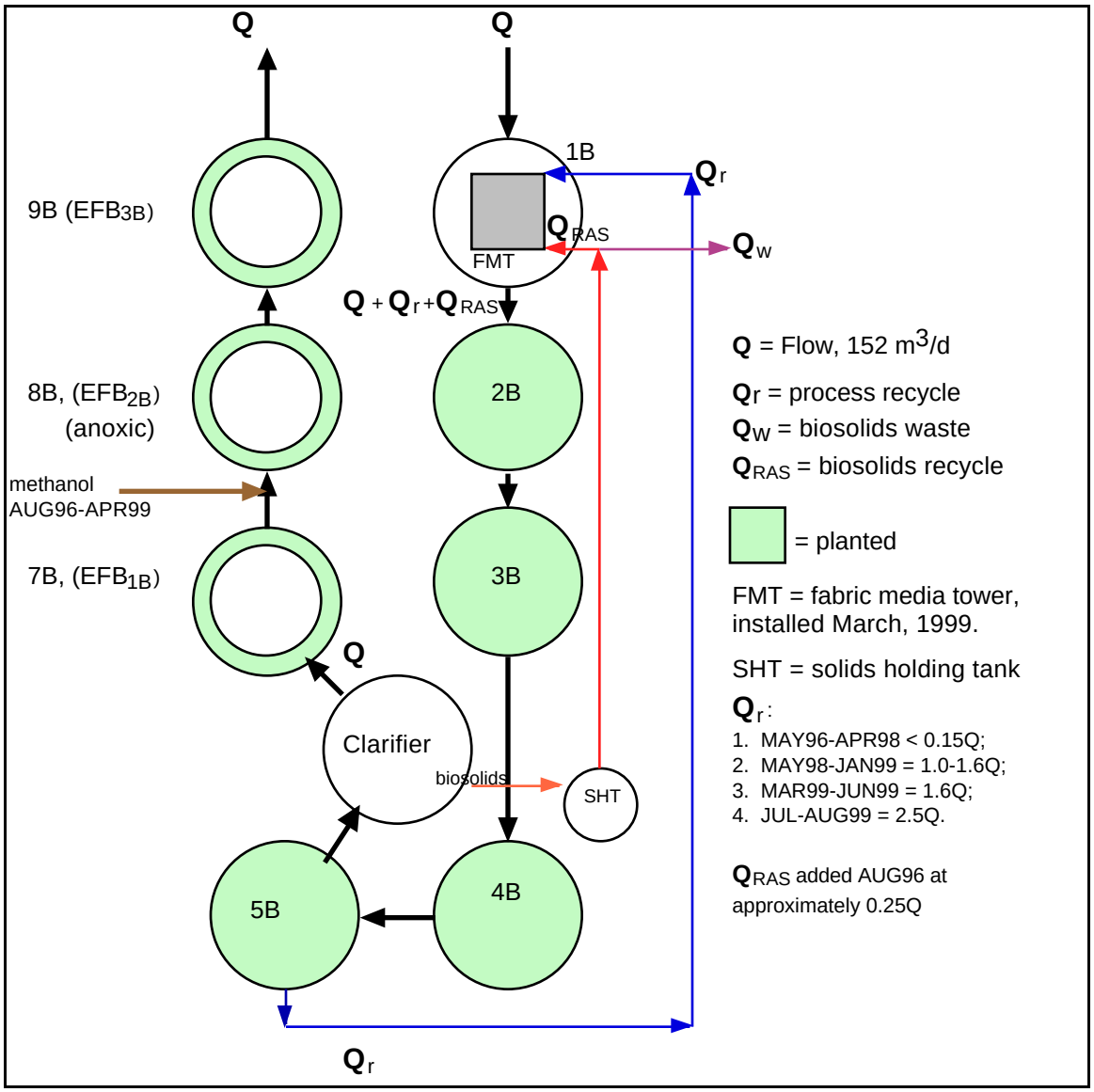

Figure 2-1. Process schematic diagram for the Vermont AEES B-train. The A-train is a mirror image of this diagram except for reactor 1A. Reactor 1A does not have an FMT.

solids holding tank (SHT). A pump located in the SHT sends biosolids to the first aerated reactor or to waste. The operator manually sets valves to waste or recycle.

Function: The clarifier settles biosolids. Biosolids are either recycled to the first reactor in each train or are wasted.

2.4.4. Ecological Fluidized Beds (EFB)

Description: Each EFB is a recirculating, downflow, vertical rock filter. The liquid volume of each EFB is approximately 39 m^3 (10,300 gallons) after displacement by media. The HRT is approximately six hours. The first EFB uses 1.5 inch nominal

Table 2-3. Process changes to aerated reactors.

Date	Design nutrient removed	Process/Change to Aerated Reactors
MAY-JULY96	$CBOD_5$, COD, TP	**Both trains**. No return activated sludge (RAS). Process recycle (reactor 5 to reactor 1) < 15 m^3/d.
AUG96-AUG99	$CBOD_5$, COD, TKN, NH_3, TP	**Both trains**: RAS started August 1996. RAS flow approximately 38 m^3/d. Process recycle < 15 m^3/d.
APR98-JAN99	$CBOD_5$, COD, TKN, NH_3, NO_3, TP	**B-train**: Reactor 1B run in anoxic mode by cycling air delivery with timer. Fine bubble diffusers in reactor 1B. Process recycle of 242 m^3/d. **A-train**: No change.
MAR-JUN99	$CBOD_5$, COD, TKN, NH_3, NO_3, TP	**B-train**: Fabric media in reactor 1B. Reactor 1B run in anoxic mode by automated air cycling control using oxidation-reduction potential (ORP) probe. Coarse bubble diffusers installed. Process recycle of 242 m^3/d. **A-train**: No change.
JULY-AUG99	$CBOD_5$, COD, TKN, NH_3, NO_3, TP	**B-train**: No change except 5B to 1B process recycle increased to 340 m^3/d. **A-train**: Reactor 1A run in anoxic mode with coarse bubble diffusers. No air cycling. Process recycle increased to 340 m^3/d.

diameter lava rock for media, the second and third use 0.75-inch lava rock. Media is cleaned by air-scour backflush. Biosolids produced from backflushing settle in a clear water annulus surrounding the media bed, then are pumped to the clarifier. There is no fluidization of media in the EFB. Other EFB designs employ buoyant media that fluidize on air-scour backflush. For this application a non-buoyant media was used.

Function: The first EFB is aerobic and was designed to nitrify clarifier effluent. Four airlift pumps achieve recirculation of wastewater through the media. The internal recirculation rate is approximately 5Q.

The second EFB is anoxic and was designed to denitrify. Wastewater originally recirculated through the medium in an upflow configuration, but was converted to a down flow configuration in 1997. The design rate of recirculation was approximately 5Q. The current recirculation rate is approximately 3Q. In August 1996, methanol dosing started to provide a carbon substrate for denitrification. Methanol dosing was discontinued by May 1999 for the B-train and July 1999 for the A-train.

The third EFB was designed for final effluent polishing of TSS. It has the same hydraulic configuration as the first EFB, but uses the 0.75-inch media.

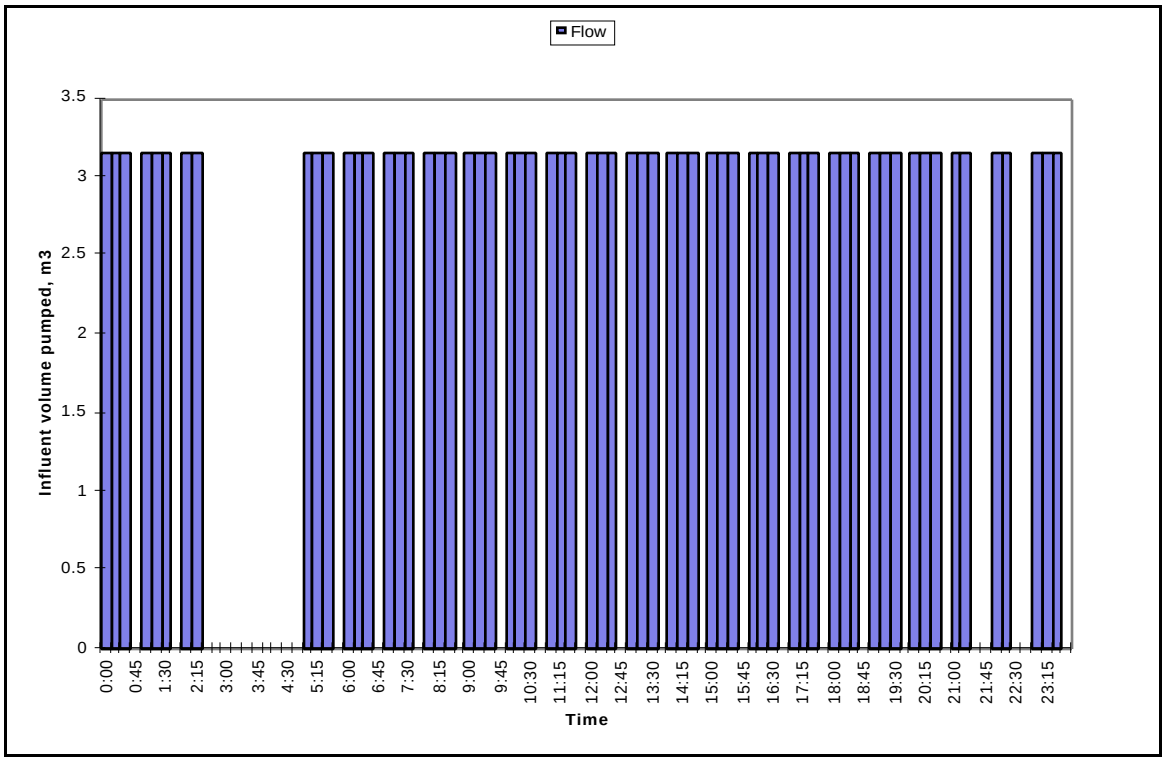

Figure 2-2. Influent flow cycling on 15-minute pin timer increments.

2.4.5. Discharge

Effluent from the final EFB flows to an effluent sump where it is pumped to the Bartlett Bay headworks downstream of the AEES influent pump.

Additional information on the process diagram is presented in the sections below.

2.5. Flows

Influent flows to each train in the Vermont AEES have been within 7% of design (Table 2-4). The frequency distribution[2] of influent flows is 50% less than design, and 50% greater than design (Figure 2-3). In Figure 2-3, a line plotted at the rate of constant design flow (152 m^3/d) crosses the data plot at the 50% value on the percent frequency axis. This value is the median for the data set. A line plotted from the flow rate of 100m^3/d to the data plot would cross at the 10% frequency. Ten percent of all influent flows were 100 m^3/d or less, and 90% of all influent flows were greater than 100 m^3/d. Although the pump should deliver the same flow every day, there have been practical limitations. The influent pump screen has been frequently clogged by rags, resulting in reduced flows. Flow imbalances between the two trains occur at least once per week. Cleaning of the

[2] Frequency distribution graphs are a principal analytical tool of this report. They are a common analytical tool used to evaluate treatment system performance. See Appendix A for more information.

Table 2-4. Mean flows. December 1996 - August 1999.

	A-train	**B-train**
Mean influent flow, m^3/d	148	140
Percent less than design flow, 152 m^3/d	2%	7%
standard deviation (σ), m^3/d	38	36

screen and purging of lines restores full and balanced influent flows. The operational response to clogging is to increase the influent pumping time to ensure that the combined A and B-train weekly total flow meets design. No attempt is made to compensate for flow imbalances. The frequency distribution of daily influent flows thus reflects both flow imbalance between the two trains, and the variability caused by clogging and compensatory pumping.

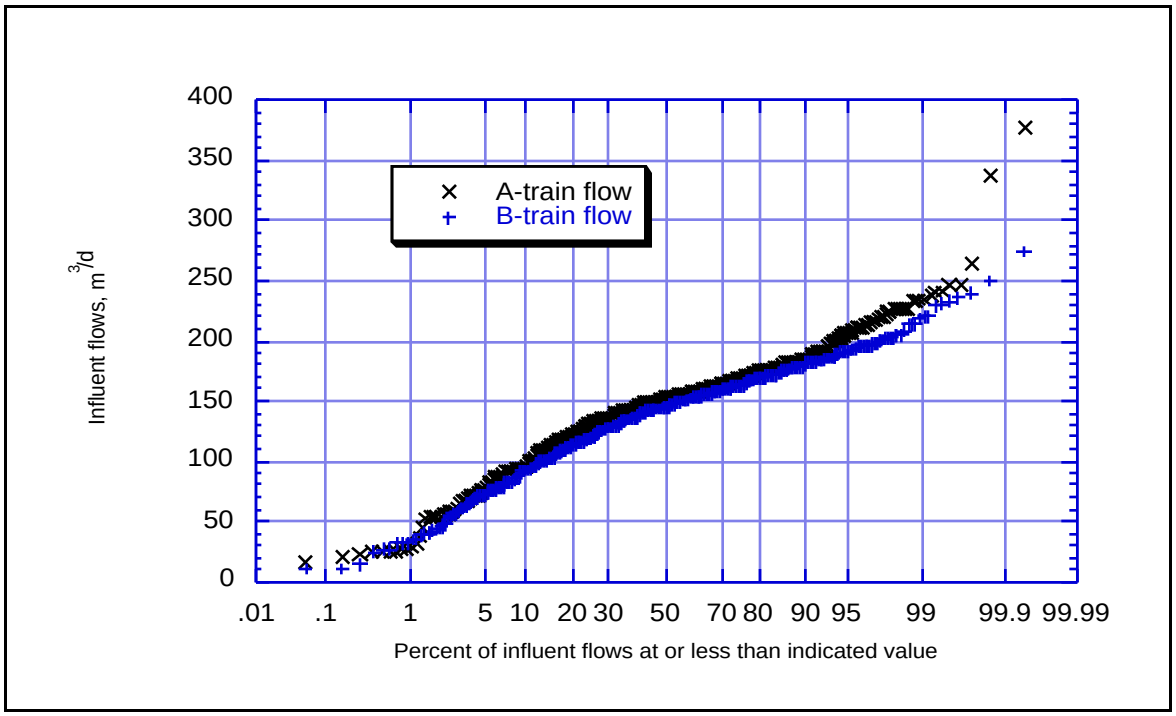

Figure 2-3. Daily influent flows for A and B-trains, December 1996-August 1999. Combined influent design flow is 303 m^3/d.

2.6. Temperatures

Temperatures have fluctuated seasonally between 11° C and 22° C since May 1996 (Figure 2-4). The coldest temperatures were in late winter and early spring when there is significant snowmelt infiltration of the South Burlington sewer system.

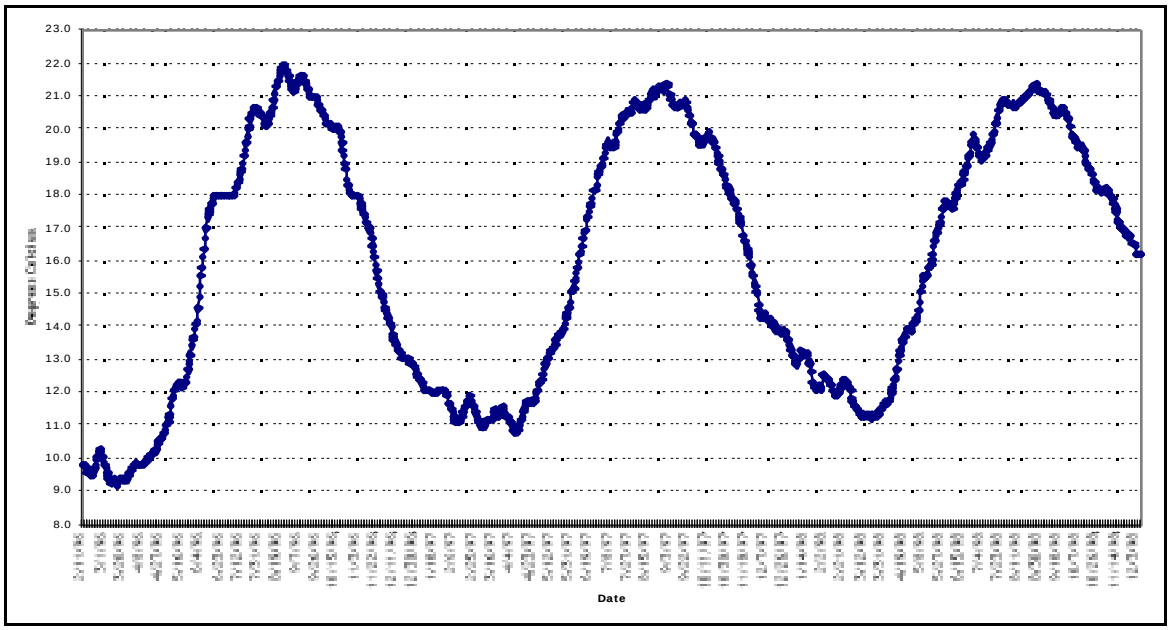

Figure 2-4. Influent temperatures.

2.7. Carbonaceous Biochemical Oxygen Demand Removal

Summary: The Vermont AEES facility reliably meets $CBOD_5$ design effluent concentrations of less than 10 mg/L when methanol is not added for denitrification. For periods when no methanol was added, $CBOD_5$ effluent concentrations were less than 10 mg/L for all monthly average values, and for 98% of total effluent samples. Methanol addition increased effluent $CBOD_5$ values. While methanol was added, 18% of monthly average $CBOD_5$ effluent values exceeded 10 mg/L. Process tests in the summer of 1999 demonstrated that methanol is not needed for denitrification with the current process design.

Introduction

The $CBOD_5$ design influent and effluent concentrations for the Vermont AEES are 350 mg/L and 10 mg/L, respectively. On a mass basis, design for each treatment train is 53 kg/d influent and 1.5 kg/d effluent at design flows of 152 m^3/d (40,000 gpd) per train.

During periods without methanol addition for denitrification, actual mean influent concentration was approximately 200 mg/L, and mean effluent concentrations were less than 5 mg/L per train. Actual mass loading was approximately 38 kg/d per train. Effluent mass flux was less than 0.7 kg/d per train.

Of the forty months of $CBOD_5$ removal performance presented in this report, eight months are from samples collected in two periods when methanol was not used: May-August 1996 and May-August 1999. As described below, effluent $CBOD_5$ excursions greater than 10 mg/L (and 1.5 kg/d) were directly related to the addition of methanol.

Table 2-5. A and B-train effluent $CBOD_5$ concentration values with and without methanol addition for denitrification. Data are from all samples. Data for A-train reflect dosing dates.

$CBOD_5$ (mg/L)	Without methanol May-August 1996, May-August 1999			With methanol September 1996 - April 1999* * A-train methanol dosing continued into June 1999.		
	Influent	A-train effluent	B-train effluent	Influent	A-train effluent	B-train effluent
Mean	206.6	4.2	3.4	245.7	9.8	5.2
σ	77.9	2.4	2.0	77.7	7.1	4.3
n	41	39	41	103	98	101

2.7.1. $CBOD_5$ removal performance on a concentration basis

Addition of methanol for denitrification significantly raised $CBOD_5$ effluent concentrations of the Vermont AEES. Methanol was used for denitrification throughout the project except for two periods: May-August 1996 and May-August 1999. There is a sharp distinction in $CBOD_5$ effluent concentration values between the periods of methanol and no methanol addition, as analyzed by mean values, time series data and frequency distributions.

Effluent $CBOD_5$ concentration mean and standard deviation values for all samples during periods without methanol addition were approximately half of those during the period of methanol addition (Table 2-5). During periods with no methanol addition, effluent $CBOD_5$ concentrations one standard deviation above the mean were approximately 7 mg/L and 5 mg/L for the A and B-train, respectively, whereas with methanol addition the values increased to approximately 17 mg/L and 10 mg/L, respectively.

Monthly mean $CBOD_5$ effluent values were significantly higher during the period with methanol addition than the two periods without methanol addition (Figure 2-5). Addition of methanol is strongly associated with effluent values greater than 10 mg/L, absence of methanol is associated, without exception, with effluent $CBOD_5$ values less than 10 mg/L.

Heterotrophic fouling of media in the second and third EFB in each train contributed significantly to $CBOD_5$ effluent concentration excursions greater than 10 mg/L. When media beds were refurbished in January 1998, $CBOD_5$ effluent concentrations dropped below 10 mg/L even though there was no reduction in methanol addition (Figure 2-5).

Excursions of $CBOD_5$ concentration values within three months of the addition of methanol in 1996, and within two months after the refurbishment of media beds in 1998,

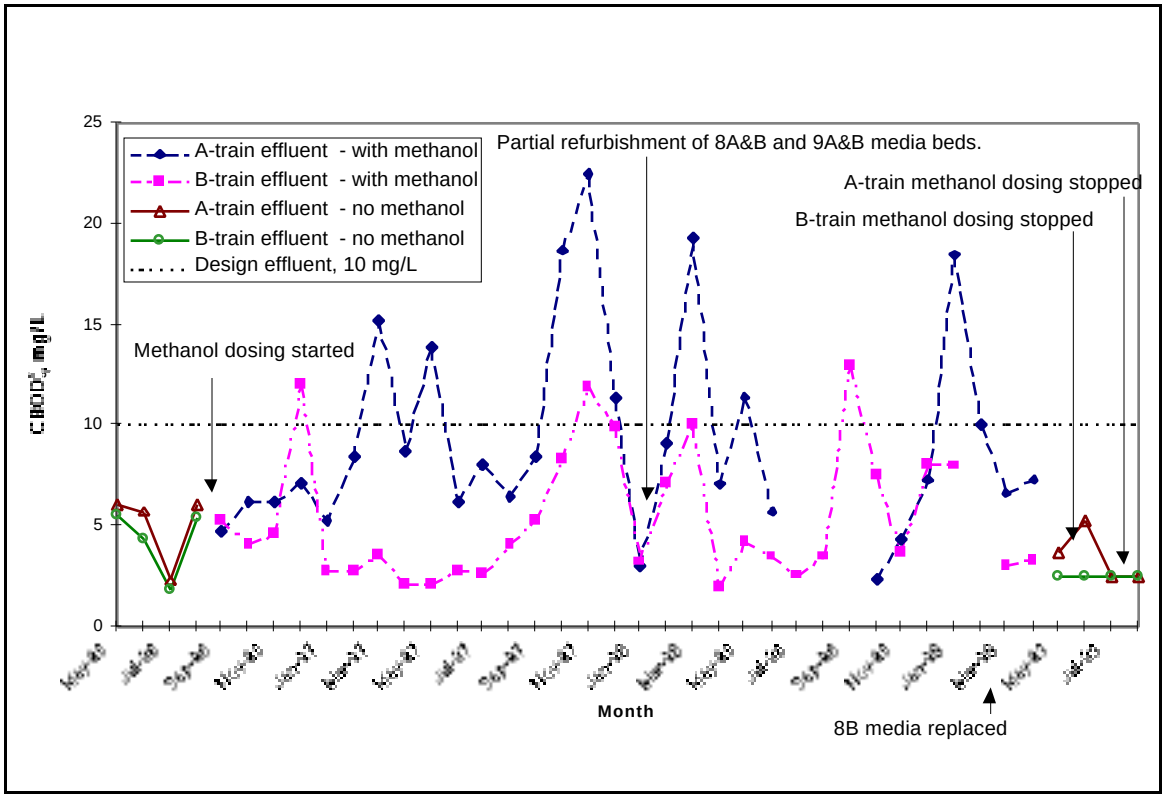

Figure 2-5. Effects of methanol addition and filter clogging on monthly average effluent $CBOD_5$ values.

coincide with the progression of media bed clogging observed during the same time frames. The difference in performance between the A and B-trains is probably due both to flow imbalances, which tended to favor the A-train (Figure 2-3), and more severe media clogging observed in EFB2A and EFB3A than in the corresponding EFBs in the B-train.

The frequency distributions of $CBOD_5$ effluent values for periods with and without methanol addition are markedly different. When no methanol was used, 98% of all $CBOD_5$ effluent samples were less than or equal to 10 mg/L (Figure 2-6). In contrast, only 60% of A-train and 85% of B-train effluent concentration values were less than 10 mg/L while methanol was added for denitrification (Figure 2-7). The frequency distribution of influent concentrations was virtually identical for both the period of methanol addition and the period without methanol addition (Figure 2-6, Figure 2-7). Therefore the difference in the distribution of effluent values between periods with and without methanol addition cannot be attributed to a difference in influent strength.

The effect of methanol addition can be seen in $CBOD_5$ concentration frequency distribution values through EFB1B, EFB2B, and EFB3B from samples taken from each EFB on the same day (Figure 2-8). At EFB1B, downstream of the clarifier and upstream of methanol addition, the $CBOD_5$ concentrations were all less than 10 mg/L. Treatment performance of $CBOD_5$ is stable at EFB1B as evidenced by the shallow slope of the data

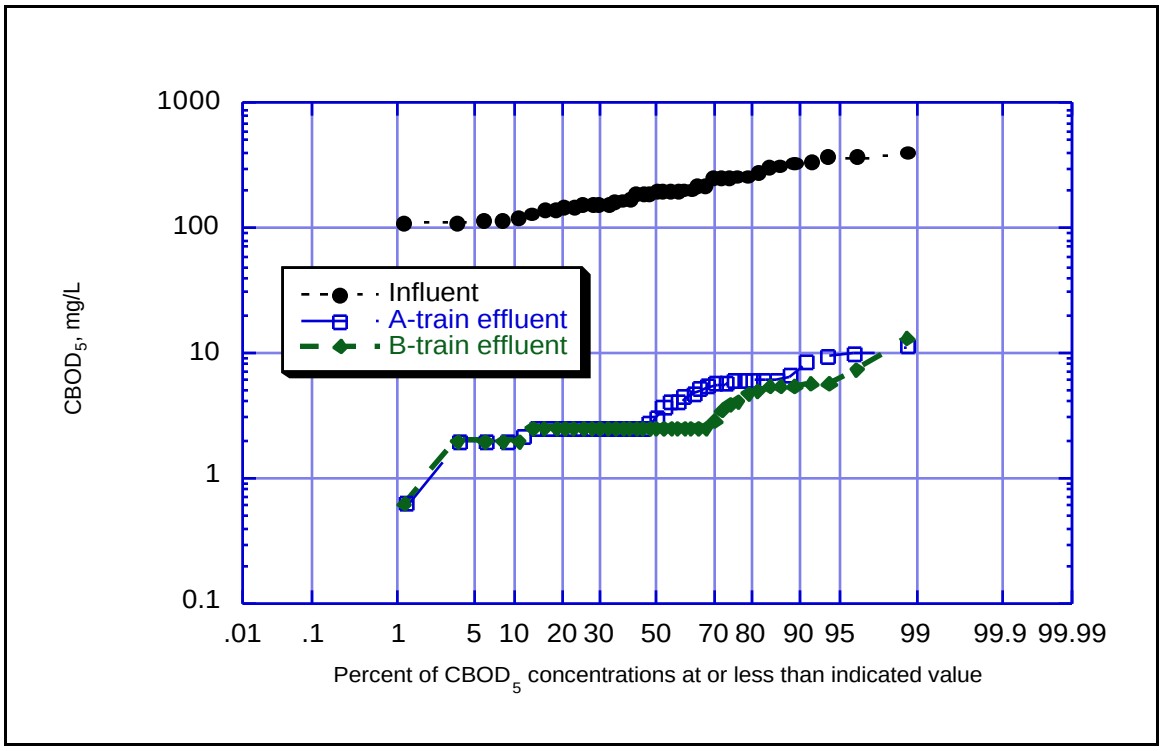

Figure 2-6. A and B-train $CBOD_5$ removal <u>without</u> methanol addition for denitrification. n=78.

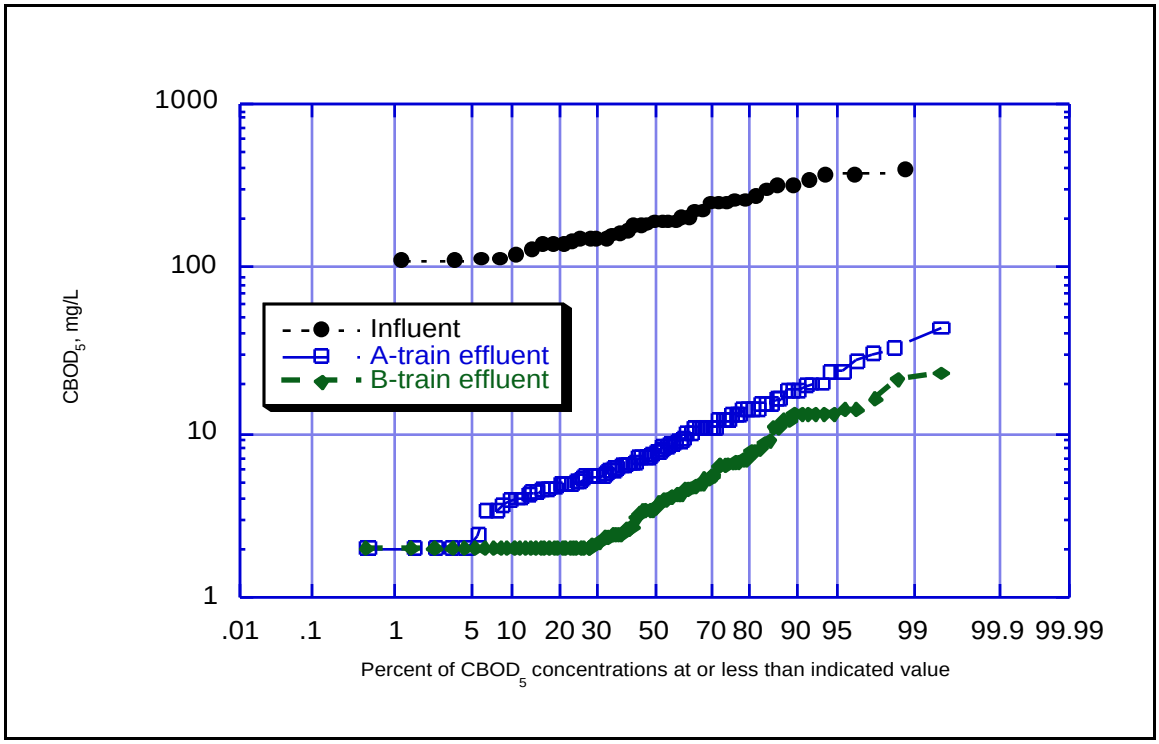

Figure 2-7. A and B-train $CBOD_5$ removal <u>with</u> methanol addition for denitrification. Data are all samples from each period specified.

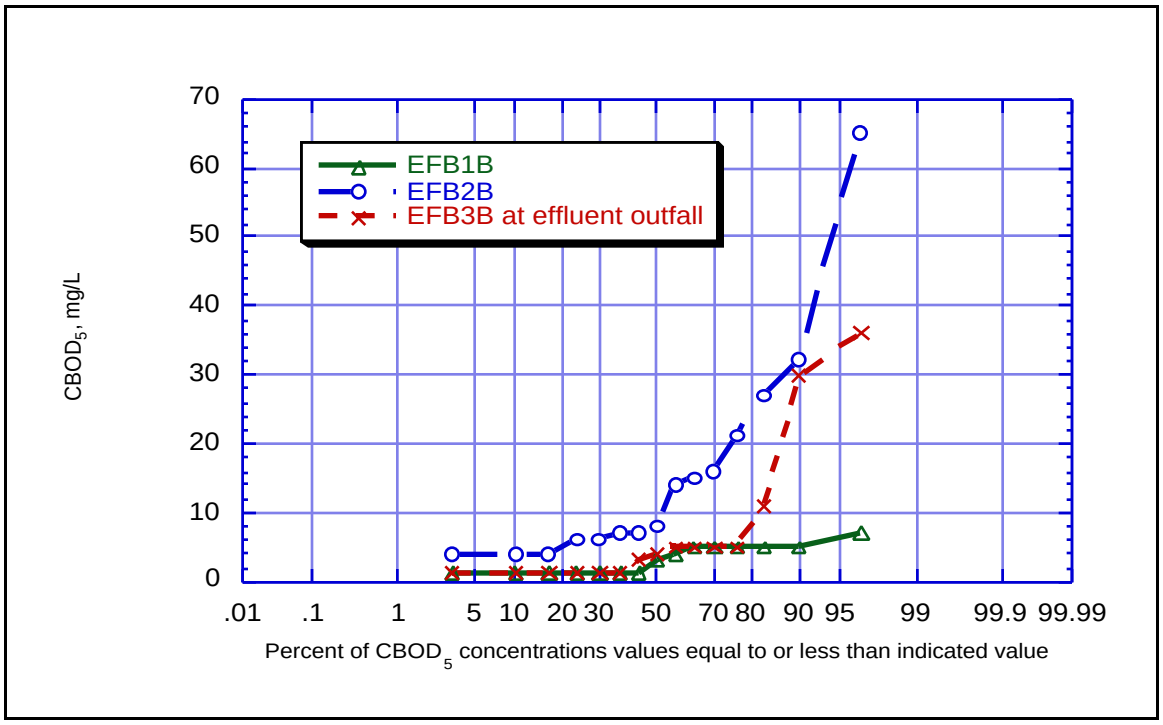

Figure 2-8. Effect of methanol addition on $CBOD_5$ effluent profile of B-train EFBs, December 1998 - April 1999. Number of sample sets, n = 15. Each sample set consists of three grab samples, one from each EFB taken on the same day. EFB1B is immediately after the clarifier. EFB2B is point of methanol addition. EFB3B is downstream of EFB2B.

plot. At EFB2B, where methanol is added, $CBOD_5$ concentrations increased an average of 658% compared to those in reactor EFB1B. Methanol added to EFB2B broke through to EFB3B, causing approximately 20% of EFB3B $CBOD_5$ concentrations at the effluent outfall to exceed 10 mg/L (Figure 2-8). Because the sample point at EFB3B is the effluent outfall, the breakthrough of $CBOD_5$ in EFB3B can be reasonably assumed to appear in samples take from the effluent composite sampler. Excursions of $CBOD_5$ effluent concentrations above 10 mg/L observed in Figure 2-8, have approximately the same frequency as the overall B-train $CBOD_5$ excursions during the period of methanol addition (Figure 2-7).

2.7.2. $CBOD_5$ removal performance on a mass basis

During periods without methanol addition, the Vermont AEES met the design $CBOD_5$ mass discharge of 1.5 kg/d for 90-97% of all samples taken. Comparison of the frequency distribution of effluent mass flux for periods with and without methanol addition reveals significant performance differences (Figure 2-9). Approximately 97% of B-train mass flux values for the B-train were less than or equal to 1.5 kg/d during periods with no methanol addition, but with methanol addition approximately 88% were less than or equal to 1.5 kg/d. The A-train exhibits even greater differences. Without methanol addition approximately 90% of effluent mass flux values are less than or equal to 1.5 kg/d, but only approximately 75% of effluent values meet the same standard during the period of

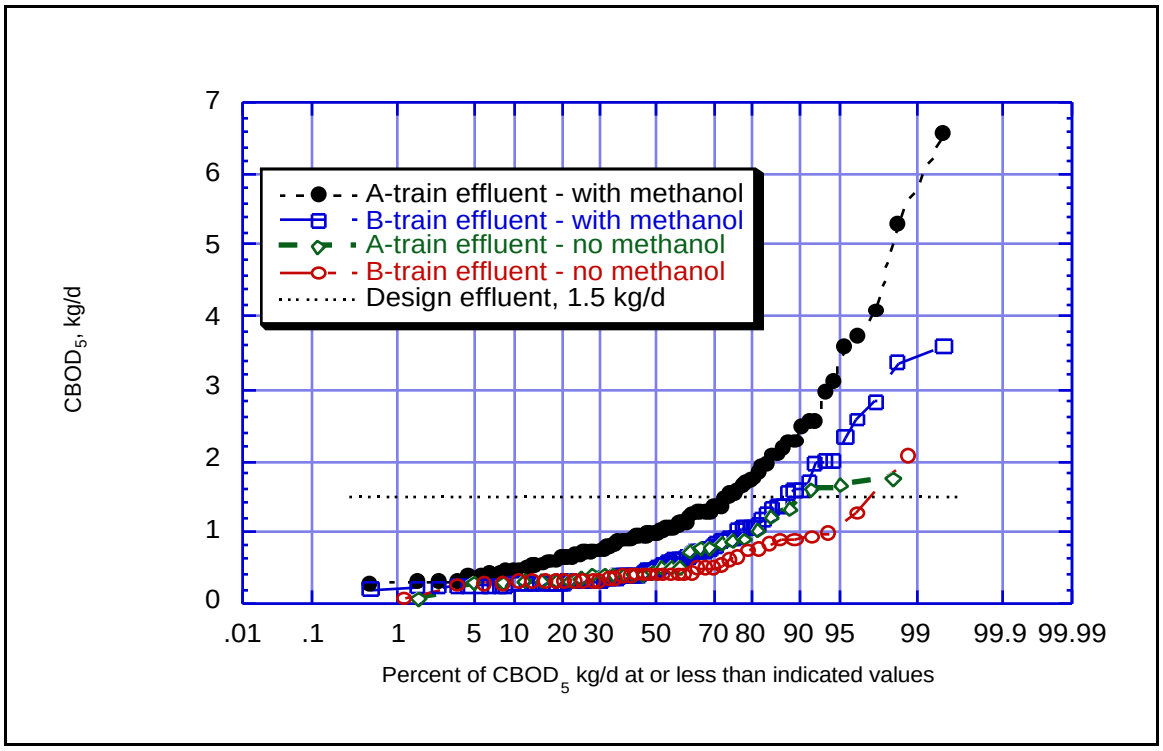

Figure 2-9. A and B-train $CBOD_5$ effluent mass flux. Data are all samples from each period specified.

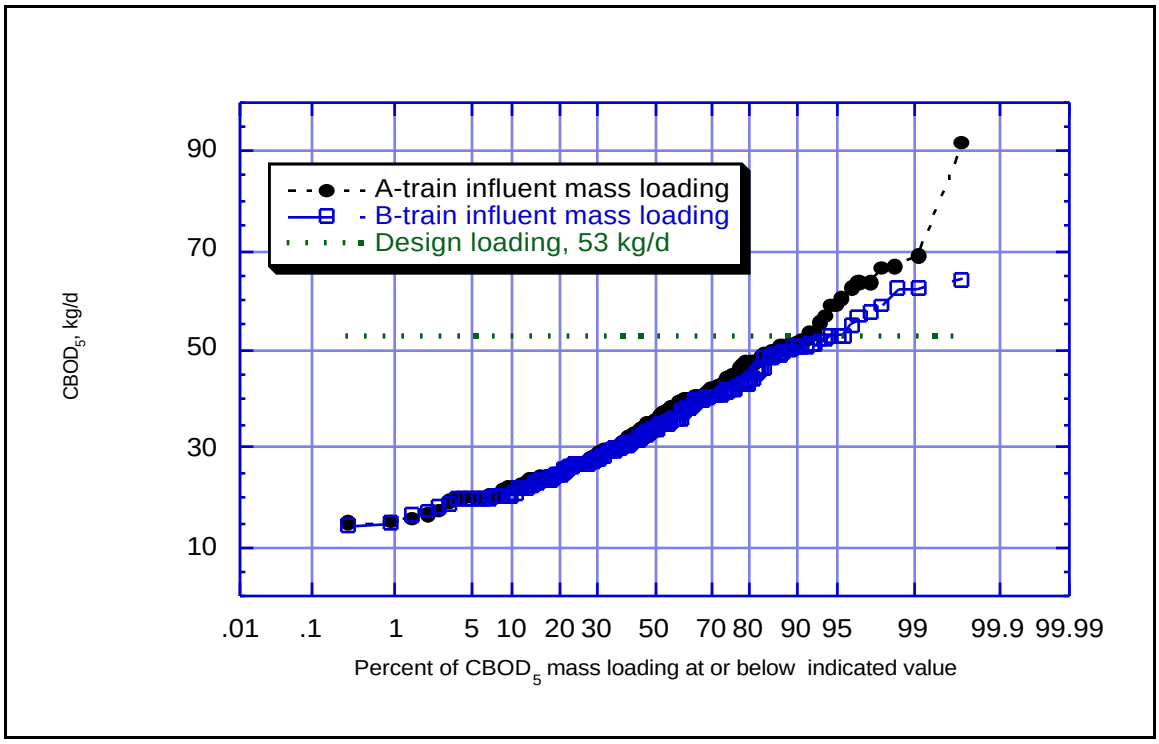

Figure 2-10. A and B-train $CBOD_5$ influent mass loading.

methanol addition. The difference in performance between the A and B-trains is probably due both to flow imbalances, which tended to favor the A-train (Figure 2-3), and the more severe media clogging observed in EFB2A and EFB3A than in the corresponding EFBs in the B-train.

Mass loading of the Vermont AEES was less than anticipated. Approximately 92% of $CBOD_5$ mass loading values were less than design (Figure 2-10). Design loading was based on influent data provided by the City of South Burlington, Vermont. The City's data was obtained from eight-hour composite samples, whereas influent data for this study were obtained from twenty-four hour composite samples, which include relatively dilute nighttime flows

The $CBOD_5$ removal performance does not appear attributable to weaker influent. Mean influent loading was 42% less than loading, but mean effluent mass flux was 66% less than design effluent mass flux (Table 2-6). Design mass $CBOD_5$ removal is 97%. Actual $CBOD_5$ mass removal is 98%. Further evidence of efficient $CBOD_5$ removal is found in the performance of reactor 1B. From December 1998 to August 1999 mean $CBOD_5$ removal in reactor 1B was approximately 85% (σ = 13%, n = 20). At the observed reaction rate coefficient of 10.7 d^{-1}, a continuous flow stirred reactor (CFSTR) in series model shows an influent concentration of 350 mg/L treated to less than 10 mg/L in reactor 4B[3]. Both actual and model performance, therefore strongly suggest that design influent $CBOD_5$ loadings could have been treated to design effluent standards.

2.7.3. Yield

Yield[4] values from the raw influent at the Vermont AEES range from 0.4 to 0.5 kg VSS/kg $CBOD_{5\ in}$. Yield at wastewater treatment facilities without primary treatment, such as at the Vermont AEES, cannot be directly compared to the yield of facilities with primary treatment. The reason why the comparison is invalid is because there is a significant contribution to yield from volatile solids in raw effluent. Primary treatment (settling or advanced screening) removes a significant fraction of volatile solids. Therefore, yield at the Vermont AEES is the product both of volatile solids in raw influent and the conversion of nutrients into biomass.

It is possible to roughly estimate what the yield would be for the Vermont AEES if it were preceded by primary treatment. Dividing by the conversion factor[5] of 1.7 (1.7 x aerated reactor yield preceded by primary settling = aerated reactor yield without primary settling), gives an approximate yield range of 0.2 to 0.3 kg VSS/kg $CBOD_{5\ in}$.

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[3] See $CBOD_5$ process model in the next section.
[4] yield = (kilograms volatile suspended solids measured at the clarifier)/(kg influent $CBOD_5$)
[5] Clark, J., Viessman, W., Hammer, M., 1977. **Water Supply and Pollution Control.** pp. 633-566., Harper and Row, New York.

Table 2-6. Comparison of mean design and actual mass loading, effluent mass flux and percent $CBOD_5$ removal in the B-train for periods when no methanol was added (n = 41).

Mean actual $CBOD_5$ loading, kg/d	Design $CBOD_5$ loading, kg/d	Percent difference between actual and design $CBOD_5$ loading	Mean actual effluent mass flux, kg/d	Design effluent mass flux, kg/d	Percent difference between actual and design $CBOD_5$ loading	Design mass removal	Actual mass removal
30.6	**52.4**	**42%**	**0.5**	**1.5**	**66%**	**97%**	**98%**

Conclusion

The Vermont AEES consistently meets effluent $CBOD_5$ concentrations of less than 10 mg/L when no methanol is added for denitrification. Effluent $CBOD_5$ excursions greater than 10 mg/L are almost entirely associated with methanol addition. Methanol addition is neither used nor needed for denitrification in the current process configuration. Yield values are 0.4-0.5 kg VSS/kg $CBOD_{5\ in}$, but would be approximately 0.2-0.3 kg VSS/kg $CBOD_{5\ in}$ if feed to the Vermont AEES were primary effluent instead of partially screened raw influent.

2.8. Chemical Oxygen Demand Removal

Summary: The Vermont AEES facility can reliably meet effluent COD concentrations less than 30 mg/L in the current configuration. Methanol dosing raised effluent COD concentrations.

Introduction

Carbon removal in the Vermont AEES was designed on a BOD basis. There is no design COD removal performance, but 30 mg/L is an adopted effluent standard. Effluent COD concentrations were significantly higher during periods of methanol addition than periods without methanol addition. The unofficial effluent target concentration was consistently met for periods of no methanol addition.

COD removal performance on a concentration basis

The effect of methanol addition on COD removal performance closely parallels that of $CBOD_5$. Methanol raised effluent COD concentrations above the unofficial effluent target of 30 mg/L (Figure 2-11). The B-train effluent COD concentration frequency distribution was approximately 95% less than or equal to 30 mg/L without methanol addition, but was approximately 50% with methanol addition. The A-train effluent COD concentration frequency distribution was approximately 82% less than or equal to 30 mg/L without methanol addition, but was approximately 35% with methanol addition. As with $CBOD_5$, the effects of methanol addition on COD removal are dramatic when viewed over forty months of operations (Figure 2-12). With the addition of methanol,

Table 2-7. A and B-train effluent COD concentrations with and without methanol addition for denitrification.

COD, (mg/L)	**With methanol** September 1996 - April 1999* * A-train methanol dosing continued into June 1999. Data for A-train reflect dosing dates.			**Without methanol** May-August 1996, May-August 1999		
	Influent	A-train effluent	B-train effluent	Influent	A-train effluent	B-train effluent
Mean	561	44	32	546	25	23
σ	176	19	11	154	8	4
n	125	123	114	38	30	40

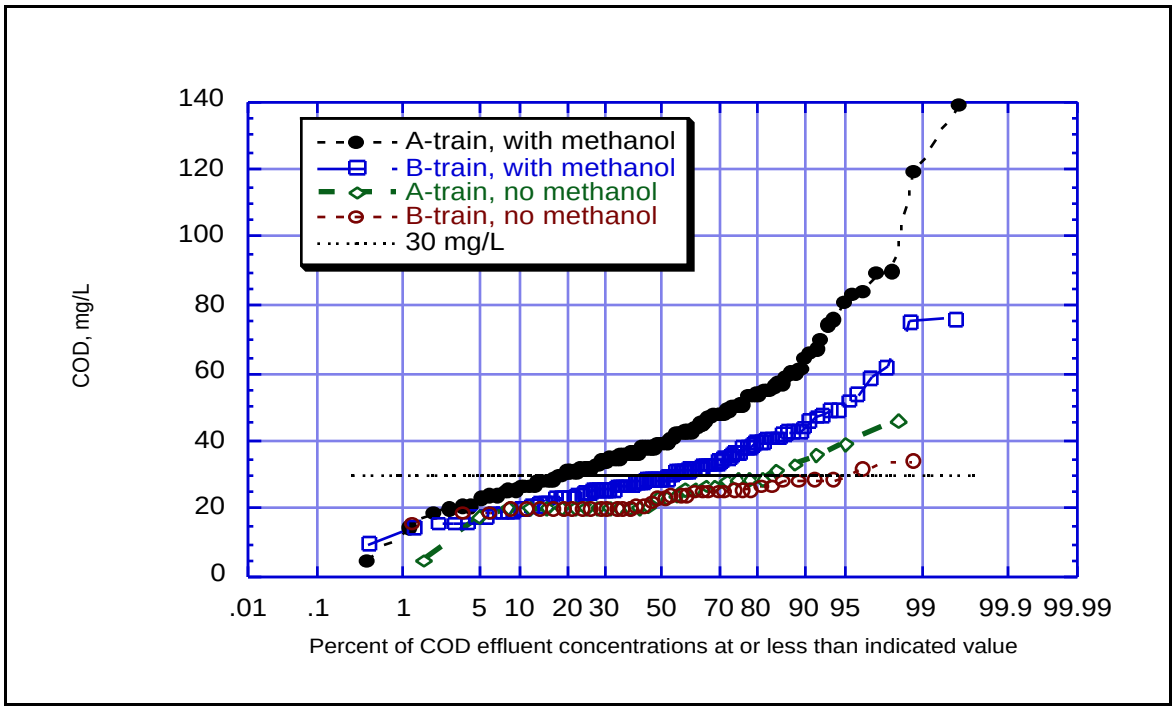

Figure 2-11. Effect of methanol addition on COD effluent concentrations. All samples plotted from period specified.

effluent COD concentrations rose to values consistently greater than 30 mg/L. After methanol addition was stopped effluent concentrations dropped to approximately 20 mg/L. The response of effluent COD concentrations to methanol addition was very similar to that of $CBOD_5$ (Figure 2-4). Effluent COD averaged 25 mg/L and 23 mg/L for the A and B-train, respectively, during periods when methanol was not added for denitrification (Table 2-7). For periods of methanol use, COD effluent concentration averaged 43.6 mg/L and 31.6 mg/L for the A and B-trains, respectively.

COD removal performance on a mass basis

There is no design basis for evaluating COD mass removal. Using the adopted 30 mg/L effluent COD standard, effluent mass flux is 4.6 kg/d per train at design flows. For

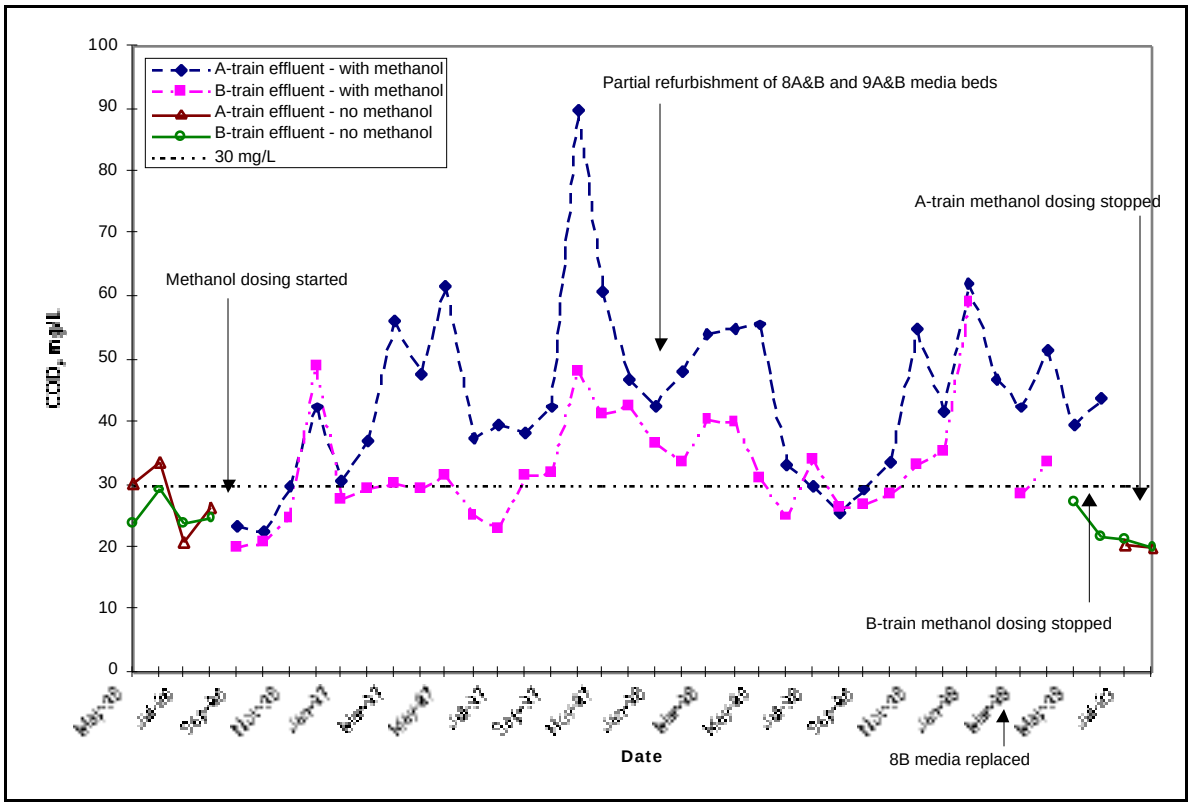

Figure 2-12. Effects of methanol addition and filter clogging on monthly average effluent COD values.

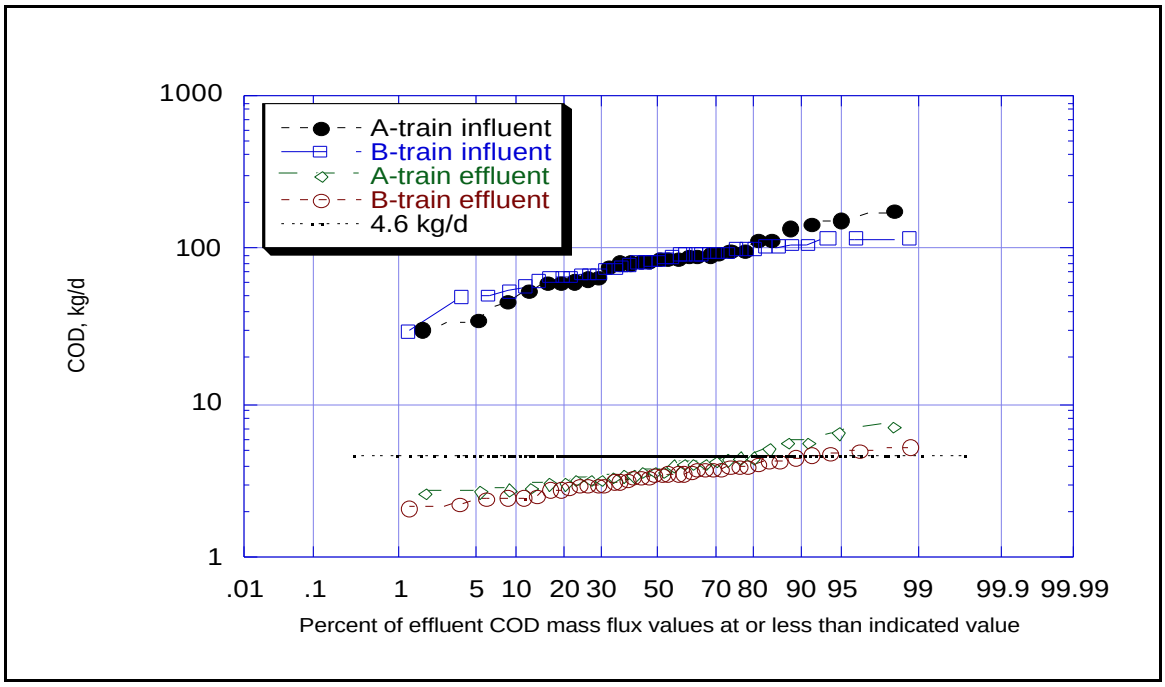

Figure 2-13. Mass basis COD removal for A and B-train for periods of no methanol use. All samples plotted from period specified.

Table 2-8. COD mass removal summary for periods with no methanol use.

COD, (kg/d)	A-train influent	B-train influent	A-train effluent	B-train effluent	A-train % removal	B-train % removal
Mean	79	80	4.0	3.5	94%	95%
σ	22	20	1.3	0.8	3%	2 %
n	27	39	29	38	27	38

periods of no methanol use, 95% of effluent mass flux values are at or less than 4.6 kg/d for the B-train, and 82% for the A-train (Figure 2-13). Mean mass removal of COD for the same periods is 94% and 95% for the A and B-train, respectively.

Conclusion

The Vermont AEES consistently removes 95% of influent COD loading to reliably meet effluent COD concentrations of 30 mg/L when no methanol is added. Effluent COD excursions greater than 30 mg/L are almost entirely associated with methanol addition. Methanol addition is neither used nor needed for denitrification in the current process configuration.

2.9. Total Suspended Solids Removal

Summary: The Vermont AEES facility meets effluent TSS concentrations less than 10 mg/L at design TSS loading in the current treatment configuration. Methanol addition for denitrification is directly related to effluent TSS concentration excursions above 10 mg/L. Without methanol addition treatment performance meets a 5 mg/L effluent standard.

Introduction

The Vermont AEES facility met and bettered design specifications for TSS removal. Influent TSS loadings were at design on average, but effluent concentrations were significantly less than 10 mg/L. As with $CBOD_5$ and COD, TSS removal is analyzed for methanol and no-methanol operations. Effluent TSS concentrations were significantly higher during periods of methanol operations than periods of no-methanol operations.

TSS removal performance on a concentration basis

Actual TSS removal performance was significantly better than design performance. As with $CBOD_5$ and COD, TSS effluent concentrations were raised significantly by methanol addition. Treatment performance from May 1996 to August 1999 exhibits a pattern of response to methanol addition and EFB media refurbishment that is similar to that of $CBOD_5$ and COD (Figure 2-14). During periods with no methanol addition, all TSS effluent concentrations were significantly less than 10 mg/L (Figure 2-15). In contrast, while methanol was added only approximately 75% of A-train and 90% of B-train effluent TSS concentrations were less than or equal to 10 mg/L. Without methanol

Table 2-9. A and B-train effluent TSS concentrations with and without methanol addition for denitrification.

TSS (mg/L)	**With methanol** September 1996 - April 1999* * A-train methanol dosing continued into June 1999. Data for A-train reflects dosing dates.			**Without methanol** May-August 1996, May-August 1999		
	Influent	A-Train effluent	B-Train effluent	Influent	A-Train effluent	B-Train effluent
Mean	220.3	8.0	4.1	189.2	3.4	2.0
σ	108.1	7.1	3.9	58.8	1.4	1.1
n	123	122	113	41	31	41

addition, one standard deviation above the mean for the A and B trains was approximately 5 mg/L and 3 mg/L, respectively (Table 2-9). During the period of methanol addition, one standard deviation above the TSS effluent mean concentration was approximately 15 mg/L and 8 mg/L for the A and B-train, respectively.

Actual TSS removal performance is better than design performance. With methanol eliminated, on a monthly average basis for both trains, and all B-train samples, the Vermont AEES meets an effluent TSS concentration standard of 5 mg/L (Table 2-9, Figure 2-14, Figure 2-15).

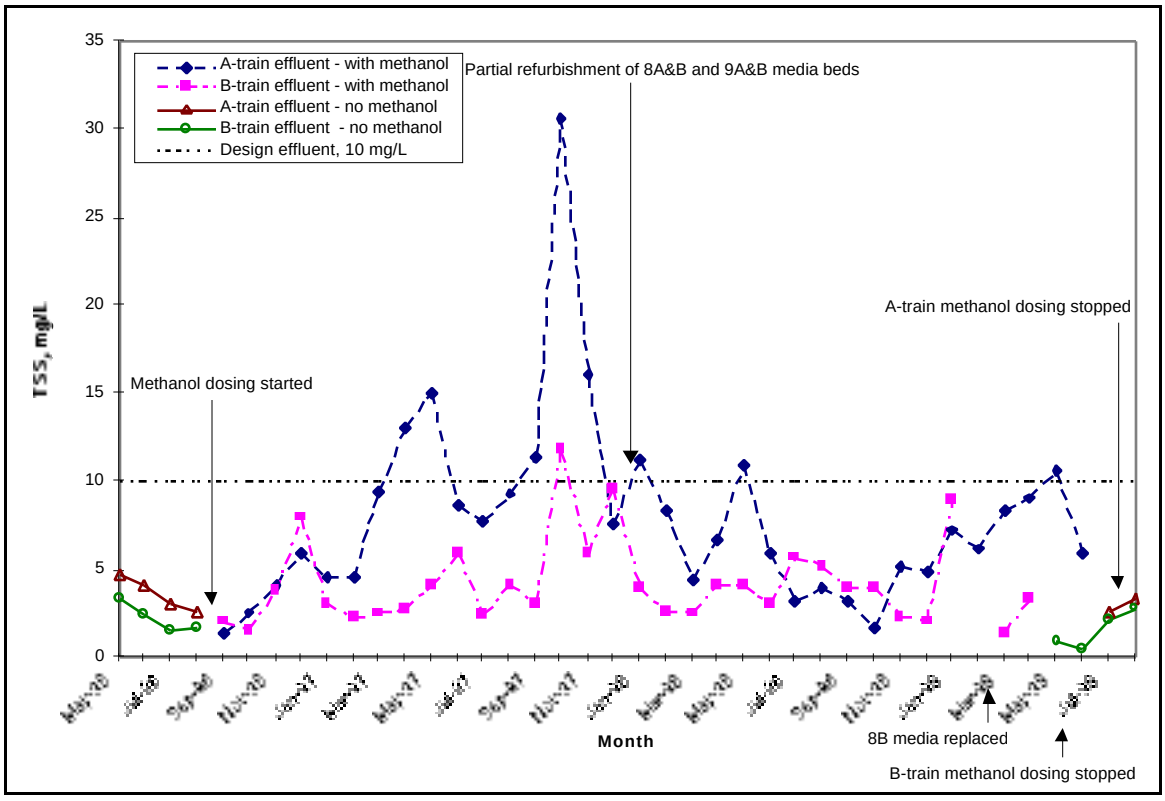

Figure 2-14. Effects of methanol addition on monthly average effluent TSS values.

Table 2-10. TSS mass removal summary for periods with no methanol use. Data are from periods of no methanol addition.

TSS, kg/d	A-train influent	B-train influent	A-train effluent	B-train effluent	A-train percent removal	B-train percent removal
Mean	30	28	0.6	0.3	98%	99%
σ	10	8	0.2	0.2	1%	1%
n	41	41	31	40	31	39

TSS removal performance on a mass basis

At design TSS mass loadings, mean TSS removal was more than twice as good as design for the A-train and approximately five times better in the B-train for periods when no methanol was added (Figure 2-16). Mean mass removal of TSS in both trains was approximately 98% (Table 2-10).

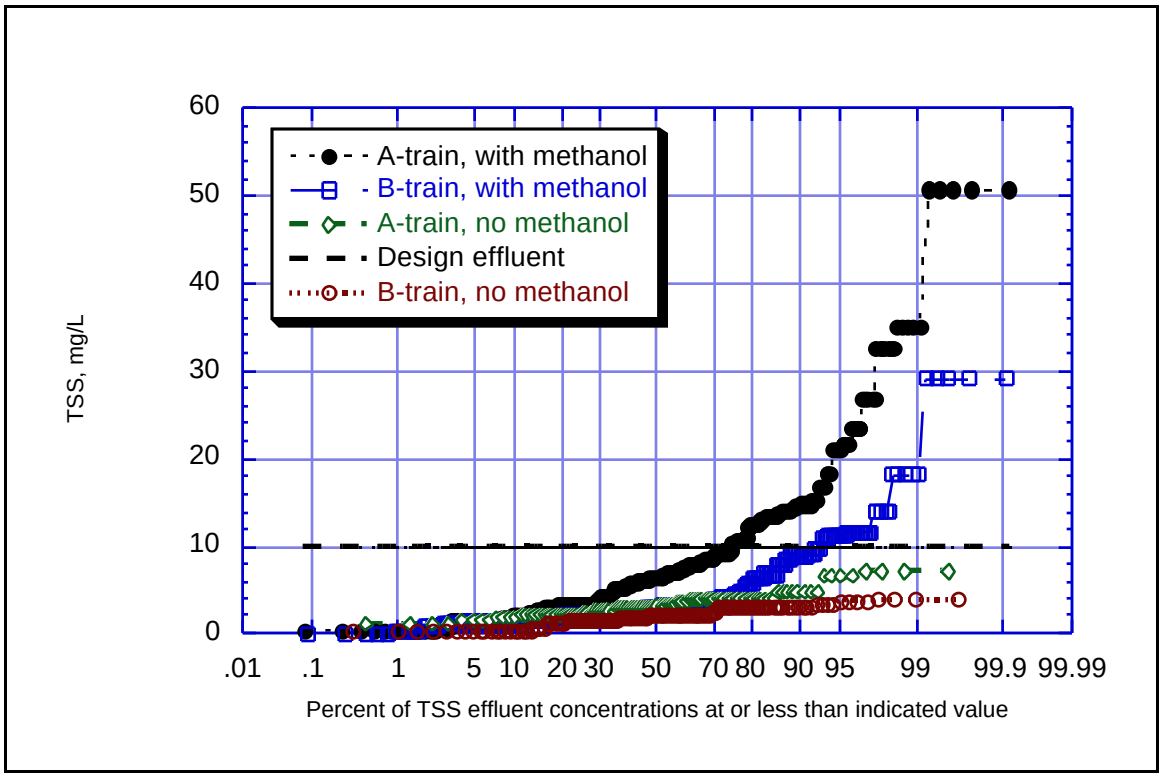

Figure 2-15. Total suspended solids effluent concentrations with and without methanol addition. All samples plotted from period specified.

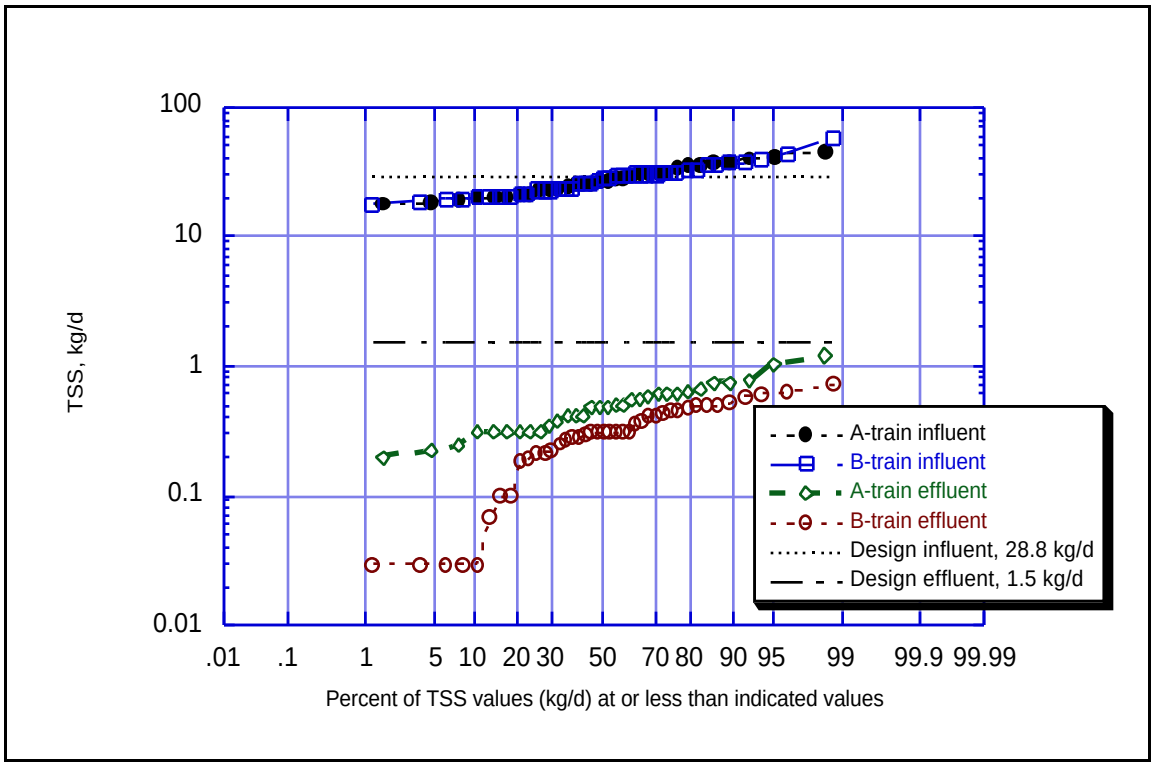

Figure 2-16. TSS influent loading and effluent mass flux for A and B-trains. Data are from periods with no methanol addition. All samples plotted from period specified.

Conclusion

The Vermont AEES facility exhibited excellent TSS removal performance in the B-train throughout the reporting period, and for both trains when no methanol was added for denitrification. With the elimination of methanol, the Vermont AEES can meet a TSS effluent standard of 5 mg/L.

2.10. Nitrogen Transformation and Removal

Summary: The Vermont AEES has consistently produced effluent ammonia concentrations less than 1 mg/L in all treatment configurations. Removal of ammonia on a mass basis is 99%. In its current configuration, the Vermont AEES reliably produces effluent nitrate and TKN concentrations of 5 mg/L or less. Total nitrogen effluent concentrations are significantly less than 10 mg/L. Total nitrogen mass removal averages 84% in the current treatment configuration.

Introduction

The Vermont AEES meets all design effluent goals for nitrogen transformation and removal in its current configuration. Total Kjeldahl nitrogen (TKN), ammonia and nitrate effluent concentrations consistently meet design criteria. Nitrification design performance has been met since the beginning of steady state operations in May 1996, with ammonia

effluent concentrations consistently less than 1 mg/L. Average ammonia removal is 99% on a mass basis. The denitrification process design evolved to meet effluent nitrate concentrations less than 5 mg/L without the use of methanol. Denitrification was initially achieved by methanol addition to an anoxic filter (EFB). In the current configuration, an initial anoxic reactor, receiving recycled nitrified effluent and MLSS, produces nitrate effluent concentrations equal to or less than 5 mg/L.

In this section, the order of performance analysis is TKN removal, nitrification (ammonia conversion to nitrate), denitrification (nitrate conversion to atmospheric nitrogen), and total nitrogen (TN) removal. The order of analysis follows the order of nitrogen transformations in the Vermont AEES. Total Kjeldahl Nitrogen (TKN) measures organic nitrogen and ammonia, which are reduced forms of nitrogen, in the waste stream. All but a very small fraction of influent organic nitrogen is converted to ammonia. The ammonia fraction of TKN is removed by conversion to nitrate (nitrification). Nitrate is removed by denitrifying bacteria. Total nitrogen is a convenient overall measure of nitrogen removal performance. It is calculated as TN = TKN + NO_3, on either a concentration or mass basis.

2.10.1. TKN Removal Performance

Summary: The Vermont AEES exhibited excellent TKN removal performance. All monthly average TKN effluent values met design on a concentration and mass basis (5 mg/L and 0.77 kg/d, respectively).

Introduction

Total Kjeldahl Nitrogen removal performance met design criteria on a concentration and mass basis throughout the forty months of Vermont AEES operation represented in this report. Removal of TKN from the waste stream depends on nitrification, which has consistently met design performance. Consequently, TKN effluent concentration and mass flux values have consistently met design targets of 5 mg/L and 0.77 kg/d, respectively.

TKN removal performance on a concentration basis

Effluent TKN mean monthly concentrations have met 5 mg/L for all forty months of this reporting period (Figure 2-17). Mean TKN effluent concentrations were approximately 2 mg/L and 1 mg/L for the A and B-train, respectively (Table 2-11). Of all samples taken, 99% of B-train and 97% of A-train concentrations have been at or less than 5 mg/L (Figure 2-18). Removal of TKN has been a highly stable process at the Vermont AEES.

Table 2-11. TKN influent and effluent mean concentrations, May 1996- August 1999.

TKN, mg/L	Influent	A-train effluent	B-train effluent
mean	27	1.7	1.3
σ	7	0.9	0.7
n	125	114	127

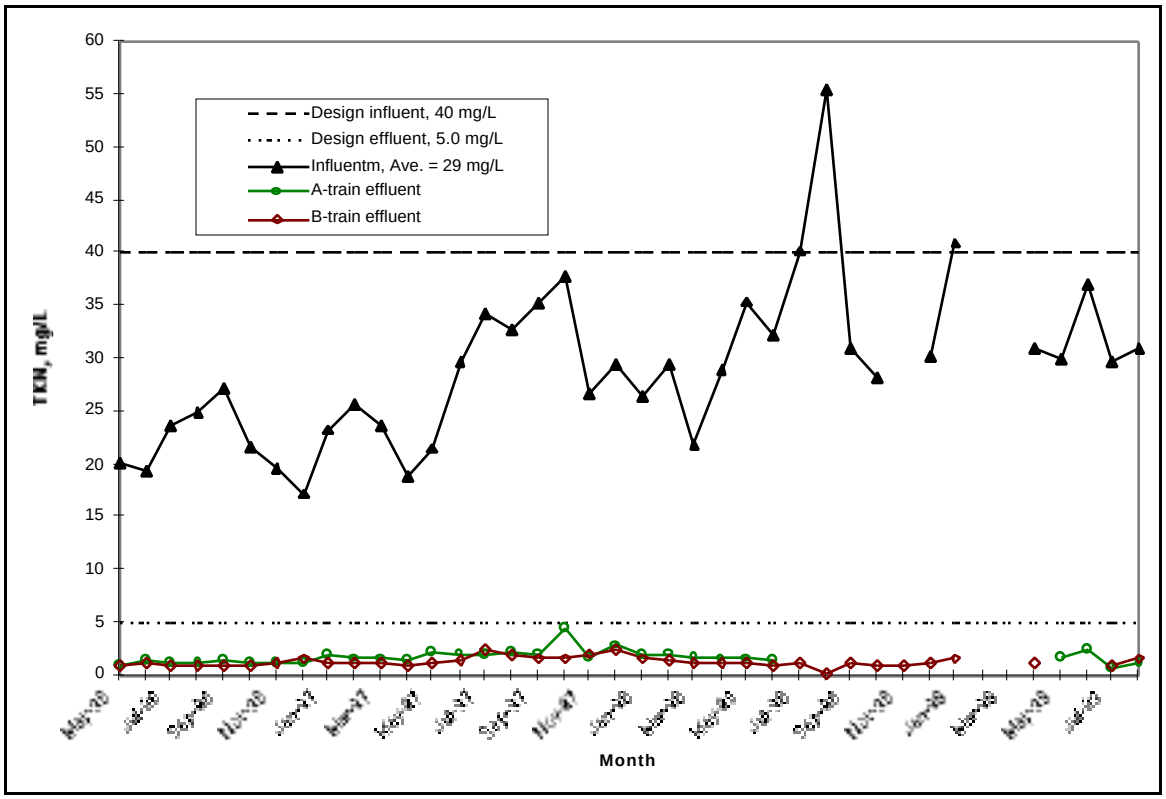

Figure 2-17. TKN monthly mean influent and effluent concentrations May 1996-August 1999.

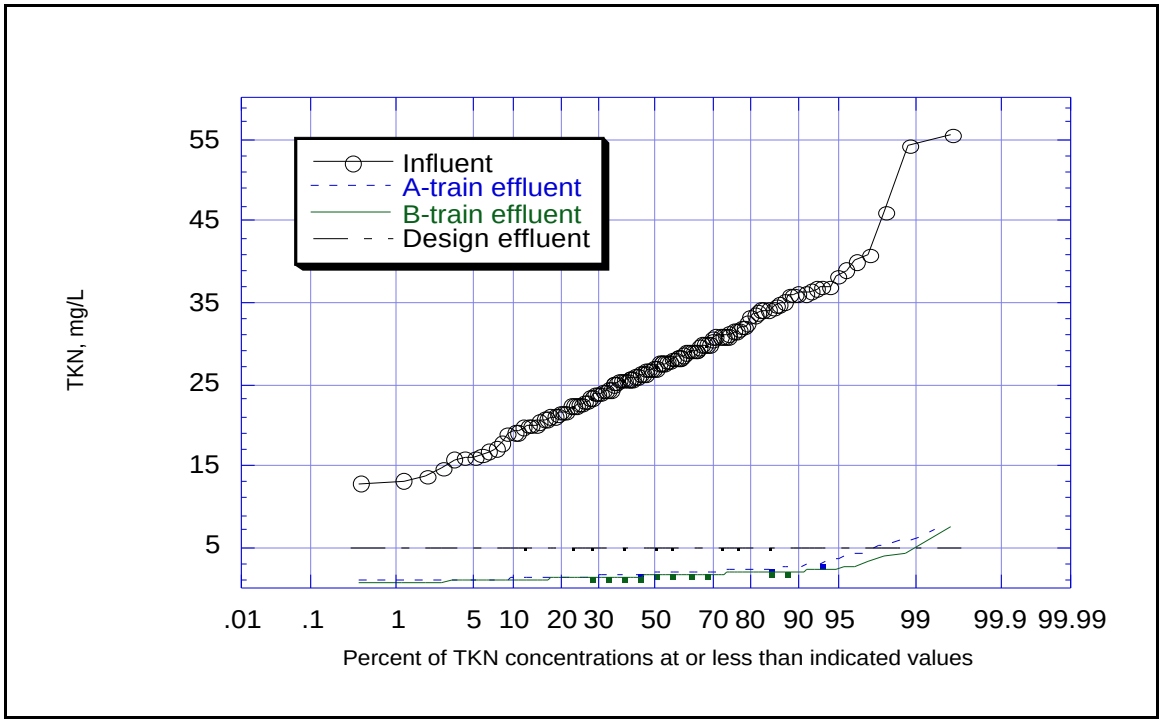

Figure 2-18. TKN influent and effluent concentrations May 1996 - August 1999. All samples plotted from period specified.

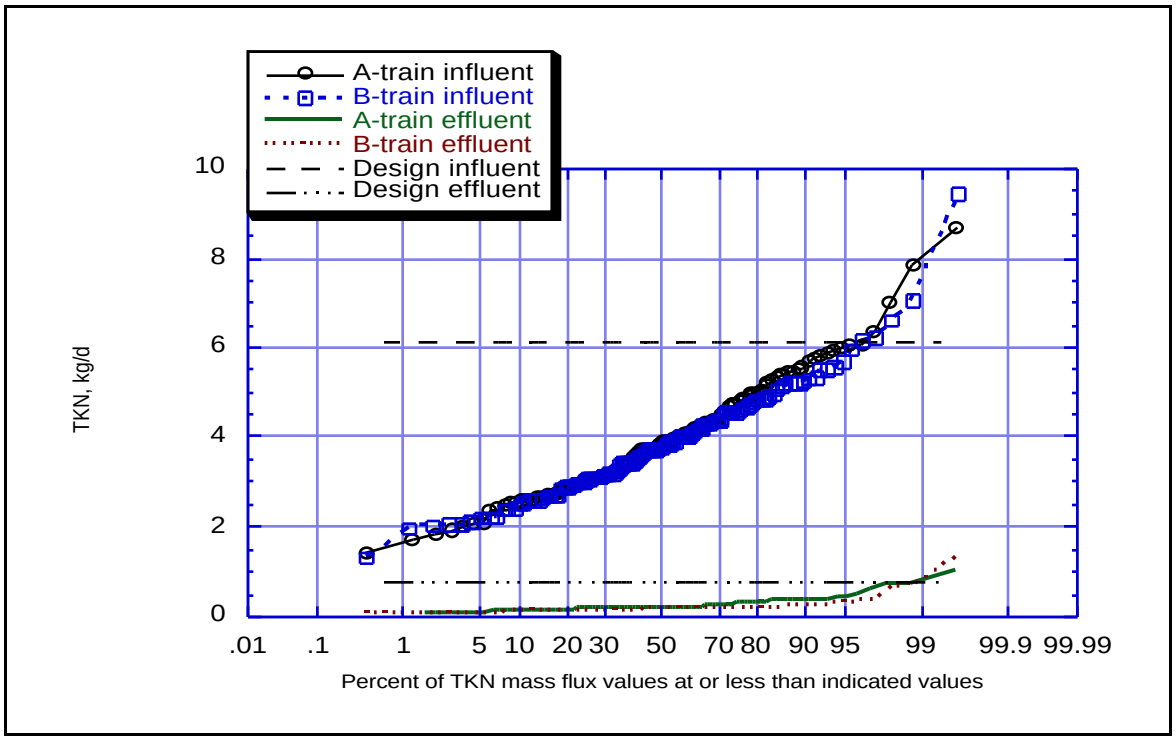

Figure 2-19. TKN removal per treatment train, May 1996 - August 1999. All samples plotted from period specified.

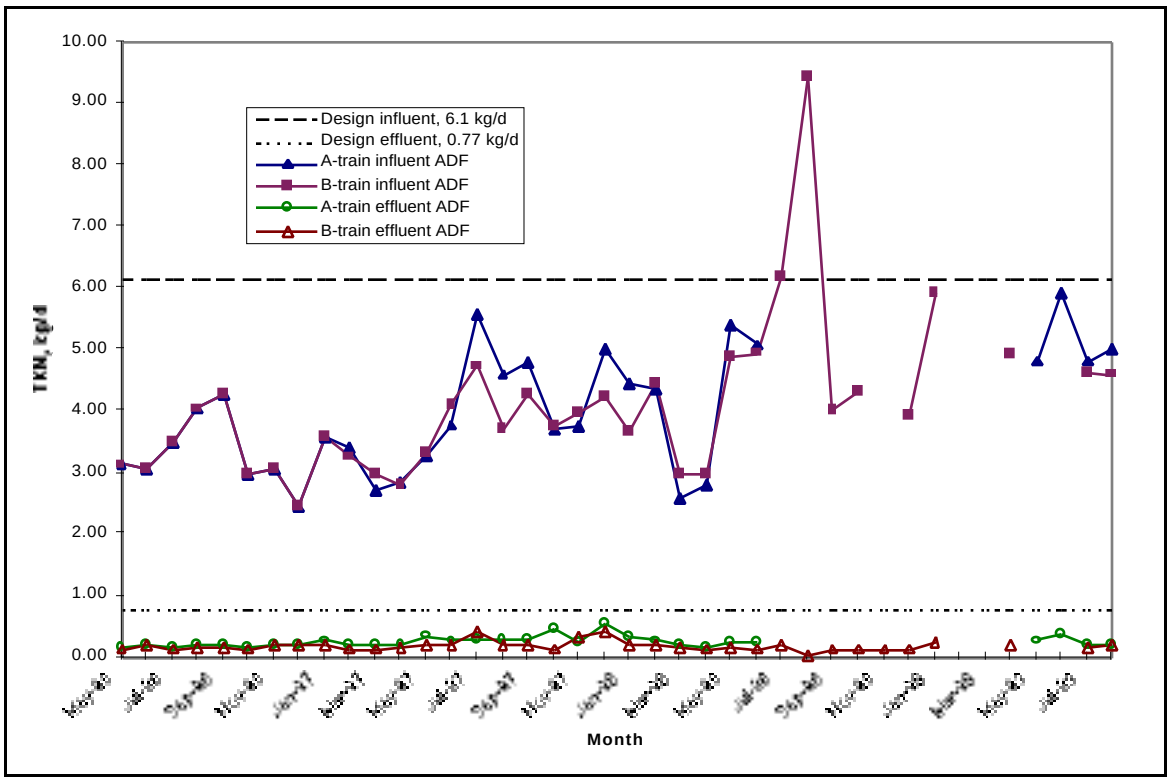

Figure 2-20. TKN monthly average influent mass loading and effluent mass flux May 1996 - August 1999.

TKN removal performance on a mass removal basis

Removal performance of TKN on a mass basis mirrors removal on a concentrations basis. Approximately 99% percent of all effluent samples were less than or equal to 0.77 kg/d (Figure 2-19). All monthly mean effluent mass flux samples are significantly less than 0.77 kg/d (Figure 2-20).

Conclusion

Treatment performance of TKN on a concentration and mass basis has been very stable. The Vermont AEES consistently met effluent concentrations at or less than 5 mg/L and effluent mass flux values of 0.77 kg/d.

2.10.2. Nitrification Performance

Summary: The Vermont AEES has consistently produced ammonia effluent concentrations of less than 1 mg/L. Mass removal of ammonia has averaged 99%. Nitrification has been a highly stable process.

Introduction

Nitrification, the conversation of ammonia to nitrate, has been a highly stable process at the Vermont AEES. One of the stated objectives for the Vermont AEES was to fully nitrify cold temperature influent. Effluent ammonia concentrations have consistently been less than 1 mg/L over the 40-month period of this report despite seasonal influent temperatures of less than 11^{o} C (Figure 2-4).

Nitrification performance on a concentration basis

Effluent ammonia concentrations have been consistently less than 1 mg/L throughout this forty month reporting period. Average effluent ammonia concentrations were 0.3 mg/L for both trains (Table 2-12). Ninety five percent of A-train and 97% of B-train weekly effluent concentrations have been at or less than 1 mg/L (Figure 2-21). Thirty-seven of 40 monthly mean ammonia effluent concentrations met design (Figure 2-22). The three monthly average excursions greater than 1 mg/L were attributable to operational difficulties. In one instance, a flow sensor malfunctioned, leading to a sudden increase of flow to the B-line of approximately 50%. The other instances occurred during a period of reduced air delivery, subsequently repaired, that limited nitrification (Figure 2-22).

Table 2-12. Ammonia influent and effluent concentrations, May 1996 - August 1999. The average influent concentration and standard deviations are approximate values to account for the organic nitrogen fraction of TKN converted to ammonia.

NH_3, mg/L	Influent	A-train effluent	B-train effluent
mean	25	0.3	0.3
σ	6	0.4	0.6
n	153	155	154

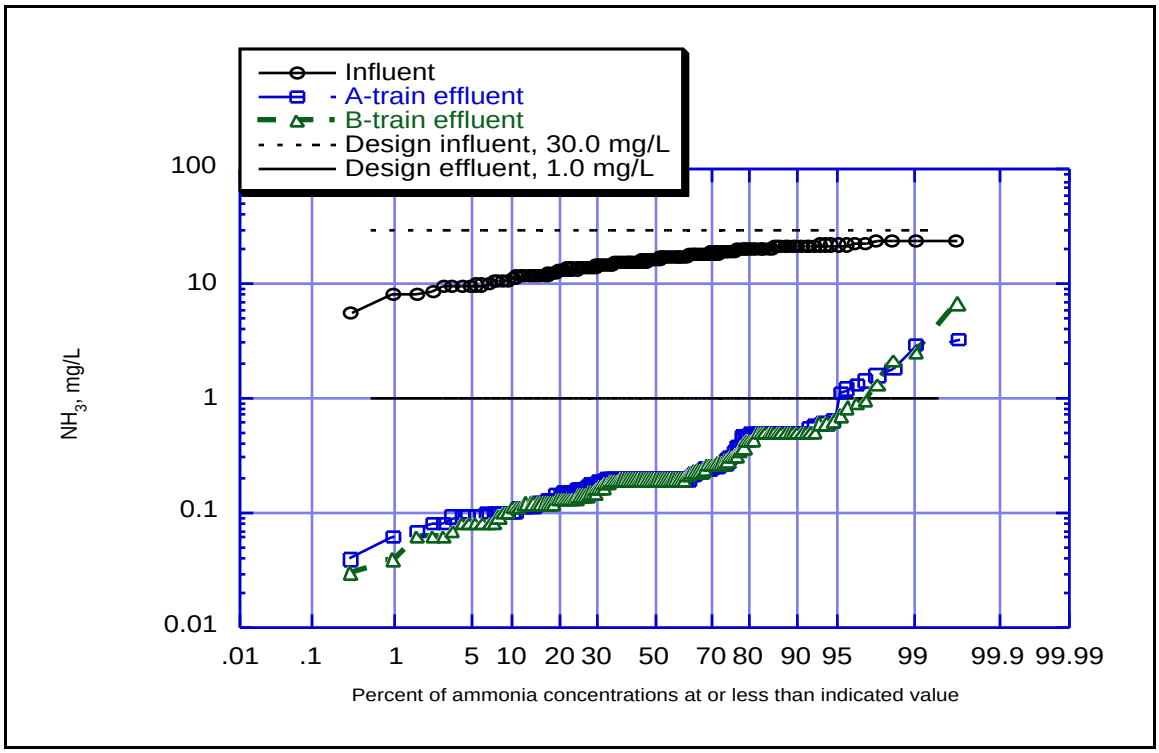

Figure 2-21. Nitrification performance on a concentration basis, May 1996 - August 1999.

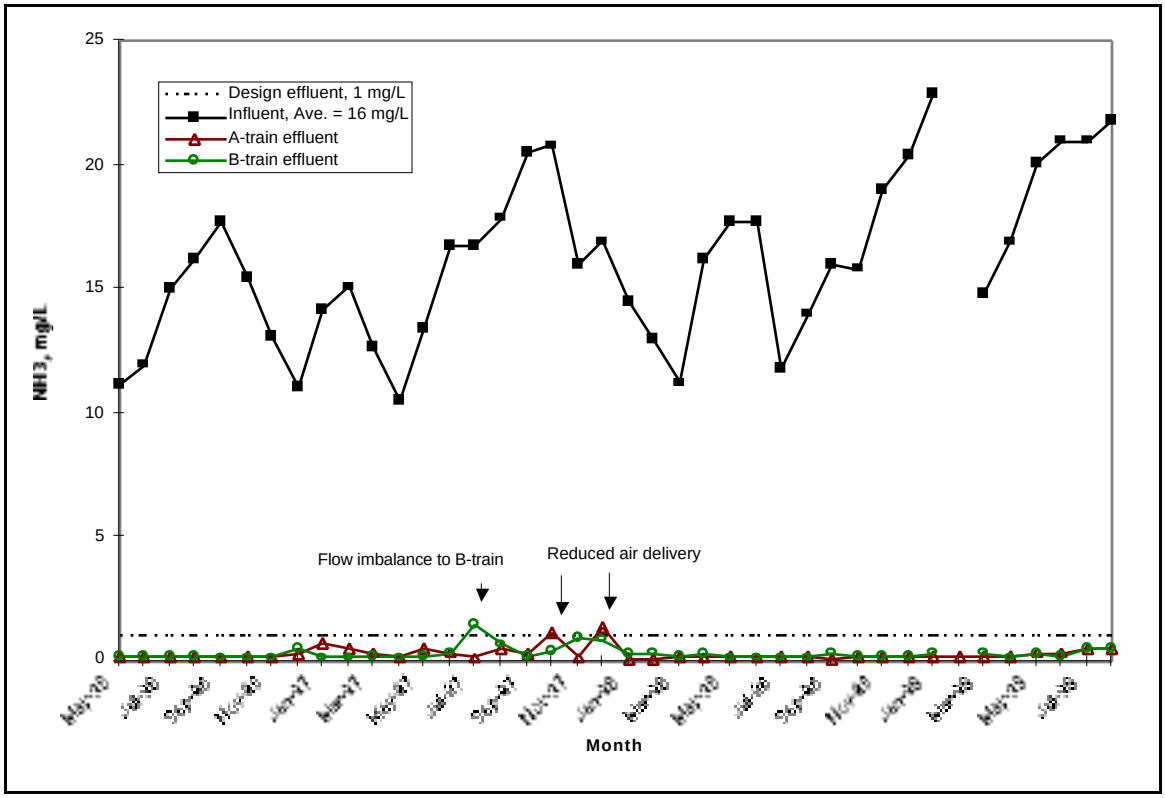

Figure 2-22. Mean monthly influent and effluent ammonia concentrations May 1996 - August 1999.

Nitrification performance on a mass removal basis

On a mass basis, mean monthly effluent ammonia values have met design (0.15 kg/d) consistently 37 out of forty months (Figure 2-23). Reasons for the three excursions from design of monthly mean effluent mass flux values have been described above.

Total ammonia loading of the treatment system is greater than in the analyses shown in Figure 2-21 and Figure 2-22. Most organic nitrogen recorded as TKN is converted from an organic form to ammonia. Total ammonia loading is calculated as:

(Equation 2.10-1), $\mathbf{Total\ NH_{3\ in} = TKN_{\ in} - (TKN_{\ out} - NH_{3\ out})}$.

For convenience, influent ammonia concentrations have been presented above as the tested, rather than calculated, influent values. Calculation of ammonia removal on a mass basis, however, must account for total ammonia loading of the treatment system. Total ammonia removal is 99% for the A-and B-trains (Table 2-13).

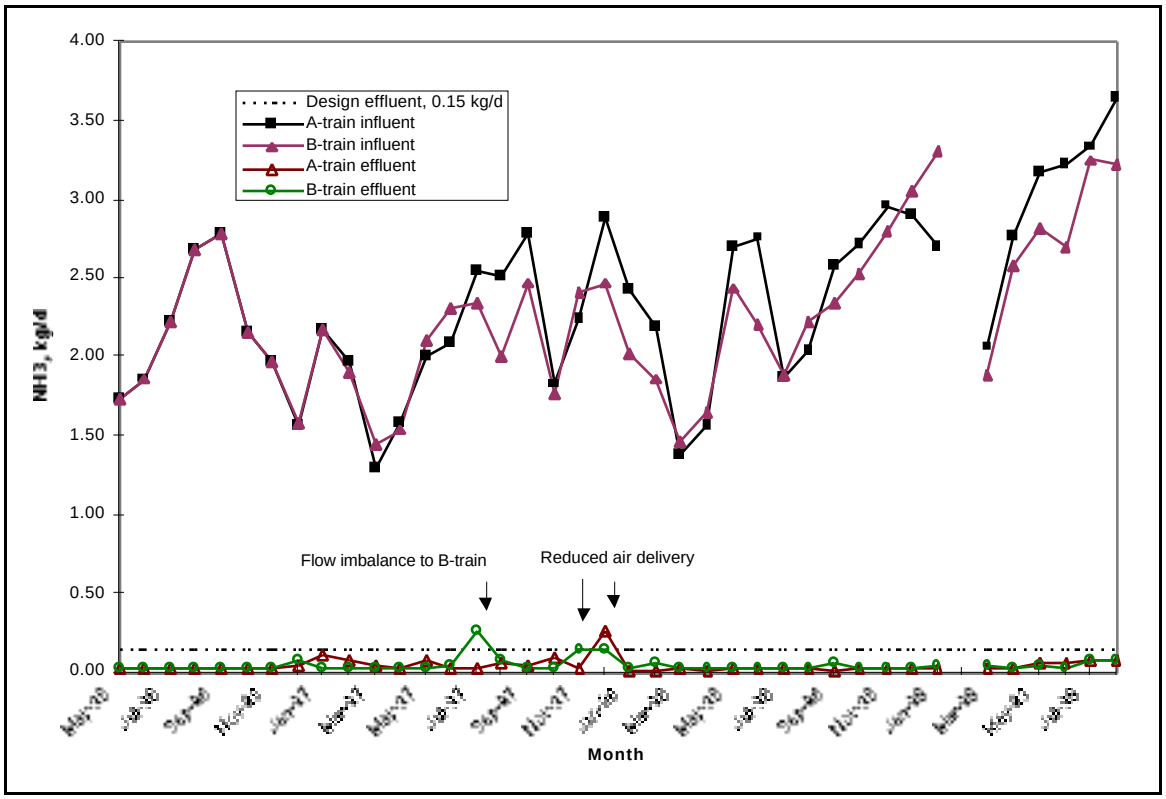

Figure 2-23. Mean monthly ammonia influent mass loading and effluent mass flux May 1996 - August 1999.

Table 2-13. Total ammonia removal. B-train period reported is October 1998 - August 1999, the period of optimized anoxic reactor operations. A-train period reported is May 1996 - June 1999.

Total NH3 in = TKN in - (TKN out - NH_3 out)						
	B-train NH3 in kg/d	**B-train NH3 out kg/d**	**B-train % removal**	**A-train NH3 in kg/d**	**A-train NH3 out kg/d**	**A-train % removal**
Mean	**5.26**	**0.06**	**98.8%**	**3.64**	**0.05**	**98.6%**
σ	**2.63**	**0.03**	**0.5%**	**1.27**	**0.08**	**1.8%**
n	**34**	**34**	**34**	**103**	**103**	**103**

Conclusion
On both a concentration and mass basis, effluent ammonia values have consistently met design throughout the forty months of this reporting period. There is no evidence that cold influent temperatures significantly degraded nitrification performance.

2.10.3. Denitrification - Methanol and Endogenous Carbon Sources.

Summary: The Vermont AEES meets design nitrate effluent concentrations of 5 mg/L without the use of methanol in its current process configuration. Prior to the processes changes instituted in 1999, design nitrate effluent concentrations were met with the addition of methanol, but with less consistency than the current process configuration.

Introduction
Upon nitrification in the Vermont AEES, all but an insignificant fraction of nitrogen in the waste stream is in the form of nitrate. Nitrogen is then removed from the wastewater by denitrification.

Denitrification is a biological process in which bacteria use nitrate as a terminal electron acceptor instead of oxygen to metabolize organic carbon. Nitrate is converted to atmospheric nitrogen in the process. Denitrification is a heterotrophic process, and so is rate limited by the availability of soluble carbon for bacteria respiration and cell growth. Denitrifying bacteria can be divided into three functional categories: obligate anaerobes that are poisoned by free oxygen; facultative bacteria that will only denitrify in an oxygen-deprived environment; and aerobic denitrifiers that are found in environments with rapidly varying availability of oxygen. To consume available nitrate, all three types require some kind of anoxic environment with sufficient organic carbon for bacterial respiration and cell growth.

Operationally, an anoxic environment is defined as one with a dissolved oxygen concentration less than 0.4 mg/l, but with nitrate concentration greater than 0.5 mg/L. Alternatively, an operationally useful definition of an anoxic environment is one with an

oxidation-reduction potential (ORP) of between -150 mV and 0 mV. Both dissolved oxygen and ORP measures of anoxic conditions were used at the Vermont AEES.

Providing an anoxic environment that is rich both in soluble organic carbon and nitrate has been a significant technical challenge for the Vermont AEES project. The original design basis for denitrification provided an anoxic environment in the second EFB in each train without sufficient carbon for denitrification. Two denitrification techniques were subsequently demonstrated: methanol addition to the second EFB in each train, and recycling of nitrified effluent and MLSS to a front-end anoxic reactor. The latter method, using endogenous carbon sources, has been significantly more successful than the former.

Process Evolution of Vermont AEES Denitrification
Denitrification design basis and performance has evolved through five process phases at the Vermont AEES. The phases are analyzed chronologically below (Table 2-14). Starting in April 1998, the A and B-train had different denitrification design bases. At that time, the denitrification design in the B-train was changed to use a single stage anoxic reactor (1B) that received process recycle of nitrified wastewater and MLSS from reactor 5B. Methanol dosing continued in the second EFB of both trains until the spring of 1999 to ensure compliance with the contract nitrate effluent standard of 5 mg/L. In March 1999, adding a fixed-film process to reactor 1B modified the denitrification process in the B-train. In July 1999, reactor 1A was changed to an anoxic reactor that received process recycle of nitrified wastewater and MLSS from reactor 5A. A description of the design basis and results for each phase follows.

Phase 1.
Phase 1 covers the period from May 5, to August 15, 1996. The original design basis was to denitrify in EFB2A and EFB2B using residual carbon in the effluent as a substrate. No design process in the aerated reactors affected denitrification.

Phase 1 denitrification failed. Before methanol dosing began on August 15, effluent values consistently exceeded 5 mg/L on a monthly average basis (Figure 2-24). Effectively no effluent nitrate concentration or effluent mass flux values met design criteria (Figure 2-25, Figure 2-26). Nitrate effluent targets were not met because of insufficient carbon substrate in EFB2A and EFB2B. Failure of denitrification forced the adoption of another denitrification design basis.

Phase 2.
Phase 2 covers the period from August 22, 1996 through March 1998 for the B-train, and from August 22, 1996 through June 1999 for the A-train. The design basis of Phase 2 used methanol as a carbon substrate for denitrifying bacteria in EFB2A and EFB2B. Methanol was dosed at 3.5 parts of methanol to 1 part nitrate on a mass basis. The RAS rate is estimated to have been 0.25Q. Denitrification from RAS was probably insignificant.

Table 2-14. Denitrification design basis summary.

Phase / Date	Denitrification Design Basis
Phase 1 **Both trains:** 05MAY-15AUG96	**Both trains**: Denitrify in anoxic filter (EFB2A and EFB2B) using residual soluble organic carbon in secondary effluent as carbon substrate.
Phase 2 **A-train**: 22AUG-20JUN99 **B-train**: 22AUG-MAR98	**Both trains**: Denitrify in anoxic filter (EFB2A and EFB2B) using methanol as the carbon substrate. Methanol dose is 3:5 parts methanol per 1 part nitrate on a mass basis.
Phase 3 **B-train:** APR98-JAN99	**A-train**: Continued methanol dosing in EFB2A. Still running in phase 2 mode. **B-train**: Evaluation of single stage anoxic denitrification process in aerated reactors. Reactor 1B run in anoxic mode by cycling air delivery on timer. Process recycle from reactor 5B to 1B at 240 m^3/d (1.6Q). Fine bubble diffusers in reactor 1B. Manual ORP process control using hand held ORP probe. ORP target ≤ -50mV. Continued methanol dosing in EFB2B as backup to meet nitrate design effluent.
Phase 4 **B-train:** MAR-AUG99	**A-train**: Continued methanol dosing in EFB2A. Still running in phase 2 mode. **B-train**: Evaluation of modified single stage anoxic denitrification process in aerated reactors. Coarse bubble diffusers installed in reactor 1B. Fabric media tower installed in reactor 1B. Process recycle from reactor 5B to 1B at 240 m^3/d. Automatic ORP control with ORP probe controlling air cycling. Methanol dosing in EFB2B stopped at the end of April 1999.
Phase 5 **A-train:** JUL-AUG99 **B-train:** JUL-AUG99	**A-train**: Methanol dosing stopped. Coarse bubble diffusers installed in reactor 1A. Process recycle from reactor 5A to 1A at 390 m^3/d. Air delivery adjusted to keep dissolved oxygen < 0.4 mg/L. No attempt for process control by ORP. **B-train**: Process same as Phase 4, but process recycle from reactor 5B to 1B increased to 380 m^3/d.

Methanol dosing continued in both trains through April 1999. Methanol dosing stopped in the B-train at the end of April 1999, but continued in the A-train through June 1999.

In Phase 2, approximately 80% of weekly effluent nitrate mass flux values met design in both the A and B-trains (Figure 2-25). Performance on a concentration basis was slightly poorer. Approximately 70% of effluent concentrations for both trains were equal to or less than 5 mg/L (Figure 2-26). Approximately 76% of monthly mean nitrate effluent concentrations met design on the A-train (26 of 34 months), and 91% on the B-train (29 of 32 months) (Figure 2-24). Denitrification was inconsistent using methanol because of two significant problems that emerged shortly after dosing began:

(1) The 0.75-inch lava rock media in EFB2A, EFB2B, EFB3A and EFB3B fouled badly from heterotrophic biofilms growing on the methanol substrate. Despite aggressive air-scour backflushing, progressive fouling eventually caused severe head losses in all EFBs downstream of methanol addition (Figure 2-1). Lack of flow through the media degraded denitrification performance;
(2) Methanol dosing was not flow proportioned. Flow imbalance between trains A and B plagued operations (see below). The train receiving flows significantly in excess of 152 m^3/d (40,000 gpd) suffered under-dosing of methanol and subsequent degradation of denitrification performance.

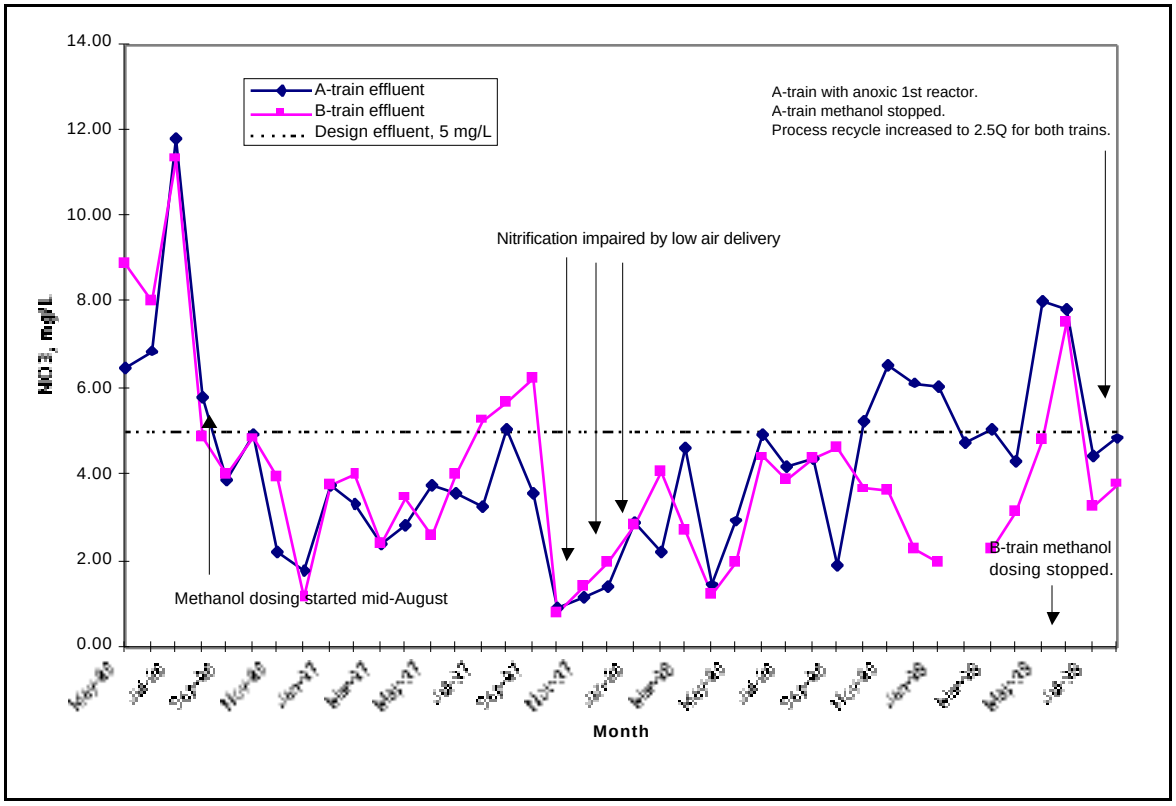

Figure 2-24. Nitrate effluent concentrations for A and B-trains. Monthly mean values plotted.

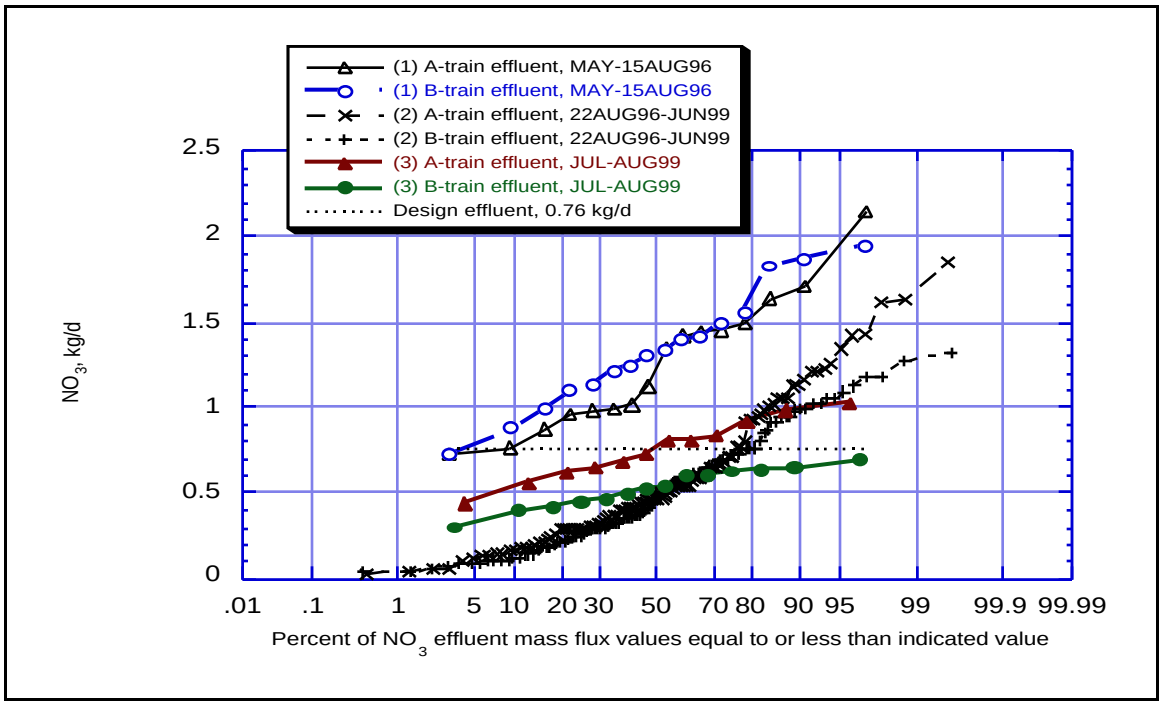

Figure 2-25. Effluent nitrate mass flux May 1996-August 1999. Periods represented are: (1) Phase 1; (2) Phase 2 for A-train. Phases 2-4 for B-train. August 22, 1996-June 30, 1999; (3) Phase 5. July 21-August 17, 1999. Data plotted are all effluent nitrate values for the period specified.

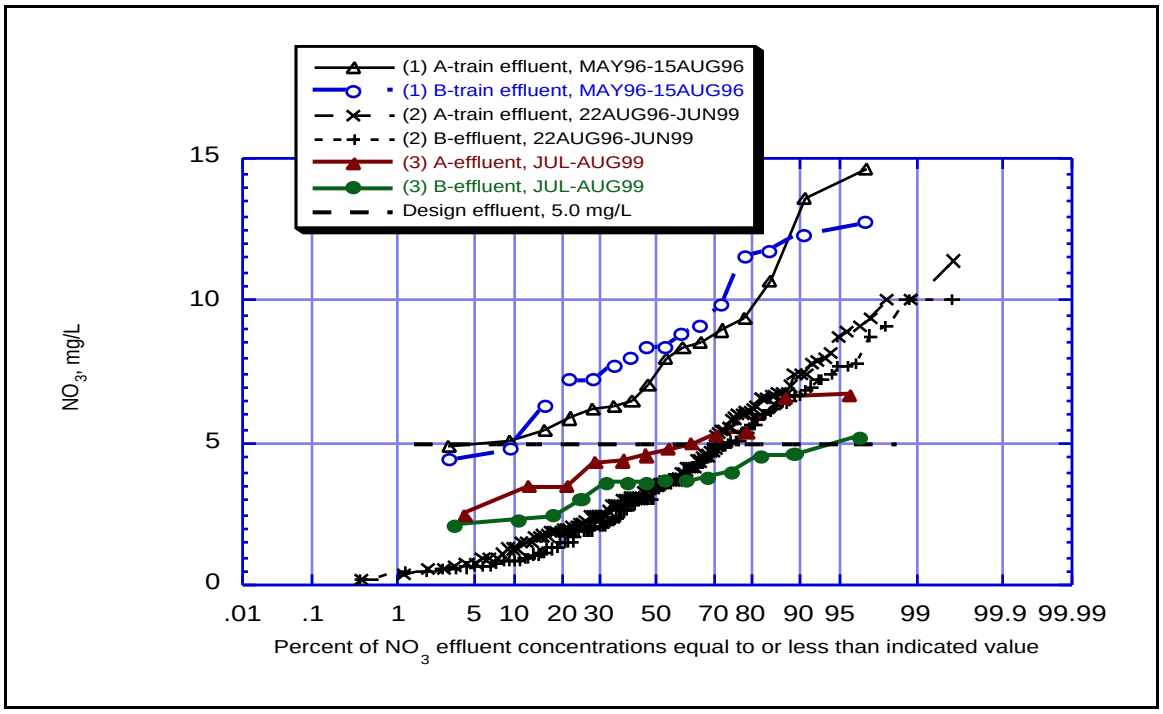

Figure 2-26. Effluent nitrate concentrations May 1996 - August 1999. Periods represented are: (1) Phase 1; (2) Phase 2 for A-train. Phases 2-4 for B-train. August 22, 1996 - June 30, 1999; (3) Phase 5. July 21-August 17, 1999. Data plotted are all effluent nitrate values for the period specified.

Phase 3

Phase 3 covers the period from April 1998 to January 1999. A modified Ludzak-Ettinger[6] (MLE), public domain, model was used as the denitrification design basis for Phase 3. The MLE model is single stage denitrification treatment comprised of an anoxic first stage reactor followed by aerobic treatment and a clarifier. There are two recycles in the MLE model: (1) Nitrified effluent containing mixed liquor suspended solids (MLSS) is recycled from the final aerobic reactor to the anoxic reactor; (2) RAS is recycled to the anoxic reactor. Actual MLSS concentrations used were significantly less than those indicated in the MLE model.

The process modifications for Phase 3 were as follows:

(1) B-train. Denitrification was achieved by recycling fully nitrified effluent ($NH_3 \leq$ 1 mg/L) and MLSS from reactor 5B to 1B at a rate of approximately 240 m^3/d (1.6Q). The RAS flow is estimated to have been 0.25Q. Air supply to reactor 1B was automatically cycled on and off, or the rate of supply reduced as needed to operate reactor 1B at an ORP of less than -50mV. A hand held probe was used to monitor the OPR in reactor 1B. The operator adjusted airflow and air cycling intervals to keep ORP at within operating ranges. Methanol dosing continued to ensure nitrate effluent concentrations of 5 mg/L or less, but the effectiveness of denitrification in the anoxic reactor was assessed by the nitrate concentration at reactor 5B. Process control of the anoxic reactor was guided by twice weekly in-house testing of nitrate and ammonia in the effluent and in reactor 5B.

(2). A-train. There was no change from Phase 2.

Nitrate concentrations in reactors 5A and 5B were significantly lower in the B-train than the A-train during Phase 3 (Figure 2-27). Approximately 75% of reactor 5B nitrate concentrations were less than or equal to 5 mg/L, whereas only 20% of reactor 5A nitrate concentrations met the same standard. The frequency distribution A-train effluent concentrations, denitrified with methanol, were approximately 55% less than or equal to 5 mg/L. In Phase 3, therefore, denitrification performance of the MLE anoxic reactor was superior to denitrification with methanol in the anoxic EFB.

In Phase 3, B-train denitrification performance measured at the reactor 5B improved significantly. The nitrate concentration in reactor 5B from April 1997 to March 1998 (Phase 2 operations) met the 5-mg/L standard for approximately 45% of values (Figure 2-28). In Phase 3, 80% of values in reactor 5B met the 5-mg/L standard. Denitrification performance measured at reactor 5B in Phase 3 was equal to the effluent denitrification performance in the previous year of Phase 2 operations.

[6] **USEPA Nitrogen Control Manual**. 1993. pp. 251-258

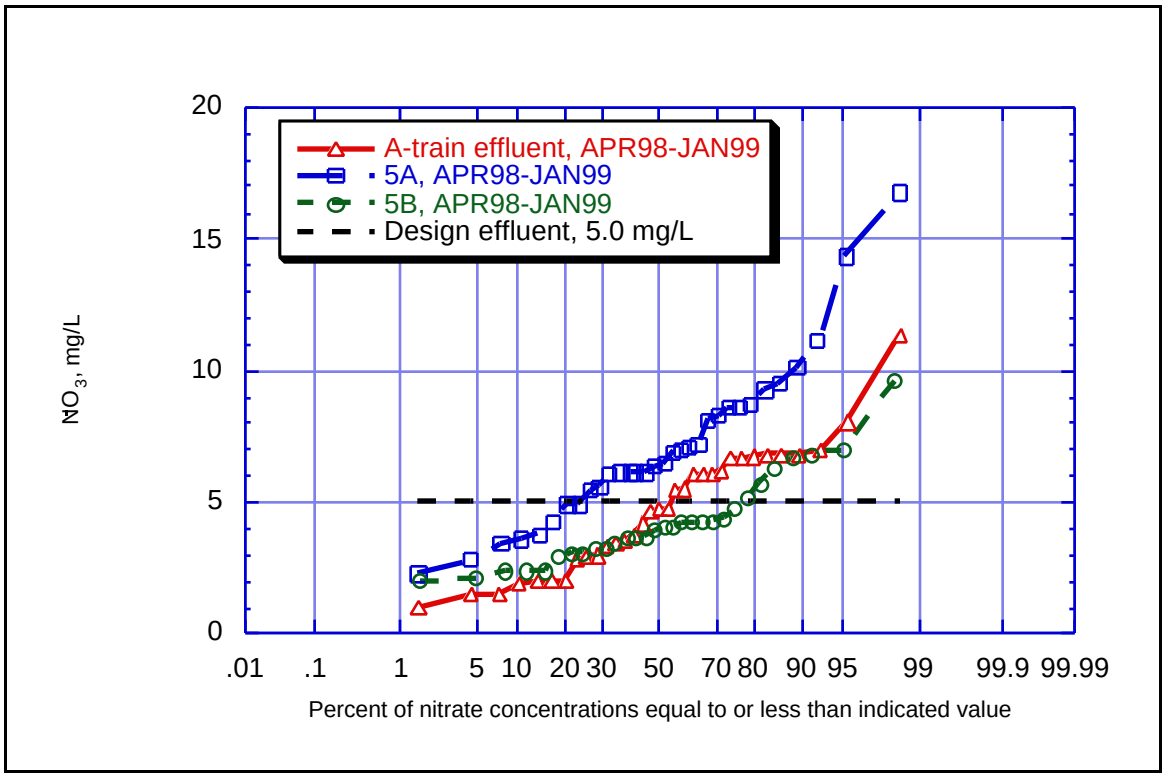

Figure 2-27. Phase 3 B-train denitrification performance compared to A-train. All samples plotted from period specified.

Phase 4

Phase 4 covers the period from March through June 1999. The design basis of Phase 3 used the MLE. The MLE model calls for MLSS concentrations at least twice that of the mean 1,000 mg/L MLSS concentration of the Vermont AEES. An important design goal of Phase 4 was to improve denitrification performance by augmenting the anoxic reactor MLSS concentration with fixed film biomass.

Buoyant fabric media (Aquamat®) augmented biomass in the anoxic reactor (Figure 2-29). The media was fixed in racks inside a square fabric media tower (FMT). The FMT was placed inside reactor 1B. Two submersible pumps placed outside the FMT cycle wastewater into the top of the FMT. Wastewater flows down through the media and out the FMT at the bottom. Coarse bubble diffusers mix wastewater outside of the FMT.

Another design goal of Phase 4 was to increase anoxic reactor efficiency by better process control. An oxidation-reduction potential (ORP) probe placed in reactor1B controls air cycling. Air is delivered while the anoxic reactor ORP is less than -50 mV.

There were no changes in A-train operation for Phase 4. The A-train continued to operate in Phase 2 mode.

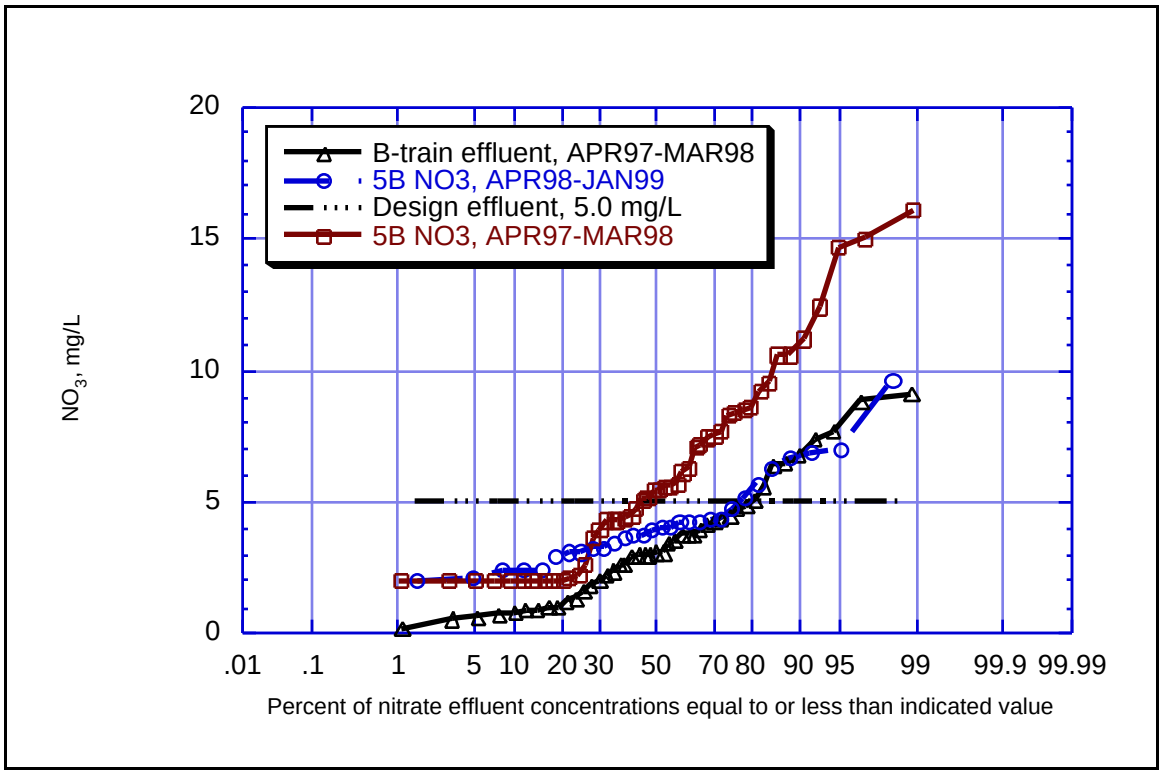

Figure 2-28. Phase 3 B-train denitrification performance compared to previous year Phase 2 B-train performance. Phase 3 - circles. Phase 2 operations - squares, triangles.

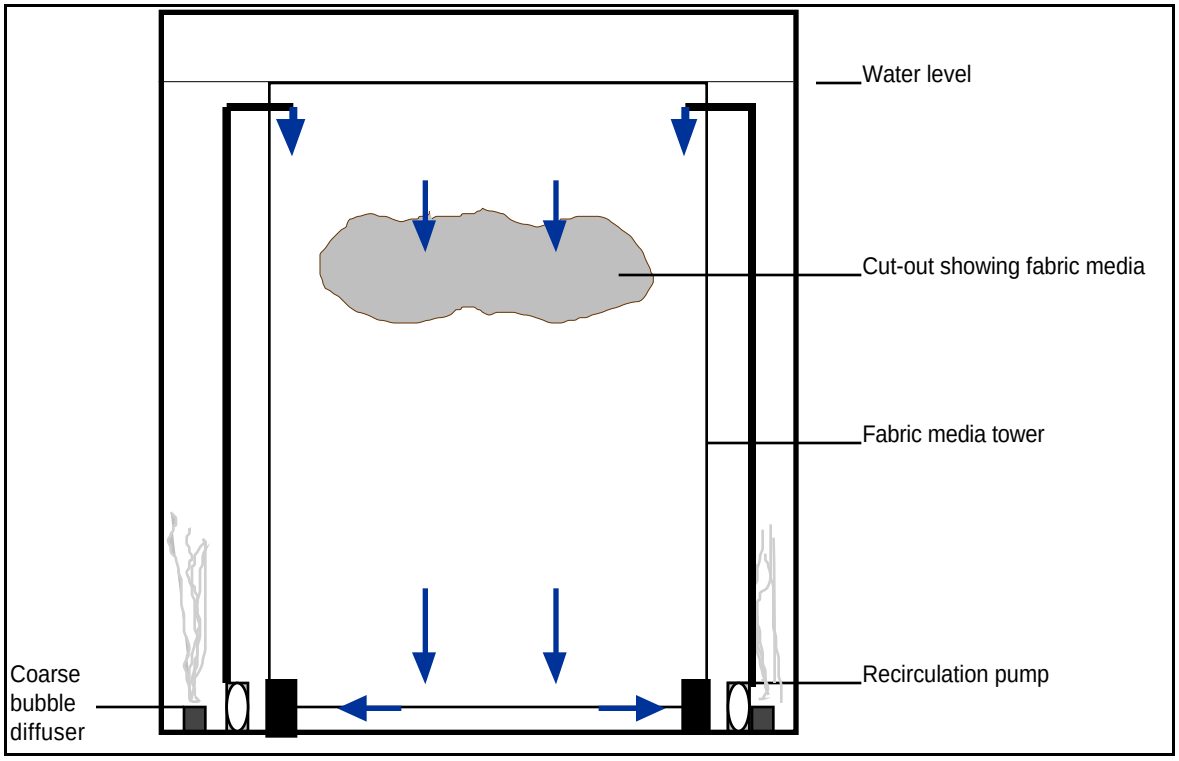

Figure 2-29. Schematic of reactor 1B with fabric media tower.

Phase 4 was a process evaluation and operational shakedown of the new B-train process configuration. April was the last month of methanol addition in the B-train. Effluent nitrate levels in the B-train averaged less than 5 mg/L in May but all values in June exceeded 5 mg/L. May values were probably the product of residual carbon in EFB2B. By June, the residual carbon was depleted. Process adjustments to improve denitrification lead to Phase 5.

Process recycle flows continued at 240 m^3/d. The rate for RAS stayed an estimated 38 m^3/d.

Phase 5.
Phase 5 covers July and August 1999. Hydraulic and monitoring modifications to the B-train improved denitrification efficiency. The A-train was converted to an MLE process using a simpler configuration of reactor 1A than used in reactor 1B.

There were two adjustments to the denitrification design basis of the B-train. The MLE model predicts greater denitrification efficiency with higher recycle rates. To increase process recycle, operators made minor adjustments to inter-reactor piping. Standpipe flow baffles at the outlet of each aerated reactor hydraulically limited flow through each treatment train. With the standpipes in place, higher process recycle rates were impossible. The standpipes were removed in both the A and B-trains. B-train process recycle was then increased to approximately 380 m^3/d.

A magnetic flow meter was installed on the B-train solids holding tank (SHT) discharge pipe to accurately measure biosolids waste and recycle flows. Return activated sludge flows averaged 38 m^3/d (0.25Q). Because there was no change in solids wasting protocol, this RAS rate has been accepted as an approximate figure for RAS flows throughout the project. Mean total recycle, which is process plus RAS recycle, for Phase 3 was 423 m^3/d (σ = 58 m^3/d).

The A-train was converted to an MLE process with approximately the same process and RAS recycle rates as the B-train. Conversion was simple. Fine bubble diffusers in reactor 1A were replaced with coarse bubble diffusers. Air delivery in reactor 1A was adjusted to maintain dissolved oxygen levels at less than 0.4 mg/L to keep reactor 1A in an anoxic condition. No other operational control for reactor 1A was used. A-train methanol dosing had been stopped by the end of June 1999. Conversion of the A-train to an MLE process allowed for a side-by-side comparison between a highly engineered (1B) and minimally engineering (1A) anoxic reactor.

Phase 5 B-train effluent nitrate values met design criteria on a concentration and mass

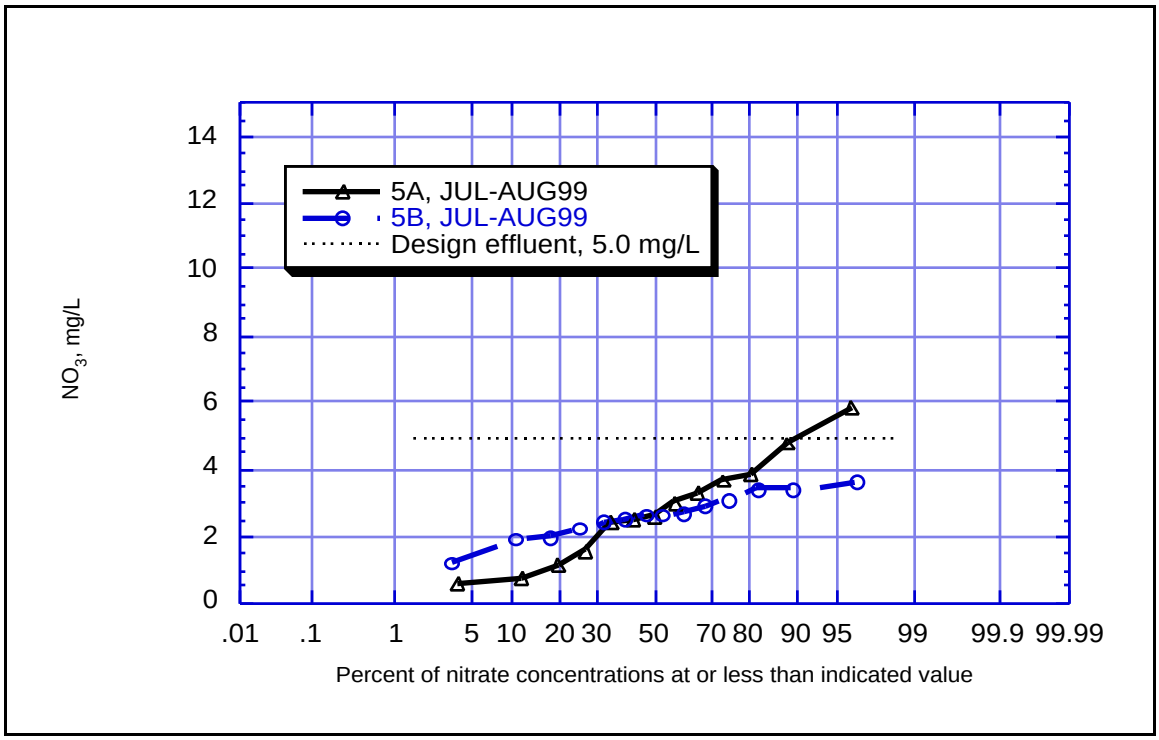

Figure 2-30. Phase 5 nitrate concentrations in reactors 5A and 5B. All samples plotted for period specified.

basis (Figure 2-25, Figure 2-26). The combined, flow-weighted mean effluent nitrate concentration for the A and B-trains was 4 mg/L (σ=1 mg/L). All of Phase 5 B-train values and 50% of A-train effluent mass flux values were equal to or less than 0.76 kg/d (Figure 2-25). Ninety five percent of B-train and 70% of A-train nitrate effluent concentrations were equal to or less than 5 mg/L (Figure 2-26).

Nitrate concentrations measured at reactors 5A and 5B are less than effluent concentrations. At reactors 5A and 5B, 90% of A-train concentrations were equal to or less than 5 mg/L, and all B-train concentrations were less than 4 mg/L (Figure 2-30). Effluent concentrations were significantly higher (Figure 2-26). Carry over of biosolids from the clarifier is the probable cause of the discrepancy between the reactor 5 and effluent nitrate concentrations.

Conclusion

The current MLE-based process meets design denitrification criteria, eliminating the need for methanol. Results from Phase 3 established that denitrification performance of the B-train anoxic reactor equaled or bettered denitrification performance with methanol. As indicated by the MLE model, increased process recycle rate improved denitrification performance. When the process recycle rate was increased from 240 to 380 m^3/d, effluent nitrate concentrations dropped below 5 mg/L in the B-train.

Evaluation of denitrification performance is best done using the nitrate concentration of reactors 5A and 5B. The design basis of the MLE process depends on completing denitrification prior to the clarifier. There is a rise of nitrate concentrations after the clarifier that is probably caused by lysis of cells carried over from the clarifier to the EFBs. A better clarifier design would minimize carryover of biosolids.

It is not clear if either the fabric media or ORP control of air delivery in 1B produce better denitrifying performance in the B-train than in the A-train. The relative effectiveness of one or the other appears to be a question of cost effectiveness. Process control of anoxic reactors using an ORP probe is a wastewater treatment industry standard and is a cost-effective way of reducing operator oversight of anoxic reactor operation. Denitrifying performance of the A-train, as measured in reactor 5A, can produce nitrate effluent concentrations of less than 5 mg/L on a monthly average basis. Given the small difference in performance between the A and B-trains, the significant expense incurred by design and construction of the fabric media tower in reactor 1B does not appear cost effective.

2.10.4. Total Nitrogen Removal Performance

Summary: The Vermont AEES produces total nitrogen effluent concentrations significantly less than 10 mg/L in the current configuration. Mass removal of influent total nitrogen is approximately 84%.

Introduction

Treatment performance of total nitrogen is a function of oxidation (nitrification) of reduced forms of influent nitrogen, followed by denitrification of the nitrified wastewater. Both nitrification and denitrification have been shown to meet or better design criteria on a mass and concentration basis. Given the consistently low TKN effluent values, it is the success of denitrification that has produced low total nitrogen effluent values well within the design basis.

Total nitrogen removal performance on a concentration basis

Nitrogen removal has consistently met the design standard. From August 1996 to August 1999, 39 of 40 monthly average total nitrogen effluent concentrations were significantly less than 10 mg/L (Figure 2-31). In the current treatment configuration, all total nitrogen effluent concentrations have been significantly less than 10 mg/L (Figure 2-32).

Total nitrogen removal performance on a mass basis

Nitrogen removal has consistently met design targets. From August 1996 to August 1999, monthly average total nitrogen effluent mass flux values have been significantly less than the effluent design standard of 1.5 kg/d (Figure 2-33). In the same period, approximately 96% of all TN effluent values have been less than or equal to 1.5 kg/d (Figure 2-34). In the current treatment configuration, in July-August 1999, all total nitrogen effluent mass flux values have been significantly less than 1.5 kg/d (Figure 2-35).

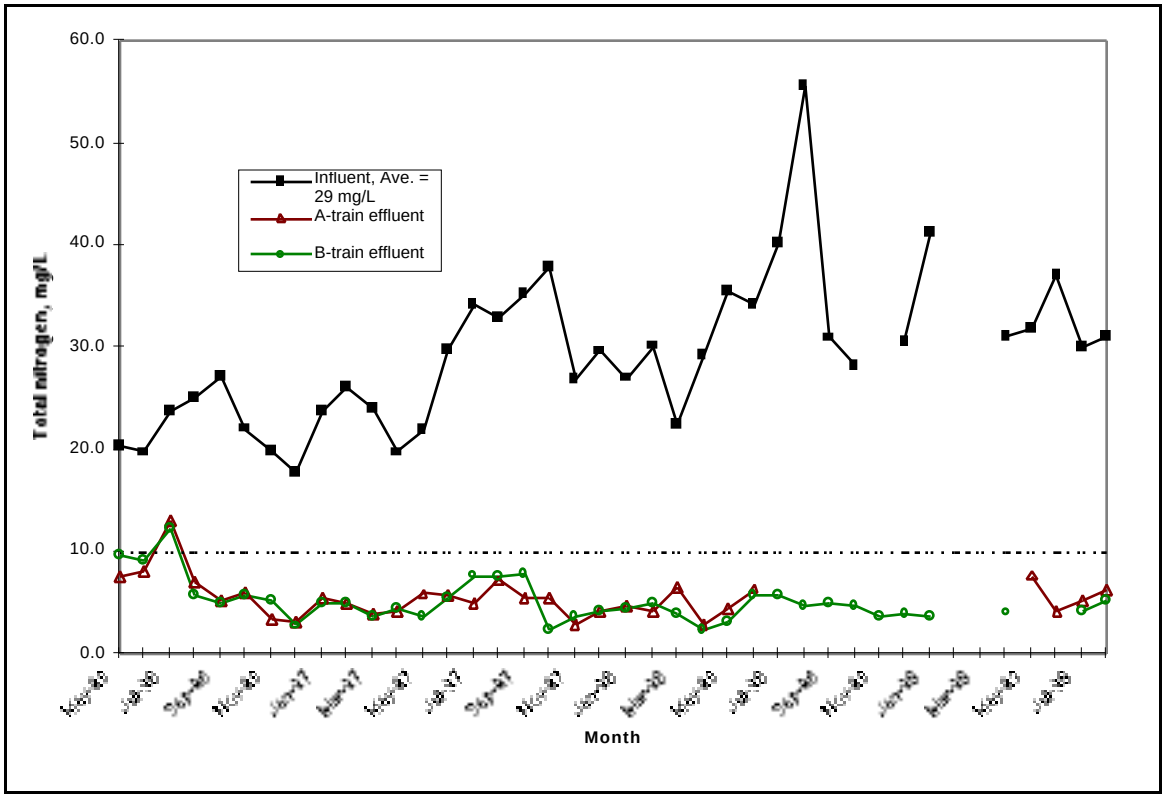

Figure 2-31. Monthly average TN removal performance May 1996-August 1999.

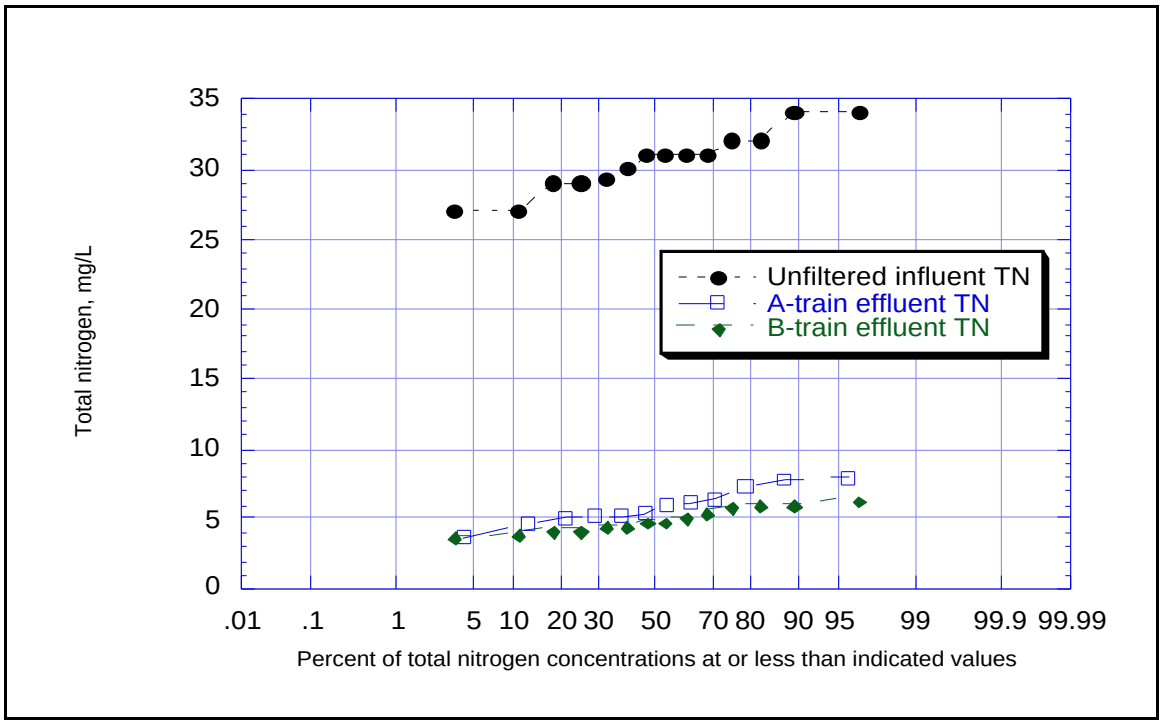

Figure 2-32. Total nitrogen influent and effluent concentrations for Phase 5, July-August 1999. All samples plotted for period specified.

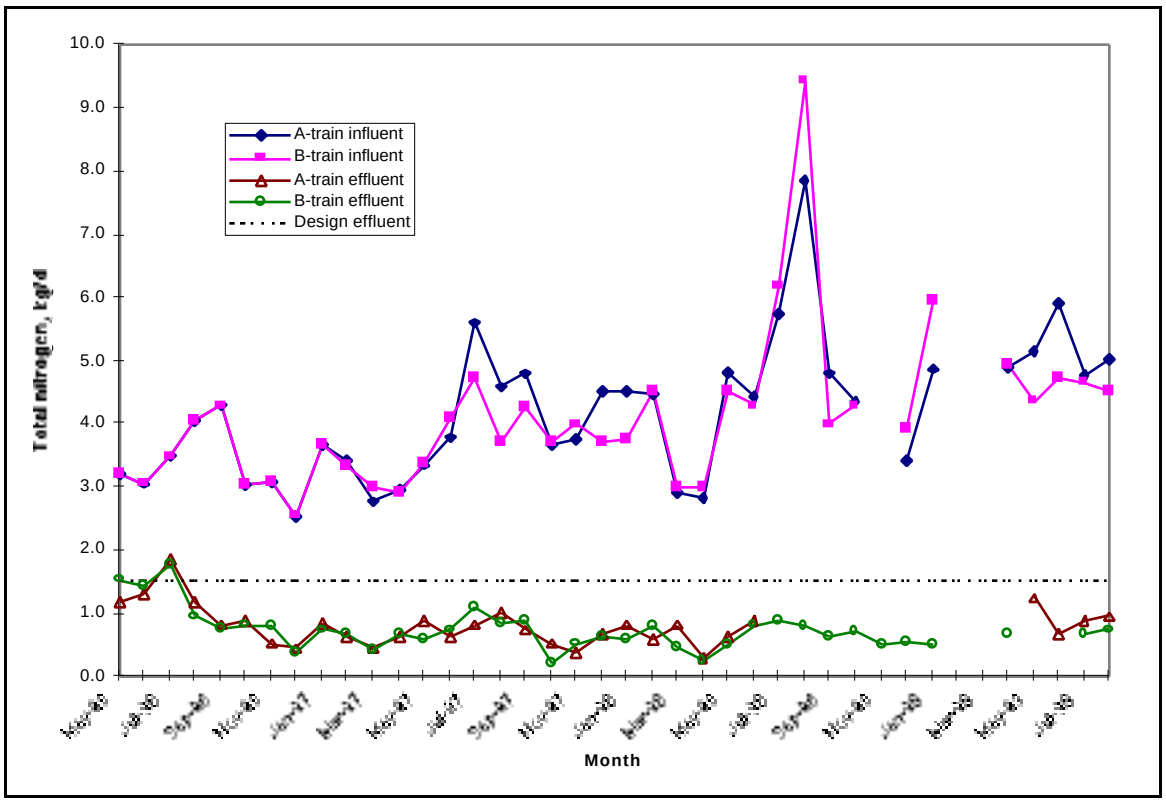

Figure 2-33. Monthly average TN influent loading and effluent mass flux values.

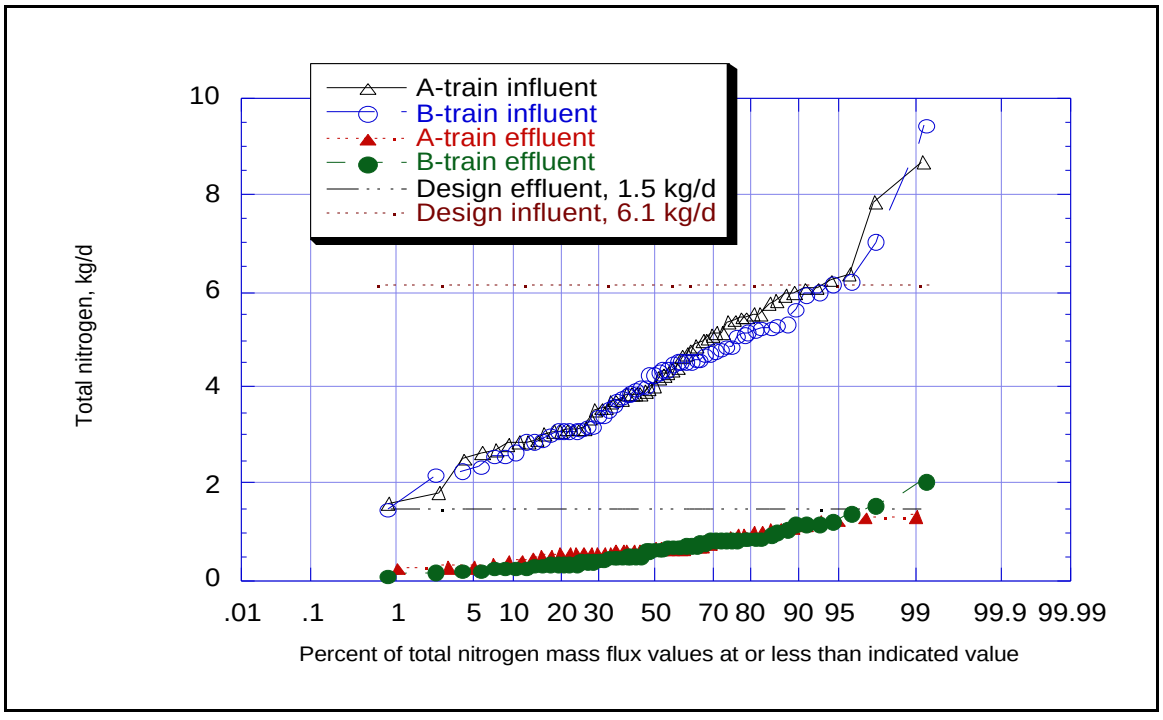

Figure 2-34. Total nitrogen influent and effluent mass flux for period of methanol addition, August 22, 1996-June 30, 1999. All samples plotted for period specified.

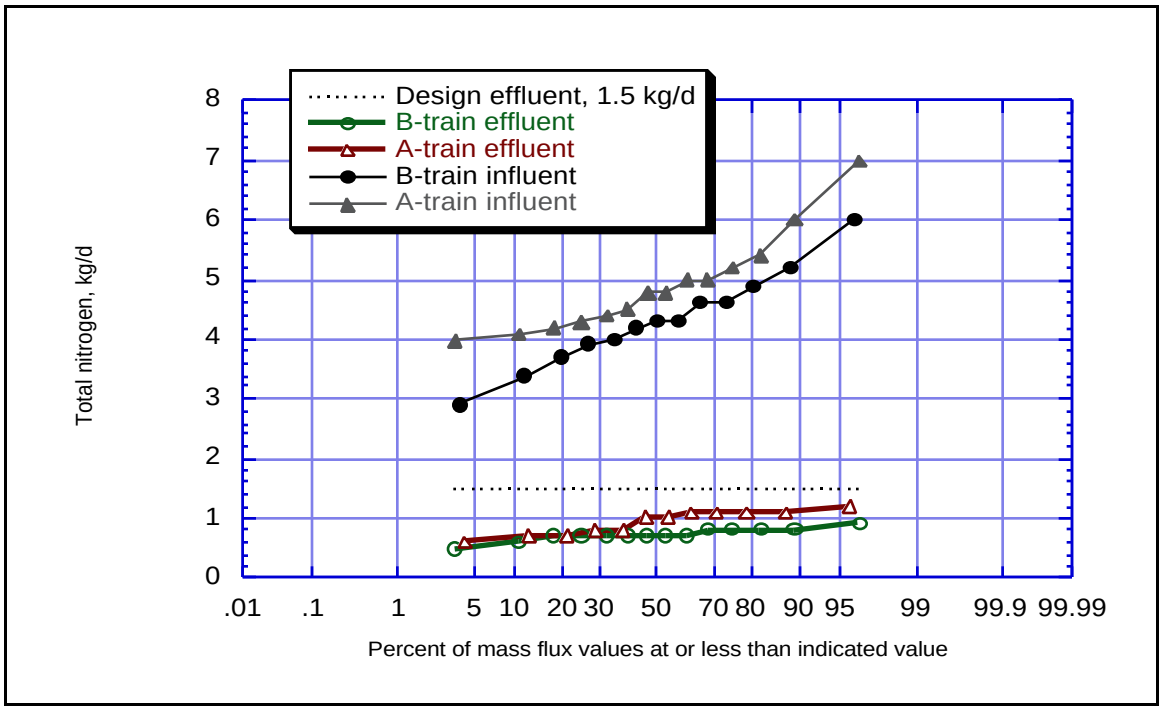

Figure 2-35. Total nitrogen influent and effluent mass flux for 2.5 Q recycle from 5A and 5B to 1A and 1B anoxic reactors, respectively, July 21-August 17, 1999. All samples plotted for period specified.

Overall mass removal was 79% and 80% for the A and B-trains, respectively. Mass removal was 81% and 85% for the A and B-trains, respectively. Design mass removal is 75%. Even though total nitrogen loading is significantly less than design, the actual mass removal performance (> 80%) is sufficient to produce design effluent at design loading (6.1 kg/d x (1 - 0.8) < 1.5 kg/d).

Conclusion

The Vermont AEES consistently produces total nitrogen effluent concentrations less than 10 mg/L and reliably removes 81 - 85% of total nitrogen influent mass. Total nitrogen removal performance has been highly stable and, in its current configuration, does not require methanol to produce the low effluent nitrate values required to meet total nitrogen design standards.

2.11. Phosphorous Removal Performance

Summary: The Vermont AEES has met the design effluent target of 3.0 mg/L total phosphorous. It cannot reliably produce effluent total phosphorous concentrations of 1 mg/L or less.

Introduction: Biological phosphorous removal is achieved by wasting biosolids. The Vermont AEES is not designed to stimulate bacteria into luxury uptake of phosphorous, and thus has a phosphorus removal capacity similar to that of conventional activated

sludge wastewater treatment. The design effluent target of 3.0 mg/L reflects limited TP removal capacity. Total phosphorus removal performance will be evaluated here in light of the common tertiary treatment effluent concentration standard of 1 mg/L.

Total phosphorous removal on a concentration basis

Total phosphorous (TP) effluent concentrations met the 1 mg/L standard seven out of forty months in this reporting period (Figure 2-36). The combined flow weighted mean effluent for this period is 2.1 mg/L (σ = 1.1 mg/L). Influent TP concentrations have been significantly less than the design concentration of 10 mg/L. It is unlikely that Vermont AEES could ever meet the 1-mg/L standard if influent phosphorous loading were significantly higher.

Total phosphorous removal on a mass basis

Total phosphorous removal has been a relatively constant percentage of TP influent loading (Figure 2-37). Average TP removal for both the A and B-trains has been 59% of influent loading from May 1995 to August 1999. The standard deviation of TP removal over the period has been 16% and 14% for the A and B-trains, respectively.

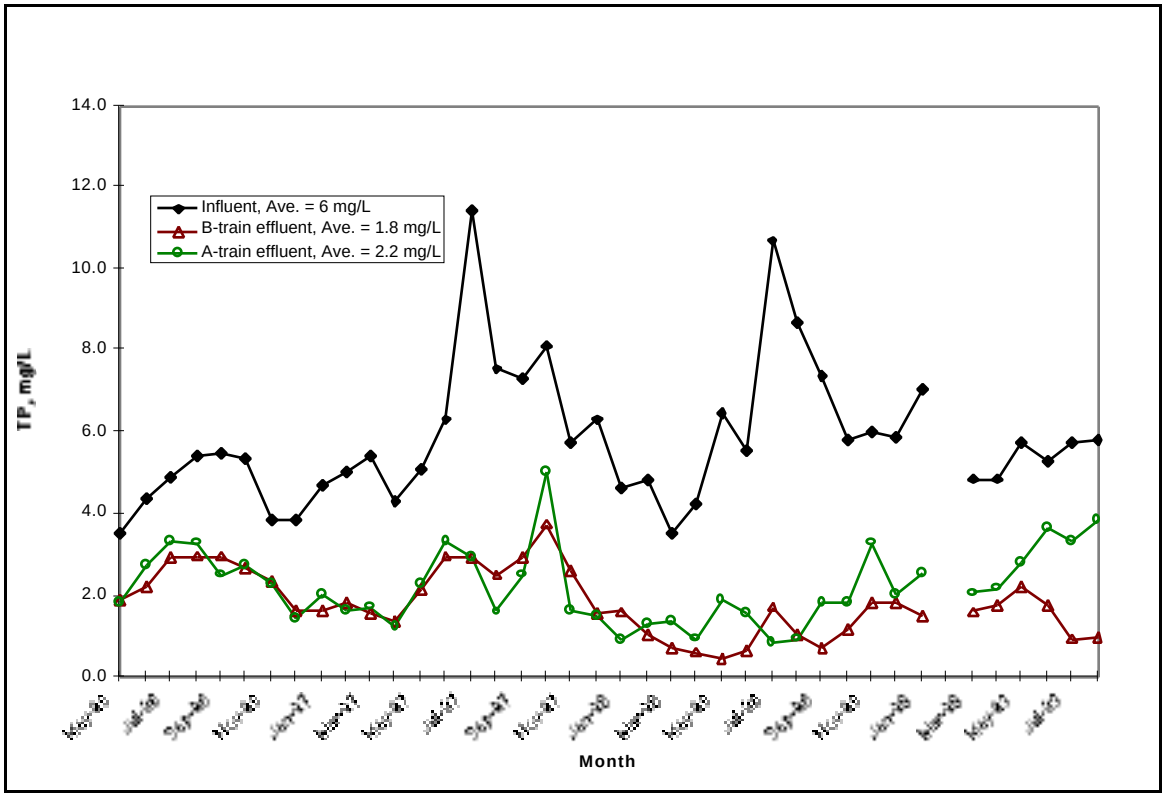

Figure 2-36. Monthly average influent and effluent total phosphorous concentrations, May 1996 - August 1999.

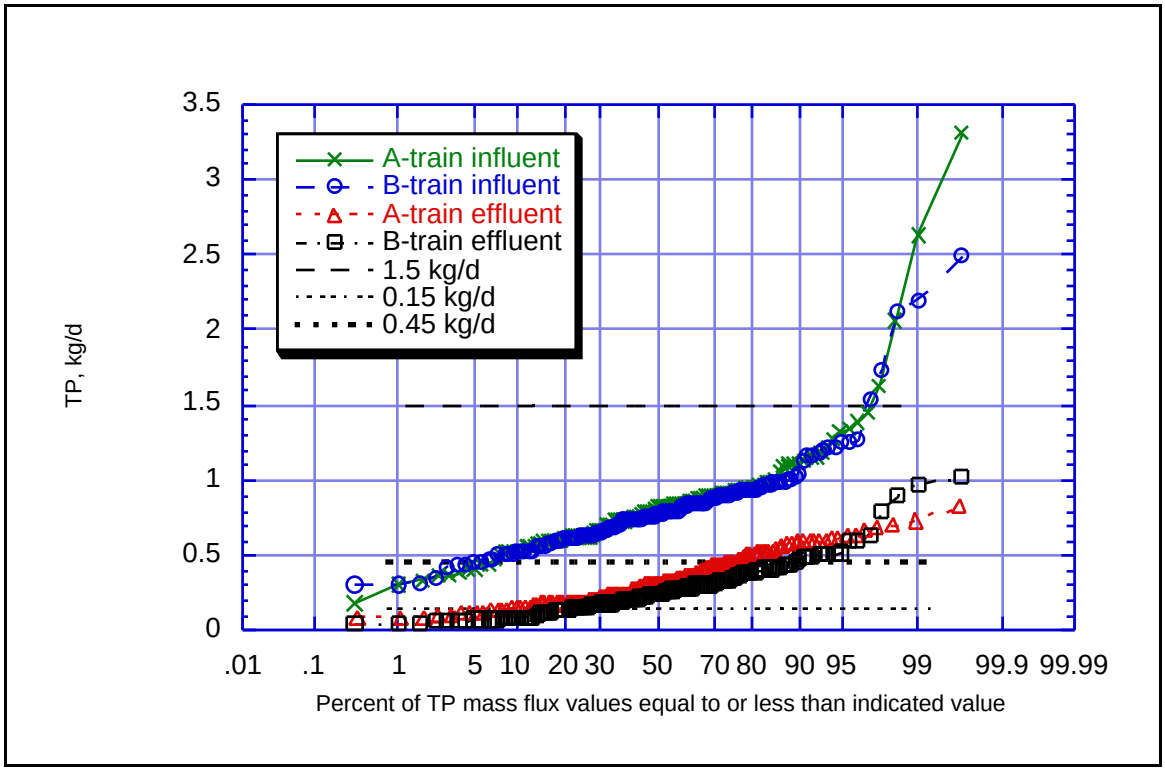

Figure 2-37. Phosphorous removal performance May 1996 – August 1999. All samples plotted for period specified. 1.5 kg/d, 0.15 kg/d, and 0.45 kg/d correspond to 10-mg/L influent concentration, and 1 mg/L and 3 mg/L effluent concentrations, respectively, at 152-m3/d influent to each train.

Conclusion

Phosphorous is removed in biosolids at a rate of approximately 60% of influent loading. To meet a total phosphorous effluent standard of 1 mg/L, additional treatment would be required.

2.12. Fecal Coliform Removal Performance

Summary: Fecal coliform removal at the Vermont AEES is highly variable, but approximately 80% of effluent values are equal to or less than 1,000 colony forming unites. Chemical or ultraviolet disinfection of Vermont AEES effluent would produce final effluent of fecal coliform counts ranging from non-detect to 100 cfu/ 100 ml.

Introduction

There is no disinfection of Vermont AEES effluent. Application of AEES technology to domestic wastewater reuse or surface discharge requires disinfection. The mechanisms of fecal coliform removal in the Vermont AEES are sedimentation in the clarifier, filtration after the clarifier and, probably, grazing by zooplankton. The performance of fecal

Table 2-15. Fecal coliform removal summary, May 1996 – August 1999.

Statistic	Influent	A-train effluent	B-train effluent
mean	8.5 x 10^6	x 10^3	x 10^2
σ	2.1x 10^7	9.6 x 10^3	2.3 x 10^3
N			
Min	3.0 x 10^5	1.1 x 10^1	2
Max.	1.6 x 10^8	9.0x 10^4	1.6x 10^4
	Mean removal	**99.97% (3.7 log)**	**99.99% (4 log)**

coliform removal in the AEES is evaluated here using the World Health Organization agricultural reuse standard of 1,000 colony forming units (cfu)/100 ml of wastewater.

Removal performance

Approximately 80% of B-train and 75% of A-train effluent fecal coliform counts meet the 1,000 cfu/100 ml standard (Figure 2-38). Treatment performance for the 2,000 mg/L design effluent standard is marginally better. The combined flow weighted mean effluent for this period is 1,700 cfu/100 ml (σ = 5,900 cfu / 100 ml). Mean removal of fecal coliform bacteria is 3.7 log and 4.0 log in the A and B-trains, respectively (Table 2-15).

Approximately 1 log removal credit can be given to the EFBs. The rest are removed in the clarifier. There is a high degree of variability in the fecal coliform data. It is not clear to what extent carry-over biosolids from the clarifier affects the variability.

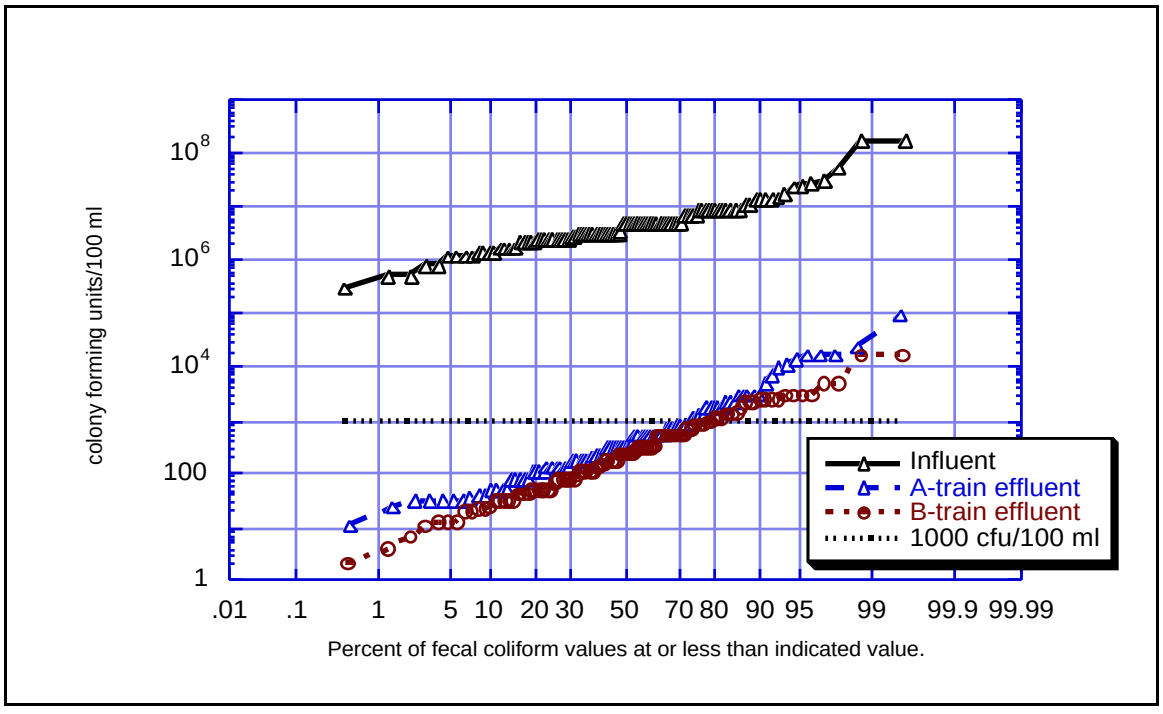

Figure 2-38. Fecal coliform removal performance May 1996 - June 1999.

Conclusion

Effluent fecal coliform values meet design criteria. Disinfection of Vermont AEES effluent to less than 1000 cfu/100 ml would for > 95% of random samples taken would require 2.0 log removal performance. Chemical or ultraviolet disinfection performance of tertiary quality effluent wastewater is typically between 3.0 and 6.0 log reduction in colony forming units, and would clearly meet the disinfection requirement.

As has been documented in the sections above, the effluent quality of the Vermont AEES meets strict tertiary effluent standards for TSS and ammonia. Effluent ammonia concentrations consistently less than 1 mg/L present negligible chlorine demand in a chlorine disinfection system. Effluent TSS concentrations less than 5 mg/L allow for effective ultraviolet and chlorine disinfection. With either chlorine or ultraviolet disinfection, therefore, Vermont AEES overall fecal coliform removal performance would range between 6.0 log and 10 log. At 6.0 log removal, the maximum recorded influent fecal coliform count would be reduced to 160 cfu / 100 ml. The mean influent fecal coliform count would be reduced to less than 10 cfu / 100 ml. With higher removal efficiencies, effluent fecal coliform counts would be at non-detect levels. The Vermont AEES is, therefore, well suited to produce effluent with fecal coliform counts of less than 100 cfu / 100 ml with ultraviolet or chlorine disinfection.

2.13. Treatment Performance Conclusion.

The Vermont AEES has met or bettered design treatment performance for $CBOD_5$, COD, TSS, TKN, NH_3, NO_3, TN, and fecal coliform bacteria (Table 2-16). Evaluation of treatment performance for $CBOD_5$, COD, and TSS is evaluated for all samples taken during period with no methanol addition. Methanol is no longer part of any AESS treatment process, but when used had significantly raised effluent concentration for $CBOD_5$, COD, and TSS. True treatment performance of the Vermont AEES for these parameters must therefore be evaluated for period without methanol addition. Treatment performance for TKN and NH_3 is evaluated over all samples taken from May 1996. Methanol addition does not affect these parameters in any way, allowing inclusion of all contract laboratory samples taken since the beginning of steady state operations. Nitrate and TN treatment performance in the current treatment configuration has allowed the elimination of methanol for denitrification.

The total phosphorous combined average effluent concentration of approximately 2 mg/L did meet design criteria. The Vermont AEES is not capable of consistently meeting total phosphorous effluent concentrations equal to or less than 1 mg/L in the current treatment configuration.

The use of methanol for denitrification has been discontinued in favor of a modified Ludzak-Ettinger (MLE) process. The MLE process uses a single-stage, front-end anoxic reactor. Fully nitrified wastewater with MLSS is pumped from just before the clarifier

Table 2-16. Treatment Summary Table. Effluent values are flow-weighted, combined means of the A and B main treatment trains.

PARAMETER	Mean Influent	Mean effluent, mg/L	Design effluent limit, mg/L
$CBOD_5$	207 (standard deviation, σ=78)	4 (σ = 2)	<10
COD	546 (σ = 154)	24 (σ = 6)	not specified
TSS	189 (σ = 59)	3 (σ = 1)	<10
TKN	28 (σ = 7)	2 (σ = 1)	<5
NH_3	14 (σ = 4)	0.3 (σ = 0.4)	<1
NO_3	-	4 (σ = 1)	<5
TN	31 (σ = 2)	5 (σ = 1)	<10
TP	6 (σ = 3)	2 (σ = 1)	<3
Fecal coliform bacteria	8.5×10^6 cfu / 100 ml ($\sigma = 7 \times 10^7$)	1,700 cfu (σ = 5,900)	<2,000 cfu

back to the anoxic reactor where it is denitrified. Process recycle is two to three time the influent flow rate.

Denitrification with methanol in the anoxic EFBs was adequate, but methanol raised $CBOD_5$, COD, and TSS effluent concentrations to regularly exceed design criteria. Cessation of methanol use eliminated the $CBOD_5$, COD, and TSS effluent concentration exceedences. The MLE process had superior denitrification performance to methanol and had no adverse impacts on any other effluent parameter concentrations. Overall treatment performance results demonstrate that Vermont AEES produces effluent of advanced tertiary quality, with the exception of fecal coliform and total phosphorus values. The effluent characteristics are well suited to achieve fecal coliform values of less than 10 cfu / 100 ml using standard disinfection technologies. To achieve effluent total phosphorous concentration less than 1 mg/L supplemental phosphorous treatment would be necessary.

3. SYSTEM COMPONENT ANALYSIS

3.1. Introduction

This chapter describes and analyses the performance of key system components of the Vermont AEES. These components are the: (1) aerated reactors 1 through 5, (2) clarifier, (3) EFBs, (4) headworks, (5) controls, and (6) blowers, (7) plants and (8) macrofauna. Components are described or analyzed in terms of their role in treatment performance and, to a lesser extent, operations. The topic of plants is expanded to include horticultural operations and data of plant performance from the test train.

3.2. Aerated reactors 1 through 5: Empirical Results and Model Analyses

Summary: The aerated reactors alone remove nutrients to design standards. Nutrient removal is completed to design standards at the fourth aerated reactor.

Introduction

The purpose of this section is to determine the effectiveness of the aerated reactors for nutrient removal. In the current configuration, the aerated reactors treat $CBOD_5$, COD, TKN, NH_3, and NO_3 to design effluent values. The parameters analyzed in this section are $CBOD_5$, COD and NH_3. Denitrification and phosphorous removal were analyzed in Section 2. A key part of the analyses that follow is a determination of the HRT required to achieve design removal of nutrients.

Profile series data are from the B-train from December 1998 though April 1999, except where noted. Influent temperatures for this period ranged from 11 to 14^{o} C.

Effectiveness of nutrient removal is analyzed using the profile of nutrient concentrations throughout the series of reactors. Part of the profile analysis is a determination of the reaction rate to be used in a first order CFSTR in series model.

A first-order, CFSTR equation is used to model $CBOD_5$, COD, and NH_3 removal in the aerated reactors.

(Equation 3.2-1), $\mathbf{C_n = C_{in} / [1+k(V_r/Q_T)]^n}$,
where:
C_n is the concentration at reactor n, mg/L;
C_{in} is the initial concentration at 1B, mg/L, to account for the effects of recycle
C_{in} = [S_o kg/d influent + S_5 kg/d recycle + S_{RAS} kg/d]/[I_n + $Q_{r.}$];
k is the reaction rate coefficient, d^{-1};
V_r is the volume of each reactor, 57 m^3;
Q_T is the flow rate through the reactor (= I_n + $Q_{r.}$), m^3/d;
n is aerated reactor sequence number;

I_n is influent flow, m^3/d;
$Q_{r.}$ is recycle flow (= Q_{RAS} + $Q_{\text{process recycle}}$), m^3/d;
S_o is influent concentration;
S_5 is concentration at reactor five;
S_{RAS} is the soluble concentration in RAS.

Variations on this equation are commonly used for wastewater modeling[7]. Principal advantages of this model equation are ease of use, excellence of fit to the data, and that recycle flows can be built into it.

3.2.1. $CBOD_5$ Removal

Mean $CBOD_5$ concentrations at 3B meet design effluent standards at influent water temperatures of 11-14° C (Figure 3-1). There are no $CBOD_5$ concentration excursions greater than 10 mg/L at reactor 4B (Figure 3-2).

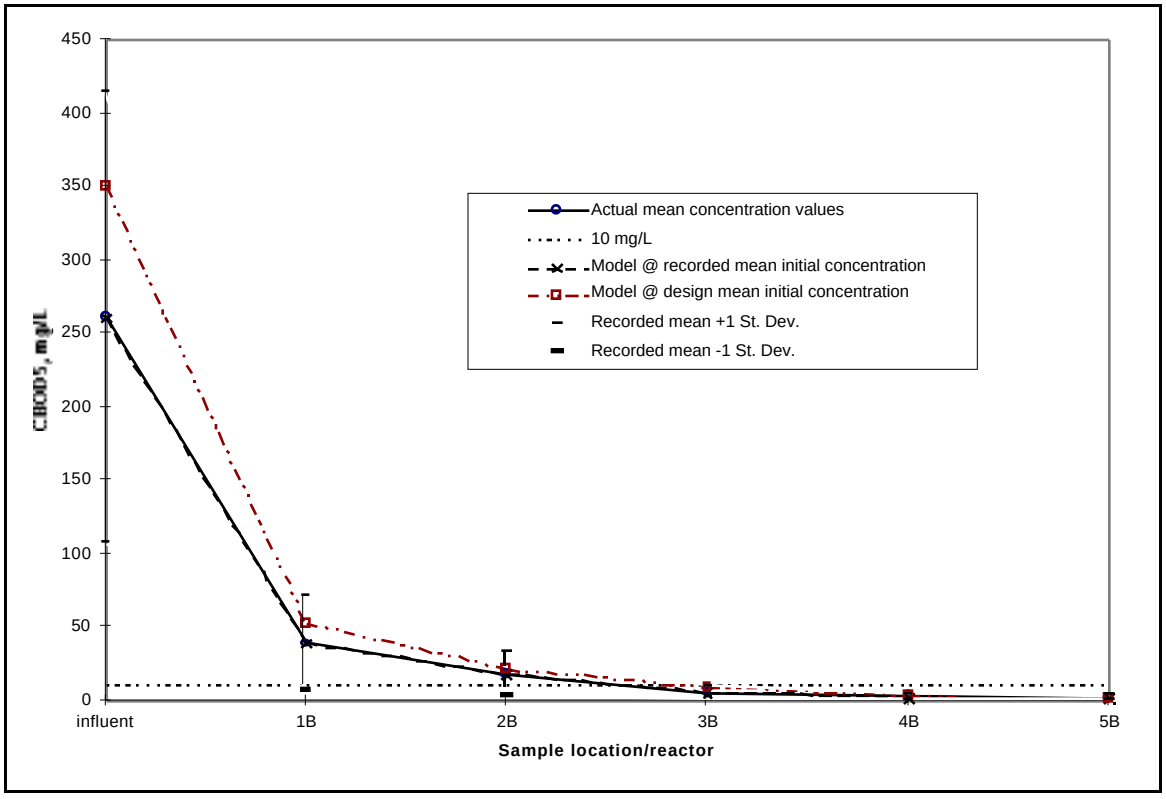

Figure 3-1. Profile of $CBOD_5$ concentrations in the B-train, December 1998-April 1999, compared to model values. Each datum of the actual values curve is a mean value (n=15) for the sample/reactor location. Error bars are one standard deviation above and below the mean. Flows during this period averaged 151 m^3/d (39,850 gpd).

[7] Crites and Tchobanoglous. 1998. **Small and Decentralized Wastewater Management Systems,** p. 458. McGraw-Hill.

3.2.2. $CBOD_5$ Removal Model

A CFSTR in series model predicts $CBOD_5$ removal performance (Figure 3-1). The model is based upon data from samples taken at cold (11-14^o C) influent temperatures. The reaction rate coefficient of 10.7d^{-1} for the model is taken from a plot of -ln($CBOD_{5\ out}$/$CBOD_{5\ in}$) concentrations versus HRT. The rate was chosen from the slope at reactor 1B. Rates evaluated from downstream reactors are lower overall and do not reasonably match actual data when used in the model. The superficial residence time in each reactor is 0.38 days (9 hours), which does not account for recycle flow to the reactors. The effective model reactor HRT used for modeling is 0.13 days (3.1 hours) at influent and total recycle flows of 151 m^3/d and 242 m^3/d, respectively. The model influent concentration is the weighted value of actual influent concentration diluted by recycle.

The model closely reproduces $CBOD_5$ removal performance (Figure 3-1, Figure 3-2, Figure 3-3). The model does not predict much of the variability of CBOD5 concentrations in reactors 1B and 2B, but reactors 3B and 4B in the model closely match actual data (Figure 3-2, Figure 3-3). Comparison of actual and model frequency distributions of $CBOD_5$ concentration at each reactor reveals that the model results are more conservative than actual data. The model can therefore be used with confidence to evaluate the $CBOD_5$ concentration profile at design loadings. At the design influent $CBOD_5$ concentration of 350 mg/L and flow of 152 m^3/d, the model predicts that an effluent concentration of 10 mg/L would be met by reactor 4B (Figure 3-1).

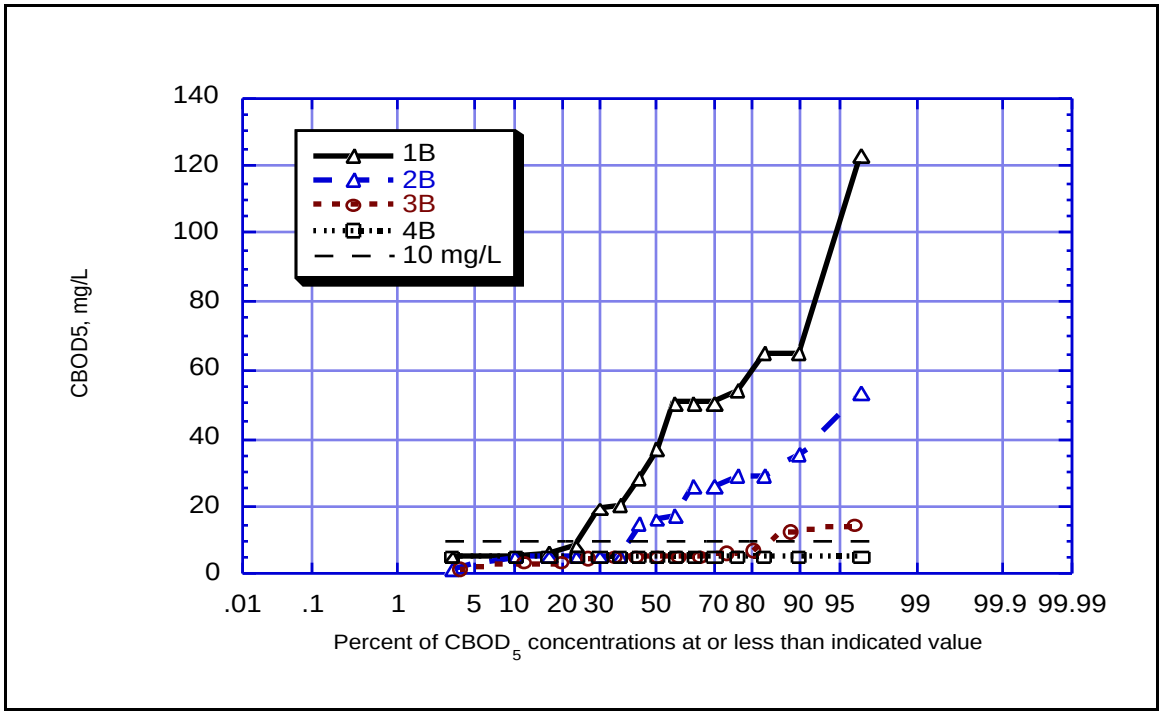

Figure 3-2. B-train $CBOD_5$ removal performance by reactor, December 1998-April 1999.

Table 3-1. Model performance predictions at design $CBOD_5$.

Sample location,	$CBOD_5$ = 350 mg/L
Influent	350
1B	56
2B	24
3B	11
4B	5
5B	2

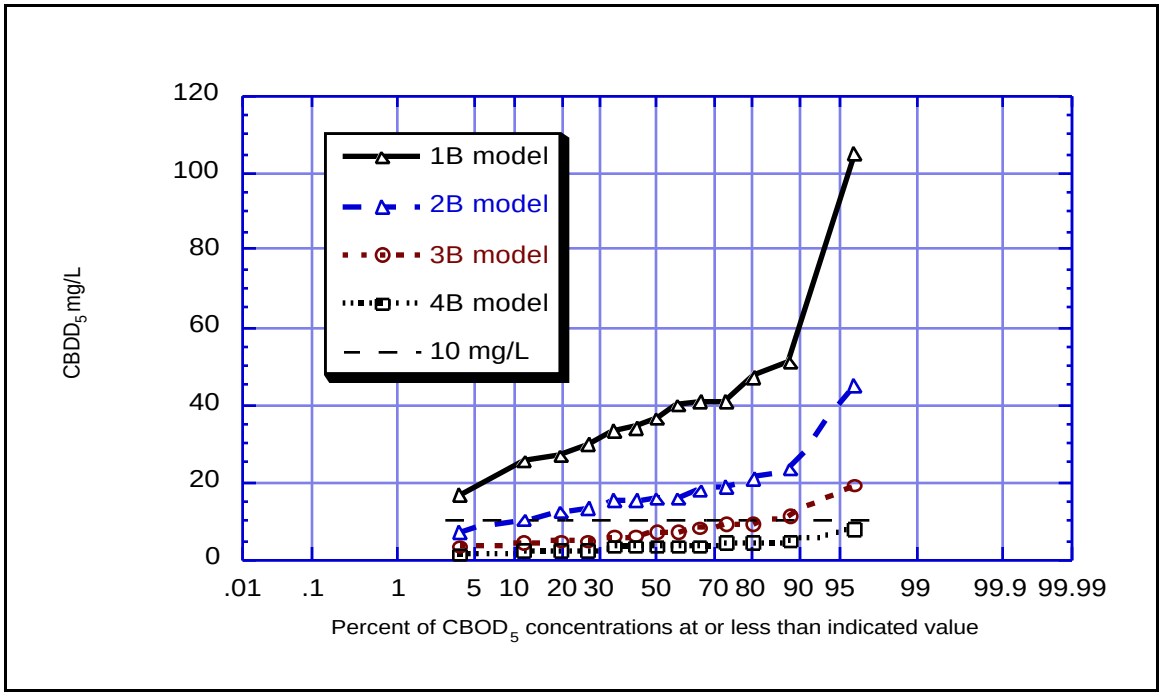

Figure 3-3. B-train model $CBOD_5$ removal performance by reactor. Influent concentrations and flows are the same as those plotted in Figure 3-2.

3.2.3. COD Removal

Mean COD concentrations meet the adapted effluent standard of 30 mg/L at reactor 3B while influent water temperatures of were 11-14^{o} C (Figure 3-4).

3.2.4. COD Removal Model

The CFSTR in series model (Equation 3.2-1) with k = 7.1 d^{-1} closely approaches the COD concentration profile in reactors 1 through 5 (Figure 3-4). As with the $CBOD_5$ model, the reaction rate coefficient, k, is the rate of reaction in the first reactor. In this

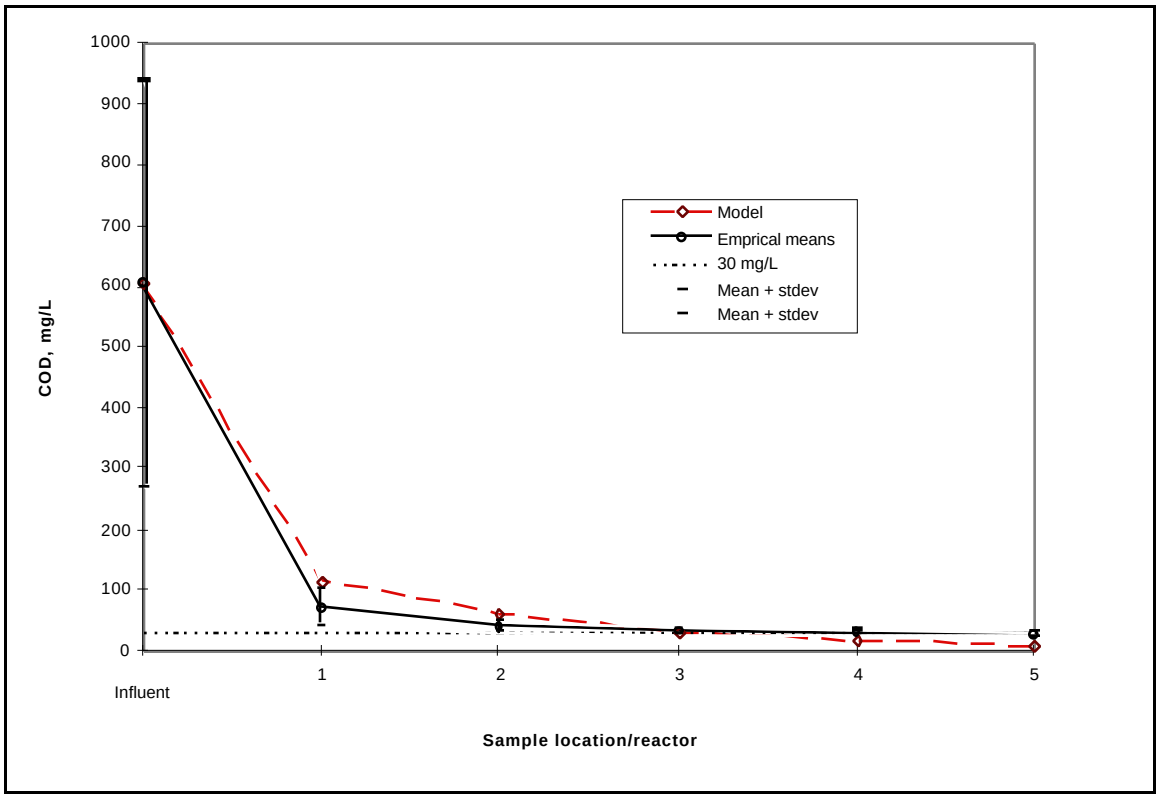

Figure 3-4. Profile of COD concentration in the B-train, December 1998 - April 1999 compared to model values. Each datum of the actual values curve is a mean value (n=15) for the sample/reactor location. Error bars are one standard deviation above and below contract laboratory means. Flows during this period averaged 151 m^3/d (39,850 gpd).

case, modeling of COD is less accurate than $CBOD_5$ because the CFSTR model does not account for the fraction of recalcitrant carbon in the influent. There is insufficient data to construct a COD removal model based on the soluble, biologically available fraction of COD.

3.2.5. Nitrification Performance

Nitrification performance in the aerated reactors is vital in the current treatment configuration. Nitrification cannot proceed at a high rate until $CBOD_5$ and COD concentrations are low. Denitrification by recycle of nitrified effluent to an anoxic first stage reactor is limited if nitrification performance is impaired.

Nitrification performance at reactors 5A and 5B permits an overall assessment of operations at the Vermont AEES (Figure 3-5). The addition of RAS to the process diagram in August 1996 significantly improved nitrification performance. Starting in early summer of 1997, nitrification performance suffered from a gradual loss of air caused by progressive fouling of diffusers and corrosion of the air delivery main. The original diffusers were replaced with surface serviced diffusers that could easily be cleaned. The air main was repaired in March 1998. Nitrification performance stabilized immediately

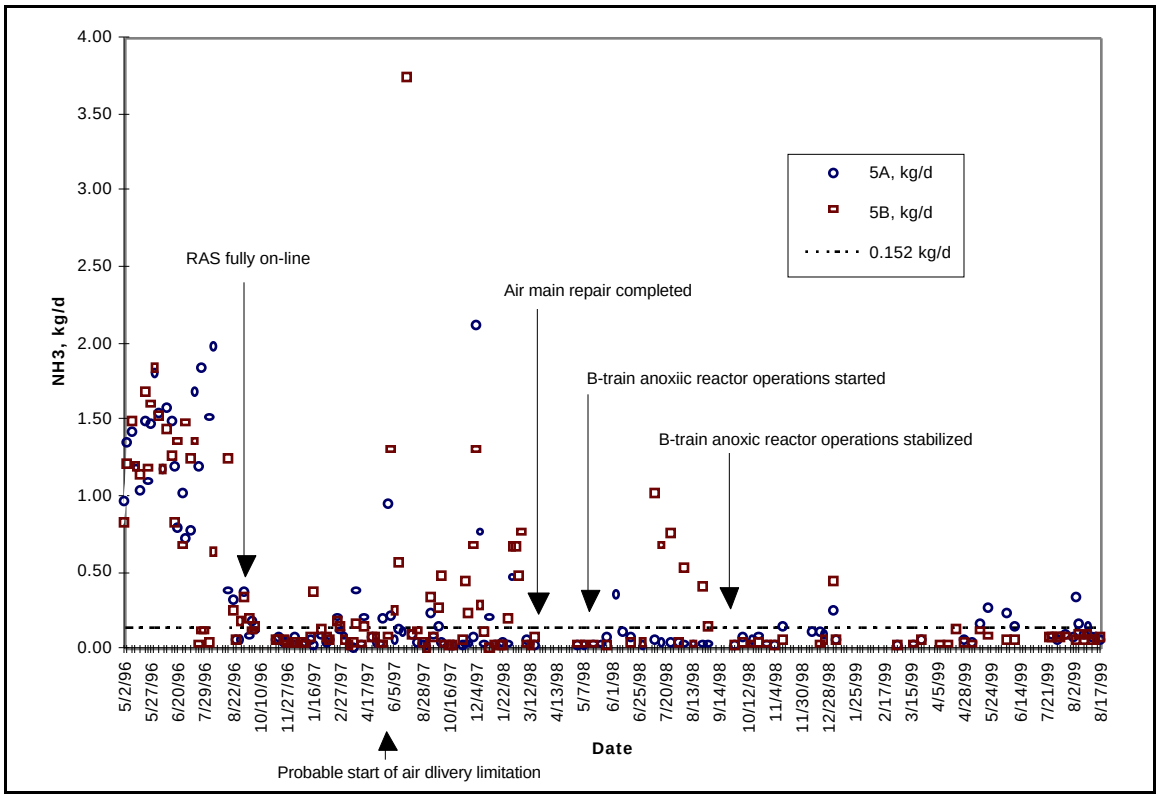

Figure 3-5. Nitrification performance at reactors 5A and 5B May 1996-August 1999. An effluent mass flux of 0.152 kg/d is 1 mg/L at a flow of 152 m^3/d. Data are from in-house process and contract laboratories.

thereafter. Nitrification performance stability in reactor 5B suffered in the summer of 1998 during experiments to determine optimal operation of the anoxic reactor.

From October 1998 to August 1999, nitrification performance of the B-train has been highly stable at mass flux values corresponding to effluent concentrations of less than 1 mg/L.

From December 1998 to April 1999, nitrification was complete by reactor 4B on an average basis, but 30% of ammonia concentrations at reactor 4B were greater than 1 mg/L (Figure 3-6, Figure 3-7). The excursions did not occur in the coldest water temperature of March-April 1999. They are clustered over the New Year holiday and the following week. An independent data set gathered from October 1998 to June 1999 shows excursions occurring exclusively in the same period. Examination of the operator's log reveals that the excursion occurred on days when the MLSS concentration in the aerated reactors had dropped below 800 mg/L. These low MLSS concentrations were the result of operator error and not indicative of a design process deficiency. It can therefore be inferred that nitrification is complete by reactor 4B.

3.2.6. Nitrification model

The model closely matches nitrification performance in cold water temperatures,

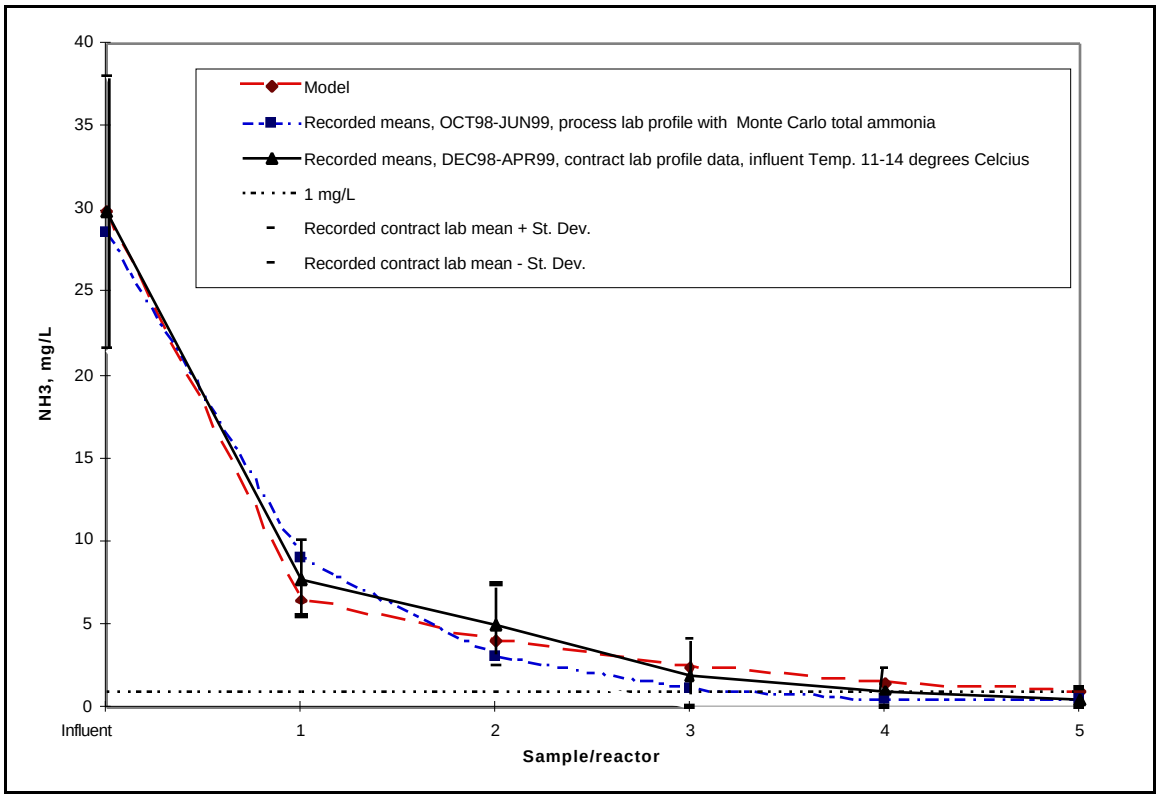

Figure 3-6. Profile of NH_3 concentration in the B-train, December 1998 April 1999 compared to model values. Error bars are one standard deviation above and below contract laboratory means.

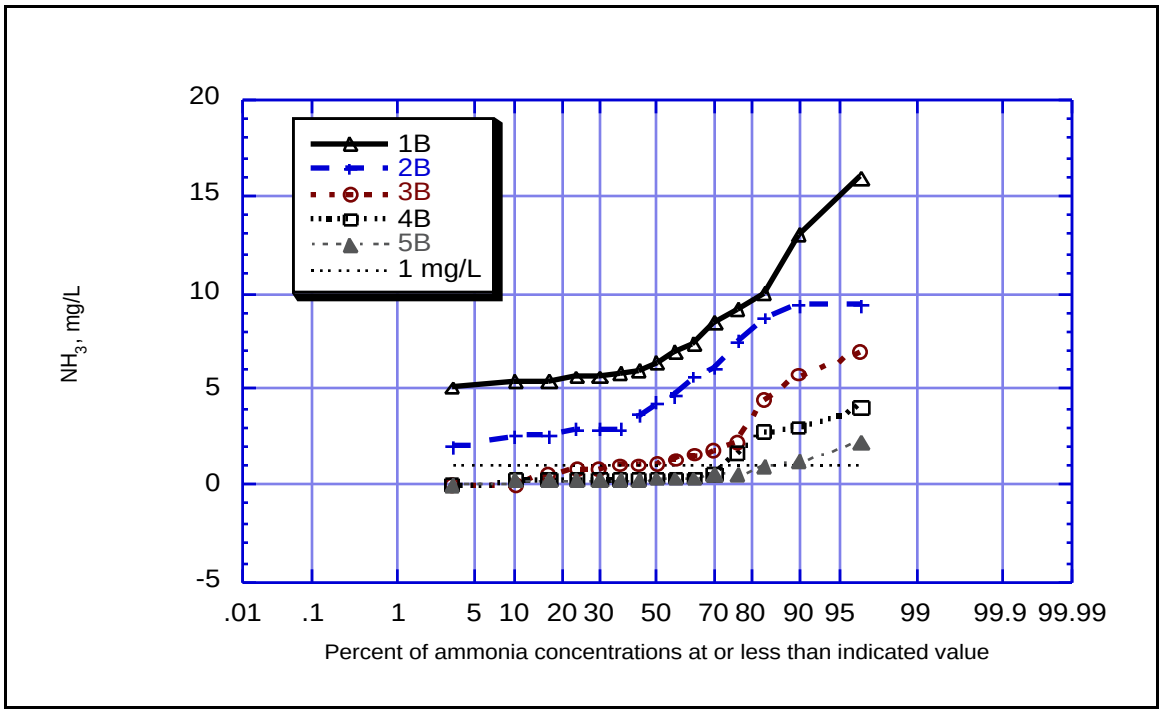

Figure 3-7. B-train nitrification performance by reactor, December 1998-April 1999.

predicting that nitrification is complete by reactor 4B (Figure 3-6). The nitrification model is a CFSTR in series (Equation 3.2-1) with a reaction rate coefficient of 5 d^{-1}. Construction and evaluation of the model required careful tracking of rate inhibition from high $CBOD_5$ or COD concentrations. As has been shown previously, most organic carbon is removed by reactor 3B, where the nitrification rate jumps from 1.2 d^{-1} to 3.1 d^{-1} in a plot of -log(C/C_{in}) vs. time. The model uses a reaction rate of 5 d^{-1} to achieve a close fit to data (Figure 3-6). The model was developed from contract laboratory data and verified with an independent data set from the in-house process laboratory.

In-house process laboratory data could not directly be used to check the model because it was not equipped to conduct TKN tests. A Monte Carlo technique was used to assign total ammonia loading values to the in-house process laboratory data, which was then used to evaluate the model fit (Figure 3-6). A random number generator with normal distribution was used create values for TKN-NH_3, the fraction of organic nitrogen in the effluent that is converted to ammonia. The random number generator was bounded by the minimum and maximum TKN-NH_3 values taken from contract laboratory data for the same period. These values were added to the process laboratory NH_3 influent values to obtain total ammonia loading.

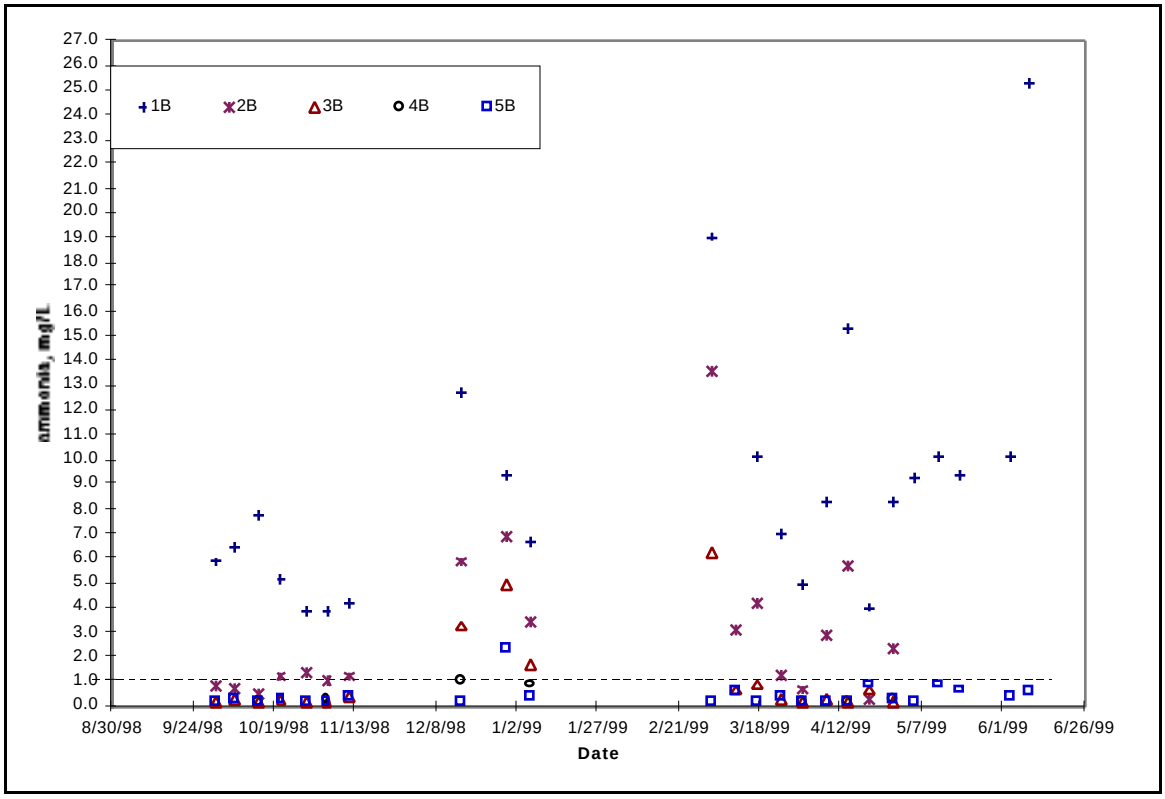

Figure 3-8. B-train nitrification performance by reactor, October 1998-June 1999. The data set is depicted is independent of data in Figures B-5 and B-6. Note ammonia concentrations in reactor 4B (circles) at or higher than 1 mg/L in late December 1998 and early January 1999.

Conclusion

Nutrient removal is completed to design standards in the five aerated reactors with influent temperatures of 11-14°C. Effluent $CBOD_5$ concentrations less than or equal to 10 mg/L are met by reactor 4B and 5 mg/L by 5B with no exceedences. Effluent COD concentrations of 30 mg/L are also met by reactor 3B. Nitrification to ammonia concentrations of 1 mg/L is complete in reactor 4B.

The total working volume of four reactors is 226 m^3. Hydraulic residence time of the four reactors at an influent flow of 152 m^3/d is 1.5 days. At cold water flows (11-14° C) a residence time of 1.5 days may be sufficient to treat to tertiary standards an influent of characteristics similar to that received at the Vermont AEES.

The nutrient removal profile data sets are supported with models. A standard CFSTR in series model closely reproduces performance data for $CBOD_5$, COD, and NH_3. Two caveats should be considered. One, the models are not integrated. A high influent COD concentration, for instance, would retard nitrification, but there is no provision in these models to link the effect of carbon loading on nitrification. Two, only the ammonia model is verified with an independent data set. With these caveats in mind, the models support the conclusion that a significantly reduced HRT in the Vermont AEES would permit treatment to design effluent standards domestic influent of composition, flow, and strength similar to that recorded over the period of this report.

3.3. Clarifiers

Summary: The clarifiers of the Vermont AEES are not a good design. Except in the summer of 1999, the clarifiers have been unable to consistently produce clarifier effluent TSS concentrations ≤ 10 mg/L. Inadequate clarifier performance forces reliance on filtration to achieve design effluent concentration. Clarifier performance dramatically improved in the summer of 1999 with the current treatment configuration.

The Vermont AEES has passive, center-feed, circular clarifiers. Side-wall depth is nine feet. Clarifier supernatant flows through a submerged, perforated pipe into the first EFB. The bottom is a 30° concrete cone with biosolids removal in the center by an airlift pump. Biosolids are pumped to a 1,000 gallon solids holding tank in each train. Biosolids are pumped from the solids holding tank to the first reactor in each train for RAS or to waste.

Solids do not move well down the 30° cone, making solids collection inefficient. Poor solids collection leads to excess gas production in the sludge blanket. Consequently, there is regular partial lifting of the sludge blanket to the surface. Floating biosolids carry over into the EFBs. Biosolids carryover is probably responsible for the increase in total nitrogen concentrations from the clarifier to the effluent (Figure 3-9). Biosolids are resettled by spraying. To prevent lifting of biosolids, the operator manually brushes

biosolids down the cone into the collection sump. Operational management of biosolids at the clarifier has been very time consuming.

Clarifier performance improved dramatically in the summer of 1999. In July and August of 1996, 1997, and 1998, no more than 50% of clarifier effluent TSS concentration were less than or equal to 10 mg/L. In contrast, in July and August 1999 all clarifier effluent TSS concentrations were less than or equal to 5 mg/L (Figure 3-10). Clarifier performance was significantly more stable in July and August 1999 than in July and August of the previous years. Prior to 1999, the best mean clarifier TSS effluent concentration for July and August was 11 mg/L with a standard deviation of 5 mg/L, but in July-August 1999 the mean clarifier effluent was 2.5 mg/L with a standard deviation of 1.1 mg/L (Table 3-2). Influent temperatures in the summer are greater than 20^{o} C. Warm water clarifier performance has typically been poor due to rising biosolids and occasional solids bulking episodes. The difference between clarifier performance in the summer of 1999 and other summers may be attributed to improved settleability of MLSS due to stable operation of the MLE process.

Conclusion

Clarifier effluent TSS concentrations recently are less than or equal to 5 mg/L. The improved performance may to be due to stable operation of the MLE process. The clarifier design is operationally time consuming.

Table 3-2. Summer clarifier performance in wastewater temperatures greater than 20^{o} C. Data are from grab samples tested at the in-house process laboratory.

TSS, mg/L	Clarifier	Effluent	
July-August 1999	**2.5**	**2.5**	**mean**
	1.1	0.8	σ
	14	14	n
July-August 1998	**11**	**3**	**mean**
	5	2	σ
	16	16	n
July-August 1997	**48**	**5**	**mean**
	122	5	σ
	15	16	n
July-August 1996	**14**	**2**	**mean**
	9	2	σ
	15	16	n

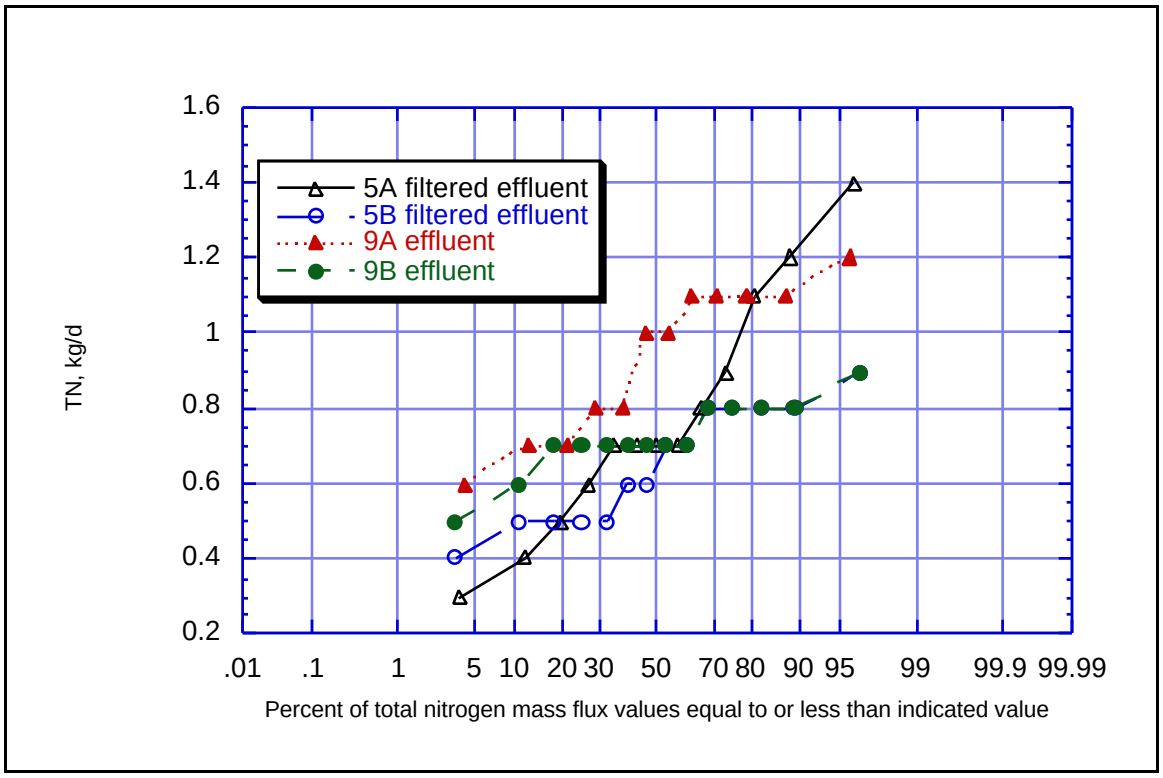

Figure 3-9. Increase in total nitrogen values from clarifier to final EFB July-August 1999. Design effluent is 1.5 kg/d.

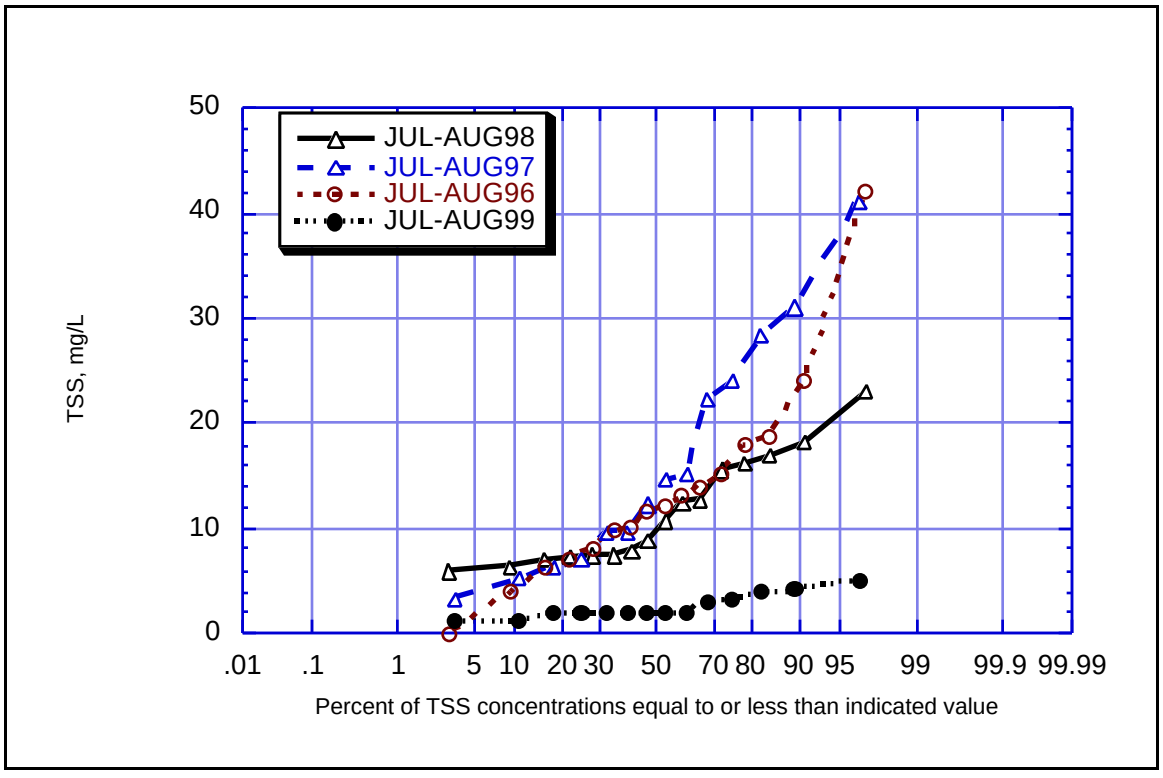

Figure 3-10. Clarifier supernatant TSS concentrations in July and August 1996-1999. All values plotted from dates specified.

3.4. Ecological Fluidized Beds

Summary: The EFBs were originally needed to nitrify to ammonia concentrations of 1 mg/L. Addition of RAS in August 1996 to the process diagram rendered them redundant for nitrification. Thereafter, the principal treatment function of the EFBs has been TSS removal.

Introduction

The EFBs are submerged, recirculating rock filters. Their design function is for nitrification when run as aerobic filters, denitrification when run as anoxic filters, and to remove TSS. In the current treatment configuration, EFBs are principally used for TSS polishing.

Performance

As nitrifying filters, EFBs have nitrified ammonia concentrations of up to 12 mg/L to non-detect levels (Figure 3-11). Addition of RAS in August 1996 significantly reduced the nitrifying demand on the EFBs.

As denitrifying filters, EFB performance has met design for 70% of weekly values with the addition of methanol to the anoxic EFB. An MLE (single stage) anoxic denitrification process, however, outperforms the denitrifying EFB.

The remaining design function of the EFB is removal of residual TSS from clarifier effluent (Table 3-3). Ninety percent of TSS samples from the first EFB1B are less than or equal to 10 mg/L from May 1996 to March 1999 (Figure 3-12). As documented above, clarifier performance was poor during this period. In the treatment current configuration, TSS loading of the post-clarifier EFB has been significantly less than in past years (Figure 3-10). Three EFBs are redundant to polish TSS effluent concentrations to less than or equal to 10 mg/L.

Media Performance

Two different sizes of lava rock were used in the EFBs. In EFB1A and EFB1B, the nominal diameter of the lava rock was 1.5 inch. The 1.5 inch media was nearly impervious to clogging. The other EFBs used 0.75 inch nominal diameter rock. During the period of methanol addition, the media bed in the second and third EFBs in each train suffered severe clogging caused by excessive biofilm growth. Fouling began within two months of the beginning of methanol addition. Within a month of the end of methanol addition, media bed head losses returned to pre-methanol addition levels.

The original depth of media was approximately ten feet in all EFBs. Three EFBs had approximately half of the media removed. There was no apparent change in treatment performance between a ten and five feet deep media bed.

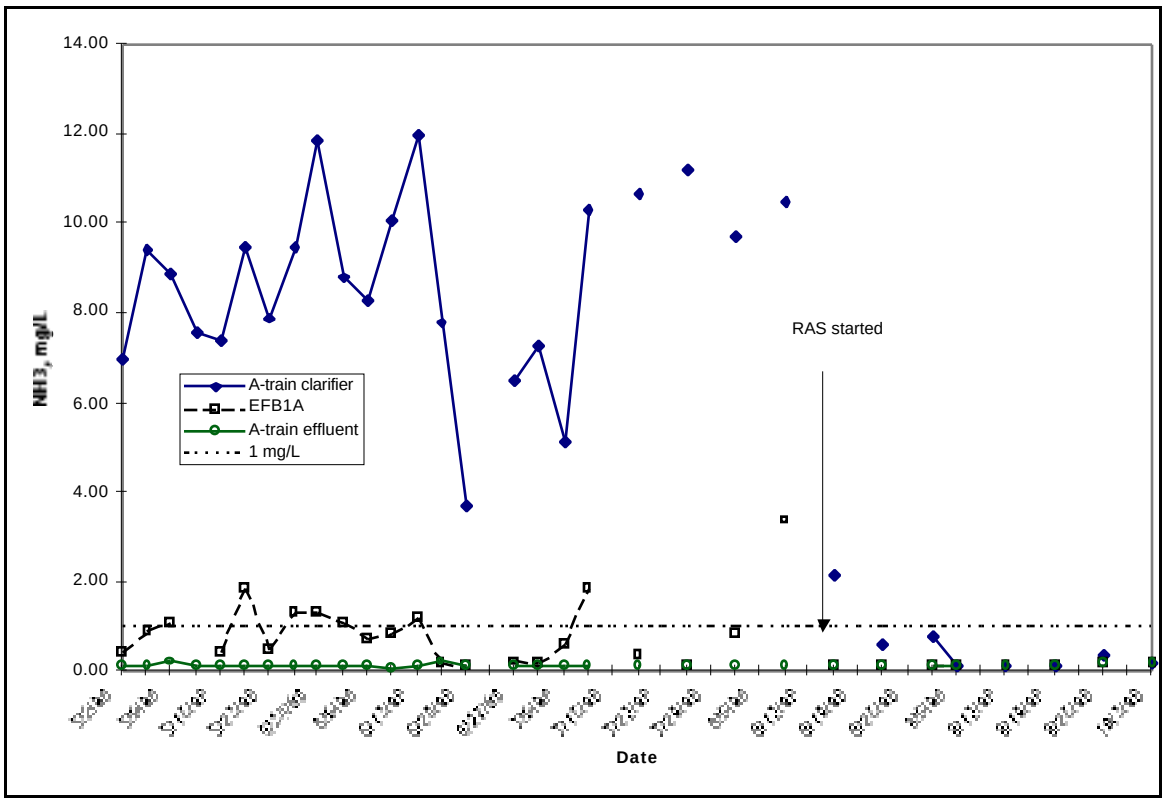

Figure 3-11. Ammonia profile from clarifier to effluent, May-October 1996**.**

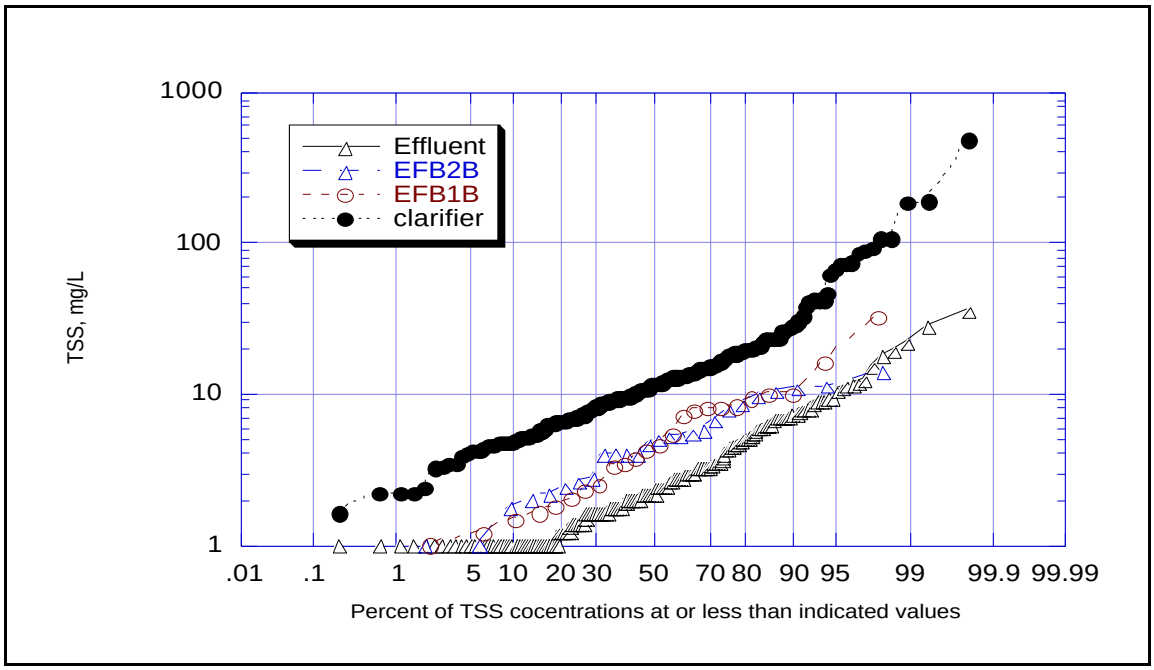

Figure 3-12. TSS profile from clarifier to effluent May 1996-March 1999. All samples are grabs tested in the in-house process lab. All samples plotted in period specified except for one noted in Table 3-3.

Table 3-3. Clarifier-EFB TSS profile, May 1998-March 1999. Data are from grab samples. Profile samples were not taken after March 1999. The data set has one day excluded from the profile series (clarifier: 28.3 mg/L, EFB1B: 431 mg/L, EFB2B: 46 mg/L, effluent: 4.2 mg/L).

Clarifier	EFB1B	EFB2B	EFB3B	TSS
19.4	6.4	5.5	3.5	**mean**, mg/L
38.4	6.6	3.5	4.1	σ, mg/L.
232	24	26	241	**n**

Conclusion

In the current AEES configuration the EFB provide TSS filtration, and limited back-up nitrification if a process upset in the aerated reactors send ammonia concentrations to the clarifier in excess of 1 mg/L.

3.5. Headworks

Summary: Treatment performance, data collection, and operations were compromised by the lack of headworks. The lack of headworks is peculiar to the Vermont AEES project.

Lack of a true headworks at the Vermont AEES has adversely affected operations by preventing consistent feed flow to the system and consistent flow split between the two treatment trains. The headworks arrangement would not be considered adequate for a stand-alone facility.

The headworks consist of a single submersible pump located at the end of the degritting channel in the City of South Burlington, Vermont Bartlett Bay Sewage Treatment Plant. The pump is located inside a screen for rag protection. It runs on a pin timer of 15 minute increments, set to reproduce the diurnal influent flow fluctuations of the Bartlett Bay treatment plant. The pump delivers influent at approximately 90 gpm (5.7 l/s) for a total flow of 80,000 gpd (303 m^3/d). Raw influent flow is split under pressure to discharge into the first reactor of each train.

The screen usually prevents rags from entering the influent pump, but clogs readily, limiting influent flow. Cleaning of the screen is done manually with a high pressure hose. Cost and logistical considerations have prevented installation of adequate screening.

Flows to the Vermont AEES split under pressure between the A and B-trains. The flow splitting arrangement is not standard in the wastewater treatment industry and has proven inadequate to achieve consistently equal flow splitting between the trains without significant, daily attention from the operating staff. Cost considerations have prevented installation of a standard gravity flow split arrangement.

Finally, the system uses simplex (single) pump stations for influent feed and effluent discharge. Duplex pumps stations are standard.

3.6. Process Controls

Summary: The Q-Com control system functions as designed, but is difficult to service. The vendor does not provide adequate technical support. In the past few years the price of programmable logic controllers (PLC) has dropped dramatically. A standard PLC would, today, be a better choice for a process control system. Process control of the B-train anoxic reactor is done with an ORP probe and controller

The automated control and monitoring system for the Vermont AEES is provided by Q-Com, a control system common to the greenhouse industry. Q-Com was chosen in 1994 as the control system because it was significantly less expensive than the programmable logic controllers (PLC) then available. Q-Com was also attractive because it is designed to control greenhouse functions such as ventilation and heating. At the time, Q-Com was a cost effective choice for a control system.

The Q-Com system is used to control clarifier biosolids pumping, EFB backflushing, and greenhouse controls. Clarifier biosolids pumping is run on a cycle timer function. Air-scour backflushing controls for the EFBs function as a sequential clock timer operating motorized valves. Once per day the backflushing sequence works as follows: air-lift recycle pumps off, air-scour on, air-scour off, a half-hour quiescent period, pumping of biosolids collected in the outer ring,, air-lift recycle pumps on. Greenhouse controls involve venting and heating governed by a thermostat.

The Q-Com system also is used for monitoring. Historical flow and temperature data can be displayed in graphical form.

There are two reasons why Q-Com would be a poor choice today for a control system. One, the price of PLCs has plummeted in the past few years. They are available from a variety of vendors and have great functional flexibility. On price alone a PLC is preferable to a Q-Com control system. Two, Q-Com vendor's level of support for hardware or software service has been unsatisfactory.

Several commercial AEES facilities now use PLCs for control systems. In very small facilities, cycle and clock timers offer adequate system control. Because of low cost and simplicity, PLCs have become increasingly common for small flow systems in general.

A Royce Instruments ORP probe and controller is used to control B-train anoxic reactor air cycling. Coarse bubble diffusers mix the outer compartment when the ORP is less

than +50 mV. At +50 mV air the controller shuts air supply. Air supply resumes at 0 mV. Performance of the ORP and controller has been excellent.

3.7. Blowers

Two Roots positive displacement rotary blowers provide aeration. Each blower delivers 165 cubic feet per minute at 12 pounds per square inch pressure and 9.1 brake horsepower. The treatment system air demand requires both blowers to be on line. Blower performance has been satisfactory.

3.8. Plants

Summary: Plants in the Vermont AEES grow on fixed racks. Plants species have been systematically evaluated to determine suitability for use in wastewater treatment. Many species have been found that produce long roots and thrive in wastewater. The depth of the aerated reactors is not optimal for plant root contribution to wastewater treatment.

Introduction

The section reports on the design basis, configuration, and management of plants as a components of the Vermont AEES. This section also describes the screening of plants for suitability in wastewater treatment. The treatment role of plants is described in Section 4.1, Parallel Comparison of Planted and Unplanted Aerated Reactors.

Plants are an integral part of the process design at the Vermont AEES. A significant, sustained effort has been made to optimize the physical arrangement of plants and to find species well adapted for use in wastewater treatment.

3.8.1. Physical Arrangement of Plants

Reactors 2 through 5 of the A and B-trains are planted. Plants grow on fixed racks that cover approximately 60% of the reactor surface (Figure 3-13). Each plant rack is baffled to prevent battering of roots by turbulence (Figure 3-14). Wastewater flows into each baffle, down through the root zone and out two large triangular openings at the bottom of each rack. Fixed plant racks permit the use of large, heavy plants that produce long, dense root systems.

The first reactor in each train had originally been planted. Plants were removed from the first reactor to permit the installation of a compost biofilter on top of reactor 1A, and to install the fabric media tower in reactor 1B.

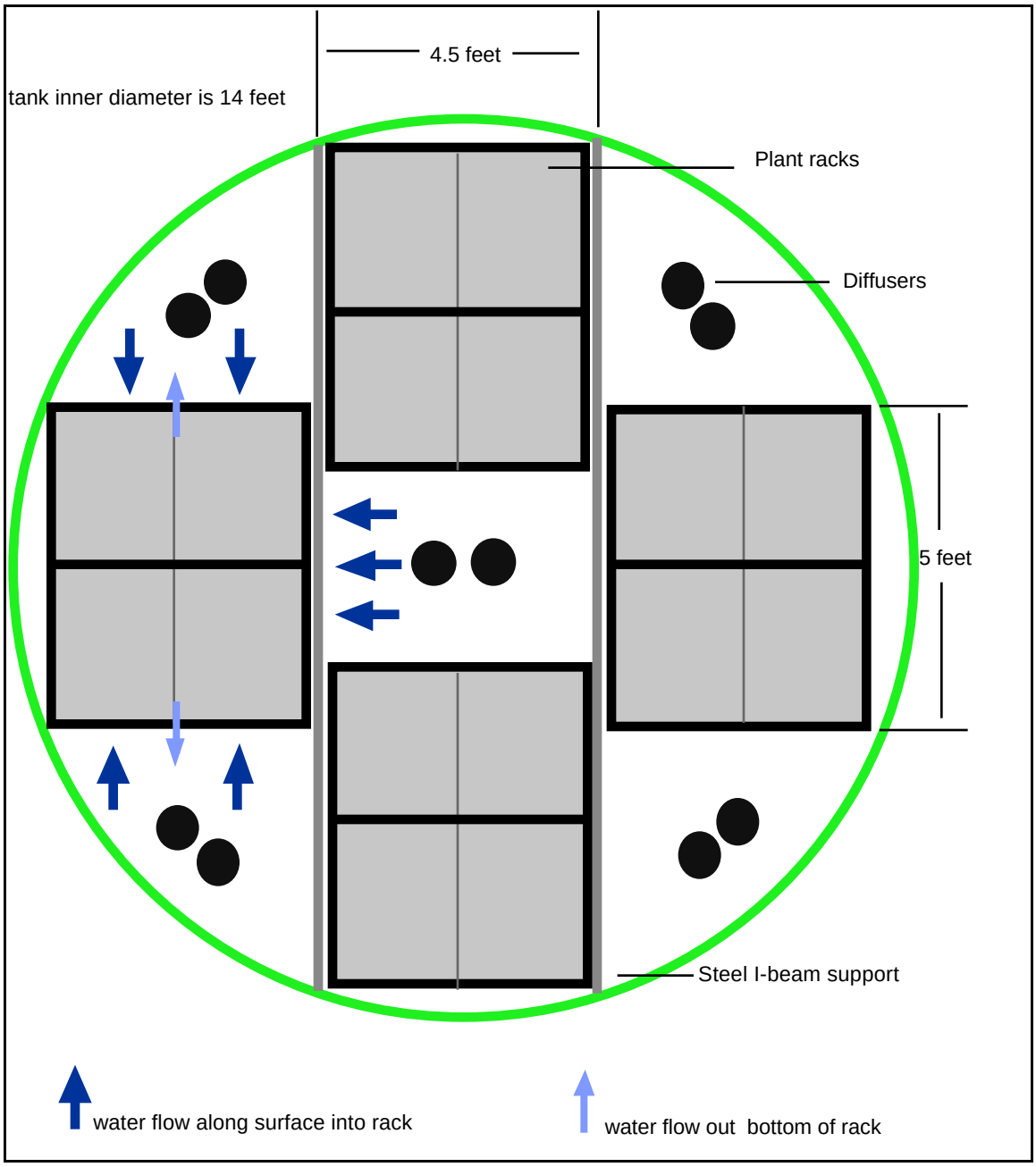

Figure 3-13. Plant rack arrangement on reactors 2-5 on A and B-trains. Plan view.

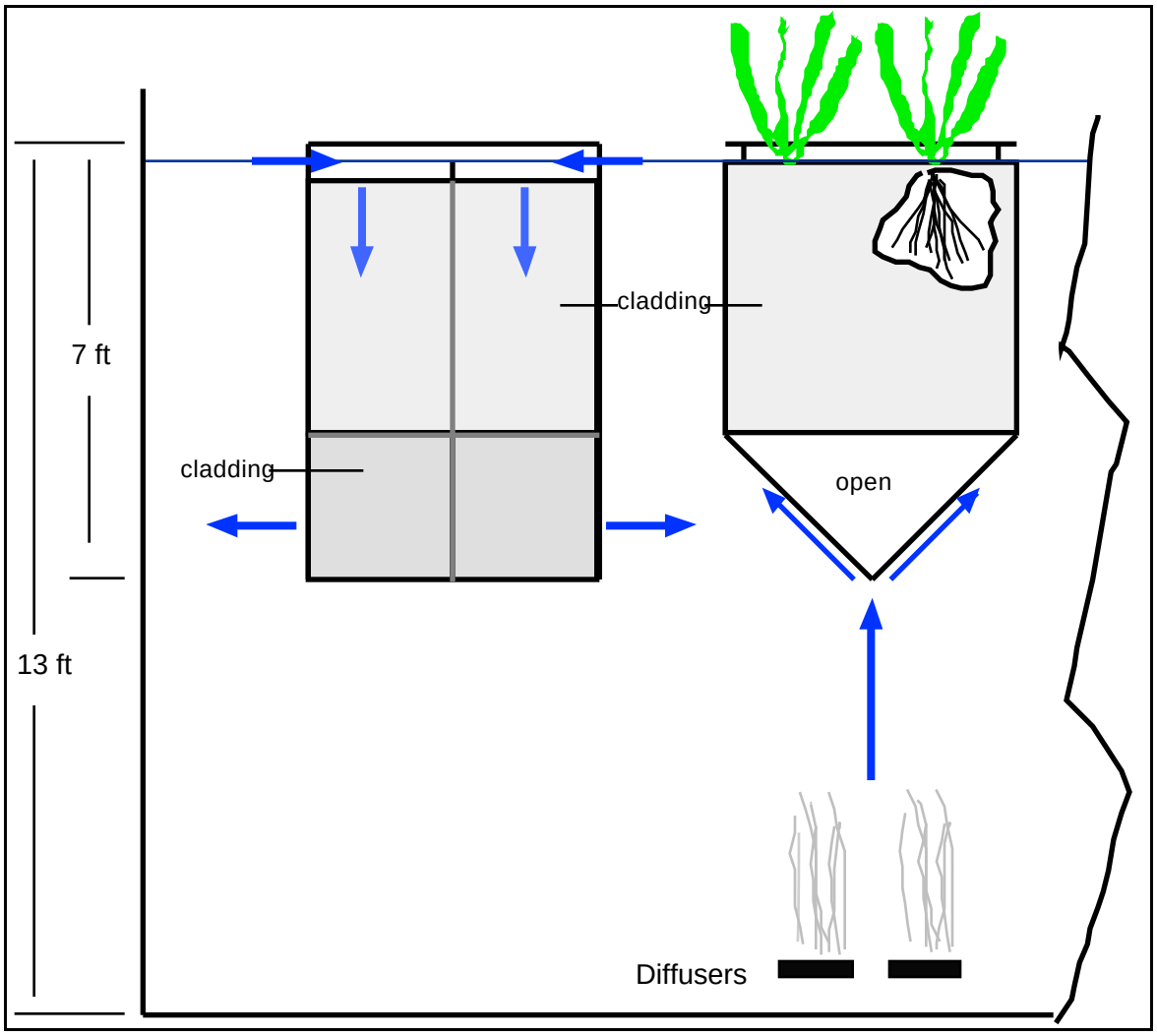

Figure 3-14. Flow and baffling schematic of individual plant racks. Front view (left), side view (right). Drawing not to scale. Cutaway (upper right) shows approximate position of roots in rack.

3.8.2. Design Basis of Plants

The design role of plants in the treatment process at the Vermont AEES is to provide a large surface area on the root mass for attachment of biofilms. Biofilms fixed on plant roots provide treatment in conjunction with MLSS. Observation of root biomass throughout the project has confirmed the formation of biofilms. The degree to which these biofilms contribute to treatment is unknown.

The actual contribution of plants to treatment at the Vermont AEES has been discussed at length at Vermont AEES project review meetings. Meeting participants have been the USEPA, The Massachusetts Foundation for Excellence in Marine and Polymer Science (MFEMPS) (project grantee and administrator), the technical advisory group (TAG) appointed by MFEMPS, Ocean Arks International (subcontractor to MFEMPS) and

Living Technologies, Inc. (subcontractor to MFEMPS). The consensus opinion of the attendees is that the plant root zone should substantially occupy approximately 20-30% of a well mixed reactor water column for the roots to play a primary role in wastewater treatment. The San Diego water hyacinth Aqua I-III treatment project has been cited by representatives of the USEPA and the TAG as a benchmark study establishing a reactor depth to root zone ratio[8].

The percent of root zone occupancy of each planted aerated reactor at the Vermont AEES is less than 20%. Reactor working depth is 13 feet (Figure 3-14). Plant roots grow to an average depth of two feet. Root zone occupancy would thus be 15% if the reactor surface were completely covered with plants. But plant racks occupy 60% of reactor surface, leaving approximately 9% of the treatment water column occupied by plant roots. It is likely, therefore, that the reactors in the Vermont AEES do not optimize use of plant roots for wastewater treatment.

3.8.3. Selection of Plants

Over 350 species of plants have been tested in the Vermont AEES. Placement of a species in categories A, B, or C is based upon a year-round performance evaluation by several criteria. Plants are observed/evaluated for a minimum of one year before being assigned to a given category. Testing of plants is usually replicated in several tanks. A year of observation is necessary because of the sharply defined seasonal environmental conditions at the Vermont AEES. The selection criteria, in order of importance, are listed below:

1) Root mass development comprised of length of penetration into the treatment water column and overall mass.

2) Fouler/stem growth, preferably evergreen year round.

3) Ease of vegetative propagation that permits economical replication elsewhere in the Vermont AEES.

4) Level of required maintenance, such as susceptibility to insect/disease, long-term, symptom-free leaf/stem appearance, pruning/debris removal requirement, and overgrowth.

5) Tolerance of 100% root submersion both at planting time (as immature specimens) and at maturity.

6) Tolerance to turbulence from diffusers.

7) General plant health/vigor in early tanks, not necessarily including root growth.

8) Tolerance to high summertime greenhouse temperatures. The South Burlington, Vermont greenhouse is not equipped with active ventilation technology, therefore

[8] **Total Resource Recovery Project, Final Report**. 1996. Prepared by the Western Consortium for Public Health with EOA, Inc.

temperatures can exceed 38°C in July and August. Plants are highly stressed by sustained high temperatures.

9) Low winter greenhouse temperature-tolerance to 10°C air temperatures and influent temperatures less than 15°C.

10) Four season and long-term aesthetic appeal.

11) Potential economic value.

12) Ability to survive planting as immature individuals.

13) Shoot growth on a year-round basis.

Category A plant species are the top performers, recommended for further use in the Vermont AEES (Table 3-4). Category B plants are strong in some categories, but suffer from an unacceptable flaw, such as intolerance to cold temperatures. Some Category B plants undergo continued evaluation. Category C plants are failures and removed from the system (Appendix M).

3.8.4. Management of Plants

Integrated pest management (IPM) and regular harvesting have been the principle tools of horticultural management at the Vermont AEES facility. The scope of this report allows only a brief description of the principle IPM practices employed to date.

Regular pruning (removal) of dead, diseased, or overgrown plant tissue has been a key IPM practice because it reduces the incidence of pest insects and disease. Decomposing plant matter can also serve as a habitat for harmful insects and disease organisms, and is best removed and composted.

There have been four primary pests encountered at the Vermont AEES: aphids, spider mites, thrips, and white flies. Each pest appears to favor specific plant types, although adjoining species may periodically be infested when population density reaches a level that can no longer be supported on the original 'target' plant. Summertime insect management is more critical than in winter. In winter, thrips, and spider mites generally disappear and aphid and white fly infestations drop dramatically.

Control of pests by synthetic chemical insecticides is unnecessary and not permitted at the Vermont AEES. Frequent inspection of plants for pests allows for timely intervention to deal with infestations before they become widespread. A forceful spray of water from a hose to dislodge insects from affected plant parts has been a highly effective tool to keep infestations in check. Removal of affected leaves also is an effective means of controlling pests and stimulates plant growth throughout the growing season. In some cases of severe and persistent infestation, insecticidal soap or other natural insecticide, such as bacteria or fungal insecticides, have proven effective.

Table 3-4. Category A plants.

Category A. Vascular Plants (Ranked alphabetically by scientific binomial)
Acorus calamus - Sweet Flag (Acoraceae)(tentative 1999 introduction)
Alnus glutinosa - European Alder (Betulaceae) (tentative 1999 introduction)
Alocasia odora - Elephant Ear (Araceae - Philodendron Family)
Arundo donax 'Variegata' - Variegated Giant Reed (Gramineae - Grass Family)
Bacopa lenagera 'Variegata' - Bacopa (Scrophulariaceae) (tentative 1999 introduction)
Bidens frondosa - Bur Marigold (Asteraceae - Aster Family)
Caltha palustris - Marsh Marigold (Ranunculaceae - Buttercup Family)
Canna x *generalis* (several cultivars, esp. green-leaf forms) (Cannaceae - Canna Family)
Carex species (Cyperaceae - Sedge Family)
Colocasia esculenta - Taro (Araceae - Philodendron Family)
Colocasia esculenta 'Euchlora' - Purple Taro (Araceae - Philodendron Family)
Colocasia gigantea (aka *C. indica*) - Giant Taro (Araceae - Philodendron Family)
Cyperus alternifolius - Umbrella Plant (Cyperaceae - Sedge Family)
Cyperus gigantius (aka *C. papyrus* var. *gigantius*) - Giant Papyrus (Cyperaceae - Sedge Family)
Cyperus papyrus - Egyptian Paper Reed or Papyrus (Cyperaceae - Sedge Family)
Cyperus papyrus 'Nanus' - Dwarf Papyrus (Cyperaceae - Sedge Family)
Decodon verticillatus - Water Willow (Lythraceae - Loosestrife Family)
Hedychium coronarium - White Butterfly Ginger (Zingiberaceae - Ginger Family)
Hibiscus coccineus - Maple Leaf Mallow (Malvaceae - Mallow Family)
Hibiscus moscheutos 'Disco Belle Mix' - Common Rose Mallow (Malvaceae - Mallow Family)
Hydrocotyle verticillata - Water Pennywort (Umbelliferae or Apiaceae - Parsley Family)
Iris pseudacorus - Yellow Flag (Iridaceae - Iris Family)
Juncus effusus - Softstem Bulrush (Juncaceae - Rush Family)
Leersia oryzoides - Cutgrass (Gramineae)
Oenanthe javanica (aka *O. japonica*) - Water Celery (Umbelliferae or Apiaceae - Parsley Family)
Oenanthe javanica 'Flamingo' - Variegated Water Celery (Umbelliferae - Parsley Family)
Phragmites australis (synonym *P. communis*)- Common Reed (Gramineae - Grass Family)
Phragmites australis 'Aurea' - Variegated Common Reed (Gramineae - Grass Family)
Phragmites australis 'Candy Stripe' - Variegated Common Reed (Gramineae - Grass Family)
Sagittaria latifolia - Arrowhead, Duck Potato (Alismataceae)(tentative 1999 introduction)
Salix x cotettii - Banker's Willow (Salicaceae - Willow Family)
Salix lucida ssp. *lasiandra* - Pacific Willow (Salicaceae - Willow Family)
Salix 'Yellow Willow' (Salicaceae - Willow Family)
Saururus cernuus - Lizard's Tail (Saururaceae - Lizard's-tail Family)
Scirpus cyperinus - Woolly Sedge (Cyperaceae - Sedge Family)
Scirpus validus - Great Bulrush (Cyperaceae - Sedge Family)
Scirpus 'Zebrinus' - Zebra Rush (Cyperaceae - Sedge Family)
Stachys palustris - Hedge-nettle (Lamiaceae - Mint Family)
Taxodium distichum - Bald Cyperus (Cupressaceae - Cedar Family)
Thalia geniculata form *ruminoides* - Violet Stem (Marantaceae - Arrowroot Family)
Wedelia trilobata - Water Zinnia (Asteraceae - Aster Family)
Zantedeschia aethiopica - Calla Lily (Araceae - Philodendron Family)
Zantedeschia aethiopica 'Hercules' - Giant Calla Lily (Araceae - Philodendron Family)
Zizania latifolia - Perennial or Manchurian Rice (Gramineae - Grass Family)

Biological control using pest predators has been a viable option for persistent problems that have not yielded to methods outlined above. Their expense, however, should be taken into account before relying on their regular use. Biological controls have been used in consultation with an IPM specialist from the University of Vermont.

Species diversity of planting appears to limit the spread of pest infestations. The natural tendency of plants to compete for light and growing space works against species diversity. Management of plants has had to balance the benefits of diversity with the effort required to maintain it.

3.9. Macrofauna

Macrofauna are animals that can be seen by the unaided human eye. Organisms found in the Vermont AEES included in this category are fish, snails, and adult crustacean plankton such as copepods and amphipods. There has been no systematic study of the role of macrofauna in wastewater treatment at the Vermont AEES, but limited observations by project personnel are worth noting.

Fish survive only in the EFBs. Several attempts were made to introduce carp into the clarifiers to see if they could survive on biosolids. Conditions in the clarifier have resulted in the death of the carp within a few weeks of their introduction. During low oxygen events, rising biosolids at the surface suffocate carp attempting to gulp air.

Snails densely cover the wetted surfaces of the clarifier, the EFBs and plants found within the EFBs. Snails also can be found in large numbers covering plant roots in the fourth and fifth reactors in the A and B-train. Populations of snails are greatest during the summer and fall. Snails graze on biofilms. The contribution of snails to treatment is not known.

Large populations of copepods inhabit the rock medium in the EFBs. Population density of adult copepods has not been quantified, but is obviously very high. Amphipods also are very numerous in plant roots. Both are known to graze on suspended bacteria and biofilms, but the contribution of copepods and amphipods to treatment is not known.

3.10. Conclusion to System Component Analysis

In the current treatment configuration, $CBOD_5$, COD, TKN, NH_3, and NO_3 concentrations meet design effluent standards at the fourth aerated reactor. Treatment performance in the aerated reactors is not a function of dilute influent. Standard CFSTR model equations that closely fit treatment data produce design effluent concentrations in or by the fourth aerated reactor at design concentrations and flows. Treatment to design effluent criteria by the fourth aerated reactor opens the possibility of reducing the hydraulic retention time prior to the clarifier by increasing influent loading.

Clarifier performance has been marginal overall at design flows, but improved dramatically in the current configuration. The current clarifier design has been abandoned. As currently sized, the clarifiers cannot take significantly increased flows, and thus prevent higher influent loading.

In the current treatment configuration, primary EFB function is limited to TSS removal. A potential secondary function of the EFB is nitrification if the effluent from the clarifier exceeds design criteria. Such exceedences, however, are very rare.

Influent flow interruptions and imbalances between the two treatment trains were caused by an incomplete headworks.

Systemic investigation of plant species suitable for wastewater treatment has discovered many that produce roots of at least two feet in length and thrive in conditions at the Vermont AEES. However, the depth of the reactors is not optimal for significant plant root contribution to wastewater treatment.

The main system components have performed close to design. Completion of design nutrient removal by the fourth reactor, and greatly contracted EFB nutrient treatment function, suggest that the overall system HRT could be significantly reduced.

4. TEST TRAIN STUDIES

Two experiments were conducted with the test train: One was a comparative investigation of the effects of plants on wastewater treatment[9]; the other was an investigation of media for EFBs. Use of a small, separate experimental train allowed flexible research because, unlike the A and B-trains, there was no set effluent performance. Investigation of an optimal media for EFB designs also required pilot scale studies in the test train due the high cost and treatment uncertainty associated with media experimentation in the main train EFBs.

4.1. PARALLEL COMPARISON OF PLANTED AND UNPLANTED AERATED REACTORS

Prior to studies described below, there had been no long-term parallel comparison of the treatment performance of planted and unplanted reactors similar in design to the Vermont AEES. A six week long parallel treatment comparison of planted and unplanted reactors had been conducted at the Frederick, Maryland AEES facility in 1994. Although the study was well designed, it was statistically confounded by external factors while it was conducted. In particular, the study was launched during a system upset that the operators failed to identify in a timely fashion. Thus, the treatment role, or potential treatment role, of plants in AEES technology remained unsubstantiated by rigorous engineering standards.

The purpose of this test train study was to determine treatment differences between planted aerated reactors and unplanted aerated reactors. The question of optimal basin geometry of planted reactors was not addressed by this study. Because of the differences in the scale of reactor geometry between the test train reactors and the main train reactors, this study could not quantify the treatment role of plants in the main trains.

4.1.1. Treatment diagram

Six aerated reactors were divided into two parallel treatment trains (Figure 4-1). Each train consists of three aerated reactors in series. The working volume of each reactor is approximately 1.2 m^3 (330 gallons). Approximately 7.2 m^3/d (2,000 gpd) of raw influent is split equally between the first reactors of each train. The HRT in each train is approximately one day. Depth and diameter of each reactor is 1.1 meters (3.5 feet) and 1.2 meters (4 feet), respectively.

Delivery of the design flow to the test train had been problematic. The test train force main branches off of the 3 inch A-train force main. The effective diameter of the test rain force main is 1 inch. When the influent pump is delivering design flow to the main trains (approximately 90 gpm) there is sufficient pressure to deliver design flows to the test

[9] Denitrification using endogenous carbon sources was funded under the category of test train experiments, but reported above with other main-train analyses for narrative clarity. The endogenous carbon study is presented in Section 2.9 as Phases 3 through 5 of the evolution of denitrification design.

train. Clogging of the influent pump screen that reduces flows to the main train then severely reduces or stops flow to the test train. Operation of the test train has often been challenged by erratic influent flows. Reliable flows could be established, but only at an level of operational attention that was unsustainable in the long term. By the time that experiments reported below were completed, the project ended, leaving no time for other experiments planned for the test train.

Flows have been measured and adjusted daily by a bucket and stopwatch method. The flow split between the planted and unplanted lines has consistently been very close with an average difference of 0.4 % (σ = 3.4 %).

4.1.2. Experimental protocol

To evaluate the contribution of plants to treatment, two parallel trains were operated with equal influent flows. One train was planted, the other left unplanted. Return activated sludge went to both trains, manually from the A-train (main train) clarifier in 1997 and, starting in June 1998, automatically from clarifiers dedicated to each of the planted and unplanted trains. The MLSS concentration was maintained equally in both trains.

Treatment performance was evaluated for COD, NH_3, and NO_3. Additionally, $CBOD_5$, TKN and TP were tested in July and August of 1999. The in-house laboratory processed all samples except those in July and August 1999, which were sent to a certified contract laboratory.

Data was gathered using composite samplers on the third reactor in each series and from grab samples from the other reactors. Effluent data samples were from the third reactor in each train. All samples were filtered prior to analysis. Samples were accepted for analysis if influent flows were between 3 and 4.5 m^3/d (800 and 1200 gpd) for the previous 24 hours.

The September-December 1997 run compared plants + RAS with RAS only. The 1998-99 run compared plants + RAS vs. RAS + fixed film media. Approximately five linear meters (1 m x 5 m) of a buoyant, polypropylene fabric media (Aquamat®) was distributed between the three unplanted reactors. According to the manufacturer, the effective surface area of the Aquamat® is forty times that of the planar surface area. The manufacturer's claims appear excessive, but roughness of the fabric undoubtedly significantly increases the effective surface area of the fabric medium.

Use of fixed film media was integral to this study to ensure operational stability of the unplanted train. Even with automated RAS, the small size of the test train left it vulnerable to biosolids wash-out. An experienced wastewater design engineer would not design an activated sludge treatment system at the micro-scale of the test train precisely

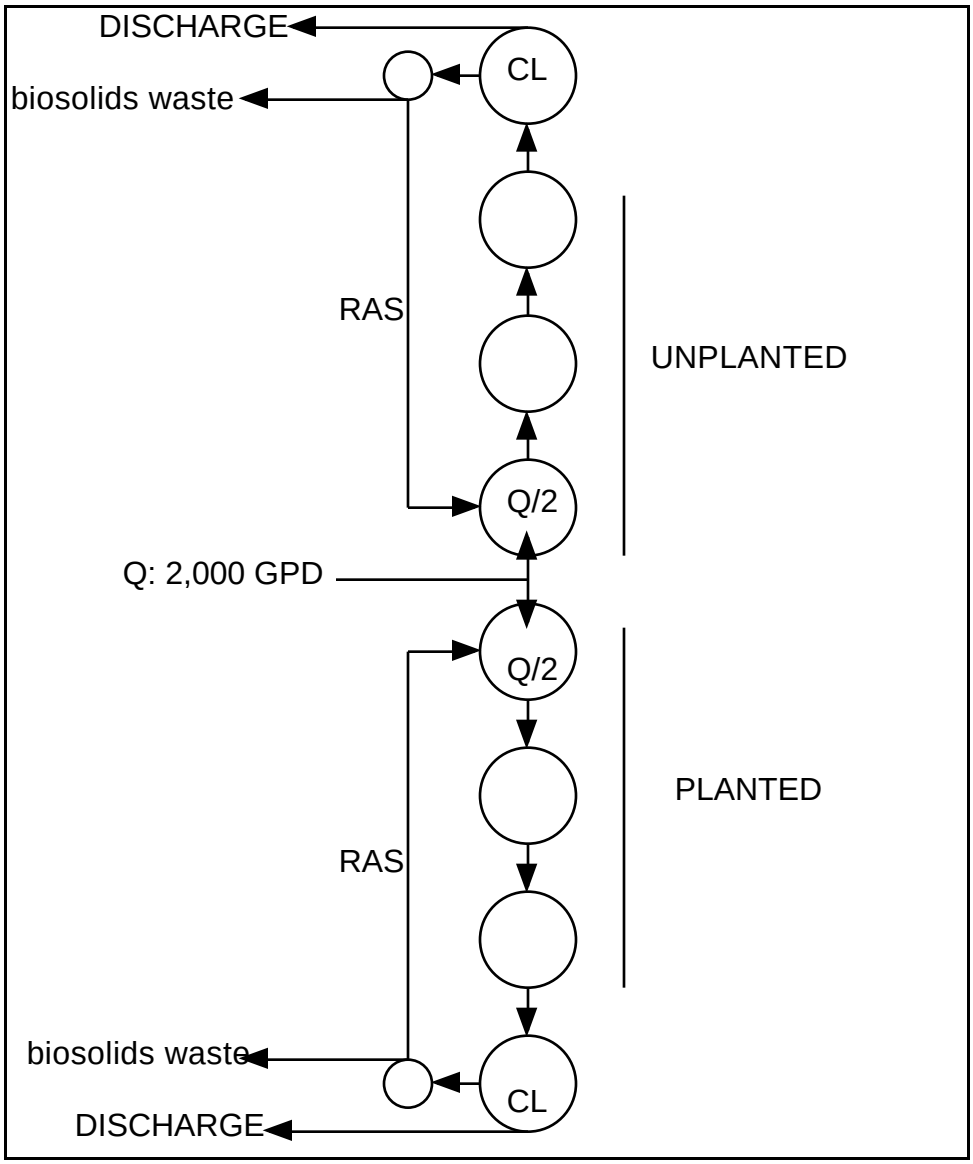

Figure 4-1. Test train treatment schematic.

because of that vulnerability to wash-out. A "straw man" comparison of a planted treatment train to an inherently poor RAS system would not have been useful. Addition of the Aquamat® media was designed to provide sufficient standing biomass to continue treatment in the event of a transient biosolids wash-out condition[10].

4.1.3. Results and Analysis

Results of the 1997 test train experiments showed a clear advantage of the planted line over the unplanted line for COD removal (Figure 4-2). Treatment performance of the

[10] Two other comparative studies originally planned were not conducted. One was a continuation of plants + RAS vs. RAS only. It was not executed for reasons stated above. The other was comparison of plants vs. artificial media only. Frequent flow interruptions to the test train prevented execution of the latter experiment.

planted line was considerably more stable than that of the unplanted line. The ratio of the COD P_{20} to P_{80} values was 0.7 for the planted line and 0.4 for the unplanted line.

During the first half of 1997 test train operations, RAS was transferred once per day from the A-train clarifier to both the planted and unplanted trains. In the latter half, RAS was transferred twice per day. For a treatment system based on RAS alone, a daily or twice daily recycle dose of biosolids is insufficient for stable COD removal performance. In contrast, the planted train exhibited stable treatment under the same operating conditions.

Even though the sharp treatment performance differences exhibited in the 1997 experiment were significant, the comparison had drawbacks. Return of activated sludge once or twice per day allowed biosolids washout in the unplanted train. Furthermore, even with regular return of biosolids, the small volume of the reactors in the test train still would leave them vulnerable to biosolids washout. Continued comparison required design and operational refinements of the test train.

The 1998-1999 experiments addressed the design and operational drawbacks to the 1997 experiment. Modifications to the test train provided clarifiers dedicated to each of the planted and unplanted line. Automation of RAS reduced variability of MLSS that degraded treatment performance in the unplanted train. Additionally, Approximately 1.7

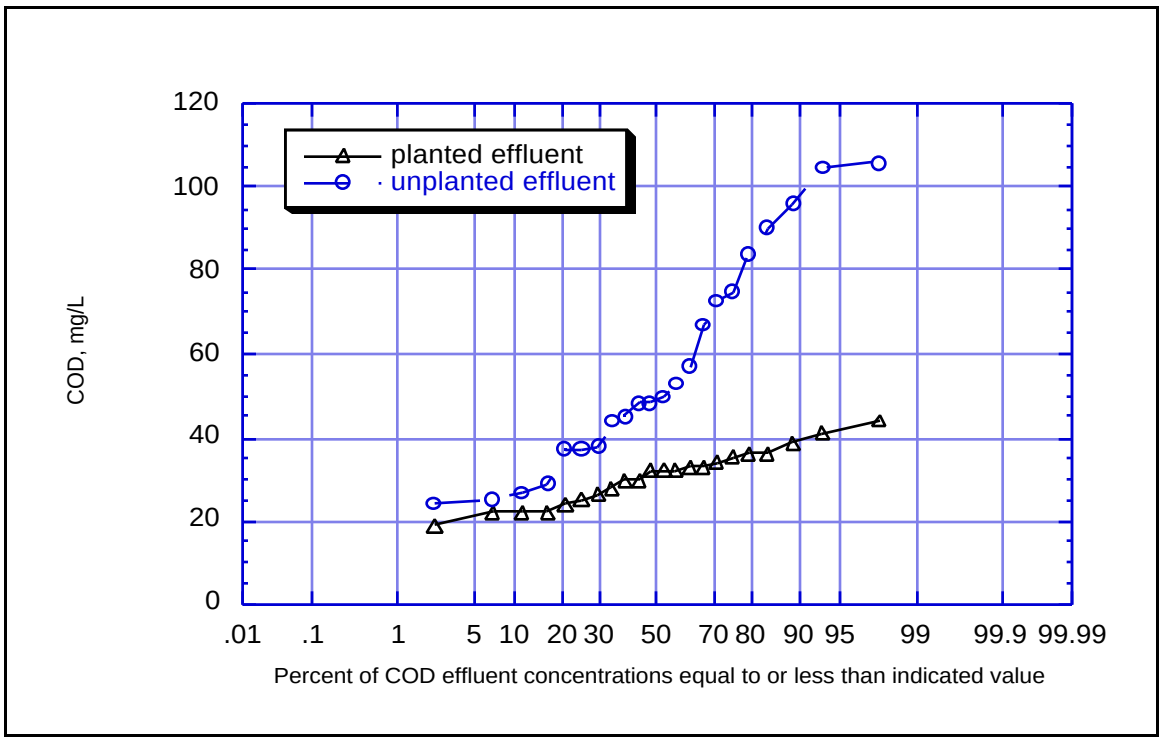

Figure 4-2. Effluent COD values of planted versus unplanted test train lines September 17-December 8, 1997.

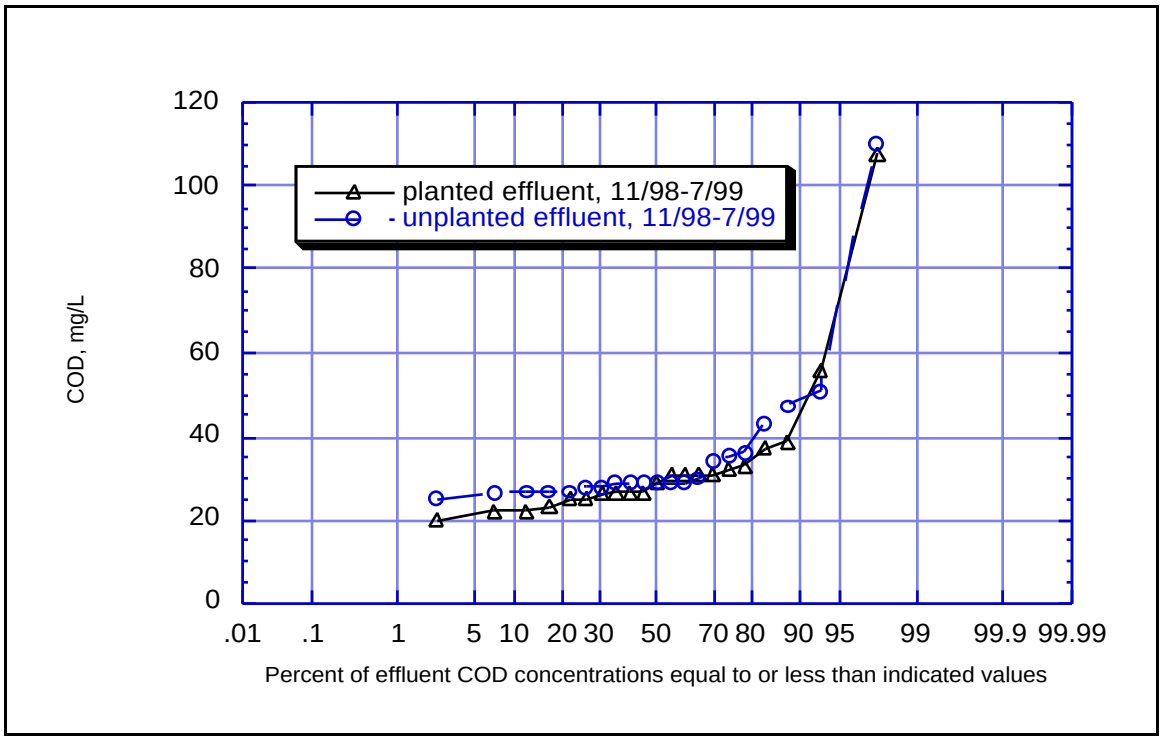

Figure 4-3. Effluent COD values of planted versus unplanted test train lines November 1998-July 1999.

meters of buoyant fabric medium (Aquamat®) was added to each of the unplanted reactors as an analog to plant roots. The nominal surface area of 1.7 meters of the medium is approximately 5 m^2, counting area exposed by fringe cuts. Using a roughness factor of 4, which is 10% of the manufacturer's value, the effective surface area of media may be as much as 20 m^2 per reactor in the unplanted train.

Under the revised design and operating conditions, COD treatment performance was essentially the same in both the planted and unplanted treatment trains (Figure 4-3). The P_{20} to P_{80} ratio for both trains was 0.7. The 1998-1999 treatment performance of the planted train did not change from the 1997 demonstration. The lack of change is significant because RAS had been automated for the 1998-1999 experiment. In the unplanted train, the combination of media and automated RAS brought the treatment performance up to that of the planted train.

Total nitrogen removal performance in the planted train was superior to that of the unplanted train. Until July 1999, no TKN test had been conducted on test train wastewater. If the organic fraction of TKN is assumed to be completely ammoniafied by the third reactor in each train, then the sum of ammonia and nitrate concentrations can serve as an approximation for total nitrogen. Approximate total nitrogen removal was significantly superior in the planted train (Figure 4-4). Results for true total nitrogen removal in July of 1999 confirmed the difference in treatment performance between the two trains (Figure 4-5).

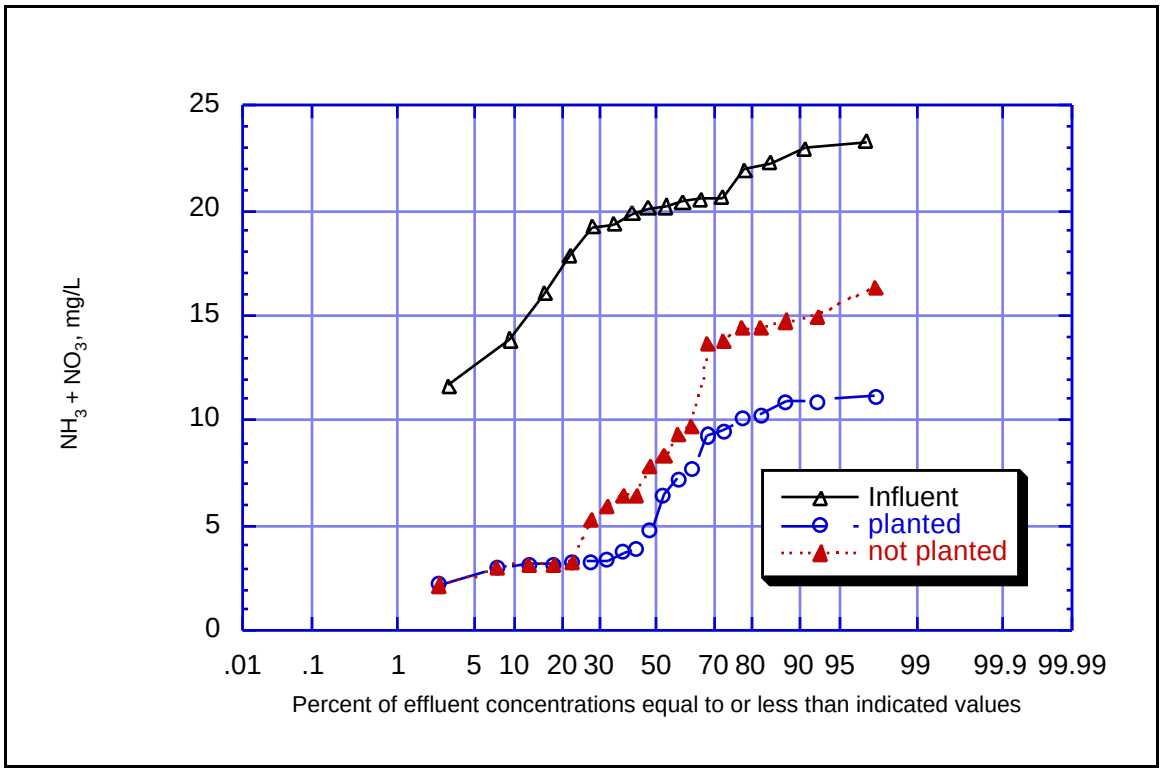

Figure 4-4. Combined nitrification and denitrification ($NH_3 + NO_3$) performance of planted and unplanted test train lines, August 1998-May 1999.

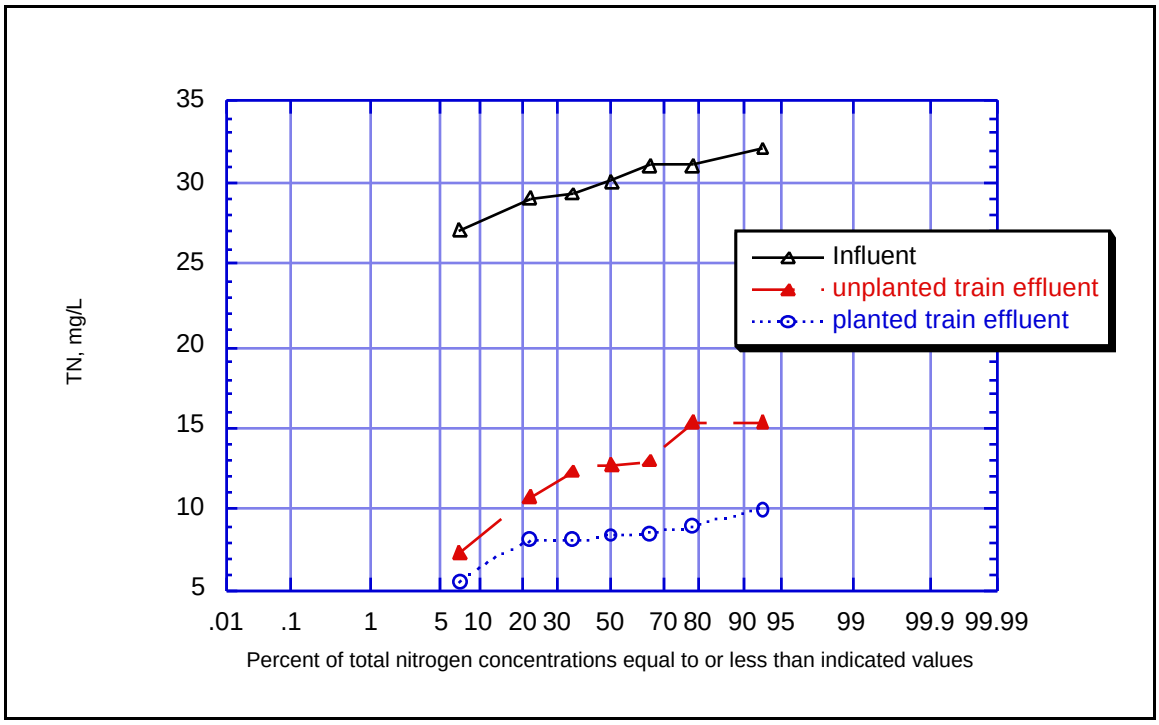

Figure 4-5. Total nitrogen treatment performance of planted and unplanted test train lines, July 1999.

No significant differences in treatment performance for $CBOD_5$ and total phosphorous removal were noted between the planted and unplanted trains.

No mass analysis is available for the test train. Without a flow meter on the test train force main, there has been no way to accurately measure flow. The bucket and stop watch method is accurate only with reliable influent delivery. Erratic flow delivery, caused by clogging of the influent strainer, makes flows set by bucket and stop watch only a rough approximation of average daily flows. A large number of unstable or unknown influent daily flow totals has made mass flux performance analysis unfeasible.

One unexpected difference between the planted and unplanted trains has emerged. The operator had continual difficulty in raising MLSS concentrations in the planted train to equal those in the planted train. Experimental protocol required that the MLSS concentration be maintained equally in the two trains. The reason for the difficulty was that the planted train appears to have produced few biosolids. Evaluation of test train operations logs revealed that from July 15 to August 12, 1999 less than 20 gallons of biosolids were wasted from the planted train whereas greater than 250 gallons were wasted from the unplanted train. Any inaccuracies in the logs are expected to be insignificant and relatively equal between the two treatment trains. Evaluation of biosolids production was not part of the experimental protocol, so no more quantitative information is available. The mechanism for the attenuation of biosolids concentrations in the planted train is unknown at this point. Biosolids do appear to be retained in roots, but accumulate only to a certain point. The root structures are densely covered with snails and other invertebrates that apparently graze on biosolids. More research is required to investigate the apparent biosolids digestion observed in the planted train.

Finally, plant roots measured in the third planted reactor were approximately 2 feet in length.

4.1.4. Conclusions to Parallel Comparison of Planted and Unplanted Reactors

Chemical oxygen demand removal performance by the planted trains did not change between the 1997 and 1998-99 operation of the test train. In contrast, COD removal performance of the unplanted line changed dramatically in 1998 with the automation of RAS and installation of fabric media. In 1997, the COD removal performance of the planted train was superior to the unplanted train. After 1998, COD removal performance was essentially equal in both trains. There is no data to differentiate the effects of automated RAS and fabric media on COD removal performance in the unplanted train. It can be concluded, therefore, that to equal the performance of the planted train the unplanted train required automated RAS, a significant quantity of a fabric medium, or both.

Total nitrogen removal performance in the planted train was consistently and significantly superior to the unplanted train.

Test train comparison of treatment performance cannot be directly related to the contribution of plants to treatment in the main trains A and B. The ratio of plant root surface area to reactor volume, although not quantified, is undoubtedly much higher in the test train than in the main trains. In the planted test train, plants cover the reactor surface completely and wastewater circulates up against the roots. In the main train, plants cover 60% of the surface and wastewater flows down through the roots. These differences of geometry and flow prevent direct comparison and leave unanswered the question of the contribution of plants to treatment in the main trains.

The test train experiment does establish some scale-specific contributions of plants to treatment. Root of two feet in length in the planted reactors of 3.5 feet in depth more than satisfies the consensus opinion described earlier that roots must occupy 20-30% of the reactor water column to significantly contribute to wastewater treatment. It is probable that the contribution of plant roots to treatment observed in the test train would also be observed in somewhat deeper reactors. Two feet long roots would occupy approximately 33% of a reactor with a water depth of six feet.

4.2. TEST TRAIN EFB MEDIA STUDY

The first generation of EFBs used a pumice media. Pumice proved to be too friable for use, eventually eroding into sand after repeated air-scour backflushing. Lava rock, a much more robust, lightweight vesicular basalt, was substituted for pumice. The lava rock, unlike pumice, is not buoyant. Fluidization in ecological fluidized beds is meant to occur when air scour backflush causes the media to sink and mix. Without a buoyant media there can be no fluidization. An EFB with lava rock media is simply a submerged, recirculating tricking filter. The test train was used to examine other buoyant media for future EFB designs.

4.2.1. Treatment Diagram

The treatment role of EFBs is post-clarifier filtration. Treatment may be sufficiently complete most of the time to obviate the need for EFBs. In a tertiary treatment system, however, variation in treatment efficiency upstream of the clarifier will occasionally require effective post-clarifier filtration. An important test of a filter media is its capacity to withstand periods of particulate or nutrient loading. Filter media must resist short-term fouling and must scour clean effectively by backflushing. The EFB media tested thus had to be subjected to sufficient loading to determine its treatment capacity and resistance to fouling.

The influent source for the test train EFBs was clarified wastewater from reactor 2A. A 24-inch diameter, eleven feet-long, stilling well was installed in reactor 2A. From the top of the stilling well, a submersible pump provides influent to the test train splitter box at

an average rate of 3,700 gallons per day. From the splitter box, influent flow is distributed equally to each of the three EFBs (Figure 4-6). Influent entering each EFB is pumped by air lift into the top central compartment containing the media. Influent then flows down through the media into the outer ring. From the outer ring wastewater is recirculated by air lift back into the media compartment. An outfall governs discharge of wastewater from each EFB.

The working volume of each EFB is approximately 1.2 m^3 (330 gallons). The media compartment volume is approximately 14 cubic feet. Approximately 9 cubic feet of media were placed in each reactor.

4.2.2. Experimental protocol

Three different types of media were tested. In EFB1, the media was comprised of balls of polypropylene netting approximately three inches in diameter. The polypropylene balls are the available in grocery stores as pot scrubbers, hence the name "scrubbies".

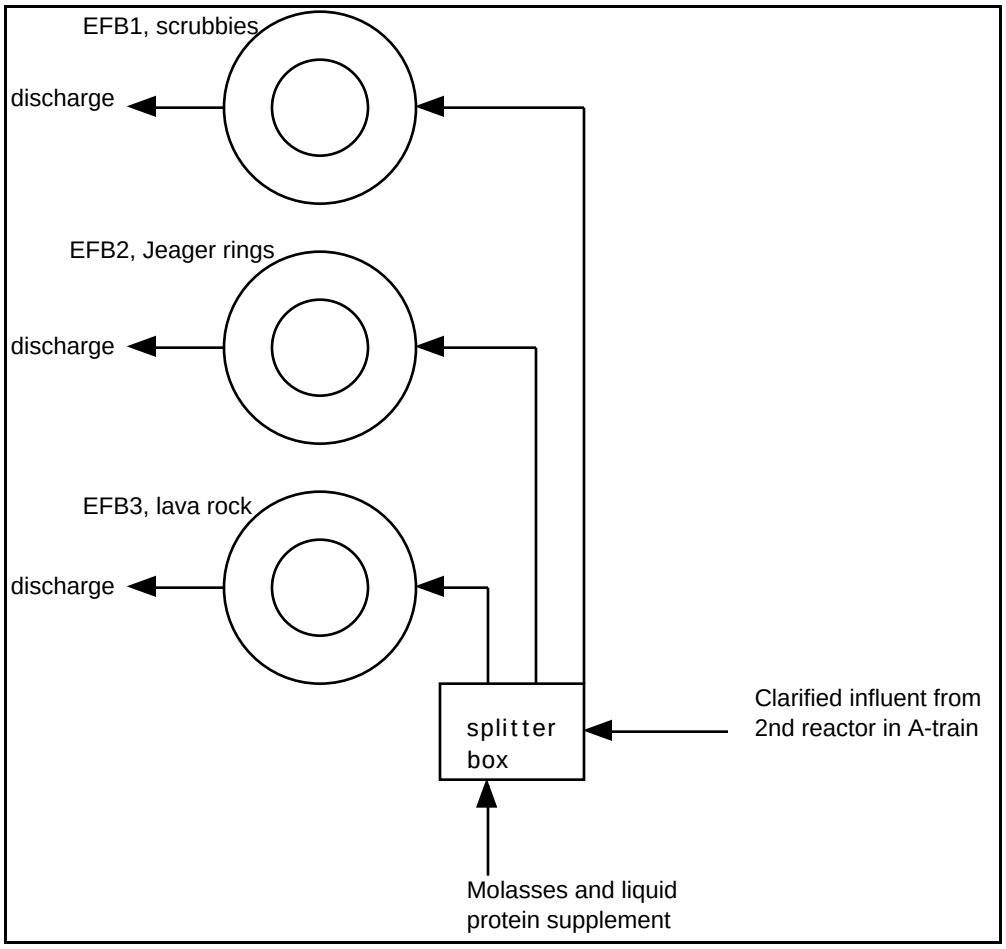

Figure 4-6. Test train EFB media study process schematic.

Cylindrical one half inch diameter rings of extruded polypropylene were placed in EFB2. The rings are stripping tower, random pack media marketed by Jaeger Products, Inc. Lava rock (1.5 inch diameter) was used in EFB3 as a control.

Both EFB treatment performance and media fouling (head loss) characteristics were the focus of this study. Treatment performance was measured by influent and effluent wastewater chemistry. Parameters tested were $CBOD_5$, TSS, TN, TKN, NH_3, and NO_3. A composite sampler at the splitter box provided influent samples. Composite samplers at each EFB provided effluent samples. All samples were tested in a certified, contract laboratory.

Influent strength was augmented by the addition of Mol Mix, a molasses and liquid protein dairy cow feed supplement. Mol Mix was metered into the splitter box at a rate of approximately 1 mL/minute. The $CBOD_5$ strength of Mol Mix is 296,000 mg/L, TSS is 26,000 mg/L, and TKN is 60,000 mg/L. When diluted with influent the approximate contribution of the Mol Mix to influent strength was 30 mg/L $CBOD_5$, 3 mg/L TSS, and 6 mg/L TKN.

Head loss was measured every three days before and after backflushing. Backflushing was by air scour, manually activated by the operator every three days. Both the Jaeger ring and scrubbies media have approximately 90% pore space. The lava rock pore space is approximately 50%.

4.2.3. Results and Analysis

Treatment performance results established the superiority of scrubbies and Jaeger ring over lava rock. Treatment of $CBOD_5$ was more stable with scrubbies and Jaeger rings than with lava rock (Figure 4-7). Differences in treatment performance between scrubbies and Jaeger rings were negligible. The treatment trends for TKN and TSS are essentially the same as those for $CBOD_5$ (Figure 4-8, Figure 4-9). Nitrification performance of Jaeger rings and scrubbies media was roughly equal, but superior to the lava rock (Figure 4-10). The difference in treatment performance between the Jaeger rings and scrubbies, on one hand, and the lava rock, on the other hand, is probably explained by differences in surface area. Lava rock porosity limits available treatment surface area in the media in comparison to the other media tested.

The lava rock media exhibited the greatest head loss before and after backflushing (Figure 4-11, Figure 4-12). As would be expected, the media with the least pore space had the greatest head loss.

4.2.4. Conclusions to Media Testing

The lava rock media was clearly inferior to both the Jaeger rings and scrubbies for all treatment performance and operational parameters tested. The Jaeger rings and scrubbies appear to be equivalent in treatment and operational performance. Random pack media of

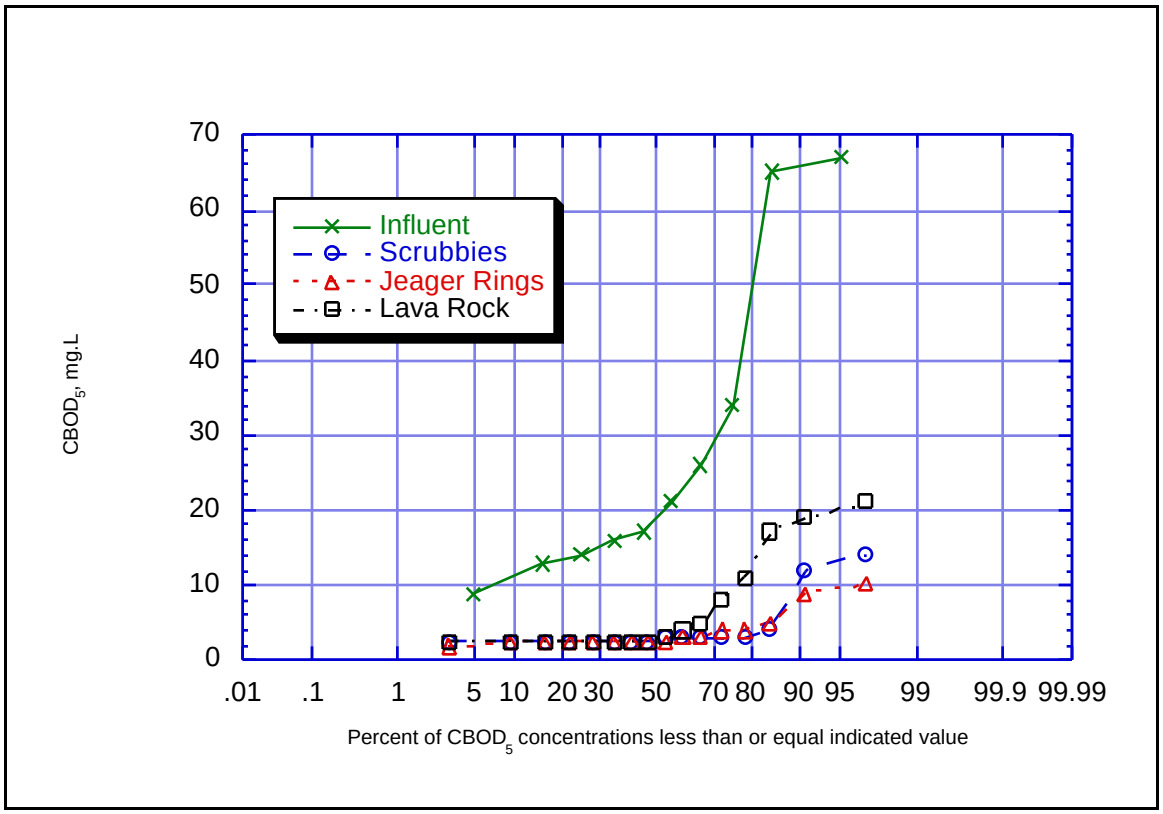

Figure 4-7. Test train EFB media study, $CBOD_5$ treatment performance.

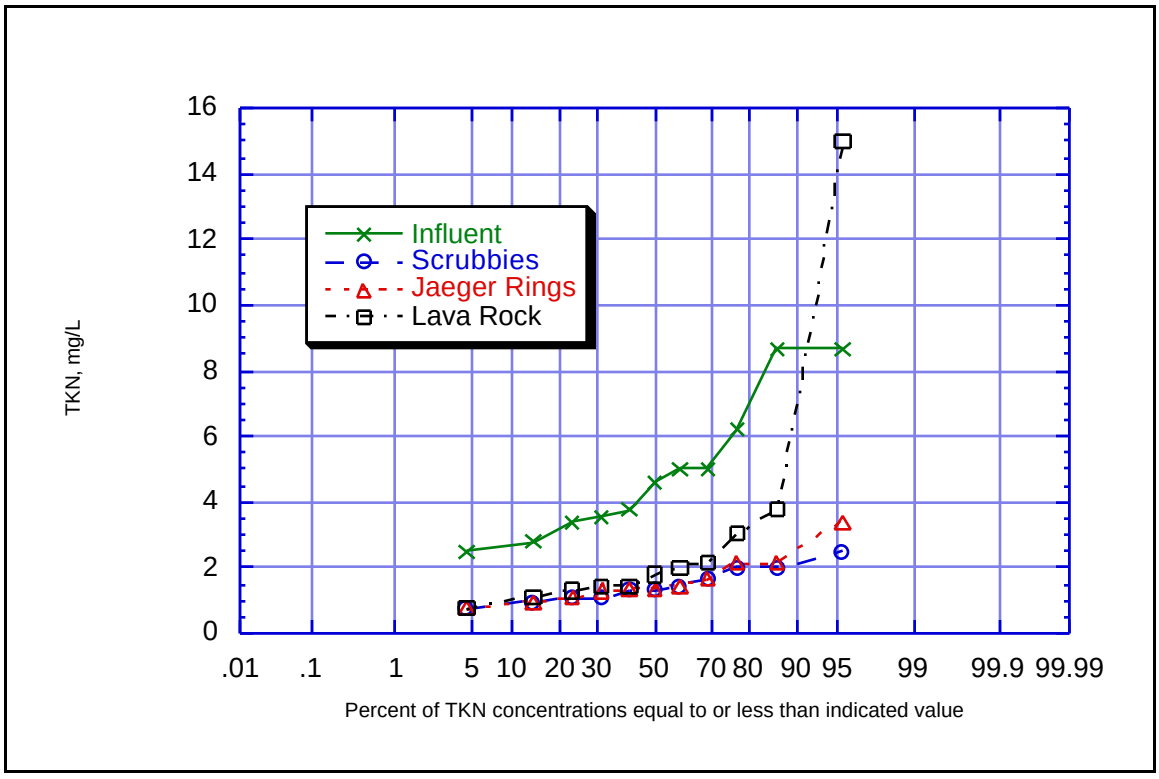

Figure 4-8. Test train EFB media study, TKN treatment performance.

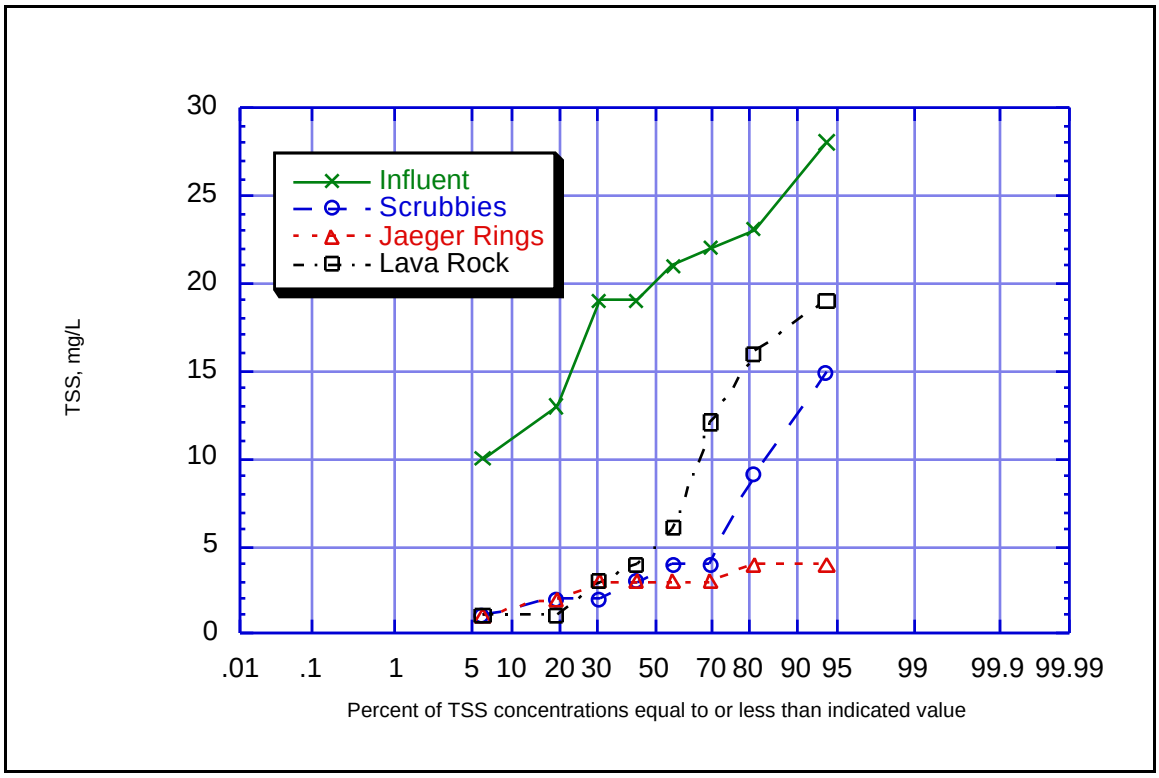

Figure 4-9. Test train EFB media study, TSS treatment performance.

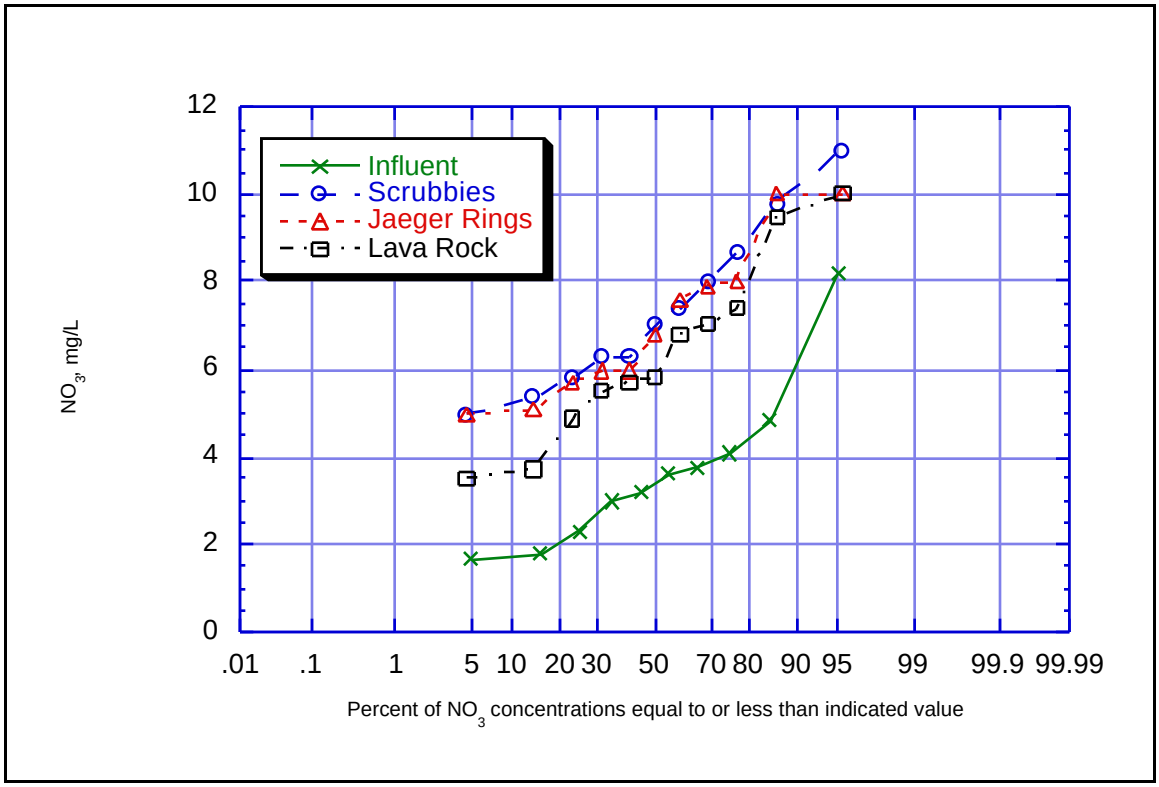

Figure 4-10. Test train EFB media study, NO_3 treatment performance.

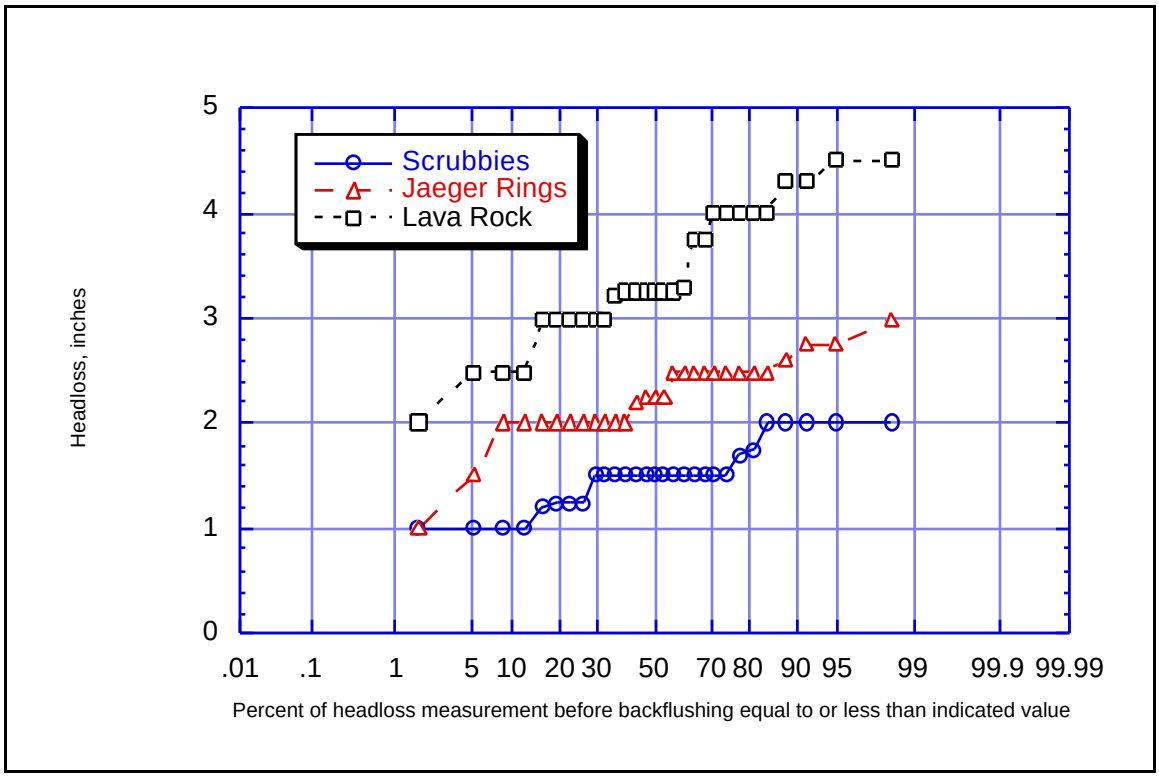

Figure 4-11. Test train EFB media study, head loss before backflushing.

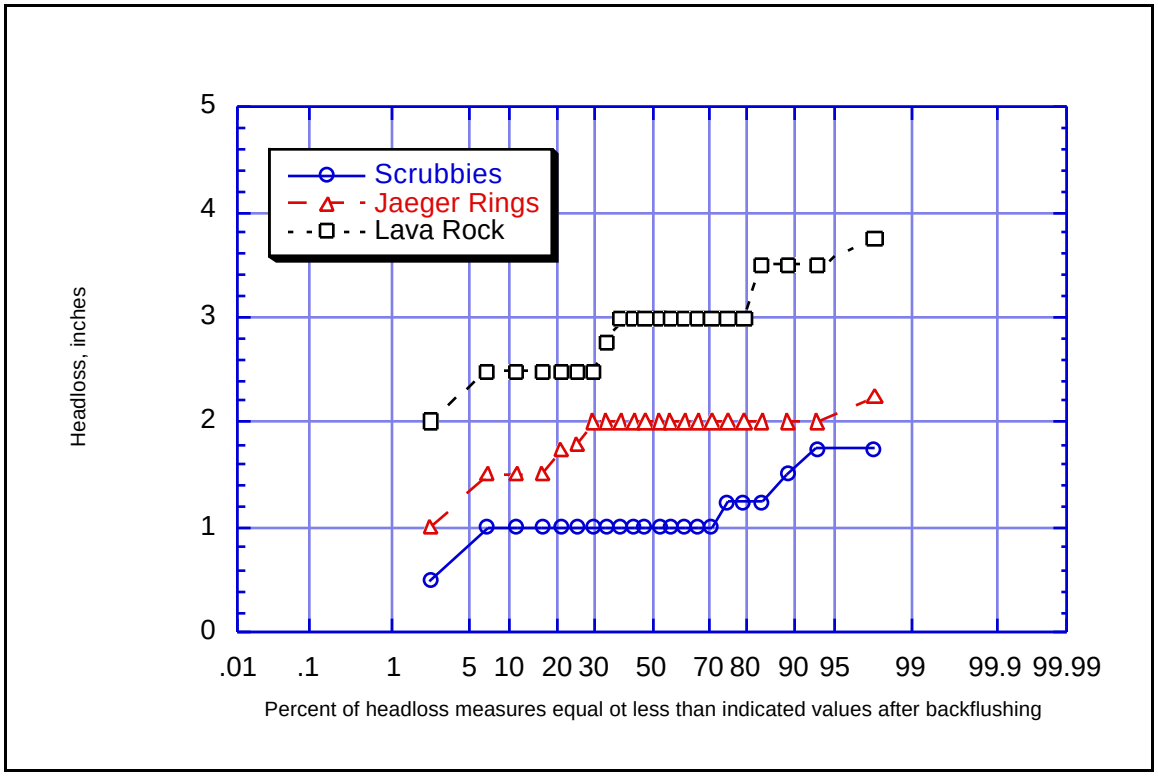

Figure 4-12. Test train EFB media study, head loss after backflushing.

the design and composition of Jaeger rings have been used in the wastewater treatment industry for about 40 years. They are robust. The scrubbies are comparatively untested for long-term use with wastewater. Therefore, Jaeger rings are the best choice of media for use in EFBs of the three media tested.

5. OTHER TOPICS

5.1. Energy Savings

Introduction

There are two principal means to save energy at the Vermont AEES. One potential energy savings is to reduce electricity costs by reducing blower output. The savings cannot be realized because the blowers operate at constant speed and both are needed. It is a useful exercise, however, to estimate an energy credit for a blower system equipped with a variable frequency drive (VFD). A VFD would allow changes in blower output to take advantage of an oxygen credit created by process recycling of nitrified wastewater. There is also a potential credit for weaker influent flows during periods of high groundwater infiltration. The other savings method, already in effect, is to reduce heating costs by setting the thermostat to lower temperatures.

5.1.1. Electricity

This section estimates the potential operational energy savings presented by recycling of nitrified wastewater to a first stage anoxic reactor at the Vermont AEES at 300 m^3/day influent flows. The cost estimates are approximate.

The recent use of a high rate recycle from the last planted open aerobic to the first tank returns oxygen to the head of the system. The oxygen is bound up in NO_3 and also is present in the form of dissolved oxygen. The two oxygen contributions were computed using July and August operational data, and are shown below:

The total oxygen demand for a nitrifying and denitrifying system is[11] (Equation 5.1-1):

Oxygen required, kg/day, = S_o - S – P_w x 1.42 + 4.6 x $(NO_3)_f$ – 2.86 x $(NO_3)_u$ + $(O_2)_r$,

Where:
S_o = influent COD kg/d = 172;
S = effluent COD kg/d = 6;
P_w = wasted cells kg/d = 57;
$(NO_3)_f$ = Nitrate formed kg/d = 2.73;
$(NO_3)_u$ = Nitrate used kg/d = 1.84;
$(O_2)_r$ is the dissolved oxygen content (kg) of the process recycle.

The total required oxygen is thus 90 kg/d, which includes a return of 5.3 kg/d for denitrification and 2.3 kg/d from dissolved oxygen.

[11] Crites and Tchobanoglous. 1998. **Small and Decentralized Wastewater Management Systems,** p. 458. McGraw-Hill.

Using the estimated annual electrical consumption of $10,000[12], and multiplying by the ratio of kilograms of oxygen returned to kilograms of oxygen required, the dollar contribution of the recycle to energy saving can be found as follows:

$10,000 x (5.3 kg/d + 2.3 kg/d)/(90.1 kg/d) = approximately $800 per year savings.

A VFD on the blowers could offer some energy costs savings. The seasonal variation in waste strength shows about a 40 percent reduction in waste strength during the winter/spring months over the summer months (Figure 5-1). This is likely due to spring melt and line infiltration, and would normally be somewhat off-set (in terms of total mass loadings) by an increase in flow rate which a system with a non-fixed influent pump rate would have experienced. The Vermont AEES received roughly the same flow no matter the season. However, the typical municipal diurnal changes in flow and loading would have allowed a reduction in the aeration rate and resulted in savings.

Projected Savings:

1. Assuming that a D.O. probe was installed and would vary the blower VFD speed so as to maintain a desired DO profile;
2. Further assuming (based on typical municipal facility operation) that the D.O. demand is reduced during the six hours from 11PM to 5AM by 50%, and that mixing is the determining factor for required air rates;
3. That mixing is roughly 2/3 of the treatment air rate;

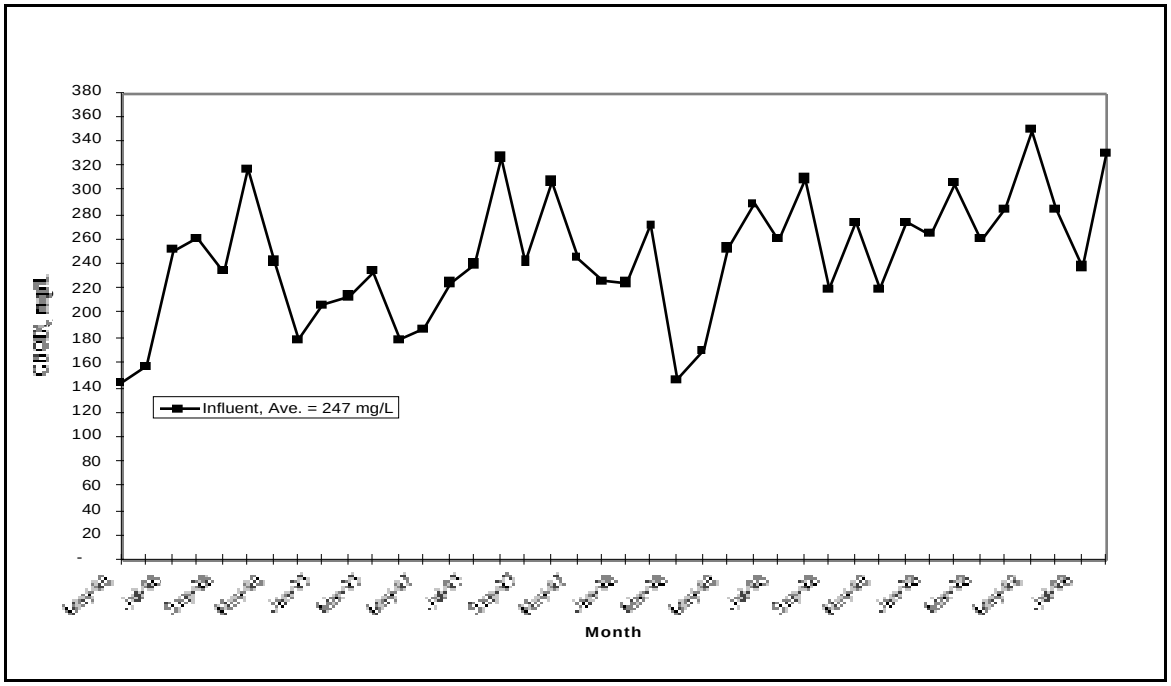

Figure 5-1. Monthly average $CBOD_5$ influent concentrations.

[12] Based on an electricity cost of $0.08 / kW.

The potential savings for one year of operation are as follows:

$10,000/yr x (1-(6 hours/24hours/day) x 1/3 =

$10,000 x 0.92 = $9,200 +/-, or a savings of about $800/year.

The total potential cost saving that could be realized by the installation of a VFD at the Vermont AEES is approximately 10% of the electricity demand of the blowers.

5.1.2. Greenhouse Heat

The cost of gas heat for the first winter of operations (1995-96) at the Vermont AEES was approximately $7,800. The thermostat had been kept near 60^{o} F for much of that winter. Heating costs were subsequently cut by more than half by lowering the thermostat to between 45^{o}F and 50^{o}F. On sunny days there is little need for supplemental heating. Heating costs have been approximately $3,500 per year.

5.2. Comparison of Operations and Maintenance Costs

This section compares yearly operations and maintenance costs between a conventional wastewater treatment facility and the Vermont AEES, each treating 80,000 gpd (Table 5-1).

All Conventional Treatment System information is taken from the January 1998 report by Parsons Engineering Science, Inc., entitled "An Evaluation Of The 'Living Machine' Wastewater Treatment Technology". The Parsons study used a sequence batch reactor (SBR) "package plant" as the conventional technology.

Costs for the Vermont AEES have the following basis:

1. **Personnel**. Personnel costs are based on the monthly average personnel operating costs over 21 months of operation in 1998 and 1999. The cost calculation assumes 8 hours per week of horticultural work, 20 hours per week of operations and 10 hours per week of sampling and miscellaneous work. These personnel costs are required to maintain purely the treatment portion of the system.
2. **Maintenance**. Maintenance costs are based on the same 21 months of data used above. This figure is conservative, given that replacing all the pumps every 5 years and replacing one blower every 7 years (interest rate of 7 percent, 20 year life cycle), provides an annual expense of about $1,300. The $10,500 figure includes a contingency of $1,300 on the main treatment components and a further $3,400 sum for maintenance of the sludge disposal equipment.
4. **Biosolids disposal**. The yield data for the Vermont AEES indicates approximately 58 kg/d of sludge produced for each 80,000 gpd treated at the Vermont AEES. The Parsons Engineering Report assumes 144 kg/d of biosolids produced for 80,000 gpd conventional treatment. A ratio of 58/144 was used to adjust the cost for conventional sludge disposal for the smaller Vermont AEES sludge yield.

5. **Other**. Costs for other categories are based on the same 21 months of data used above.

Total operations and maintenance costs for the conventional system and the Vermont AEES are approximately 15% less than the conventional system. Comparison of operations and maintenance costs is difficult. The conventional system is hypothetical. Determination of costs for the Vermont AEES requires elimination of costs unique to the contractual requirements of the Vermont AEES demonstration project. Some operational protocols required by the project contract would not be done otherwise. The comparison is therefore approximate.

Table 5-1. Comparison of yearly operations and maintenance costs of a conventional treatment system and the Vermont AEES.

Functional Category	**Conventional Treatment System** * * Parsons Engineering EPA Study	**Vermont AEES**
Personnel	$ 46,800	$ 42,900
Maintenance	$ 17,130	$10,500
Biosolids Disposal	$ 8,750	$3,535
Horticulture Supplies	N/A	$ 700
Plants	N/A	$ 500
Trash	(in maintenance cost)	$ 500
Electric Power	$ 10,000	$ 10,000
Gas Heat	N/A	$ 2,000
TOTALS	**$82,680**	**$70,625**

6. CONCLUSION

The Vermont AEES has met design treatment performance for carbonaceous biochemical oxygen demand ($CBOD_5$), total suspended solids (TSS), total Kjeldahl nitrogen (TKN), ammonia (NH_3), nitrate (NO_3), total nitrogen (TN), and fecal coliform bacteria (Treatment Summary Table). Treatment performance for total phosphorous (TP) met the design criterion of 3 mg/L, but would not meet the typical TP tertiary discharge limit of 1 mg/L.

The design basis for denitrification evolved over the course of the project. Starting in August 1996, methanol was used in an anoxic EFB near the end of treatment to denitrify the effluent. Denitrification with methanol was adequate, but methanol had other unintended, adverse impacts on treatment performance. After extensive testing of denitrification at the front of the treatment process using an anoxic, complete-mix reactor, methanol was eliminated from the treatment process. The current treatment configuration uses the front-end anoxic reactor for denitrification. Treatment performance has been superior in the current configuration to previous configurations for all parameters.

Evaluation of overall treatment performance results established that Vermont AEES produces effluent of advanced tertiary quality, with the exception of fecal coliform and total phosphorus values. The effluent characteristics are well suited to achieve fecal coliform values of less than 10 cfu / 100 ml using standard disinfection technologies. Supplemental phosphorous treatment would be necessary to achieve an effluent total phosphorous concentration of less than 1 mg/L

Treatment Summary Table. Effluent values are flow-weighted, combined means of the A and B main treatment trains.

PARAMETER	Mean Influent	Mean effluent, mg/L	Design effluent limit, mg/L
$CBOD_5$	207 (standard deviation, σ=78)	4 (σ = 2)	<10
COD	546 (σ = 154)	24 (σ = 6)	not specified
TSS	189 (σ = 59)	3 (σ = 1)	<10
TKN	28 (σ = 7)	2 (σ = 1)	<5
NH_3	14 (σ = 4)	0.3 (σ = 0.4)	<1
NO_3	-	4 (σ = 1)	<5
TN	31 (σ = 2)	5 (σ = 1)	<10
TP	6 (σ = 3)	2 (σ = 1)	<3
Fecal coliform bacteria	8.5×10^6 cfu / 100 ml ($\sigma = 7 \times 10^7$)	1,700 cfu (σ = 5,900)	<2,000 cfu

System treatment components have performed close to design. Review of data from the treatment profile throughout the AEES and model results, indicates that there is greater treatment efficiency in the aerated reactors in the current configuration than anticipated in the original design. Treatment of influent flows greater than approximately 300 m^3/day, however, was not possible because of a hydraulic limitation imposed by the clarifier size.

The greater efficiency of nutrient removal in the aerated reactors also significantly reduced loading of the EFBs. The EFBs originally were used for nitrification, denitrification, residual carbon removal, and TSS polishing. In the current configuration, the principal remaining treatment function of the EFBs is TSS polishing. Nitrification and denitrification are now completed to design standards in the aerated reactors.

The treatment role of plants in the main trains could not be adequately measured or characterized due to the depth of the aerated reactors. Systematic research of over three hundred fifty plant species has discovered many species that can reliably grow roots two feet into the wastewater column. The consensus opinion at the EPA, and within the technical group advising this project is that plant roots must occupy 20-30% of the wastewater column to provide significant treatment capacity. At the Vermont AEES the roots occupy approximately 9% of the wastewater column in the 13 feet deep aerated reactors. The aerated reactors are therefore probably too deep to optimize the contribution of plants to wastewater treatment.

Results from the test train establish some advantages of plants in shallow (3.5 feet deep) reactors. When compared in parallel to activated sludge treatment, planted reactors receiving the same biosolids recycle exhibited significantly more stable COD treatment performance. Addition of artificial media and optimization of the biosolids recycle to the unplanted parallel line brought its performance on par with that of the planted line, but no better. The planted test line also had better total nitrogen removal than the unplanted line.

Operations and maintenance costs of the Vermont AEES are slightly less than a comparably sized conventional system. Parsons Engineering, under contract to the USEPA, estimated that O&M costs were $83,000 per year for a sequenced batch reactor (SBR) treating 80,000 gpd to advanced tertiary standards. Yearly O&M costs for the Vermont AEES determined from actual costs were $71,000 per year.

This four year long project has demonstrated the ability of the AEES to treat municipal wastewater to tertiary standards in cold climates. The system has been proven to have highly stable treatment performance, and has lower operating and maintenance costs than conventional small treatment systems.