NITRIFICATION AND DENITRIFICATION: BIOLOGICAL NITROGEN REMOVAL AND SLUDGE GENERATION AT THE YORK RIVER TREATMENT PLANT

by

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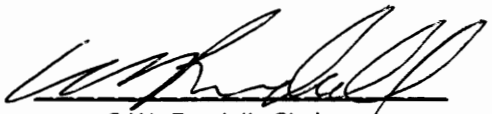

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Keywords: Nitrification, Denitrification, Biological Nitrogen Removal, Wastewater Treatment

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(ABSTRACT)

Data from Hampton Roads Sanitation District was used to calculate nitrification and denitrification rates for the A^2/O mode (1987) and the VIP mode (1988) of operation. Nitrification and denitrification rates compared to literature values for similar sludge ages. The mean VIP nitrification rate was eight percent less compared to the A^2/O mode. Denitrification varied with the amount of nitrate loading to the anoxic zone and the rate of total nitrate recycle. The amount of denitrification that occurred in each zone during the different operations was determined. Process mode variations caused different percentages in each zone. Anaerobic and anoxic denitrification was a linear function of the mass of nitrate recycled to the anoxic zone. Fifty to seventy-five percent of the denitrification took place in the aerobic basin during both process modes, but more aerobic denitrification occurred for the A^2/O operation. Secondary clarifier nitrate varied inversely with the nitrate recycle similarly for both process modes. The differences in sludge production between the VIP and A^2/O process could be explained by the differences in mean cell residence time.

Keywords: Nitrification, Denitrification, Biological Nitrogen Removal, Wastewater Treatment

DEDICATION

I would like to dedicate this work to my late mother and father.

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Acknowledgement is due to my husband Tom Mosca for his love and support, patience, encouragement and advice during this project. My family also contributed in that regard. The author wishes to thank Dr. C. W. Randall for his guidance in this project and for serving as committee chairman. Thanks are also due to my committee members, Dr. Nancy G. Love and Dr. John C. Little. Other acknowledgements are due to Dave Waltrip and Bob Jones of Hampton Roads Sanitation District for their advice and the support and encouragement of my co-workers and friends at Hampton Roads Sanitation District and the Department of Environmental Quality - Water Division.

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CHAPTER 1
INTRODUCTION

Domestic wastewater is treated by municipal wastewater treatment plants and released to available waterways. The wastewater discharged from treatment plants may still, under current regulations, only undergo primary (sedimentation) treatment, plus secondary (biological) treatment and no nutrient removal. Biological Nutrient Removal (BNR) was developed in response to increasing problems of eutrophication of waterbodies due to the addition of sewage effluent. The use of BNR has become more critical in recent years due to the difficulty in arresting eutrophication through regulation of point sources of pollution and additional considerations such as of the toxicity to aquatic life of unionized ammonia.

BNR research was pioneered by Ludzack and Ettinger (1962), Barnard (1973), and Wurhmann (1964), as discussed by Wanielista and Eckenfelder (1979). Full BNR involves alternating anaerobic, anoxic and aerobic wastewater treatment activated sludge zones to provide conditions favoring phosphorus removal, denitrification and nitrification.

The nitrogen in municipal wastewaters is primarily in the forms of ammonia and organic nitrogen. Microbially catalyzed hydrolysis reactions break down organic nitrogen, with amino NH_2 groups removed from amino acids, and ammonia is formed as an end product. Nitrification is a two step microbial process wherein ammonia is oxidized through nitrite to nitrate. Denitrification is a process wherein nitrate or nitrite are reduced to nitrogen gas. *Nitrosomonas*, *Nitrobacter*, and *Pseudomonas*, are genera of bacteria important to these processes.

Frequently in the literature, all the autotrophic bacteria which oxidize ammonia to nitrate are lumped under the term *Nitrosomonas*. The genera actually include *Nitrosomonas*,

Nitrosospira, *Nitrosococcus*, and *Nitrosolobus*. The autotrophs that oxidize nitrite to nitrate are termed *Nitrobacter*. These genera include *Nitrobacter* and *Nitrospira*. The same convention will be adhered to in this paper.

Nitrification is essential to the process of denitrification, by which *Pseudomonas* and other heterotrophic bacteria are responsible for the reduction of oxidized nitrogen to nitrogen gas, thereby eliminating nitrogen from the waste treatment system. While primary and secondary treatment typically achieves approximately a 20% reduction in nitrogen, the utilization of BNR can eliminate up to 90% of the wastewater nitrogen, depending on process configuration and operating specifics (U. S. EPA, 1975).

Separate sludge systems involve two or more biological treatment trains to remove first carbon, and then oxidize and remove nitrogen. The carbon removal and nitrogen oxidation steps are combined into one train in a two sludge system, but occur separately in a three sludge system. Denitrification under these conditions requires the addition of an external carbon source, usually methanol. Single sludge systems, in contrast, use the carbon in the wastewater for this purpose and denitrification and carbon removal occur simultaneously. The remaining organic carbon is removed in the aerobic zone where nitrification also occurs.

A complete understanding of the processes of nitrification and denitrification is still evolving and the study of optimizing these reactions and their kinetics is ongoing. Painter in 1970 and Sharma and Ahlert in 1977 published literature reviews to consolidate information (see Chapter 2, Review of Literature). In the economically preferable single sludge systems of wastewater treatment, nitrification takes place in the presence of heterotrophic organisms that are energetically better suited to exploit the ecosystem (Snoeyink and Jenkins, 1980). Since the

nitrifiers are disadvantaged in this regard, they are slow growers and, therefore, longer detention times must be used to keep them in the system (U. S. EPA, 1975). However, their slower growth rates (Gaudy and Gaudy, 1988) result in less biomass for disposal.

Sufficient oxygen must be provided for both the autotrophs and heterotrophs. Oxygen is necessary to satisfy biochemical oxygen demand, to convert ammonia to nitrate, and to satisfy endogenous respiration needs of the biomass (Beckman *et al.*, 1972). Although the bulk liquid dissolved oxygen (DO) may indicate sufficient oxygen for nitrification, mass transfer resistance in the floc may limit the amount available to nitrifiers (Hart *et al.*, 1986). Nitrifier reactions are susceptible to toxics which typically may enter the treatment plant in discrete high concentrations (slugs) (Painter, 1970). Nitrifiers are also sensitive to temperature (Painter, 1970). Larger tank volumes and longer detention times (Sharma and Ahlert, 1977), pH optimization and increased mixed liquor concentrations (Wild, 1975) are methods of accommodating nitrifying systems to seasonal temperature changes.

Denitrification, or nitrate dissimilation, by the *Pseudomonas* heterotrophic bacteria is inhibited by oxygen. Oxygen molecules are used (in heterotrophic metabolism) preferentially by the organism instead of the bound oxygen in the nitrate form (denitrification) when oxygen is present (WPCF, 1983), because denitrification produces less energy for the organism (U. S. EPA, 1975). Like the nitrification reaction, the energy disadvantage translates into less biomass produced. Further, still less sludge is produced when the substrate to be consumed is less highly oxidized (Gaudy and Gaudy, 1988).

Nitrification will require additional oxygen over and above the amount of oxygen necessary to satisfy biochemical oxygen demand in a treatment plant. The use of denitrification

in a single sludge treatment system will result in a typical net "return" of 15 to 20 per cent of the total system requirement for oxygen. By stabilizing additional influent organic matter with nitrate instead of oxygen, the denitrification reaction "recovers" about two thirds of the oxygen used to produce the nitrate by nitrification (Ekama *et al.*, 1984).

The York River Demonstration Project was initiated at the Hampton Roads Sanitation District (HRSD) York River wastewater treatment plant (WWTP) for the purpose of altering the existing process scheme to one of biological nutrient removal and developing greater understanding of BNR in a full scale treatment plant (Randall *et al.*, 1990). The process ran from 1986 to 1989, using, at first, the A/O process for phosphorus removal, then the A^2/O process for phosphorus and nitrogen removal and, lastly, the VIP process for phosphorus and nitrogen removal. The latter process is a high rate system similar to the University of Cape Town (UCT) process. The UCT mode of operation addressed concerns of incomplete phosphorus removal associated with the A^2/O configuration with the addition of a denitrified recycle to maintain the anoxic character of the anoxic basin (Lamb, 1988). The VIP process is operated at a higher loading rate, has lower sludge ages and a greater food : microorganism ratio than the UCT process, and thereby requires much less reactor volume (CH2M Hill, 1987).

The purpose of this study was to analyze the data collected during the York River BNR Demonstration Project to gain additional insights into where nitrification and denitrification occurred within the activated sludge system, and how the extent, rates and locations of nitrogen transformation differed between modes of operation. The objectives were to derive the necessary equations in order to quantify the nitrification and denitrification rates and associated processes for the York River Demonstration Project system.

CHAPTER 2
REVIEW OF LITERATURE

A Chemical and Microbiological Description of Nitrification

Nitrification is the oxidation of NH_3 to NO_2, and subsequently to NO_3. It is part of the nitrogen cycle in nature, and is microbially catalyzed by two groups of chemoautotrophic bacteria. The overall reaction is:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O. \tag{2.1}$$

Nitrosomonas oxidizes ammonia according to the reaction:

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O. \tag{2.2}$$

Nitrobacter catalyzes the reaction:

$$2NO_2^- + O_2 \rightarrow 2NO_3^-. \tag{2.3}$$

Nitrosomonas and *Nitrobacter* are genera names commonly used to refer to the two groups of the family Nitrobacteraceae that oxidize ammonia and nitrite, respectively, for energy, although other genera can perform the same functions. Bergey's manual (Buchanan and Gibbons, 1974) lists *Nitrosomonas*, *Nitrosospira*, *Nitrosococcus*, *and Nitrosolobus* as genera which oxidize ammonia to nitrite. *Nitrobacter* and *Nitrospira* oxidize nitrite to nitrate (Buchanan and Gibbons, 1974).

The nitrifiers were first described by Winogradsky in the late 19th century (Gaudy and Gaudy, 1988). They are common to soil and both salt and fresh water. Typically they are found wherever their nutritional requirements of ammonia, carbon dioxide and oxygen are present. Nitrifiers require calcium, magnesium, copper, iron and molybdenum (Brezonik, 1968). The latter three may be limiting in natural waters (Brezonik, 1968), but may be expected in sufficient concentration in municipal wastewater (Painter, 1970). These bacteria are also correctly described as lithotrophs (autotrophs) in that they use inorganic carbon for their carbon source.

In natural waters, nitrifiers are most often found on suspended particulate matter (Tuffey, 1974). Early attempts at culturing these bacteria supposed particulates to be a requirement; Engel and Alexander (1958) showed this to be untrue. Before a better understanding of pH optima was attained, it was noted that calcium carbonate suspensions allowed better growth of the organisms; enhanced growth was thought to be due to the presence of these particulates (Painter, 1970).

The most common inhibitors to nitrifier growth are excessive amounts of ammonia and nitrite, their substrates. *Nitrosomonas* and *Nitrobacter* are each more sensitive to the other's substrate than to their own (Painter, 1970). This is not due to total ammonia or nitrite concentrations, but to the presence of unionized ammonia and nitrous acid (Anthonisen *et al.*, 1976). Nitrifier populations have a reputation of being vulnerable to environmental changes (Sutton, *et al.*, 1975), e.g., pH and available micronutrients. Nitrification inhibition in municipal wastewater treatment plants is rare however (Anthonisen *et al.*, 1976), and is most often the result of industrial discharge (Ekama, *et al.*, 1984). Acclimation may decrease this effect. Industrial discharges most frequently pass through a treatment plant in slugs however,

and this does not allow acclimation. Temperature inhibition of *Nitrobacter*, resulting in a build up of nitrite, is fairly common (Randall and Buth, 1984a).

A pathway of electron transfer has been proposed in regard to the ammonia oxidation step:

$$NH_4^+ \rightarrow NH_2 \rightarrow NH_2OH \rightarrow NHOH \rightarrow NOH \rightarrow NO \rightarrow NO_2^- \tag{2.4}$$

(Doetsch and Cook, 1973). Some of the short term intermediates between hydroxylamine and NO_2, e.g. hyponitrite and nitroxyl, do not leave the enzyme complex since they are unstable and toxic (Brezonik, 1968); they have not been found in culturing experiments in the laboratory (Painter, 1970). The oxidation of nitrite to nitrate is thought to be a single step (Doetsch and Cook, 1973), because no intermediates have been found (Painter, 1970).

The change in oxidation state through the nitrification process is as follows:

$$\underset{-3}{NH_4^+} \rightarrow \underset{+3}{NO_2^-} \rightarrow \underset{+5}{NO_3^-} \tag{2.5}$$

(Turk and Mavinic, 1987). The ammonia oxidation step provides the reducing power of 3 pair of hydrogen ions; the nitrite oxidation step is equivalent to one pair of hydrogen ions (Painter, 1970).

During oxidation, electrons are released, so an electron acceptor is required. Molecular oxygen serves this purpose for the nitrifiers; nitrification is therefore an aerobic process. The amount of oxygen required is 4.57 mg per mg NH_4^+- N oxidized (U.S. EPA, 1975). Thus, a substantial nitrogenous oxygen demand may be realized during nitrification of a wastewater.

Energy considerations

The nitrification reaction is endothermic and exergonic. Painter (1970) reports a range of values of free energy released by the ammonia oxidation step of 66 to 84 Kcal, with 17 Kcal generated in nitrite oxidation. The reaction, being exergonic, provides energy to the bacteria. Reactions used for energy production exert a greater effect on the environment than those used for growth (Gaudy and Gaudy, 1988). Growth related nutrients are consumed in reactions, whereas those involved in energy reactions are only altered, in accordance with Newton's first law, into metabolic by-products which are released into the environment. The by-products of nitrification provide an electron acceptor, oxidized nitrogen, for heterotrophs denitrifying under anoxic conditions.

Nitrosomonas realizes more free energy per mole of substrate by oxidation of ammonia than does *Nitrobacter* through nitrite oxidation. If it is assumed that an equal number of cells is produced per unit of energy, then more *Nitrosomonas* cells are created than *Nitrobacter* cells per mole of ammonia nitrogen oxidized (U. S. EPA, 1975). Ratios of experimental yields of cell dry weights to weight of oxidized nitrogen are reported to vary from 0.06 to 0.13 for *Nitrosomonas* and 0.02 to 0.07 for *Nitrobacter* (Painter, 1970).

Compared to heterotrophic organisms, both groups of nitrifiers are energetically disadvantaged. The cellular yield, Y, is a measure of the cell weight produced per weight of substrate consumed. Cell yields for nitrifying organisms are on the order of 0.1 cell equivalents per equivalent electrons exchanged in the ammonia oxidation redox reaction. By comparison, a heterotrophic organism oxidizing fructose to carbon dioxide and water can produce a cell yield of 0.74 with a free energy change of -28.7 Kcal (McCarty, 1971, in Snoeyink and Jenkins, 1980).

One might therefore expect heterotrophs to overgrow nitrifiers in the natural environment, or single sludge systems, providing the appropriate nutrients are available to each.

The maximum rate of growth, μ_{max}, is also proportional to the amount of energy generated by the different redox reactions (Snoeyink and Jenkins, 1980). The growth rate per unit of biomass is the specific growth rate, μ, with units of time.$^{-1}$ A specific growth rate of 1.6 to 2.0 day^{-1} at 30 °C has been reported for ammonia oxidation performed by *Nitrosomonas*. By way of comparison, a typical heterotrophic specific growth rate in municipal activated sludge ranges from 9.6 to 14.4 day^{-1} at 18 to 25 °C (Gaudy and Gaudy, 1988).

Nitrifier Kinetics

The implications of reduced cellular yield and growth rate of nitrifiers are critical to an understanding of the interaction of nitrifier and heterotroph kinetics in sewage treatment. Since nitrifier growth occurs so much more slowly than that of heterotrophs, the Biological Solids Retention Time (BSRT) of the mixed liquor must be long enough to accommodate both sets of microorganisms. If care is not taken to ensure an adequate nitrifier BSRT (proportional to the inverse of nitrifier growth rate), the heterotrophic growth rate may settle at a value that exceeds the maximum nitrifier growth rate, μ_{max_N}. This will cause a phenomenon called "washout" where the population of nitrifiers decreases relative to the total mixed population of nitrifiers and heterotrophs until they are effectively eliminated from the system (U.S. EPA, 1975).

Monod and First Order Models

The Monod model is generally accepted as describing the kinetics of the conversion of

substrate to biomass (Gaudy and Gaudy, 1988). The Monod model has been applied specifically to nitrification (e.g., Knowles, *et al.*, 1965, Stratton and McCarty, 1967, Poduska and Andrews, 1975, Sutton *et al.*, 1975, Hall and Murphy, 1980). This model has a utility in that it covers first order kinetics when the limiting nutrient is in short supply and zero order kinetics when substrate concentrations are in excess of greater than ten times the saturation constant K_s (Sharma and Ahlert, 1977). The applicability of the Monod model to substrate kinetics was confirmed in a batch radiochemical study which took into account the growth of the biomass during the experiments (Speitel and DiGiano, 1988). However, Michaelis-Menten kinetics, which are similar in form to Monod kinetics and are used to describe enzyme kinetics, are also used to describe the kinetics of the nitrification reaction (Shieh and Lamotta, 1979). The Monod relationship is as follows (Benefield and Randall, 1980):

$$\mu = \mu_m \frac{S}{K_s + S} \tag{2.6}$$

where, μ = the specific growth rate, 1/time,

μ_{max} = the maximum value of μ obtained when substrate is in sufficient supply to be not limiting, 1/time,

S = concentration of the growth-limiting substrate, mass/volume,

K_s = A constant equal to the concentration at which $\mu = (\mu_{max}/2)$, mass/volume.

The growth limiting nutrient in the Monod equation is most often considered to be the energy source, ie., ammonia in the application to nitrification. The carbon source, calcium carbonate (as alkalinity) is thought to be rarely limiting. Some researchers have considered the limiting nutrient to be the electron acceptor (dissolved oxygen) (e.g., Sharma and Ahlert, 1977).

Cellular yield, the ratio of the rate of biomass production to the rate of substrate utilization, is a somewhat different concept for nitrifier and heterotrophic bacteria. Heterotrophic bacteria use organic carbon as an energy and carbon source and the yield is considered to be the proportion of substrate converted to new biomass during growth. Nitrifiers mainly use ammonia as an energy source and to a much lesser degree, for synthesis. Carbon dioxide is the carbon source for synthesis (Poduska and Andrews, 1975).

In the literature, reports on the order of the nitrification reaction differ. Some believe that nitrification follows first order kinetics, and that the Monod equation is a mathematical approximation of bacterial growth over the growth portion only (McKinney, 1975). Only at the start up of a wastewater treatment plant, or in an emergency where microbial solids are being lost, would there be a need for the design engineer to consider a mass limiting model, in which the rate of reaction is limited by the bacteria themselves. In this school, Monod must be modified for endogenous respiration to be applicable to activated sludge systems and that not all phases of growth may be modeled by equations. In the practical design and operation of activated sludge systems, the mixed microbial culture operates in a food limiting condition as long as the organic matter is being added to the system. Should the food input stop, the population stops growing, and through endogenous respiration, declines. The operating modes that correspond to "log growth" and declining growth in the batch system are food excess and food limiting, respectively. A mathematical expression for first order kinetics is:

$$dS/dt = K\,[S]^{1} = KS, \tag{2.7}$$

where dS/dt = the removal over time of substrate (S) by the organisms and K = a constant.

Non-limiting Zero Order Kinetics

Many researchers believe the nitrification reaction to be of order zero with respect to substrate. This corresponds to substrate being non-limiting, i.e., sufficient substrate for the progress of the reaction to be unimpeded. A mathematical expression for zero order kinetics is:

$$dS/dt = K\,[S]^{0} = K\,, \tag{2.8}$$

where dS/dt = the removal over time of substrate (S) by the organisms and K = a constant. Zero order kinetics have been seen at influent ammonia concentrations of 100 to 1200 mg/L (Wong-Chong and Loehr, 1975). The nitrification reaction may be of order zero when dissolved oxygen and ammonia concentrations are greater than 1 mg/L (Eckenfelder and Argaman, 1978). Influent ammonia concentrations of 6 to 60 mg/L may be characteristic of a zero order reaction (Wild, *et al.*, 1971). Zero order kinetics were found in a study of trickling filter kinetics (Huang and Hopson, 1974). It was shown that other researchers have found first order kinetics by ignoring the effect of pH on the reactions, or through the evaluation of data as "percent remaining" or "percent removal." The mass transfer resistance can also mask the true order of the reaction (Shieh and Lamotta, 1979). Zero order kinetics have been found through the simplification of the Monod equation if ammonia concentration is much greater than the half saturation constant (Hall and Murphy, 1980). A first order equation for nitrification modeling may result from a composite series of zero order equations (Eckenfelder, 1975). The constant that results from this is therefore pseudo-first order, having the units of volume/mass-time. In this manner, the overall reaction may follow first order kinetics since it is dependent on substrate and nitrifying organism concentration, although for ammonia concentrations greater than 1 mg/L, the reaction is of the zero order (Poduska and Andrews, 1975). The nitrification rate can change from zero to first order solely due to a change in temperature of the reaction

(Randall and Buth, 1984a). The extent of nitrification is dependent on the ammonia : alkalinity ratio as found in a study of trickling filters (Huang *et al.*, 1989). The results indicated that as the ratio got smaller, the reaction order increased. As ammonia is oxidized to nitrate, 6.0 to 7.4 mg alkalinity is destroyed per milligram nitrogen oxidized (U.S. EPA, 1975).

Lawrence and McCarty (1970) have developed equations for describing activated sludge kinetics based on material balances. This concept was used in this thesis for the calculation of nitrification and denitrification rates. Into these relationships, equations which follow zero, first order or Monod kinetics may be substituted.

Most initial work was done on pure cultures with pure substrates. Higher growth rates may be seen in continuous pure culture experimentation than in activated sludge systems (Sharma and Ahlert, 1977). Competition by other species could be a limiting factor (Painter, 1970) (Lijklema, 1973). The chelating effect of muds, sludges and sewages would affect the concentration of free ions and would partially explain how results obtained in the laboratory could differ from those of the "real world" (Painter, 1970). Also, differences in diffusional resistances of oxygen in the floc affect how much oxygen is available for use (Painter, 1970) and thereby limit the system in that regard. These effects may explain many conflicting results found in the literature that will be discussed.

Common Model Assumptions

The inclusion in models of mass transfer resistances in flocs negates a common assumption in the development of kinetic models that the bulk liquid and the microbial population located within is homogeneous. Nitrification can be initiated in a full scale domestic

wastewater treatment plant primarily through adjustment of oxygen controls (Hart *et al.*, 1986). The increased DO in the bulk liquid serves the purpose of increasing the amount of oxygen available in the floc and to make the population distribution seem more homogeneous (Hart *et al.*, 1986). Determining Monod constants with flocculated cultures leads to overestimation of the half saturation coefficient (Shieh, 1980). It is possible to eliminate diffusional characteristics by agitation and observe the nitrification process (Shieh and Lamotta, 1979). The diffusional resistance of the floc matrix has an effect on the half saturation constant: if the resistance decreases, the saturation constant decreases; the diffusional chemistry of ammonia is very different than that of nitrite (Chudoba *et al.*, 1985). Ammonia diffuses into the floc matrix from the bulk liquid, while nitrite is formed in the matrix and diffuses out into the bulk liquid. This explains why ammonia in the floc matrix may be lower than in the bulk liquid, while nitrite in the floc is higher than in the bulk liquid. Lau (1984) proposes calculating the growth function, μ, as the volume averaged rates of substrate diffusion at the outside of the floc and the inside of the floc.

Carrying the floc diffusional discussion further, Stenstrom and Song (1991) maintain that multiple substrate limiting conditions within the flocs can affect the nitrification process while single substrate limiting conditions were apparent in the bulk liquid. The multiple limitation of substrates referred to is the energy source, ammonia, and the electron acceptor, dissolved oxygen. Single substrate limiting conditions is a common model assumption. Okey and Albertson (1989) found that the ability of oxygen to diffuse across a biofilm controlled the rate of nitrification for a range of ammonia concentrations that were not limiting. The electron donor of the nitrification reaction, inorganic carbon from the wastewater, is only rarely found to be in limiting supply (Painter, 1970). For each mg/L of ammonia oxidized, 7.14 mg of

alkalinity as $CaCO_3$ are destroyed. (U. S. EPA, 1975).

Another common model assumption is that nitrifiers are unaffected by the heterotroph population (Hockenbury *et al.*, 1977). An increase in the proportion of heterotrophs and nitrifiers takes place with an increase in organic carbon to the system. Increased food to microorganism ratios at several temperatures correlated to lowered nitrification rates (Beckman, *et al.*, 1972). Increased organic carbon, in the form of an organic shock load, can interfere with nitrification when heterotrophs utilize of all the oxygen, thereby depriving the nitrifiers of adequate amounts (Stenstrom and Song, 1991).

Another assumption is the effluent concentration is unaffected by the influent concentration of substrate. This assumption does not necessarily hold for heterotrophic populations (Grau, 1975) (Randall and Benefield, 1977). Grau's work showed the dependence of the effluent upon the influent substrate concentration for variable strength, organic substrates of multicomponent mixtures. Since the breakdown of organic nitrogen results in the formation of ammonia by hydrolysis, this observation for substrate kinetics could be carried over to nitrification kinetics (Sheih and LaMotta, 1979). These researchers found that for a constant contact time, the effluent substrate concentration could be changed by a change in the influent substrate amount. Nitrite buildup in a trickling filter system was found to be dependent on influent ammonium loading and effluent ammonium concentration (Huang *et al.*, 1989).

Nitrification can be considered a single step for modeling purpose, since the limiting step is thought to be the first one, ammonia oxidation. The oxidative capacity of the nitrate formers is greater than that of the nitrite formers. Thus, the equations would simplify to: the rate of nitrate formation is equal to the rate of ammonia oxidation (Wong-Chong and Loehr, 1975). However, this is not true when the temperature decreases to low values, because the nitrate

formers are more strongly inhibited by temperature than are the nitrite formers. Consequently, nitrite concentrations build up (Randall and Buth, 1984a).

A widespread assumption is that the viability and activity of sludge is best measured by the measurement of the volatile suspended solids (Weddle and Jenkins, 1971). Much of the available data is based on the VSS measurement due to the difficulties associated with the measurement of the nitrifier population (Sharma and Ahlert, 1977). The fraction of nitrifiers (f) in the biomass may be calculated in proportion to the BOD/TKN ratio (Gupta, 1985)(Sharma and Ahlert, 1977).

$$f = \frac{1}{S_o/N_o * Y_H/Y_N + 1} \tag{2.9}$$

S_o and N_o refer to the initial heterotrophic and nitrifier substrate values, respectively, while the Y's denote heterotrophic and nitrifier yields. This fraction may be multiplied by the VSS to calculate the nitrifier portion of the biomass.

$$f = \frac{X_{VN}}{X_V} = \frac{a_N N_{ox}}{aS_r + a_N N_{ox}} \tag{2.10}$$

The fraction of nitrifiers in the biomass may be calculated above as compared to the whole (Eckenfelder and Argaman, 1979). X_{VN} is the concentration of nitrifying organisms, as mg/L VSS, X_V is the total concentration of the biomass, a_N is the nitrifier yield coefficient, N_{ox} is the amount of ammonia oxidized to nitrate, a is the heterotrophic yield, and S_r is the BOD removed, mg/L. A low viability of the population or a high maintenance need may explain why growth and substrate utilization do not always coincide and the Monod equation

does not always apply (Jones, 1975). Studies performed by Jones with continuous cultures indicated that low specific growth rates (where the maximum growth rate, μ_{max}, is less than or equal to 10%) have a cell viability of 10 per cent or less; viability increases to 100 per cent as the growth rate increases to 30 to 50 per cent. This hypothesis is similar to that of Pirt (1975, as discussed by Benefield and Randall, 1980) in which substrate is used for the energy of maintenance. Herbert (1958, discussed in Benefield and Randall, 1980) disagreed in that instead of substrate being used to satisfy this requirement it is biomass that is oxidized and thus accounts for the decrease in cellular mass seen. Pirt's constant is typically referred to as "b" and Herbert's coefficient is usually denoted by K_d. These are related through

$$b=K_d/Y_T \tag{2.11}$$

(Benefield and Randall, 1980).

These models discussed all assume that equilibrium conditions prevail. With this assumption, in a material balance, the overall change in substrate or biomass is considered to be zero, allowing one to solve for quantities unknown. One may assume this state when process control parameters change little with time (McClintock *et al.*, 1988). Where there is a lag in the growth rate corresponding to a change in the dilution rate, the Monod relationship did not fit nitrification kinetics (Mateles *et al.*, 1965). Growth rate hysteresis is a situation in substrate kinetics in which the growth rate is not predictable according to the Monod function; this is likely to occur during the "transient" state in which the amount of substrate is changing (Storer and Gaudy, 1969). Assuming that the system is in equilibrium negates the need to have the growth rate respond instantly to a change in substrate concentration. It is logical that a certain time period would be required for acclimation to any change. The limiting amount of oxygen needed by the nitrifiers may change while the system is transient as opposed to while the system

is in steady state (Stenstrom and Song, 1991). Due to mass transport resistance and competition from heterotrophs for oxygen, during an organic shock load to the system, the apparent limiting amount of oxygen for nitrifiers could be as much as 4.0 mg/L, whereas under corresponding equilibrium conditions, the demand for oxygen was found to be 0.5 to 2.5 mg/L (Stenstrom and Song, 1991). Nitrification is dependent on the MCRT during steady state conditions. However, while the system is in a transient state, the hydraulic retention time becomes more important and short detention times must be accompanied by large increases in MLVSS to maintain the same rate (Lijklema, 1973).

The Effect of the Reactor on Nitrification

The reactor type influences and is influenced by the kinetics of the nitrification reaction. Nitrification modeling can be applied to completely mixed reactors from experimentation on batch reactors (Wong-Chong and Loehr, 1975). Nitrification rates were found to be equivalent for separate and combined sludge systems operating at the same HRT. Conflicting results have been noted in the literature, for reasons that may be explained in part by chelation chemistry and the diffusional effects of oxygen, and other factors in experimental design. For combined systems, the nitrification rate is unaffected by the mixing regime or the reactor configuration (Sutton *et al.*, 1975). Increased rates of ammonia removal were found in hyacinth systems as hydraulic conditions become more and more like those of the completely mixed regime (Weber and Tchobanoglous, 1985). Fewer nitrifiers were found in a completely mixed reactor than a plug flow reactor due to an intractable amount of ammonia that is unoxidized in this type of reactor while the plug flow type oxidizes completely (Lijklema, 1973). Presumably the

difference is because of the greater amount of contact time with the ammonia in the plug flow reactor. Theoretical calculations show that the true plug flow reactor is more efficient, taking less time for stabilization than the complete mix system. In the complete mix reactor, the initial high driving force (the concentration of the substrate) is immediately transformed to the low driving force of the effluent (Benefield and Randall, 1980). At high rates of sludge wasting, approaching the washout point for the nitrifiers, this effect is diminished. The biomass of the nitrifiers increases down the length of the reactor. Due to their slower growth, they are not to be found in significant numbers until the far end of the reactor, and as sludge is wasted from this area, a disproportionate number of nitrifiers compared to heterotrophs are removed (Lijklema, 1973).

A further explanation may be provided involving the parameter of dissolved oxygen. An inhibition could also be taking place with a low oxygen level in the front of the basin due to the low dissolved oxygen levels in primary effluent, the high demand due to carbonaceous loadings there and additional oxygen depression if the return sludge has been allowed to go anaerobic (Hockenbury *et al.*, 1977).

Sheih and Lamotta (1979) report that contact time has an affect on k, (a constant they call the maximum rate of substrate utilization per unit mass of floc). Larger k values were found for smaller contact periods. They found the Michaelis constant K_s was not affected by contact time. The substrate utilization rate was found to be greater for the nitrifiers than the heterotrophs. Intuitively, the longer the heterotrophic and nitrifier growth period, the shorter the contact time for nitrifiers proportionately for the same amount of actual time in the system, because of the nitrifier slower growth. The shorter the detention time, the more sensitive nitrifiers appeared to be in response to changes in the detention times. Small decreases in the

detention time under these conditions must be compensated for with large increases in the MLVSS (Lijklema, 1973). For combined and separate sludge systems, as the hydraulic retention time increases, temperature sensitivity was reported to decrease (Sutton *et al.*, 1975). Hall and Murphy (1985) however, found temperature sensitivity to be independent of hydraulic retention time. The minimum MCRT to avoid the washout phenomena is affected by the DO and the prevalence of mass transport resistance. A high MCRT may be necessary to compensate for low DO and high mass transport resistance (Stenstrom and Song, 1991). In water hyacinth treatment systems, increased aeration and mixing increased ammonia removal rates through nitrification and denitrification. This was thought to increase oxygen transfer across the water surface to the bulk liquid and to decrease mass transport resistance (Weber and Tchobanoglous, 1985).

The Effects of Temperature, Oxygen and pH on Nitrification

According to Lijklema (1973), any and all of the conditions of high temperature, low organic loadings, plug flow conditions, high TSS, steady state conditions and long detention times are considered beneficial conditions for nitrifiers. Temperature affects the nitrification rate through the growth rate. The effect of temperature on the nitrifier growth rate corresponds such that for every 6 degree drop in temperature, the growth rate will halve, making the minimum sludge age for the nitrifiers double (Ekama *et al.*, 1984). Most researchers have adopted a modification of the Arrhenius relationship to describe how this rate changes with temperature. Metcalf and Eddy (1972) show this relationship as:

$$\frac{K_T}{K_{20}} = \theta^{(T-20)} \tag{2.12}$$

where K_T is the reaction rate at the temperature at which the reaction takes place, K_{20} is the reaction rate at 20 degrees C, θ is the temperature activity coefficient, and T is the temperature in degrees C. Shammas (1986) has compiled a list of the temperature activity coefficients (as theta = e^b) found in the literature as they affect the maximum nitrification rate constant k. They range from 0.028 to 0.121 in activated sludge.

Painter (1970) reports that the optimum temperature range for nitrifiers is 30-36 degrees C. Shammas, 1986, found the range of nitrification to be 5-35 degrees C with a maximum at 30 degrees. Wild (1975) studied nitrification over the range of 5-30 degrees C and found that the rate increased with increased temperature over the entire range. Temperature, inside of the range over which nitrification is likely to occur, affects the nitrification rate mainly through the growth rate (Lijklema, 1973). Low temperatures may be compensated for operationally through: design for changes in tank capacity, design for winter conditions, use biological nitrogen removal during the summer months only and alternate methods of nitrogen removal during the winter (Sharma and Ahlert, 1977). Temperature was found to affect the order of reaction of the nitrification reaction, and the use of the Monod equation for modeling was recommended since it may be applied for both first order and zero order reactions. The order of the reaction changed from zero to first order in the temperature range of 10 to 17 degrees C (Randall and Buth, 1984). Temperature was found to affect the nitrification rate more at low hydraulic detention times for combined and separate sludge systems (Sutton *et al.*, 1975). Nitrification can be sustained at lower temperatures with increased mixed liquor concentrations and optimum pH (Wild *et al.*, 1971). Stratton and McCarty (1967) stated that while they did

not see any variation in the Monod half saturation constant with temperature, the changes seen with the substrate utilization constant k were significant. Substrate utilization increased with increased temperature. Shammas (1986) reported that with a MLVSS of 430 mg/L, K_s decreased with increased temperature (4 to 33 degrees C) and pH (7.0 to 8.3). However, with a MLVSS of 3200, K_s values at 25 and 33 degrees C were higher than at 10 and 17 degrees C. Shammas (1986) further found that with the 430 mg/L MLVSS system, pH 8.3 was the critical pH at which K_s was at its lowest for all temperatures tested. For the 3200 mg/L system, however, low K_s values developed at pH 8.3 or 7.0 depending on whether or not the temperature was above or below 17 degrees C. This investigator also found that the maximum rate constant varied logarithmically with temperature from 4 to 33 degrees C in the 430 mg/L system, and from 4 to 25 degrees C for the higher MLVSS concentrations.

Considering the nitrifier organisms separately, their reactions to temperature differ as a result of different optimal and inhibitory ranges. The temperature range over which *Nitrosomonas* experiences inhibition is broader than that of *Nitrobacter*. Although at 20 degrees C the growth rate of *Nitrosomonas* is greater than that of *Nitrobacter*, as the temperature decreases, this situation is reversed and results in concentrations of nitrite not being oxidized to nitrate and accumulating in the medium (Randall and Buth, 1984). A similar situation exists at the upper end of the range. Increased temperatures benefit the growth rate of *Nitrosomonas* over *Nitrobacter* and lead to the build up of toxic nitrite (Quinlan, 1980).

As for other effects, differences in the literature are found. No association between pH and temperature may be assumed in their ability to effect the nitrification rate (Shammas, 1986). Temperature has been shown to be more critical to nitrification activity than a pH outside of the optimum (Randall and Buth, 1984a). Levels of oxygen are more important to the

nitrification rate than the temperature (Okey and Albertson, 1989). Both DO and temperature may be considered to be the principal parameters in the induction of seasonal nitrification in a domestic wastewater treatment plant (Hart *et al.*, 1986).

The optimum pH for nitrification to proceed is between 8 and 9 (U.S. EPA, 1975), 8.2 (Benefield and Randall, 1980), 8.4 (Wild, *et al.*, 1975). An inhibition of greater than 50 per cent of the nitrification rate outside the pH range of 7.0 to 9.8 was found (Wild, *et al.*, 1975). Metcalf and Eddy (1972) recommend a design range of 7.4 to 8.6. Shammas (1986) concludes that there is a wide range of optima published in the literature and states that the general trend is that when pH is decreased, the nitrification rate is decreased.

A hypothesis was developed to describe how pH affects the rate of nitrification (Anthonisen *et al.*, 1976). The degree of ionization of the substrates for the nitrifiers, ammonia and nitrite, is affected by the pH of the medium. When the pH inside the nitrifier organism is lower than that of the surrounding environment, "free ammonia" or "free nitrous acid" diffuses into the cell from the medium, while ionized ammonia and ionized nitrite remain. These organisms are inhibited by the free portions of these substrates' ability to permeate the cells. It is possible to use this free ammonia inhibition and nitrite buildup to shorten the nitrogen removal pathway for highly nitrogenous wastes (Turk and Mavinic, 1989).

The amount of oxygen that will be available to the nitrifiers, as discussed previously, will vary with the amount of oxygen in the bulk liquid and the mass transfer resistance presented by the floc. This amount has been shown to be 4.57 mg oxygen required to oxidize 1 mg of ammonia (Benefield and Randall, 1980). Factors discussed by Stenstrom and Poduska (1980) that affect the amount of oxygen inside the floc include the bulk DO concentration, the size and shapes of flocs, bulk mixing, turbulence and intensity and the oxygen uptake rates of

heterotrophic organisms and nitrifiers in the floc. Not only is oxygen needed in the conversion of ammonia to nitrate but is also required in the system to react with carbonaceous oxygen demanding substances and for endogenous respiration of the biomass. Nitrification was found to be complete at oxygen concentrations of 1.6 mg/L (Beckman, *et al.*, 1972). While oxygen amounts in the bulk liquid might predict effective nitrification, the mass transfer resistance within the floc limits the amounts available to the nitrifiers. Nitrification may be established in a full scale sewage treatment plant through increasing aeration rates and in effect increasing the proportion of the floc volume available to the nitrifiers, thus making their population dispersion more homogeneous (Hart *et al.*, 1986). Oxygen levels of less than 1.0 mg/L were limiting to nitrification in water hyacinth treatment systems (Weber and Tchobanoglous, 1985). This figure was found to be the cutoff point in nitrification pilot plant work. No inhibition was seen at dissolved oxygen concentrations above 1.0 mg/L (Wild, *et al.*, 1971). The most likely D.O. concentration range required for nitrification is concluded to be between 0.5 and 2.0 mg/L (Stenstrom and Poduska, 1980). The lowest concentration necessary to establish nitrification is 0.3 mg/L (Stenstrom and Poduska, 1980).

A Chemical and Microbial Description of Denitrification

Denitrification is the nitrate dissimilation or respiration in which nitrate instead of oxygen is used as the final electron acceptor, and the nitrate is transformed to nitrous oxide or nitrogen gas instead of being incorporated into cells (Painter, 1980). Since this process uses nitrate formed in nitrification and removes it from the system, it is the companion step to nitrification in biological nitrogen removal. Whereas nitrification takes place in an aerated system by

autotrophic bacteria, denitrification for nitrogen removal requires an environment without oxygen. The bacteria which denitrify are facultative and can also utilize oxygen, to greater energy gain. The presence of oxygen therefore inhibits denitrification. Denitrifiers may also perform what is termed nitrate assimilation in which nitrate is transformed to ammonium. This reaction is not typically used unless sufficient ammonium is not present for growth (WPCF, 1983).

Microorganisms which denitrify are classified as chemoorganotrophs, using energy from chemical bonds for growth and metabolism. The reason denitrifiers are termed heterotrophs is that the carbon source of the denitrifiers' energy is organic carbon, in contrast to the nitrifiers reliance upon inorganic carbon. A similarity of nitrifiers and denitrifiers is that their energy source serves as the source of reducing power (Doetsch and Cook, 1973). The electron donor function is more important for the chemolithotrophs (nitrifiers) than the chemoorganotrophs since the former's use of inorganic substrates makes them reduce a more highly oxidized substance (CO_2) to cell material (CH_2O) (Gaudy and Gaudy, 1988). The chemoorganotroph substrate is typically not as highly oxidized. The less oxidized the substrate for the chemoorganotroph, the lower the cellular yield that is realized and the less the sludge production (Gaudy and Gaudy, 1988). Future studies should take place concerning the implications the choice of substrate has on the selection between different genera of denitrifiers (Gaudy and Gaudy, 1988).

Most previous work with denitrification involves the use of methanol as a carbon source. Nitrification has to take place first to supply the nitrate for the denitrification reaction and this typically takes place in the aeration basin. Organic carbon to drive the denitrification reaction

is depleted after this step, and methanol is a relatively cheap and convenient source of carbon to add to an anoxic basin in the treatment train at this point. More current thought, for municipal applications especially (Eckenfelder and Argaman, 1979), includes using the influent organic carbon in the wastewater as the carbon source and adding nitrates to an anoxic basin with a nitrate recycle, prior to aeration. The U. S. EPA Process Design Manual for Nitrogen Control (1975) states that for wastewater organics, the rate of denitrification is about one third less than that obtained using methanol and larger basins must be employed for longer retention times.

Many different genera of bacteria can accomplish denitrification, including, *Pseudomonas, Micrococcus, Archromobacter and Bacillus* (U.S. EPA, 1975). The Water Pollution Control Federation's Manual of Practice for Nutrient Control (1983) additionally lists *Brevibacterium, Enterobacter, Lactobacillus, Paracalobactrum, and Spirillum.* Painter (1970) gives mineral requirements for these bacteria as including sulfate, phosphate, chloride, sodium, potassium, magnesium and calcium ions and trace metal ions as molybdenum, iron, copper and manganese. Inhibitors which interfered with nitrate reductase in the denitrification reaction were metal chelating agents such as potassium cyanide, dithiol, chlorate and copper. Other enzymes were inhibited with chemicals such as p-chloromercuribenzoate, hydrazine, acetaldehyde oxime, pyruvic acid oxime and hydroxylamine (Painter, 1970).

The oxidation reaction performed by the denitrifiers is (Manahan, 1990):

$$\frac{1}{4}\{CH_2O\}+\frac{1}{4}H_2O \leftrightarrow \frac{1}{4}CO_2+H^+ +e^- \qquad (2.13)$$

This step is the same whether the denitrifiers use nitrate or oxygen as the final electron acceptor. The reduction reaction associated with denitrification is (Manahan, 1990):

$$\frac{1}{5}NO_3^- + \frac{6}{5}H^+ + e^- \leftrightarrow \frac{1}{10}N_2(g) + \frac{3}{5}H_2O \tag{2.14}$$

The reduction reaction for aerobic respiration is given by Manahan (1990) as:

$$\frac{1}{4}O_2(g) + H^+ + e^- \leftrightarrow \frac{1}{2}H_2O \tag{2.15}$$

The pathway upon which nitrogen gas is formed from nitrate is (Doetsch and Cook, 1973):

$$NO_3 \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \rightarrow N_2(g)\uparrow \tag{2.16}$$

Since nitrite is formed from nitrate in the conversion process, prior to the formation of nitrogen gas, denitrification may be thought of as a two-step process. As illustrated in equation (2.13), organic matter acts as the electron donor, supplying electrons for the reduction reaction in which the nitrate or the nitrite is the electron acceptor. Turk and Mavinic (1987) have done much work with the possibilities of eliminating the step in nitrification from nitrite to nitrate, the ability to denitrify using nitrite and how to maintain a system in that mode. These researchers list the change in the oxidation state from NO_3^- at +5 to NO_2^- at +3 to nitrogen gas at the 0 state.

The use of oxygen as the electron acceptor instead of nitrate or nitrite produces more energy for the microorganism and will be used preferentially if any is present. Therefore, the presence of oxygen inhibits denitrification and must be avoided for process optimization. The EPA Process Design Manual for Nitrogen Control (1975) shows a comparison of energy yields for nitrate dissimilation versus oxygen respiration for glucose. The equations and indication of energy yields are shown below (U. S. EPA, 1975).

Nitrate Dissimilation, Energy Yield per Mole Glucose, Kcal 570

$$5C_6H_{12}O_6 + 24KNO_3 \rightarrow 30CO_2 + 18H_2O + 24KOH + 12N_2 \quad (2.17)$$

Oxygen Respiration, Energy Yield per Mole Glucose, Kcal 686

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O \quad (2.18)$$

Both nitrate dissimilation and aerobic respiration use the same metabolic pathway, with nitrate replacing oxygen in the electron transport chain (Benefield and Randall, 1980).

The Effect of Temperature, Oxygen and pH on Denitrification

The nitrification reaction destroys alkalinity at the rate of 7.14 milligram calcium carbonate for each milligram of ammonia changed to nitrate. The denitrification reaction produces bicarbonate and decreases carbonic acid concentration, though only at approximately half the rate of destruction by nitrification, with 3.57 milligram bicarbonate formed for each

milligram nitrate reduced (U.S. EPA, 1975). This phenomena usually will maintain sufficient alkalinity and the optimum pH for nitrification to take place in low alkalinity wastewaters (those below 200 mg/L as $CaCO_3$) if denitrification is included in the treatment train (Ekama *et al.*, 1984). The design or optimum range of pH for denitrification is given as 6.5 to 7.0 (Metcalf and Eddy, 1972), 7.0 to 8.0 (WPCF, 1983), 7.0 to 7.5 (U.S. EPA, 1975). At a pH of 7.0, oxygen inhibition was found to be greatest, while at a pH of below 6.0, oxygen concentration had little effect on the denitrification rate (Painter, 1970).

Although oxygen is an inhibitor of denitrification, some researchers discussed by Painter (1970 to include Skerman and Macrae, 1957 a,b) and reported by the WPCF Manual of Practice (1983) and the U. S. EPA Process Manual for Nitrogen Control (1975) have noted the phenomena of aerobic denitrification and explain it as the result of an oxygen gradient in the medium which results in some fraction of the microbial mass seeing an absence of dissolved oxygen.

Randall (1984) discussed a concept of oxygen "recovery" through denitrification. Since the number of electrons lost in the denitrification reaction from nitrate to nitrogen gas is 5, and 8 electrons are required to form the nitrate through nitrification, 5/8 or 62.5 per cent retrieval is achieved. This recovery may be presented through the comparison of stoichiometric equivalence amounts of nitrate and oxygen. One milligram of nitrate is equivalent to 2.86 milligrams of oxygen, while for the production of nitrate from ammonia, 4.57 milligrams of oxygen are required per milligram nitrogen. The division of 2.86 by 4.57 produces the fraction 0.625 which can be converted to 62.5 per cent (Ekama *et al.*, 1984). Since the oxygen required for nitrification is about 25 to 35 per cent of the total amount required for sewage treatment, the addition of denitrification to the treatment train will reduce the total oxygen requirement for

municipal wastewater treatment by 15 to 20 percent (Ekama *et al.*, 1984).

The WPCF Manual of Practice (1983) reports that the denitrification reaction has been observed at a temperature range of 0-50 degrees C. No difference in denitrification rates was noted at 20 or 30 degrees C., but decreased biological activity was evidenced at 10 degrees C (Stensel *et al.*, 1973). Benefield and Randall (1980) cite work by Dawson and Murphy (1973), that provides a temperature correction for the specific denitrification rate between 5 and 30 degrees C. This work provides a lower limit of 3 degrees C for the denitrification reaction.

$$(q)_{DN}=0.07(1.06)^{T-20} \qquad (2.19)$$

where,

$(q)_{DN}$ = the specific rate of denitrification $\left(\frac{\text{mg/L } NO_3\text{-N removed}}{\text{mg/L MLVSS-h}}\right)$, and

T = operating temperature.

Most researchers have adopted a modification of the Arrhenius relationship to describe how this rate changes with temperature. This relationship was shown as equation 2.12 in the previous section. Metcalf and Eddy (1972) gives values for theta, the temperature coefficient, as 1.14 to 1.16 for denitrification. Eckenfelder and Argaman (1979) state that theta values for suspended growth systems are typically 1.07 to 1.20.

Denitrification Kinetics

Stensel *et al.* (1973) citing work performed by Moore and Schroeder (1971) stated that since nitrite does not build up during the denitrification reaction in a completely mixed reactor, the reaction may be thought of as a single step for design purposes. Moore and Schroeder,

(1971) who assumed first order kinetics, concluded that the denitrification rate was limited by the nitrite and nitrate concentrations.

Stensel *et al.* (1973) utilized a Monod relationship to model the denitrification reaction. These investigators expected that the rate limiting species would be methanol (the organic carbon source) or nitrate, assuming sufficient nutrients are available. The results of their study indicated that unless the nitrate values approached zero, the system will be limited by the amount of the organic carbon source.

Moore and Schroeder (1970) and Engberg and Schroeder (1975) worked with nitrate limited systems. Systems operated for low nitrate effluents may be predicted to be nitrate limited. For reasons of economy, it is desirable to minimize the methanol added to the system (Engberg and Schroeder, 1975). However, no effect due to methanol was seen upon the denitrification rate at concentrations greater than 5 mg/L (Moore and Schroeder, 1970). The EPA Process Design Manual for Nitrogen Control (1975) indicates that with 1 mg/L of methanol in the effluent, 90% of the maximum denitrification rate may be maintained. Panzer (1984) used a first order relationship in his work using oxygen uptake rates to predict nitrogen removal rates from the system, but found that the Monod model would have been better, based on the type of results obtained. A combined expression for Monod-type kinetics assuming either nitrate or the organic carbon source to be limiting is (U. S. EPA, 1975):

$$\mu = \hat{\mu}_D \left(\frac{D}{K_D + D} \right) \left(\frac{M}{K_M + M} \right) \tag{2.20}$$

where,

$\hat{\mu}_D$ = the maximum rate of denitrifier growth at temperature, T, and pH,

μ = the observed rate of denitrifier growth affected by nitrate, methanol, T and pH,

D = concentration of nitrate nitrogen, mg/l,

K_D = half saturation constant, mg/l NO_3-N,

M = concentration of methanol, mg/l, and

K_M = half saturation constant, mg/l of methanol.

Since neither nitrate nor organic carbon source is expected to be limiting in a treatment system operating for nitrogen removal, the system is virtually at zero order kinetics (WPCF, 1983) (Eckenfelder and Argaman, 1979).

Alleman and Irvine (1980) reported a 92 per cent loss of total nitrogen using no added carbon source in a Sequencing Batch Reactor (SBR); Palis and Irvine in 1985 found 86-94 per cent removal using a concentrated BOD feed stock for a carbon source in an SBR.

Sludge Production

A high rate process that operates at a low mean cell residence time or high F: M ratio produces a maximum of sludge (Sherrard, 1984). This was characteristic of the York River Demonstration Project. Due to the low hydraulic retention times and the high F:M experienced at the plant, one would expect to find high sludge production and a larger portion of active biomass.

In addition to these process considerations, other characteristics attributable to the BNR configuration affect the sludge production. Comparison of two identically operated pilot scale activated sludge reactors, one set up in a conventional mode and the other configured for biological nutrient removal, indicated that the yield coefficients (Y) were the same. However, the endogenous decay rate, b, was lower for the BNR process than the conventional one,

explaining why greater solids concentrations are seen at BNR plants (McClintock *et al.*, 1992). Additionally, for a BNR plant operated in the VIP mode, the MLVSS concentrations are greater in the anoxic and aerobic sections than in the anaerobic section because of the recycle configurations. In this case, even though the aerobic volume is 50% of the total volume, the aerobic mass fraction is always greater than 50% (McClintock *et al.*, 1993). The use of fixed film media in activated sludge basins (e.g., Rotating Biological Contactors or RBCs) has been proposed to decrease the amount of secondary clarification volume necessary to settle the otherwise higher mixed liquor concentrations from BNR plants (Sen *et al*, 1992). Large amounts of biomass are maintained on the media and are not suspended in the mixed liquor. This increased total biomass concentration allows for nitrification in a decreased aerobic volume than required for a conventional plant; however, greater total volume is required to accomplish nitrification in the BNR plant. The need for the greater total volume becomes more pronounced at lower temperatures (Randall *et al.*, 1991).

The growth yield Y is a measure of the biomass produced per substrate consumed and may be calculated to describe sludge production. The growth yield Y has been determined to be dependent on (1) the carbon source and nutrient elements' oxidation state (2) substrate polymerization (3) metabolic pathways (4) growth rate (5) physical characteristics present in the growth vessel (Ribbons, D. W, 1970, in Metcalf and Eddy, 1972). The true yield may be calculated through the linear relationship:

$$\left(\frac{dX}{dt}\right)_g = Y_t\left(\frac{dS}{dt}\right)_u - K_d X \qquad (2.21)$$

where, dX/dt = the growth rate of the biomass, mass/volume-time, dS/dt = the removal over time of substrate (S) by the biomass, mass/volume-time, K_d = the decay coefficient, 1/time,

and X = the viable biomass concentration (mass/volume) (Benefield and Randall, 1977).

A net growth rate, Y_{obs}, may be determined from the following relationship:

$$\left(\frac{dX}{dt}\right)_g = Y_{obs}\left(\frac{dS}{dt}\right)_u. \tag{2.22}$$

Equation 2.22 describes the observed yield after the decay coefficient has been subtracted and therefore may be expected to be less than the true yield (Benefield and Randall, 1980). At the low sludge ages seen at the York River Demonstration Project, however, the decay coefficient is not expected to be significant. Its effect is seen more at higher sludge ages (Randall *et al*, 1991).

A mass balance for the net rate of microbial mass in the system may be written, incorporating the positive and negative influences to the system. Expressions for $\left(\frac{dX}{dt}\right)_g$ may be derived, using the assumption of system equilibrium where the sludge wasting rate is equal to the growth in the system (Lawrence and McCarty, 1970). In this event, the sludge wasted plus the solids lost in the effluent will equal the growth rate of the biomass. The substrate utilization may be calculated by subtracting the effluent substrate from the influent substrate. Y_t or Y_{obs} may then be calculated from these two expressions.

Biological Nitrogen Removal

The single sludge removal of nitrogen using biological processes is comparatively inexpensive when compared to physical-chemical methods (Barth, 1978) (Morales *et al.*, 1991) such as breakpoint chlorination. Other methods include nitrogen removal by air stripping or tertiary removal (separate stage nitrification and denitrification activated sludge, requiring organic

carbon addition) (U. S. EPA, 1975). Nitrification is performed in an aerobic basin with activated sludge. This basin contains high levels of nitrate, oxidized by the nitrifiers using ammonia and low levels of biochemical oxygen demand (BOD) and soluble total Kjeldahl nitrogen (TKN). TKN hydrolyses to form the ammonia used in the nitrification process. BOD is low because it is consumed in the heterotrophic energy reaction also taking place in this basin.

Nitrate is removed by denitrification in a non-aerated activated sludge basin. BOD from influent waste may be available for use as the organic carbon source if the anoxic basin precedes the aerobic one. Nitrate instead of oxygen is the oxidant. The nitrate is reduced to nitrogen gas (Panzer, 1984).

The earliest work in single sludge biological nitrogen removal was done by Ludzack and Ettinger in 1962 (Ekama *et al.*, 1984). The process configuration consisted of influent waste introduced to an anoxic reactor, then flowing to an aerobic reactor, with clarification providing for sludge recycle to the aerobic reactor. Conclusions were drawn that nitrification and denitrification using the influent BOD as the energy source for denitrification was possible, producing increased quality effluent with lower air needs (Ludzack and Ettinger, 1962).

Wurhmann in 1964 developed a process similar to Ludzack and Ettinger's, but with endogenous respiration providing the carbon source for the denitrification reaction. In this mode, the aerobic reactor precedes the anoxic reactor and the return activated sludge is again recycled to the aerobic reactor from the clarifier. Denitrification capability is lessened because the energy released through the endogenous respiration process is so low (Ekama *et al.*, 1984).

Barnard modified the Ludzack and Ettinger process configuration, recycling the clarifier sludge to the anoxic basin instead of the aerobic reactor. Another recycle was added from the aerobic to the anoxic zone. All of the waste stream is not denitrified because only the portion

recycled from the aerobic to the anoxic basins undergoes denitrification (Ekama *et al.*, 1984). To overcome this, Barnard proposed the Bardenpho system in which an anoxic basin precedes an aerobic basin, and this configuration is repeated, prior to clarification. Sludge is recycled from the effluent end of the first aerobic basin back to the first anoxic basin. The sludge return line recycles sludge from the clarifier to the first anoxic basin (Lamb, 1988). The second anoxic basin provides for complete denitrification and the second aeration basin is present to strip nitrogen gas from the mixed liquor and thus prevent rising sludge in the clarifier (Ekama *et al.*, 1984).

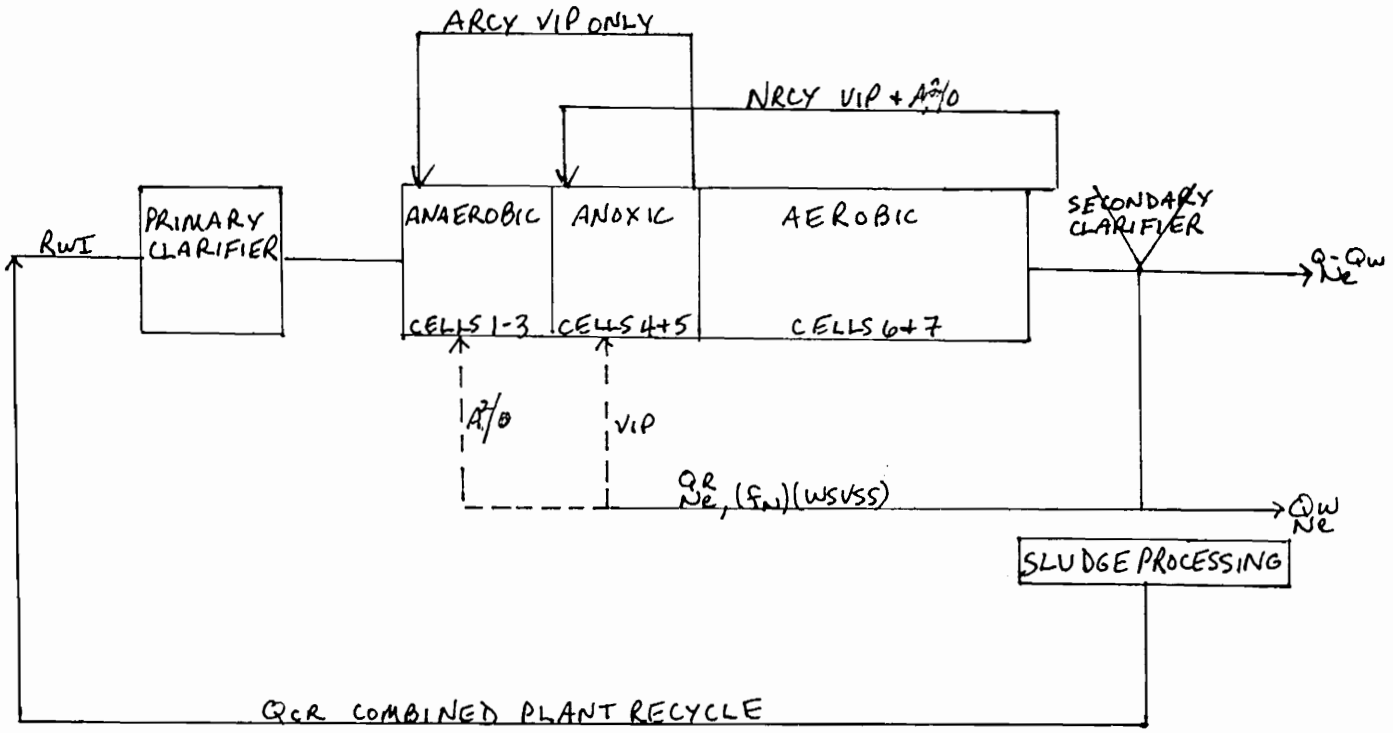
The A^2/O process marketed by Air Products and Chemicals, Inc., is a modification of the A/O process, configured to remove both nitrogen and phosphorus. This system includes multiple process basins for reaction to avoid short circuiting. Anaerobic basins (for phosphorus removal) precede anoxic basins, aeration basins and clarification. An effluent recycle is provided from the second aeration effluent line to the first anoxic basin for denitrification. Sludge is recycled from the clarifier to the first anaerobic basin (Lamb, 1988).

The University of Cape Town (UCT) process addresses concerns with incomplete phosphorus removal with the A^2/O process. This mode of operation consists of an anaerobic basin followed by an anoxic basin, aeration reactor and clarification. Three recycles are used. Denitrified sludge from the anoxic basin (without nitrate which interferes with the phosphorus removal process) is returned to the anaerobic basin. Part of the aeration effluent is recycled to the anoxic basin for denitrification while the return sludge from the clarifier is also introduced back to the anoxic reactor (Lamb, 1988).

The Virginia Initiative Project (VIP) process is a high loading rate version of the UCT process. It was developed jointly by CH2M Hill and Hampton Roads Sanitation District

(HRSD) for smaller sludge ages and greater Food : Microorganism ratios (F:M) than the UCT process and thereby requires smaller tankage (CH2M Hill, 1987). Reactors are divided into two or three stages to profit kinetically from higher loading rates and multiple complete mix reactors in series, as in the A^2/O configuration (Waltrip, 1990).

The York River Demonstration Project was operated for biological nutrient removal from August 1986 to 1990. Nitrogen removal began with the onset of nitrification in July 1987; in August of 1987, the plant began to be operated as an A^2/O process with the initiation of the nitrate recycle. VIP operation of the plant began in March 1988 (Randall, 1988). Please refer to Figure 2.1 for illustration of the activated sludge treatment train under both modes of operation.



YORK RIVER DEMONSTRATION PROJECT
1986 through 1989

FIGURE 2.1

CHAPTER 3
MATERIALS AND METHODS

Background

Staff of the Hampton Roads Sanitation District (HRSD) gathered data and samples at the York River Treatment Plant of HRSD in 1987 and 1988 and delivered the samples to HRSD's North Shore Laboratory for analysis. Samples were preserved, stored and analyzed within holding times according to EPA guidelines (Federal Register Vol. 44 No. 244 - Dec. 18, 1979). Some sampling necessary for Virginia Pollutant Discharge Elimination System (VPDES) reporting and related process control was performed daily. These types of analyses included 5-day Biochemical Oxygen Demand (BOD_5), Total Suspended Solids (TSS) and Total Volatile Suspended Solids (TVSS), and were performed on flow weighted 24-hour composites prepared at the treatment plant. BOD_5, TSS and TVSS analyses were performed at the laboratory according to the methodology presented in Chemical Methods for the Analysis of Wastewater (U. S. EPA, 1979).

Sampling related to the York River BNR Demonstration Project, such as the automated nutrient analyses, was performed for three nonconsecutive days per week in 1987 and twice a week in 1988. Grab samples were taken between 8 and 10 A.M. The samples were preserved according to EPA standards, as above. Automated methods for the determination of nutrients (Total Kjeldahl Nitrogen [TKN], Ammonia [NH_3], Nitrate/Nitrite [NO_x], Nitrate [NO_3]) were developed from the Technicon instruction manual for autoanalyzer analysis.

The York River BNR Demonstration Project continued for four years and the activated sludge process was operated in three BNR modes. The modes of operation and the time span of operation are summarized in Table 3.1.

Process performances were evaluated by comparing biosolids generation and substrate

Sludge Fraction and Basin Volumes
for Selected Time Periods

Table 3.1

York River Demonstration Project
Hampton Roads Sanitation District

Operational Mode	A/O Mode			No Anoxic Zone, Complete Nitrification	A^2/O Mode	VIP Mode		
Time Period	11/1-11/30/86	2/10-3/31/87	5/1-5/31/87	7/1-7/31/87	9/1-10/31/87	4/1-4/30/88	7/1-8/31/88	7/1-7/31/89
Sludge Mass Fraction (Anaerobic, Anoxic, Aerobic)	50-0-50	33-0-67	33-0-67 from 5/1-5/14 25-0-75 from 5/15-5/31	25-0-75	25-25-50	25-25-50	25-25-50	25-25-50
Volume Anoxic + Aerobic Zones, MG	0 + 0.674	0 + 0.903	0 + 0.903 0 + 1.01	0 + 1.01	0.337+ 0.674	0.337+ 0.674	0.337+ 0.674	0.337+ 0.674
Nitrate Recycle (NRCY) Flow, MGD	None	None	None	None	7.26	7.26	4.00	7.49
Return Sludge Recycle (RAS) Flow, MGD	4.36	8.32	7.83	9.59	5.58	7.84	5.98	7.54
Sludge Age (Eqn 3.20) (days)	3.0	3.0	3.7	7.5	6.9	4.3	4.2	4.1

removal rates in addition to nitrification and denitrification rates. The months of July, 1987 through October, 1987 were selected for nitrification and denitrification calculations in order to compare the A/O and the $A^2/0$ processes. The nitrate recycle was not initiated until August, 1987 and, therefore, during July, 1987 the plant was technically under the A/O process and not the A^2/O mode of operation.

The criteria used to judge if nitrification was occurring was the occurrence of minimal values of ammonia in the aeration effluent. A state of incomplete nitrification was judged to exist if the effluent ammonia was consistently higher than 1 mg/l, or the minimal aeration effluent ammonia values did not consistently turn up in every sample. Nitrification at the plant began during the month of June, 1987, and was complete by July, 1987. Complete nitrification continued through October, 1987, with some minor fluctuations during the last weeks of August and October.

In order to study nitrification and denitrification in the VIP process (from "Virginia Initiative Plant"), data from the months April, 1988 through July, 1988 were evaluated. Nitrification was established during the month of April, 1988, but did not become complete until the last week in April. Complete nitrification continued through May and June, but fluctuated during the beginning and end of July, 1988, although it was steady during the middle of the month. The relative stability of the nitrification process and other operational parameters during these months led to their selection for analysis and comparison.

Process control and nutrient study data that were collected were entered into the nitrification and denitrification equations to compute daily values. Monthly values are averages of the approximately 13 daily values for the months of 1987 discussed (A/O and A^2/O operation) and the approximately 9 daily values for the 1988 months (VIP mode of operation).

The periods of November, 1986, February 10 through March 31, 1987, May, 1987, July, 1987, September 1 through October 31, 1987, were selected for the biosolids generation analysis and comparison. November, 1986, February 10 through March 31, 1987, and May, 1987 were selected to analyze the A/O operating mode. July, 1987 was of interest because the plant had no anoxic zone and was nitrifying completely. The data for the period from September 1 through October 31, 1987 is descriptive of the A^2/O mode of operation, while the periods April, 1988, July 1 through August 31, 1988 and July, 1989 were selected for the VIP operation.

Averages of process control data for the time periods discussed in the biosolids study were used to calculate the observed yield, sludge wasted, biomass under aeration, pounds of BOD_5 consumed, specific growth rate and fractional substrate utilization rate.

Manipulation of data was performed with Lotus 1-2-3, version 2.0, and ASEASYAS, version 4.0, on an IBM PC compatible computer.

Explanation of Terms

The acronyms, symbols and abbreviations used in this and subsequent chapters are explained in Appendix I.

Mass Balance Equations for the System

A mass balance of biomass for the activated sludge system was performed using equation 3.1 for calculation of heterotrophic bacterial sludge production (Please refer to Figure 2.1 for a process flow diagram):

$$\left(\frac{dX}{dt}\right) V_a = X_{PCE}Q_{PCE} + \left(\frac{dX}{dt}\right)_g V_a - (Q_w X_w) - [(Q_{PCE} - Q_w)(X_e)]. \qquad (3)$$

At equilibrium, $\left(\frac{dX}{dt}\right) V_a = 0$ and since $X_{PCE} = 0$,

$$\left(\frac{dX}{dt}\right)_g V_a = (Q_w X_w) + [(Q_{PCE} - Q_w)(X_e)]. \quad (3.1)$$

The net growth of biomass in the system ultimately is equal to the waste sludge flow times the biomass leaving as waste sludge plus the solids discharged in the treatment plant effluent, quantity multiplied by the effluent flow. The effluent flow is equal to the primary clarifier effluent flow minus the waste sludge flow.

Calculations for nitrification and denitrification rates were determined for spreadsheet insertion by a nitrogen balance, as follows (Figure 2.1):

$$\left(\frac{dN}{dt}\right) V_a = N_{PCE} Q_{PCE} - [N_e(Q_{PCE} - Q_w) + N_e Q_w] - \left(\frac{dNO_3 - N}{dt}\right)_{denit} - \left(\frac{dNH_3 - N}{dt}\right)_{net\ growth}. \quad (3.2)$$

The net rate of change in nitrogen $\left(\frac{dN}{dt}\right)$ equals the influent nitrogen ($N_{PCE}Q_{PCE}$) minus the nitrogen leaving in the secondary clarifier effluent $\left(N_e(Q_{PCE} - Q_w) + N_e Q_w\right)$ minus the soluble nitrogen leaving with the waste sludge ($N_e Q_w$) minus the loss due to denitrification $\left(\frac{dNO_3 - N}{dt}\right)_{denit}$ and minus the uptake by heterotrophs and nitrifiers (assimilation) $\left(\frac{dNH_3 - N}{dt}\right)_{net\ growth}$. The assimilation of nitrogen by the nitrifiers is assumed negligible since the proportion of nitrifiers in the bacterial population is so low. Stratton and McCarty (1967) estimated this figure to be 4% of the total nitrogen oxidized. At equilibrium operation, nitrification should produce a conversion of ammonia to nitrate; denitrification should lead to a net loss of nitrate.

The influent nitrogen, the waste sludge nitrogen and the nitrogen leaving in the secondary clarifier effluent may be measured directly. Nitrification must precede denitrification;

values for both rates were calculated according to equations which follow. These rates as calculated were compared for the months of different modes of operation of the treatment plant during the nutrient removal phase of treatment. The July, 1987 data represent nitrification without nitrate recycle. These data were contrasted with August, September, and October, 1987 in which the plant operated in the A^2/O mode, *i.e.*, nitrifying and with nitrate recycle. The 1988 data represent operation in the VIP mode. In the VIP mode, waste sludge was returned to the anoxic zone of the mixed liquor basins instead of the head of the tanks, at the anaerobic section. The nitrate recycle continued, from the effluent of the aeration basin to the influent to the anoxic zone. An additional recycle carried denitrified mixed liquor from the effluent of the anoxic zone to the influent to the anaerobic zone.

The uptake of nitrogen by heterotrophs and nitrifiers (assimilation) in pounds per day is equal to the waste sludge flow (in MGD) times the volatile suspended solids (VSS) of the waste sludge in mg/l, times the fraction of nitrogen in the waste sludge VSS (WSVSS). Heterotrophic uptake of nitrogen was calculated in g/day, as follows:

$$\left(\frac{d\mathrm{NH_3{-}N}}{dt}\right)_{\text{net growth}} = (\mathrm{F}_n)\Big(\text{ WSVSS mg/L}\Big)\Big(\mathrm{Q}_w \text{ MGD}\Big)\Big(8.34\frac{\text{lb L}}{\text{mg MG}}\Big)\Big(453.6 \text{ g/lb}\Big), \qquad (3.3)$$

where, F_n = the fraction of nitrogen in the waste sludge or return sludge mgN/mgVSS, and may be represented by

$$\mathrm{F}_n = \frac{\mathrm{N}_{\mathrm{WSVSS}}}{\mathrm{WSVSS}}. \qquad (3.3a)$$

This fraction was assumed to be 0.105 as determined for the VIP Pilot Study by CH2M Hill for HRSD (CH2M Hill, 1987). At equilibrium state, growth is equal to the wasting rate.

The daily WSVSS was derived from the daily total suspended solids value in mg/l for the return aeration sludge (RAS) times the daily fraction of volatile suspended solids determined for the mixed liquor (or aeration effluent) suspended solids. The monthly figure is an average of all the daily values for the month.

The sludge age (θ_c), biological solids retention time (BSRT), and mean cell retention time (MCRT), were defined to be equal for this thesis. Equation 3.4 and 3.4a is taken from the HRSD process control calculation. The daily sludge age in days (for the different modes of operation),

$$\theta_c \text{ VIP} = \frac{(.337 \text{ MG}*8.34*\text{ANA TSS})+((.337 \text{ MG}+.674 \text{ MG})*8.34*\text{ARE TSS})}{(8.34* Q_w * \text{WS TSS})+(8.34 * \text{RWI Q} * \text{FNE TSS})} \quad (3.4)$$

$$\theta_c \text{ A/O and A}^2\text{/O} = \frac{((.337 \text{ MG}+.337 \text{ MG}+.674 \text{ MG})*8.34*\text{ARE TSS})}{(8.34* Q_w * \text{WS TSS})+(8.34 * \text{RWI Q} * \text{FNE TSS})} \quad (3.4a)$$

The basin volumes used in the calculations were 0.337 million gallons each for the anaerobic and anoxic basins, and 0.674 million gallons for the aeration basin. For VIP, the aeration effluent TSS were equal to the anoxic TSS, but not the anaerobic TSS. In the A^2/O mode of operation, the TSS concentration in the three basins were approximately the same.

The nominal hydraulic retention time, (HRT), is the amount of time the flow remains in the anaerobic, anoxic and aeration basins.

$$\text{HRT (Days)} = \frac{\text{Basin Volume}}{\text{Influent Flow Rate}} = \frac{\text{Million Gallons}}{\text{Million Gallons per day}} \quad (3.4b)$$

Mass Balance Equations for Nitrification

Total System Nitrification was calculated in the units of grams of nitrogen per day (gN/day) using equation 3.5, below.

$$\left(\frac{dNH_3}{dt}\right)_{\text{nitrified}} = \left((PCE_{TKN})(Q_{PCE})\left(3.785\frac{g\ \ L\ 1x10^6gal}{1000mg\ MG}\right)\right)$$

$$- \left((SCE_{TKN})(Q_{PCE} - Q_w)\left(3.785\frac{g\ \ L\ 1x10^6gal}{1000mg\ MG}\right)\right) \qquad (3.5)$$

$$- \left(\text{The uptake of N by biomass}\right)$$

where the conversion factor $\left(3.785\frac{g\ \ L\ 1x10^6gal}{1000\ mg\ MG}\right)$ yields g/day when multiplied by mg/L × MGD and the uptake of nitrogen by the biomass is calculated as in equation 3.3.

Completeness of nitrification is gaged by the proportion of nitrification to the amount of ammonia available for nitrification.

$$\left(\frac{dNH_3}{dt}\right)_{\text{available for nitrification}} = \left((PCE_{TKN})(Q_{PCE})\left(3.785\frac{g\ \ L\ 1x10^6gal}{1000mg\ MG}\right)\right)$$

$$- \left((SCE_{\text{Soluble Organic N}})(Q_{PCE} - Q_w)\left(3.785\frac{g\ \ L\ 1x10^6gal}{1000mg\ MG}\right)\right) \qquad (3.5a)$$

$$- \left(\text{The uptake of N by biomass}\right)$$

where the uptake of nitrogen by the biomass is calculated as in equation 3.3 and the Soluble Organic Nitrogen in the SCE is the difference between the Soluble SCE TKN and the Soluble SCE Ammonia. Since the Total and not Soluble SCE TKN was measured, an estimate of the Soluble SCE TKN was calculated from the following equation:

Soluble SCE TKN = Total SCE TKN − (3.5.a.1.)

(SCE TSS × decimal fraction aeration volatile solids × 0.105 Nitrogen fraction of the VSS (3.3a)).

The nitrification that occurred in only the aeration zone was calculated using equation 3.6.a, by subtracting the nitrification that occurred in the secondary clarifier from the system nitrification.

$$N_{\text{nit-aer}} = N_{nitrified} - SC_{nitrified} \tag{3.6a}$$

where the system nitrification ($N_{nitrified}$) was calculated as in 3.5 and the secondary clarifier nitrification ($SC_{nitrified}$) was calculated as below,

$$N_{\text{nit-SC}} = \left(SCE_{NO_x} \times (Q_{PCE} - Q_w) \times \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right)$$

$$- \left((Q_{PCE} + Q_r + Q_{NRCY})(\text{CELL } 7_{NOX}) \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right) \tag{3.6b}$$

$$+ \text{ Secondary Clarifier Denitrification,}$$

and where the Secondary Clarifier Denitrification was calculated as shown below:

$$\left(\frac{dNO_3 - N}{dt}\right)(V_{AER}) \frac{\text{g N}}{\text{day}} = \left((Q_{PCE} + Q_r)(\text{Cell } 7_{NOX}) \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right) -$$

$$\left((Q_{PCE} - Q_w)(SCE_{NOX}) \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right) - \left((Q_r)(SCE_{NOX}) \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right) \tag{3.6c}$$

$$- \left((Q_w)(WS_{NOX}) \left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right) \right)$$

The aeration nitrification or ammonia nitrified in the aeration basin, expressed in mg/L-hr, was converted from the units of gN/day as shown in equation 3.6d.

$$\text{Ammonia nitrified in the aeration basin, mg/l-hr} = \left(\frac{\text{gN}}{\text{day}}\right)_{\text{nitrified}} \times \frac{1000\text{ mg}}{\text{g}} \times \frac{\text{day}}{24\text{ hr}} \times \frac{1}{2550000\text{L}_{\text{Vol Aerobic Zone}}}. \quad (3.6d)$$

Equation 3.6d was shortened to:

$$\text{Ammonia nitrified in the aeration basin, mg/l-hr} = \left(\frac{\text{g}}{\text{day}}\right)_{\text{nitrified}} \times \left(1.63 \times 10^{-5}\ \frac{\text{mg day}}{\text{g hr L}}\right). \quad (3.6.d1)$$

The total system nitrification was alternatively calculated as the nitrate appearance in the system, expressed as gN/day. This equation was found to be analytically equal to the total system nitrification calculated as the ammonia disappearance in the system, equation 3.5. See Appendix II.

$$\text{Total ammonia converted to nitrate} = \left(\text{SCE}_{\text{NO}_x} \times (Q_{\text{PCE}} - Q_w) \times \left(3.785\frac{\text{g L } 1\text{x}10^6\text{gal}}{1000\text{ mg MG}}\right)\right) - \left(\text{PCE}_{\text{NO}_x} \times Q_{\text{PCE}} \times \left(3.785\frac{\text{g L } 1\text{x}10^6\text{gal}}{1000\text{ mg MG}}\right)\right) + (\text{system denitrification}). \quad (3.7)$$

The system denitrification was calculated as expressed in equation 3.9.

The ammonia nitrified in the aeration basin per unit biomass, or, the specific nitrification rate, expressed as $\frac{\text{g NH}_3}{\text{g MLVSS-day}}$, was calculated using equations 3.8 and 3.8a, and

the volume of the aerobic zone, where nitrification occurred. The MLVSS concentration value used in 3.6e was that of the aeration portion of the biological treatment basin.

$$\left(\frac{\text{gN}_{\text{nitrified}}}{\text{gMLVSS-d}}\right) =$$

$$\frac{\text{g}}{\text{day}} \times \frac{\text{L}}{\text{mg}}\ \text{MLVSS}_a \times \frac{\text{gal}}{3.785\text{L}} \times \frac{1000\text{mg}}{\text{g}} \times \frac{1}{674000\ \text{gal}_{\text{Vol aerobic zone}}}. \qquad (3.8)$$

The quantity $\left(3.92 \text{ x } 10^{-4}\ \frac{\text{mg}}{\text{g L}}\right)$ is generated in the unit analysis above so that 3.8 may be shortened to

$$\text{gN}_{\text{nitrified}}/\text{grams MLVSS-day} = (\text{gN/day}\ /\text{MLVSS}_a) \times \left(3.92 \text{ x } 10^{-4}\ \frac{\text{mg}}{\text{g L}}\right). \qquad (3.8a)$$

Mass Balance Equations for Denitrification

System denitrification was calculated as follows:

$$\left(\frac{d\text{NO}_3\text{-N}}{\text{dt}}\right)_{denit} = \text{N}_{\text{PCE}}\text{Q}_{\text{PCE}} - [\text{N}_e(\text{Q}_{\text{PCE}} - \text{Q}_w) + \text{N}_e\text{Q}_w] - \left(\frac{d\text{NH}_3\text{-N}}{dt}\right)_{\text{net growth}}, \qquad (3.9)$$

where the biomass uptake of nitrogen, $\left(\frac{d\text{NH}_3\text{-N}}{dt}\right)_{\text{net growth}}$, is described in equation (3.3). The loss of nitrogen due to total system denitrification and volatilization $\left(\frac{d\text{NO}_3\text{-N}}{\text{dt}}\right)_{denit}$, is equal to the nitrogen entering the plant ($\text{N}_{\text{PCE}}\text{Q}_{\text{PCE}}$) minus the nitrogen leaving in the secondary clarifier effluent $[\text{N}_e(\text{Q}_{\text{PCE}} - \text{Q}_w)]$ minus the nitrogen leaving in the waste sludge flow (N_eQ_w) minus the uptake by heterotrophs and nitrifiers (assimilation)$\left(\frac{d\text{NH}_3\text{-N}}{dt}\right)_{\text{net growth}}$. Volatilization was assumed to be zero.

The spreadsheet equation for total system denitrification, expressed in gN/day, was:

$$\frac{dN}{dt} = \text{Total Nitrogen (PCE)} - \text{Total Nitrogen (SCE)} - \text{Nitrogen Uptake of the Biomass} \quad (3.10)$$

where,

$$\text{Total Nitrogen (PCE)} = \left((\text{PCE}_{NO_x} + \text{PCE}_{TKN}, \text{mg/L})(Q_{PCE})\left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) \quad (3.10.a)$$

and

$$\text{Total Nitrogen (SCE)} = \left((\text{SCE}_{NO_x} + \text{SCE}_{TKN}, \text{mg/L})(Q_{PCE} - Q_w)\left(3.785 \frac{\text{g L } 1\text{x}10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) \quad (3.10.b)$$

and the nitrogen uptake of the biomass was calculated with equation 3.3.

The unit conversion for system denitrification was performed as follows,

$$\left(\frac{g}{day}\right)_{\text{denitrified}} = \frac{mg}{L} \times \frac{MG}{day} \times 3.785 \frac{L}{gal} \times \frac{g}{1000 \text{ mg}} \times \frac{10^6 \text{gal}}{MG}. \quad (3.10.c)$$

The nitrogen uptake of the biomass was calculated according to equation 3.3.

Alternatively, the system denitrification was calculated summing the separate basin denitrification values in Equation 3.11, below. The basin denitrification values are calculated in 3.12 and 3.12 a, 3.13, 3.13a and b, 3.14, 3.15 and 3.15a, and 3.6c as explained below.

$$\text{System Denitrification} = \text{ANA}_{denit} + \text{ANX}_{denit} + \text{AER}_{denit} + SC_{denit}. \quad (3.11)$$

The process of denitrification was examined as to location in the system and quantification of nitrogen loss in the system. The loss in the anaerobic basin was quantified for August through October, 1987 for the A^2/O system and April through July, 1988 for the VIP system using equations 3.12 and 3.12a, respectively. The calculations were different due to

differences in the A^2/O and the VIP processes. In the A^2/O process, the return sludge recycle returned sludge to the anaerobic basin and there was no denitrified recycle. In the VIP process, sludge was returned to the anoxic zone and the denitrified recycle recirculated sludge from the anoxic basin to the anaerobic basin. It was decided to use zero for the anaerobic effluent (Cell 3) since nitrate will not persist under anaerobic conditions when BOD is present, and thus it is reasonable to assume complete nitrate removal in the anaerobic zone (Randall, 1993a).

$$\left(\mathrm{ANA}\right)_{\text{denit } A^2/O} = \left(\frac{d\mathrm{NO_3-N}}{dt}\right)\left(V_{\text{ANA-}A^2/O}\right)\frac{\text{g N}}{\text{day}} =$$

$$= \left((Q_{PCE})(PCE_{NOX}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) \tag{3.12}$$

$$+\left((Q_r)(RAS_{NOX})\left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) - \left((Q_{PCE}+Q_r)(\text{CELL } 3_{NOX})\left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right).$$

$$\left(\mathrm{ANA}\right)_{\text{denit VIP}} = \left(\frac{d\mathrm{NO_3-N}}{dt}\right)\left(V_{\text{ANA-VIP}}\right)\frac{\text{g N}}{\text{day}} =$$

$$\left((Q_{PCE})(PCE_{NOX}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{10 \text{ mg MG}}\right)\right)$$

$$+ \left((Q_{ARCY})(\text{CELL } 5_{NOX}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{\text{mg MG}}\right)\right) \tag{3.12a}$$

$$-\left((Q_{PCE}+Q_{ARCY})(\text{CELL } 3_{NOX}) \left(3.785\frac{\text{g L } 1x1^6\text{gal}}{1000 \text{ mg MG}}\right)\right).$$

The units of equations 3.12 and 3.12a are g N/day. In equation 3.12a, Cell 5_{NOX} was

used for the $ARCY_{NOX}$ since the recycle was not sampled for this parameter. The origin of this recycle was Cell 5, the second and final cell of the anoxic zone.

The loss due to denitrification in the anoxic basin for August through October, 1987 (A^2/O) is shown in equation 3.13. As before, Cell 3 nitrate values were assumed to be zero. The loss due to denitrification in the anoxic basin for April through July, 1988 (VIP) is described by 3.13a, which follows.

$$\left(\mathrm{ANX}\right)_{\text{denit } A^2/O} = \left(\frac{d\mathrm{NO_3-N}}{dt}\right)\left(V_{\text{ANX-}A^2/O}\right)\frac{\text{g N}}{\text{day}} =$$

$$\left((Q_{PCE}+Q_r)(\mathrm{CELL3_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right)$$

$$+\left((Q_{NRCY})(\mathrm{CELL7_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) \qquad (3.13)$$

$$-\left((Q_{PCE}+Q_r+Q_{NRCY})(\mathrm{CELL\ 5_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1 \text{ mg MG}}\right)\right).$$

$$\left(\mathrm{ANX}\right)_{\text{denit VIP}} = \left(\frac{d\mathrm{NO_3-N}}{dt}\right)(V_{\text{ANX-VIP}})\frac{\text{g N}}{\text{day}} =$$

$$\left((Q_{PCE} + Q_{ARCY})(\mathrm{CELL3_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) \qquad (3.13a)$$

$$+\left((Q_r)(\mathrm{RAS_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right) + \left((Q_{NRCY})(\mathrm{CELL\ 7_{NOX}}) \left(3.785\frac{\text{g L } 1x10^6\text{gal}}{1000 \text{ mg MG}}\right)\right)$$

$$-\left((Q_{PCE}+Q_r+Q_{NRCY}+Q_{ARCY})(\mathrm{CELL\ 5_{NOX}}) \left(3.785\frac{\text{g L } 1x^6\text{gal}}{1000 \text{ mg MG}}\right)\right).$$

July, 1987 is a special case and considered by itself since there was no nitrate (NRCY)

recycle. It is unknown if the assumption concerning complete denitrification in the anaerobic zone is a valid one for July, 1987. Due to this lack of data, an expression was developed that describes denitrification in the combined anaerobic and anoxic zones. No nitrate data were taken for Cell 5 of the anoxic zone. Cell 5 is assumed to have no nitrate since it is believed that complete denitrification would have taken place after travel through both the anaerobic and anoxic zones (Randall, 1993b).

$$\left(\text{ANA/ANX}\right)_{\text{denit A/O}} = \left(\frac{d\text{NO}_3\text{- N}}{\text{dt}}\right)\left(\text{V}_{\text{ANA/ANX-July 87}}\right)\frac{\text{g N}}{\text{day}} =$$

$$\left(\left(\text{Q}_{\text{PCE}}\right)\left(\text{PCE}_{\text{NOX}}\right)\left(3.785\frac{\text{g L 1x10}^6\text{gal}}{1000\text{ mg MG}}\right)\right) \tag{3.13b}$$

$$+\left(\left(\text{Q}_r\right)\left(\text{RAS}_{\text{NOX}}\right)\left(3.785\frac{\text{g L 1x10}^6\text{gal}}{1000\text{ mg MG}}\right)\right) - \left(\left(\text{Q}_{\text{PCE}}+\text{Q}_r\right)\left(\text{CELL5}_{\text{NOX}}\right)\left(3.785\frac{\text{g L 1x10}^6\text{gal}}{1000\text{ mg MG}}\right)\right).$$

Denitrification in the aerobic zone was calculated using equation 3.14. It states that the amount of denitrified nitrogen, as total nitrogen in the system that had been nitrified, plus the nitrate entering the aerobic zone from anoxic zone Cell 5, minus the nitrate leaving the aerobic zone in Cell 7 equals aerobic denitrification. The aeration basin nitrified nitrogen is used in the calculation to account for the nitrate produced since all of it is produced in the aeration basin. The assumption is made that all the heterotrophic uptake of nitrogen occurs in the aeration basin and none in the anoxic basin. The heterotrophic uptake of nitrogen was never nitrified and therefore could not be denitrified.

$$\left(\text{AER}\right)_{\text{denit}} = \left(\frac{d\text{NO}_3\text{-N}}{\text{dt}}\right)(\text{V}_{\text{AER}})\,\frac{\text{g N}}{\text{day}} = \text{aeration nitrified nitrogen} \tag{3.14}$$

$$+\left((\text{Cell } 5_{\text{NOX}})(Q_{\text{PCE}}+Q_r+Q_{\text{NRCY}})\left(3.785\frac{\text{g L } 1\text{x}10^6\text{gal}}{1000\text{ mg MG}}\right)\right)$$

$$-\left((\text{Cell } 7_{\text{NOX}})(Q_{\text{PCE}}+Q_r+Q_{\text{NRCY}})\left(3.785\frac{\text{g L } 1\text{x}10^6\text{gal}}{1000\text{ mg MG}}\right)\right)$$

where the aeration nitrified nitrogen is calculated by 3.6a and the nitrogen uptake by the biomass is calculated by 3.3.

Alternatively, the aerobic denitrification is calculated by subtracting the remaining component parts of denitrification from the total system nitrification, where the total system nitrification (Equation 3.5) must first be multiplied by the following proportion:

$$\text{Denit Fraction} = \frac{\sum\text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum\text{Total System Nitrification Daily Values}}{n}. \tag{3.15}$$

where n is the number of daily values calculated in a month and the total system denitrification is found by Equation 3.9 and the total system nitrification is described by Equation 3.5.

$$\left(\text{AER}\right)_{\text{denit}} =$$

$$N_{\text{nitrified}} \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n}$$

$$-\text{ANA}_{denit} - \text{ANX}_{denit} - SC_{denit}, \tag{3.15a}$$

where anaerobic denitrification was calculated using 3.12a and b, anoxic denitrification by 3.13a, b, and c and the secondary clarifier denitrification by equation 3.6c.

Basin denitrification expressed as grams N/grams MLVSS-day, was calculated for each basin as follows:

$$\text{Specific Denitrification Rate} = \left(\frac{\text{grams N}}{\text{grams MLVSS-day}}\right)_{\text{denitrified}} =$$

$$\frac{gN_{\text{denit}}}{\text{day}} \times \frac{L}{mg} \text{ Aeration MLVSS} \times \frac{\text{gal}}{3.785L} \times \frac{1}{\text{gal}_{\text{basin}}} \times \frac{1000mg}{g} \tag{3.16}$$

The volume of the two clarifiers in service during the months of interest was 1,100,000 gallons.

Waste Sludge Production

A study was performed to examine waste sludge production and related characteristics for various modes of operation of the York River Treatment Plant during the Biological Nutrient Removal Demonstration Project.

Data from the following time periods were examined: November 1-November 30, 1986, February 10-March 31, 1987, May 1-May 31, 1987, July 1-July 31, 1987, September 1-October 31, 1987, April 1-April 30, 1988, July 1-July 31, 1988 and July 1-July 31, 1989. The first three

time periods corresponded to the A/O mode of operation (phosphorus removal only, no anoxic zone) at the York River plant. During July 1 - July 31, 1987, the plant operated in full nitrification, but without an anoxic zone. The A^2/O mode of operation was studied using the time period of September 1 - October 31, 1987. The VIP mode was represented by the time frames April 1 - April 30, 1988, July 1 - July 31, 1988 and July 1 - July 31, 1989.

The waste sludge calculation, ΔX or $(dX/dt)_g$, was derived from the heterotrophic mass balance (equation 3) and is actually the working form of equation 3.1. It is calculated in lb/day as below:

$$\Delta X = \left((\text{Avg } Q_w) \times \text{Avg } WS_{vss} \times \frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}} \right)$$

$$+ \left((\text{Avg } FNE_{vss}) \times \text{Avg } (Q_{PCE} - Q_w) \times \frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}} \right). \qquad (3.17)$$

For this exercise only (Equations 3.17 through 3.24), since the primary clarifier underflow data was not available for all time periods under consideration, the primary clarifier effluent flow was calculated without it by adding the combined plant recycle to the raw flow. This did not result in any significant differences between the calculated parameters of this section for the months for which the primary clarifier underflow was available and when it was not.

The computation (in pounds) of the biomass in the reactors was generated by multiplying the combined volumes of the aeration and the anoxic basins by the average aeration effluent VSS and the conversion factor $\frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}}$. Biomass production in the anaerobic basin was assumed to be negligible due to the near absence of oxygen and nitrate in the basin. Heterotrophic biomass production is very low without the presence of a terminal electron acceptor for the electron transport chain.

Biomass in the reactors, XV_a (lb) =

$$\left(\text{Volume}_{\text{aeration + anoxic}}, \text{MG}\right)\left(\text{ARE VSS}, \frac{\text{mg}}{\text{l}}\right)\left(\frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}}\right). \tag{3.18}$$

The mass fraction of the biomass per zone and the volume of the zones varied over the life of the demonstration project. Refer to Table 3.1 for a list of the volumes used in the XV_a calculation. May, 1987 had two configurations of the mass fraction of the biomass per zone. For this month, the ARE VSS was calculated for each configuration. A separate XV_a was calculated for each ARE VSS, summed and divided by two. This average figure was used in calculations calling for XV_a for this particular time period.

The specific growth rate μ, was calculated by dividing the pounds of sludge wasted, ΔX, (Equation 3.17) by the pounds of biomass, XV_a (Equation 3.18).

$$\mu \text{ (1/Day)} = \Delta X/XV_a. \tag{3.19}$$

The sludge age, θ_c was calculated by dividing the pounds of biomass, XV_a (Equation 3.18) by the pounds of sludge wasted, ΔX, (Equation 3.17).

$$\theta_c \text{ (Days)} = XV_a/\Delta X. \tag{3.20}$$

Pounds of substrate per day were calculated by multiplying the average of the primary clarifier effluent flows for the time period in question by the average of the primary clarifier BOD_5 times $\left(\frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}}\right)$. The food : microorganism ratio (F:M) was then calculated by

dividing pounds per day of substrate by the biomass in the reactors as calculated in equation 3.18.

$$F : M = \frac{(Q_{PCE})(S_o)}{XV_a}. \tag{3.21}$$

The pounds of BOD_5 consumed were calculated by subtracting the average final effluent BOD_5 in mg/l multiplied by the conversion factor $\frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}}$ and the average effluent flow in MGD, from the average primary clarifier BOD_5 in mg/l multiplied by the conversion factor 8.34 and the average primary clarifier effluent flow in MGD.

$$\text{Lb } BOD_5 \text{ consumed, } \Delta S =$$

$$\left(\text{Avg PCE } BOD_5 - \text{ Avg FNE } BOD_5\right)\left(\frac{8.34 \text{ lb}}{\text{mgL}^{-1}\text{MG}}\right)\left(Q_{PCE}\right) \tag{3.22}$$

The fractional, or specific, substrate utilization rate, q (1/Day) was calculated by dividing the pounds of BOD_5 consumed (ΔS, as in Equation 3.22) by the pounds of biomass, XV_a (as in Equation 3.18).

$$q = \Delta S/XV_a. \tag{3.23}$$

The specific growth rate μ (or, $\Delta X/XV_a$) is divided by the fractional substrate utilization rate q (or, $\Delta S/XV_a$), to obtain the ratio of the observed sludge yield, Y_{obs}.

$$Y_{obs} = \frac{\text{lb biomass produced}}{\text{lb } BOD_5 \text{ removed}}. \tag{3.24}$$

The ratio of the BOD_5 to the TKN was averaged for the months under evaluation and compared. The BOD/TKN ratios were calculated by dividing the influent BOD_5 in the primary clarifier effluent by the primary clarifier effluent TKN, as below,

$$BOD_5/\text{TKN ratio} = \frac{\text{PCE } BOD_5}{\text{PCE TKN}}. \tag{3.25}$$

The ratio of the COD to the TKN was averaged for the months under evaluation and compared. The COD/TKN ratios were calculated by dividing the influent COD in the raw influent by the raw influent TKN, as below,

$$\text{COD/TKN ratio} = \frac{\text{RWI COD}}{\text{RWI TKN}}. \tag{3.26}$$

CHAPTER 4
RESULTS

Nitrification Rates

The monthly average nitrification results (Table 4.1) were found to be equal for all months except April 1988 when calculated both in terms of ammonia disappearance (equation 3.5) and nitrate appearance (equation 3.7). Daily calculated values were also equal for the months in question. The two equations were examined for analytical equality. This exercise is presented in Appendix II. For the month of April, 1988, the nitrification result based on ammonia disappearance (206,076 gN/day) is less than that obtained based on nitrate appearance (218,456 gN/day). Three of the eight daily values obtained with equation 3.5 were negative due to incomplete nitrification during this month. The negative results were replaced with zeroes since it is not possible for nitrification to be negative. When the corresponding values in equation 3.7 were replaced with zeroes, the same mean nitrification result of 206,076 gN/day was obtained.

System (overall) nitrification amounts were highest in the warmer two study months of 1987, July and August (Table 4.1). In 1988, the overall nitrification rate did not increase with increasing temperature, but peaked in May 1988 and decreased in June and July. The true population standard deviation for the overall nitrification (equation 3.5) was 55,094 gN/day in for the A^2/O months examined, with the total nitrification ranging from 576,392 gN/day to 705,247 gN/day. For the VIP mode of operation, including the low April 1988 value, the true population standard deviation was 175,153 gN/day; excluding that datapoint, the standard deviation was 83,550 gN/day. VIP total nitrification ranged from 206,076 to 678,047 gN/day. The mean VIP nitrification (574,448 gN/day) was 9 percent less than the mean A^2/O mode value (629,243 gN/day).

Total Nitrification Calculations York River STP 1987 and 1988
Table 4.1

Mode	July 1987 Nitrification without Nitrate Recycle	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Nitrification gN/day (Eqn 3.5 Ammonia Oxidized)	634100	705200	576400	606100	206100	678000	571900	473400
Ammonia Available for Nitrification (Eqn 3.5a)	645800	693800	601900	633900	461500	717900	602500	531100
Percentage Ammonia Oxidized/ Available	98	102	96	96	45	94	95	89

The month of July, 1987, is excluded from discussions and totals of the A^2/O processes unless noted even though nitrification was complete because there was no nitrate recycle in operation. The month of April, 1988 experienced incomplete nitrification, and thus provided low values for nitrification and denitrification processes. Only the values representative of the processes will be included in the discussion.

System nitrification did not always take place entirely in the aeration basin, but sometimes extended further along the treatment train into the secondary clarifier. During September and October, 1987 and April, 1988, 96, 99.9 and 92 per cent, respectively, of the total nitrification calculated, occurred in the aeration basin (Table 4.2). The rate values for nitrification differed in Table 4.2 from Table 4.1 for the months of September and October, 1987 and April, 1988 because the values in Table 4.2 represented nitrification that was taking place in the aeration basin only whereas Table 4.1 described system nitrification. In terms of rate per unit of biomass for the A/O and A^2/O modes, the highest aeration basin nitrification rate observed was in the month of October, 1987 (Table 4.2). The aeration basin nitrification rate for the VIP process, expressed in terms of units of biomass, peaked in May of 1988, then decreased in June, 1988 and increased again slightly in July, 1988.

Operational Comparisons: Nitrification

Consulting Table 4.2, the average specific aeration nitrification rates were the same for the VIP months and the 1987 months (0.116 gN/gMLVSS-d). When the A^2/O months alone were considered, the average specific nitrification rate (0.117 gN/gMLVSS-d) slightly exceeded the VIP rate.

Aeration Basin Nitrification Rates[1] York River STP 1987 and 1988
Table 4.2

Mode	July 1987 A/O Mode Nitrifi-cation without Nitrate Recycle	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Basin Nitrifi-cation gN/day eqn. 3.6a	634100	705200	556100	605500	190400	678000	571900	473400
Percent-age of Total Nitri-fication	100	100	96	99.9	92	100	100	100
Percent-age of Ammonia Avail-able for Nitri-fication	98	102	92	96	41	94	95	89
Basin Nitrifi-cation mg/L-hr eqn. 3.6d	10.36	10.91	9.22	9.89	3.11	11.08	9.34	7.65
Specific Nitrifi-cation Rate gN/gMLVSS-day eqn 3.6d1	0.1129	0.1158	0.1153	0.1192	0.0252	0.1230	0.1116	0.1125
gN/ gMLVSS-d		average 0.1168						
gN/ gMLVSS-d	average 0.1158					average 0.1157		

[1]This table differs from Table 4.1 in that only the nitrification which took place in the aeration basin is described. Table 4.1 describes the total system nitrification.

Denitrification Rates

Denitrification rates were considered as system (overall) quantities in gN/day (Table 4.3), and rates per basin in gN/day (Table 4.4) and per unit of biomass (gN/gMLVSS-d) (Table 4.5). The sum of the basin rates was calculated for an overall denitrification rate expressed per unit of biomass. The true population standard deviation for the overall denitrification rate (equation 3.10) was 45,743 gN/day for A^2/O, with the denitrification rate ranging from 460,051 gN/day to 571,772 gN/day. For the VIP mode of operation, including the low April 1988 value, the true population standard deviation was 146,013 gN/day; excluding that datapoint, the standard deviation was 85,558 gN/day. Denitrification was low in April, 1988, peaked in May, 1988 and then decreased due to changes in the nitrate recycle. In 1988, the rate expressed per units of biomass increased until May, 1988, and decreased in June and increased again in July, 1988, ranging from 0.0233 gN/gMLVSS-d to 0.1109 gN/gMLVSS-d. The true population standard deviation was 0.0192 gN/gMLVSS-d for the A^2/O data and 0.0144 gN/gMLVSS-d for VIP. The A^2/O mean rate of denitrification expressed per units of biomass was 153 percent of the VIP rate because of the lowered nitrate recycle in 1988.

Completeness of Anoxic Denitrification

From the data that was collected, it appears that generally complete denitrification was accomplished in the anoxic zone during both modes of operation. During July, 1987, no nitrate-nitrite data was collected from Cell 5, the final cell of the anoxic zone. With the exclusion of mid-September, 1987, when results rose to 7.16 mg/l over the course of about one week before dropping to 0.04 mg/l, under A^2/O operation, average nitrate-nitrite results leaving the anoxic zone were 0.15 mg/l compared to VIP results of 0.18 mg/l.

A Comparison of System Denitrification Rate Calculations
York River STP 1987 and 1988
Table 4.3

Mode	July 1987 Nitrification without Nitrate Recycle	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Denitrification Rate gN/day (Eqn 3.10)	451900	571800	460100	508500	135500	540900	401600	335600
Denitrification Rate gN/day (Eqn 3.11, using Eqn 3.14)	521100	555800	449400	463800	133500	464400	275300	297300
% Difference, (Eqn 3.11 from Eqn 3.10)	+15.3	-2.8	-2.3	-8.8	-1.5	-14.1	-31.4	-11.4

A Comparison of Basin by Basin and System Denitrification Rates York River STP 1987 and 1988
Table 4.4

Mode	July 1987 No NRCY	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
System Denit. Rate gN/day Eqn 3.10	451900	571800	460100	508500	135500	540900	401600	335600
ANA Basin Denit. Rate gN/day Eqn 3.12 & 3.12 a	151400; 34% of System Denit.	65660; 11% of System Denit.	2354; 0.5% of System Denit.	3978; 0.8% of System Denit.	9907; 7.3% of System Denit.	6482; 1.2% of System Denit.	15094; 3.8% of System Denit.	5605; 1.7% of System Denit.
ANX Basin Denit.Rate gN/day Eqn 3.13 & 3.13 a	(No anoxic zone)	145000; 25% of System Denit.	101900; 22% of System Denit.	97970; 19% of System Denit.	41870; 31% of System Denit.	97300; 18% of System Denit.	50400; 13% of System Denit.	64200; 19% of System Denit.
Total ANA+ ANX,%	34	36	22.5	19.8	38.3	19.2	16.8	20.7
Aerobic Basin Denit.Rate gN/day Eqn 3.14	198000; 44% of System Denit.	267200; 47% of System Denit.	320500; 70% of System Denit.	336900; 66% of System Denit.	53150 39% of System Denit.	273400; 51% of System Denit.	155100; 39% of System Denit.	198300; 59% of System Denit.
Aerobic Basin Denit.Rate gN/day Eqn 3.15a	278600; 62% of System Denit.	283200; 50% of System Denit.	331100; 72% of System Denit.	381600; 75% of System Denit.	68310; 50% of System Denit.	349800; 65% of System Denit.	281400; 70% of System Denit.	236600; 71% of System Denit.
%Difference Eqn 3.14 from 3.15a	-28.9	-5.7	-3.2	-11.7	-22.2	-21.9	-44.8	-16.2
Secondary Clarifier Basin Denit.Rate gN/day Eqn 3.6 c	21890; 4.8% of System Denit.	77950; 14% of System Denit.	24640; 5.4% of System Denit.	25000; 4.9% of System Denit.	29310; 22% of System Denit.	87260; 16% of System Denit.	54660; 14% of System Denit.	23310; 7.0% of System Denit.
Percent Total (with Aer. Denit. eqn 3.14)	82.9	97.0	97.7	91.6	99.3	86.2	69.8	86.7
Denit. in Clarifier Sludge Blanket and RAS Line, %	17.1	3.0	2.3	8.4	0.7	13.8	30.2	13.3

Basin Denitrification Rates, gN/gMLVSS-d York River STP 1987 and 1988
Table 4.5

Mode	July 1987 Nitrification without Nitrate Recycle	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
ANA Basin Denitrification Rate Eqn 3.12 &3.12a	0.0541	0.0270	0.0010	0.0016	0.0026	0.0023	0.0056	0.0027
ANX Basin Denitrification Rate Eqn 3.13& 3.13a	(No anoxic zone)	0.0601	0.0424	0.0401	0.0107	0.0352	0.0214	0.0309
Aerobic Basin Denitrification Rate Eqn 3.14	0.0357	0.0549	0.0669	0.0643	0.0078	0.0502	0.0290	0.0476
Aerobic Basin Denitrification Rate Eqn 3.15a	0.0497	0.0585	0.0687	0.0735	0.0101	0.0640	0.0544	0.0568
Secondary Clarifier Basin Denitrification Rate Eqn 3.6c	0.0023	0.0100	0.0014	0.0035	0.0022	0.0094	0.0064	0.0038
Total (with Aeration Denitr. Eqn 3.14)	0.0921	0.1520	0.1132	0.1095	0.0233	0.0971	0.0624	0.0850
		average 0.1249				average 0.0815		

Denitrification and Nitrate Availability to the Anoxic Zone

A significant, positive linear relationship was found between the total denitrification rate (in gN/gMLVSS-d) and the anoxic zone nitrate loading (Table 4.6,4.7, Figure 4.1). The correlation coefficient was 0.83 and the significance value was 0.01. The VIP and A^2/O modes produced less significant relationships if the modes of operation were examined separately, and both correlation coefficients were reasonably high (0.66 and 0.77, respectively). If the change in total denitrification, the difference in denitrification in gN/day between successive months, was examined versus the anoxic loading rate, a very strong and significant relationship appeared for the VIP mode (Figure 4.2), with no apparent relationship overall. The month of April, 1988 was included for this graph. Since there was no anoxic zone for July, 1987, the A^2/O mode was not examined by itself because the difference between three successive points produces only two points, which would not have produced a meaningful graph.

A positive and significant relationship was found between the total denitrification rate (in gN/gMLVSS-d) and the total nitrate recycle ($r^2 = 0.65$, $P = 0.03$)(Figure 4.3). The total nitrate recycle is the sum of the return sludge recycle and the NRCY. The A^2/O mode regression analysis produced a higher correlation coefficient and lower significance value ($r^2 = 0.89$, $P = 0.05$)(Figure 4.4) compared to the VIP data ($r^2 = 0.80$, $P = 0.30$)(Figure 4.5). Both modes separately produced higher correlation coefficients than the modes combined. If the total denitrification and the NRCY alone is analyzed, the data for both modes combined again produced a lower correlation coefficient than the data for each mode separately. The overall correlation coefficient was 0.42 (Figure 4.6), the A^2/O mode correlation coefficient was 0.65 (Figure 4.7) and the VIP data produced a correlation coefficient of 0.90 (Figure 4.8). None of the three relationships were significant, but significant relationships may be produced if more

Nitrate Loading to the Anoxic Zone York River STP 1987 and 1988
Table 4.6

Mode	July 1987 Nitrifi-cation without Nitrate Recycle	August-October A_2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Average monthly nitrate loading, lbs	151400	172200	139600	100400	61650	126800	62120	67690
Nitri-fied Recycle (NRCY), MGD	None	7.26	7.26	7.26	7.26	7.36	1.96	3.97
Return Sludge (RAS) Recycle, MGD	9.59	8.37	5.32	5.83	7.84	7.89	6.51	6.13
Total Recycle, NRCY + RAS, MGD	9.59	15.63	12.58	13.09	15.15	15.25	8.47	10.10

Table 4.7

York River Demonstration Project

Total Denitrification Regression Analysis Relationships

Relationship	Overall	A^2/O	VIP
Total Denit. (gN/gMLVSS-d) vs. Anoxic Zone Nitrate Loading (gN/day)			
r^2=	0.83[1,2]	NA--too few points	0.66[1]
P =	0.01		0.39
Y =	$6.041 \times 10^{-7} X + 0.0317$		$4.002 \times 10^{-7} X + 0.0473$
Change in Total Denit.[3] (gN/day) vs. Anoxic Zone Nitrate Loading (gN/day)			
r^2=	0.11[2,3]	NA--too few points	0.99[3]
P =	0.66		0.03
Y =	-0.042X - 49995		8.237 X -637858
Total Denit. (gN/gMLVSS-d) vs. Total NO_3 Recycle (MGD)			
r^2=	0.65[1]	0.89	0.80[1]
P =	0.03	0.05	0.30
Y =	$7.99 \times 10^{-3} X + 0.0048$	$9.64 \times 10^{-3} X - 5.92 \times 10^{-3}$	$4.45 \times 10^{-3} X + 0.0313$
Total Denit. (gN/gMLVSS-d) vs. NRCY Recycle (MGD)			
r^2=	0.42[1]	0.65	0.90[1]
P =	0.12	0.35	0.20
Y =	$5.89 \times 10^{-3} X + 0.0721$	$4.52 \times 10^{-3} X + 0.092$	$6.127 \times 10^{-3} X + 0.0544$

[1]Excludes April, 1988 data due to incomplete denitrification, unless noted.

[2]Excludes July, 1987 data since no anoxic zone was present.

[3]The overall and VIP statistics include the difference between May and April, 1988 denitrification paired with the May 1988 anoxic zone nitrate loading.

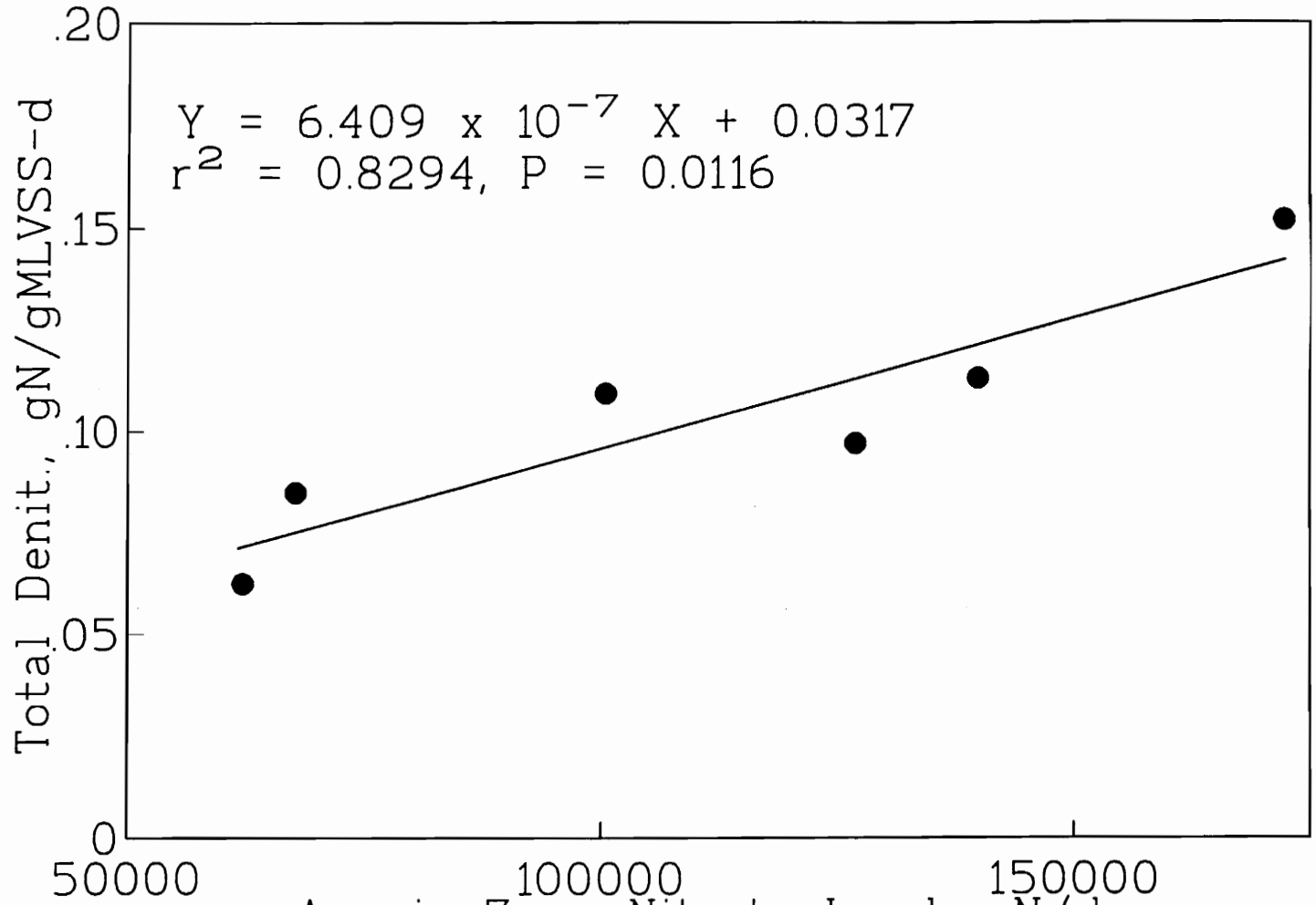


Figure 4.1. Total Denitrification vs. Anoxic Zone Nitrate Load

data were examined.

System and Aerobic Denitrification Rates

System denitrification rates were calculated by several different equations. One calculation, equation 3.10, is independent. The component equation (equation 3.11) adds up all the individual basin denitrification rates.

The system denitrification rate values shown in Table 4.3 and computed using the component equation 3.11 and the aerobic denitrification equation 3.14 are typically less than those calculated with the mass balance equation 3.10 or the component equation 3.11 where the aerobic denitrification equation 3.15a is inserted. The mass balance equation 3.10 and the component equation 3.11 when the aerobic denitrification equation 3.15a is inserted produce the same result. Aerobic denitrification results computed with equation 3.14 are always less than those calculated with equation 3.15a and would produce a lower result in the component system denitrification equation. July, 1987, is an exception to the above observations as seen in Table 4.3 (System Denitrification). Perhaps combining the anaerobic and anoxic basin denitrification rate calculations for July, 1987 led to an overestimation in one or both of their values and inflated the component system denitrification value (equation 3.11) compared to that calculated with equation 3.10 .

In every case except for July, 1987, equation 3.10 for system denitrification produced a higher result that agreed reasonably well with the sum of the individual basin denitrification figures, if equation 3.14 for aerobic denitrification is used (Table 4.4) compared to component equation 3.11, using aerobic denitrification equation 3.14 (Table 4.3). For this reason, equation 3.10 is chosen as the most representative system denitrification description.

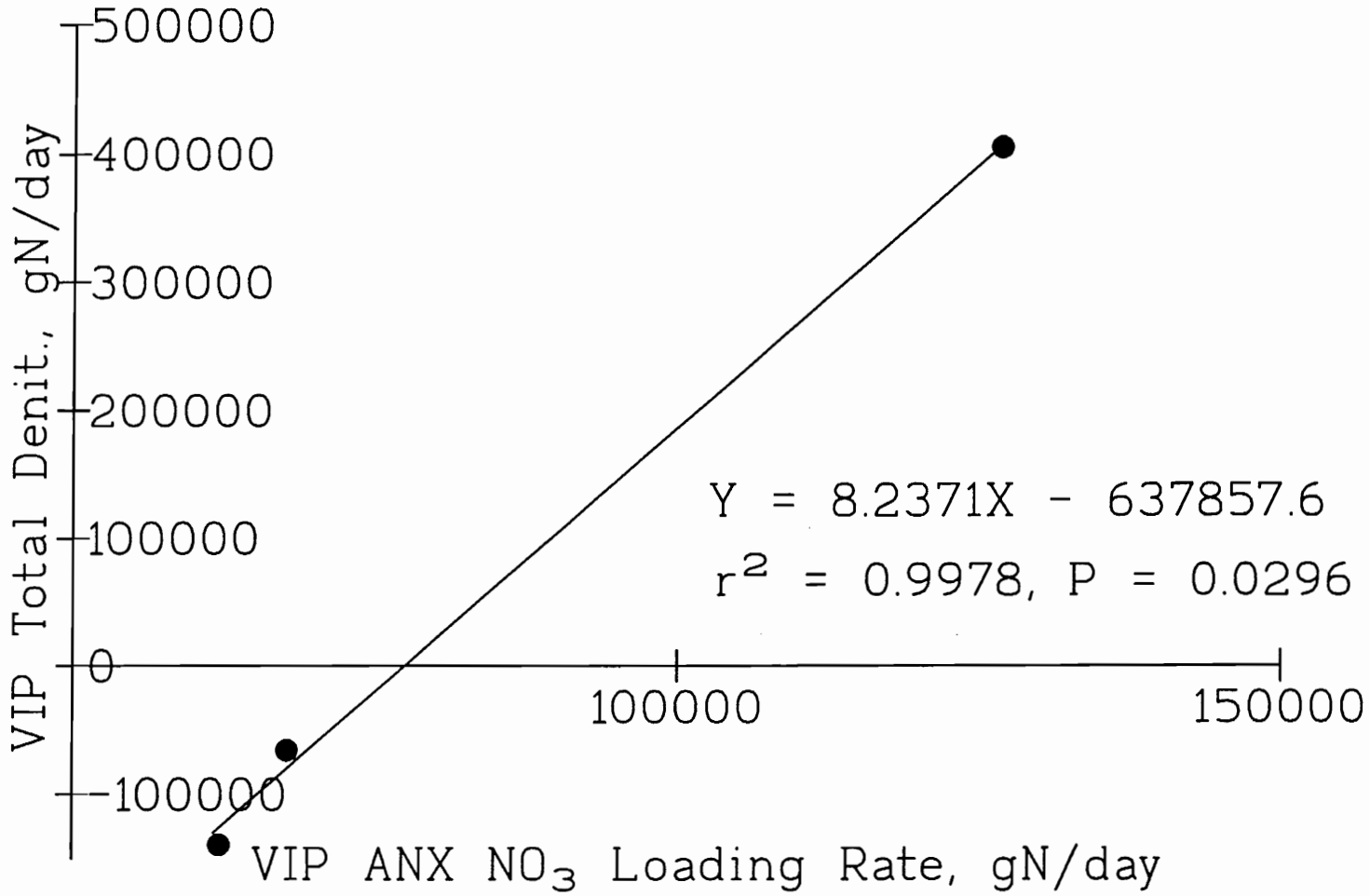


Figure 4.2. Change in VIP Total Denitrification vs. ANX NO_3 Load Rate

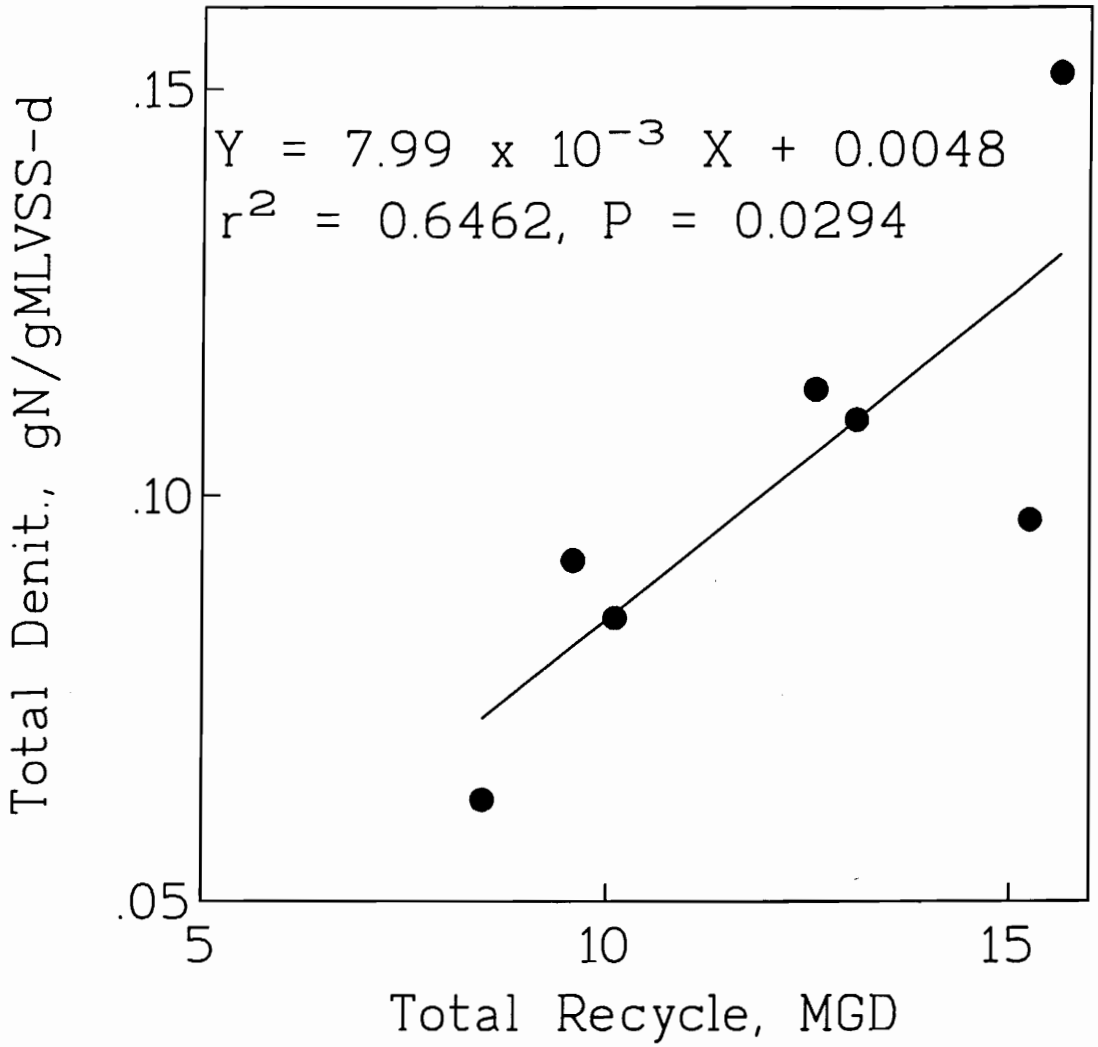


Figure 4.3. Total Denitrification vs. Total NO_3 Recycle

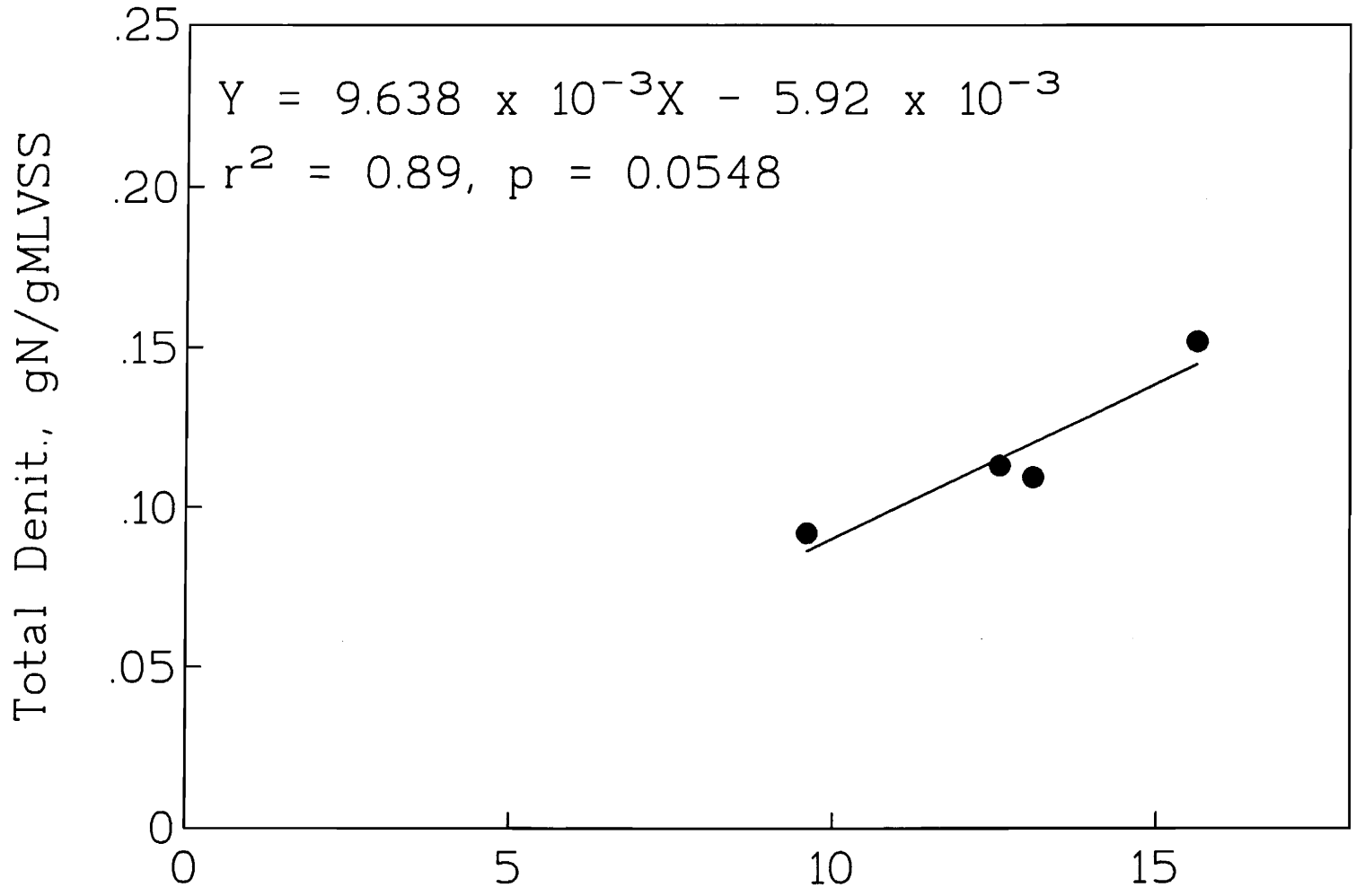


Figure 4.4. A^2/O Total Denitrification vs. Total NO_3 Recycle

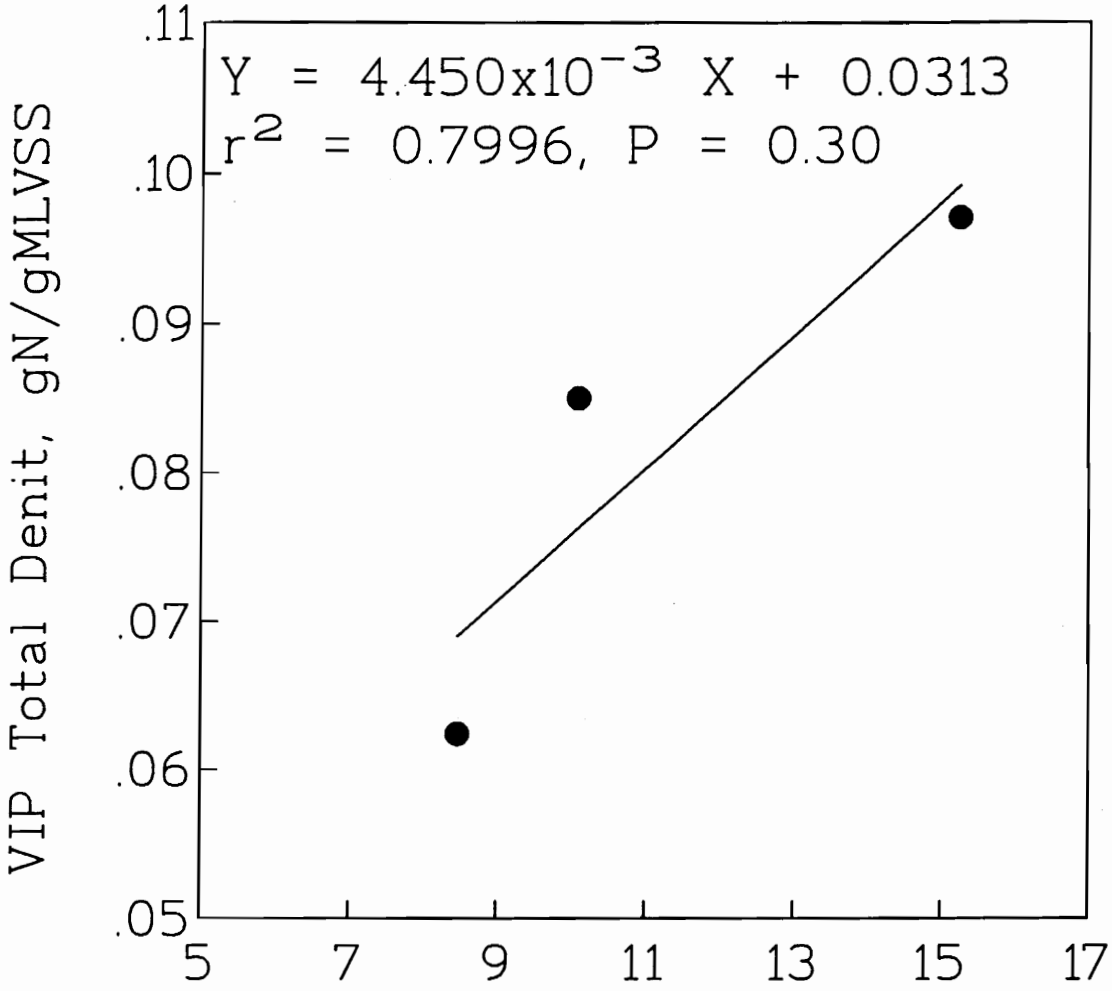


Figure 4.5. VIP Total Denitrification vs. Total NO_3 Recycle

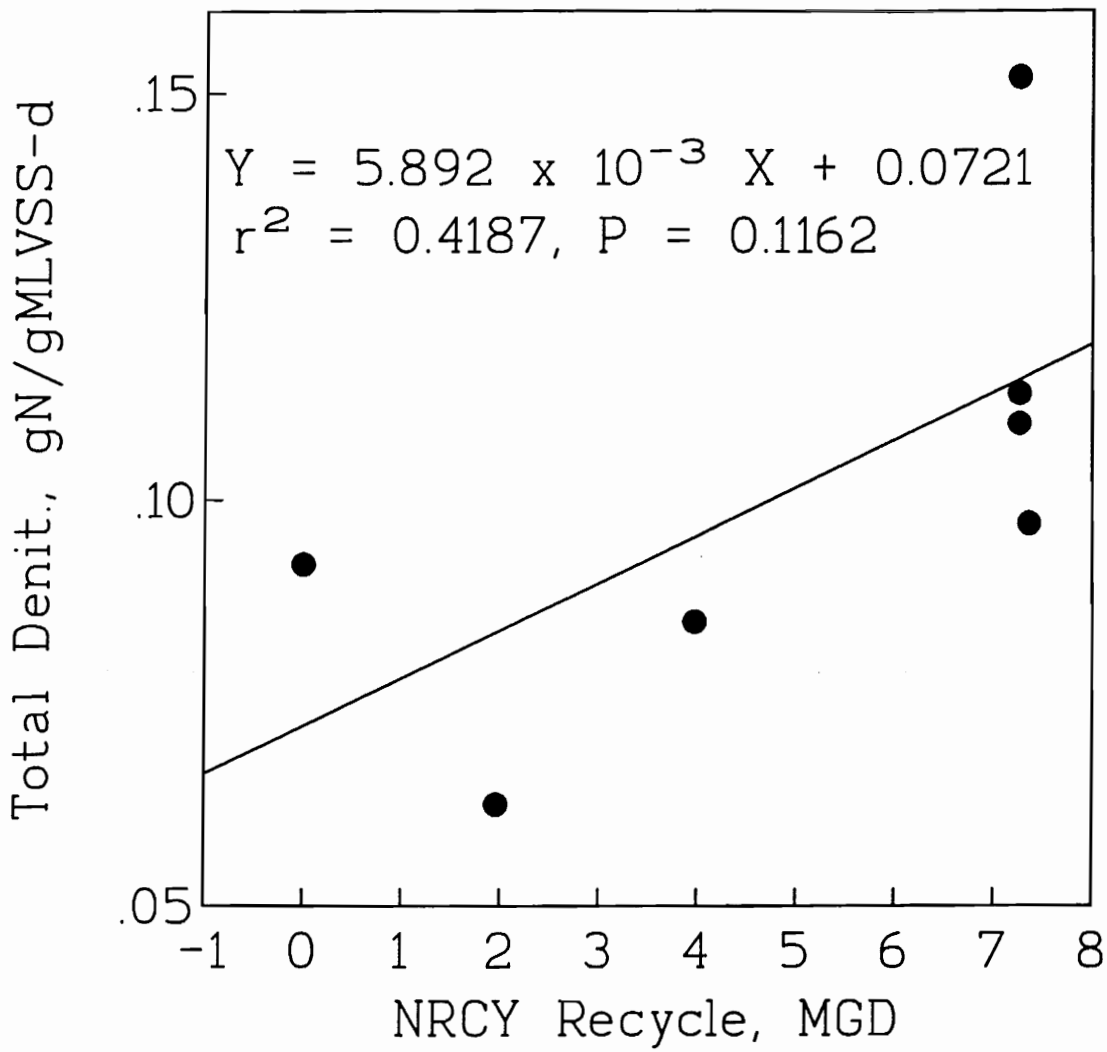


Figure 4.6. Total Denitrification vs. NRCY Rate

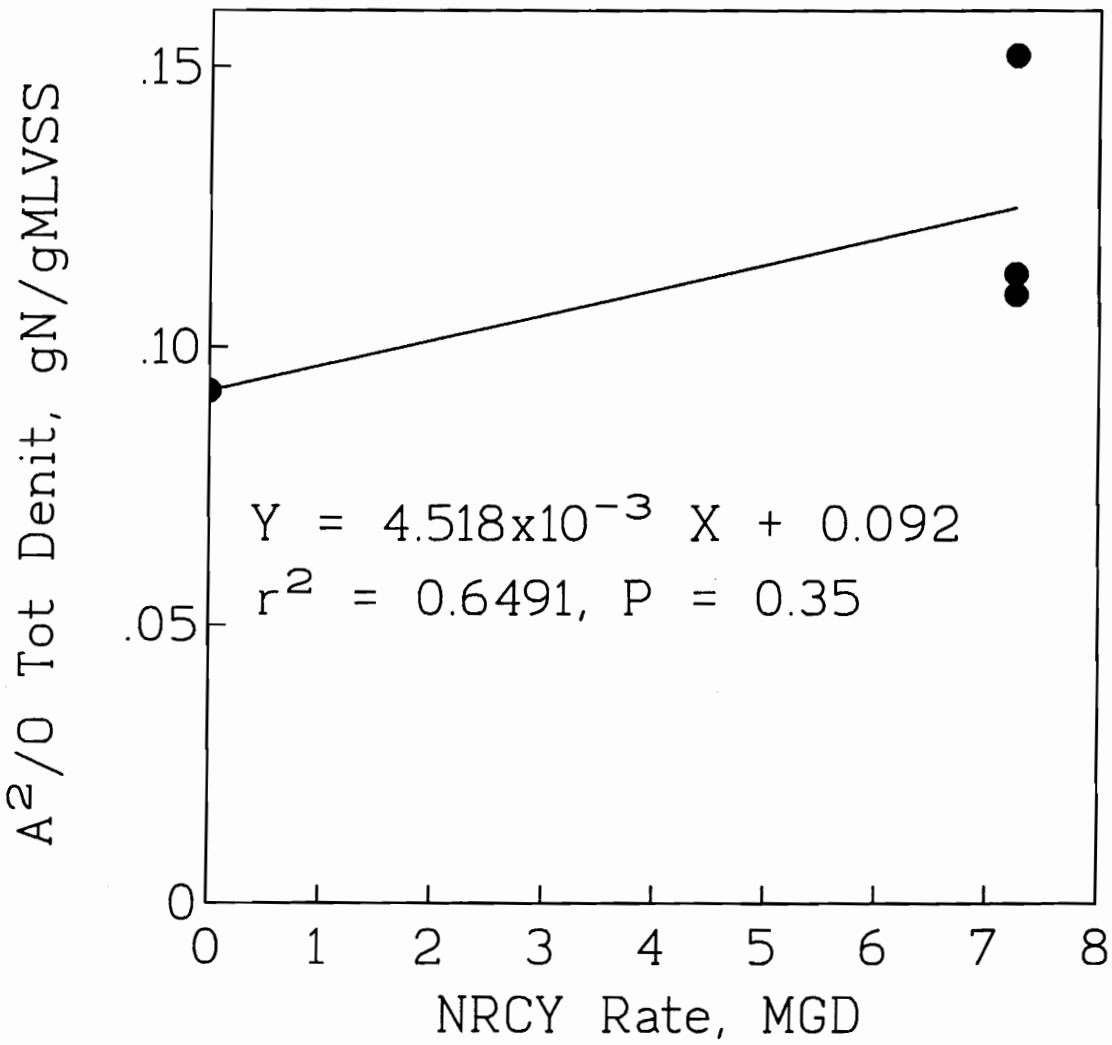


Figure 4.7. A^2/O Total Denitrification vs. NRCY Rate

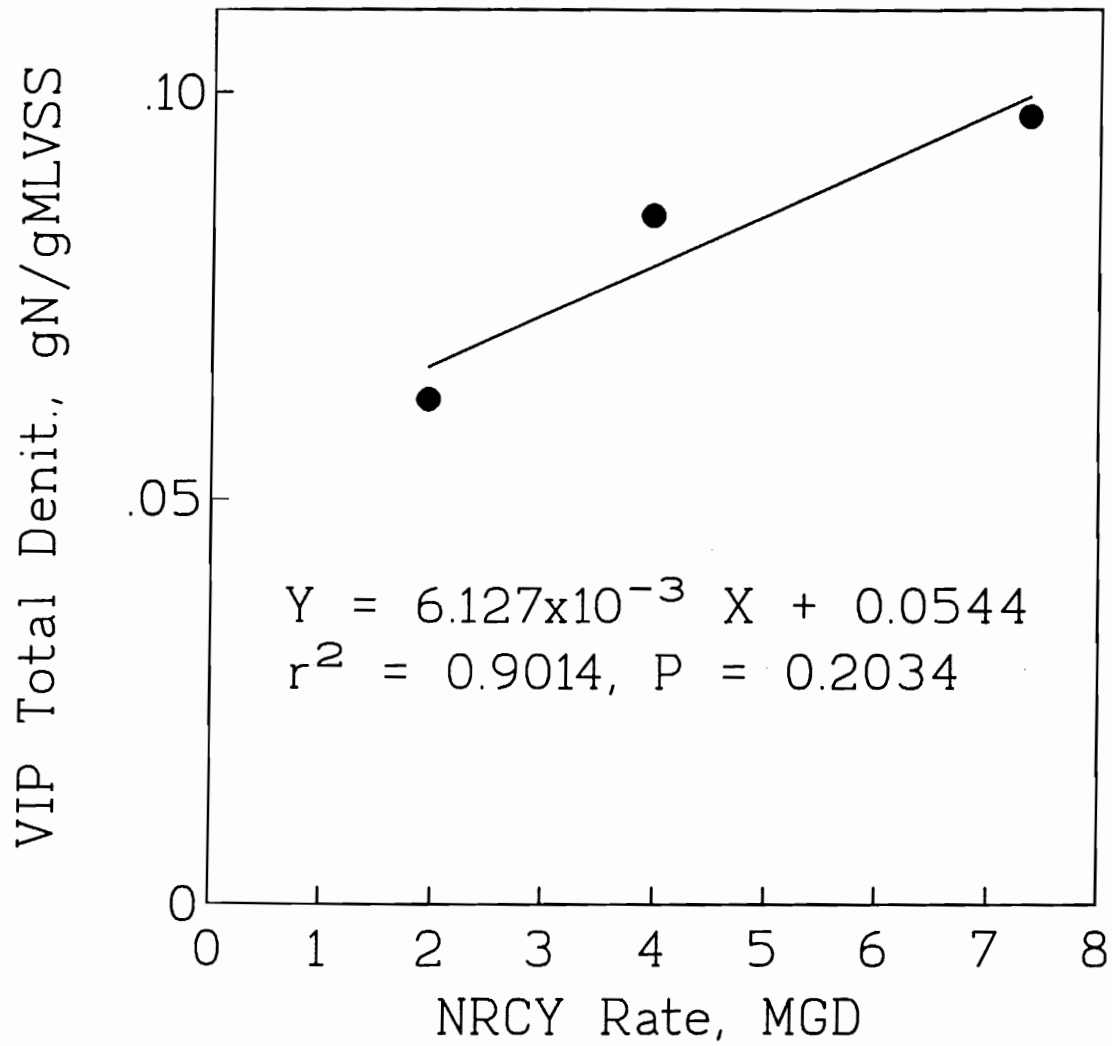


Figure 4.8. VIP Total Denitrification vs. NRCY Rate

If the basin denitrification rates are expressed as percentages of the system denitrification rate, in some cases the sums do not approach 100 percent. The percent totals of the basin denitrification rates in Table 4.4 are lower in 4 cases than 90 percent, for July, 1987, and May through July, 1988. This is due to the aerobic denitrification values produced by equation 3.14 which, when added into the total, are less than expected (Table 4.4). When aerobic denitrification is calculated using equation 3.15a, in every case the value is greater than the result calculated with equation 3.14. When the results generated by equation 3.15a are substituted into the percent totals of the basin denitrification rates in Table 4.4, in all but one case does the total sum to nearly 100 per cent. Because equation 3.15a uses the other basin denitrification rates in the calculation, it is expected that the system totals using this equation would be closer to 100 percent than those calculated with equation 3.14, which is independent from the other totals. This suggests that aerobic denitrification calculated with equation 3.14 may be low.

The oxygen utilization rate (OUR) in the aeration basin (measured in Cell 6 for July, 1987 and in Cells 6 and 7 in April through June, 1988) increased half again as much from June, 1987 to July, 1987 and similarly again from April, 1988 to May, 1988. The OUR remained at that higher level for the month of June, 1988 before decreasing. If the OUR was high, lowered D.O. was seen in the aeration basin, especially during VIP operation. Lower aeration D.O. during VIP operation compared to the A^2/O mode of operation could explain why equation 3.15a has a better recovery for the VIP data than equation 3.14. Less nitrification was taking place during the VIP mode of operation, and though the nitrification figure is used (as both the aeration and system nitrification, which were equal for May through July 1988) in both equations 3.14 and 3.15a, the nitrate loading to the aeration basin from Cell 5 under A^2/O

operation (7.71 lb/day) was greater than two times the corresponding VIP loading of 3.43 lb/day. In equation 3.14, the addition of the greater loading to the greater nitrification value for A^2/O data would result in higher values generated compared to the VIP data.

Aerobic denitrification was calculated with two equations: 3.14 (an independent mass balance equation) and, 3.15a (a component equation in which the other basin denitrification values are subtracted from the total system denitrification). In all cases, equation 3.15a resulted in greater values than equation 3.14 (Table 4.4). Since the aerobic denitrification calculated with equation 3.15a is the remainder after the subtraction of the other components from the system denitrification and it is greater than the results obtained with equation 3.14, a greater recovery was implied by its use. Equation 3.15a is used therefore to describe the most representative results for aerobic denitrification. However, due to correlation, this value cannot be used in conjunction with the component equation for the determination of system denitrification with equation 3.11.

When component aerobic denitrification equation 3.15a is substituted into the component system denitrification equation 3.11 (Appendix III), the mathematics simplifies to:

$$\text{System Denitrification} = \frac{\sum \text{Total System Denitrification Daily Values}}{n} \quad .$$

Since all the components cancel due to correlation and system denitrification is left to equal system denitrification, only the independent system denitrification equation 3.10 and the component equation 3.11 with aerobic equation 3.14 were expressed in Table 4.3.

Since aerobic denitrification calculated with equation 3.15a was considered the more representative result, it is included in a summation of all the denitrification components in

Table 4.4. However, this summation is the same as the calculation for system denitrification using equation 3.11, using 3.15a for the aerobic denitrification component. The mass balance recoveries are shown in this Table 4.4 with equation 3.10 and equation 3.11 and 3.15a, with the understanding that though the recoveries with equation 3.11 and 3.15a are superior, they are due to correlation and are presented for comparison with equation 3.10 only.

The A^2/O aerobic denitrification rate ranged from 50 to 75 percent of the total system denitrification, with a mean of 66. The VIP aerobic denitrification rate ranged from 50 to 71 percent, with a mean of 69. Ranges were 236608 to 381564 gN/day to 0.0544 to 0.0735 gN/gMLVSS-d. Thirteen percent more denitrification took place under A^2/O than VIP. Nine percent of the difference could be attributable to the difference between the VIP nitrification and the A^2/O.

Anoxic Basin Denitrification

The anoxic basin denitrification consisted of 13 to 25 percent of the total system denitrification rate for both modes of operation overall. The means were 22 and 17 percent of the total system denitrification for months representative of A^2/O and VIP processes, respectively. The overall range was 50429 to 144975 gN/day or 0.0241 to 0.0601 gN/gMLVSS-d. The month of highest anoxic denitrification (August, 1987) corresponds with the month of highest nitrate loading to the anoxic zone.

Anaerobic Denitrification

With the exception of July, 1987, in which there was no anoxic zone, the anaerobic basin contributes 11 per cent or less to the system denitrification rate considering both modes of operation. The A^2/O anaerobic denitrification, in gN/day, averaged 4.1 percent of the total system denitrification. VIP anaerobic denitrification, in gN/day, averaged 3.5 percent. The overall range was 2254 to 65658 gN/day or 0.010 to 0.0270 gN/gMLVSS-d. The A^2/O average anaerobic denitrification (23997 gN/day) was three times the average VIP anaerobic denitrification (9060). However, if August 1987 was not included (since the anoxic and anaerobic system was equilibrating after establishment of the anoxic zone) the A^2/O rate drops to one third of the VIP rate (3166 gN/day).

Anoxic and Anaerobic Denitrification and Nitrate Availability to the Anoxic Zone

A significant, positive linear relationship was found between the anoxic and anaerobic denitrification rate (in gN/gMLVSS-d) and the anoxic zone nitrate loading (Table 4.8, Figure 4.9). The correlation coefficient was 0.69 and the significance value was 0.04. The VIP and A^2/O modes produced less significant relationships if the modes of operation were examined separately, and the A^2/O mode ($P = 0.35$) was slightly less significant than the VIP mode ($P = 0.38$). However, the correlation coefficients for the overall relationship and the VIP operation were the same (0.69). The correlation coefficient for the A^2/O mode was slightly higher (0.73). If the change in anaerobic plus anoxic denitrification, the difference between successive months in the sum of the anaerobic and anoxic denitrification, was examined versus the anoxic loading rate, a strong (but significantly insignificant) relationship appeared for the VIP mode ($r^2 = 0.84$, $P = 0.26$), with no apparent relationship overall or for the A^2/O mode. If more VIP data

Table 4.8

York River Demonstration Project

Regression Analysis Relationships

Relationship	Overall[1]	A^2/O	VIP[1]
ANA+ANX Denit. (gN/gMLVSS-d) vs. Anoxic Zone Nitrate Loading (gN/day)			
r^2=	0.68	0.66	0.69
P =	0.02	0.19	0.38
Y =	3.912×10^{-7} X + 5.07×10^{-4}	5.665×10^{-7} X - 0.0233	1.228×10^{-7} X + 0.0222
Change in ANA+ANX Denit.[2] (gN/day) vs. Anoxic Zone Nitrate Loading			
r^2=	0.06	0.10	0.84
P =	0.65	0.79	0.26
Y =	0.340 X -43103	0.738 X - 117928	1.154 X -92656
ANA+ANX Denit. (gN/gMLVSS-d) vs. Total Recycle (MGD)			
r^2=	0.31	0.35	0.82
P =	0.19	0.41	0.23
Y =	3.96×10^{-3} X - 0.0015	5.04×10^{-3} X - 0.0076	1.358×10^{-3} X + 0.0174
ANA+ANX Denit. (gN/gMLVSS-d) vs. NRCY Recycle (MGD)			
r^2=	0.057	0.006	0.916
P =	0.61	0.92	0.18
Y =	5.5×10^{-4} X + 0.0386	4.55×10^{-4} X + 0.0541	1.81×10^{-3} X + 0.0245

[1]Excludes April, 1988 data due to incomplete denitrification, unless noted.

[2]The overall and VIP statistics include the difference between May and April, 1988 denitrification paired with the May 1988 anoxic zone nitrate loading.

[3]Eliminating the difference between April and May denitrification and the May 1988 anoxic zone nitrate loading datapoint yields an r^2= 0.38, P = 0.26, which is not significantly better than leaving it in.

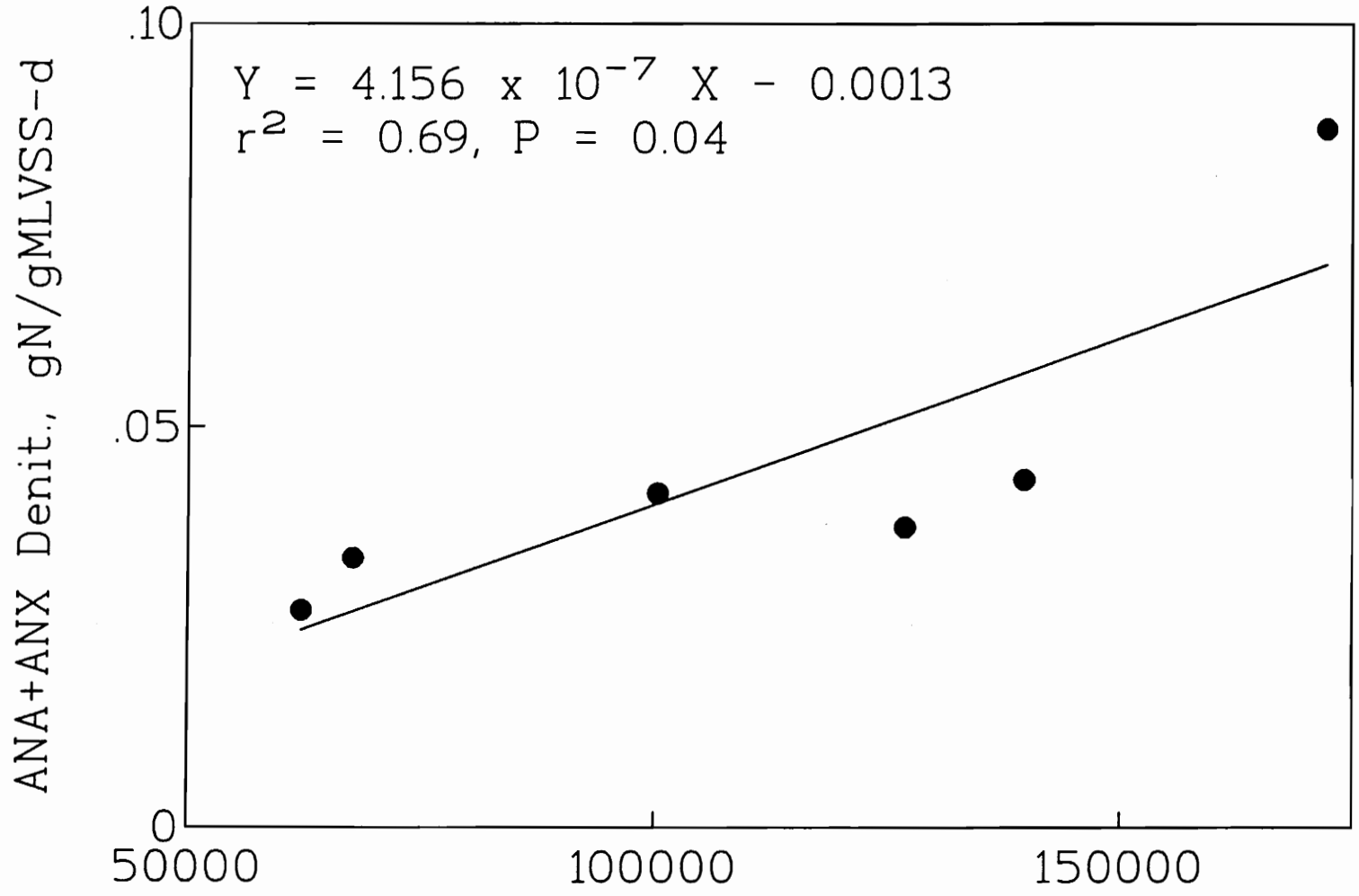


Figure 4.9. ANA+ANX Denitrification vs. Anoxic Zone Nitrate Loading

points were available, the statistical significance may increase.

No significant relationship overall was found between the anaerobic and anoxic denitrification rate (in gN/gMLVSS-d) and the NRCY (r^2 = 0.057, P = 0.61). Most of the x values in the overall and the A^2/O data were the same, i.e., NRCY flow of 7.26 MGD. The A^2/O mode of operation did not produce the same denitrification for each month, so some other factor or factors must be affecting A^2/O anaerobic and anoxic denitrification more strongly than the NRCY. The VIP data showed a high degree of positive correlation (r^2 = 0.92) (Figure 4.10), though the A^2/O mode analysis did not (r^2 = 0.006). None of these three regressions were significant, as all of the significance values were greater than 0.05.

No significant relationship overall was found between the anaerobic and anoxic denitrification rate and the total nitrate recycle (r^2 = 0.31, P = 0.19). The total nitrate recycle is the sum of the return sludge recycle and the NRCY. The VIP data showed a high degree of positive correlation (r^2 = 0.82) (Figure 4.11), though the A^2/O mode regression analysis did not (r^2 = 0.35). None of these three regressions were significant, as all of the P values were greater than 0.05. Since the NRCY recycle was approximately half of the total recycle in the A^2/O mode and was the same value for three out of four months examined, this suggests that the anaerobic and anoxic denitrification was not dependent on the return sludge recycle. A regression of the A^2/O mode anaerobic and anoxic denitrification (in gN/gMLVSS-d) versus the corresponding average return sludge flow produced a correlation coefficient of 0.34 and a significance value of 0.41.

Secondary Clarifier Denitrification

The secondary clarifier contributed 5 to 16 percent of the system denitrification rate

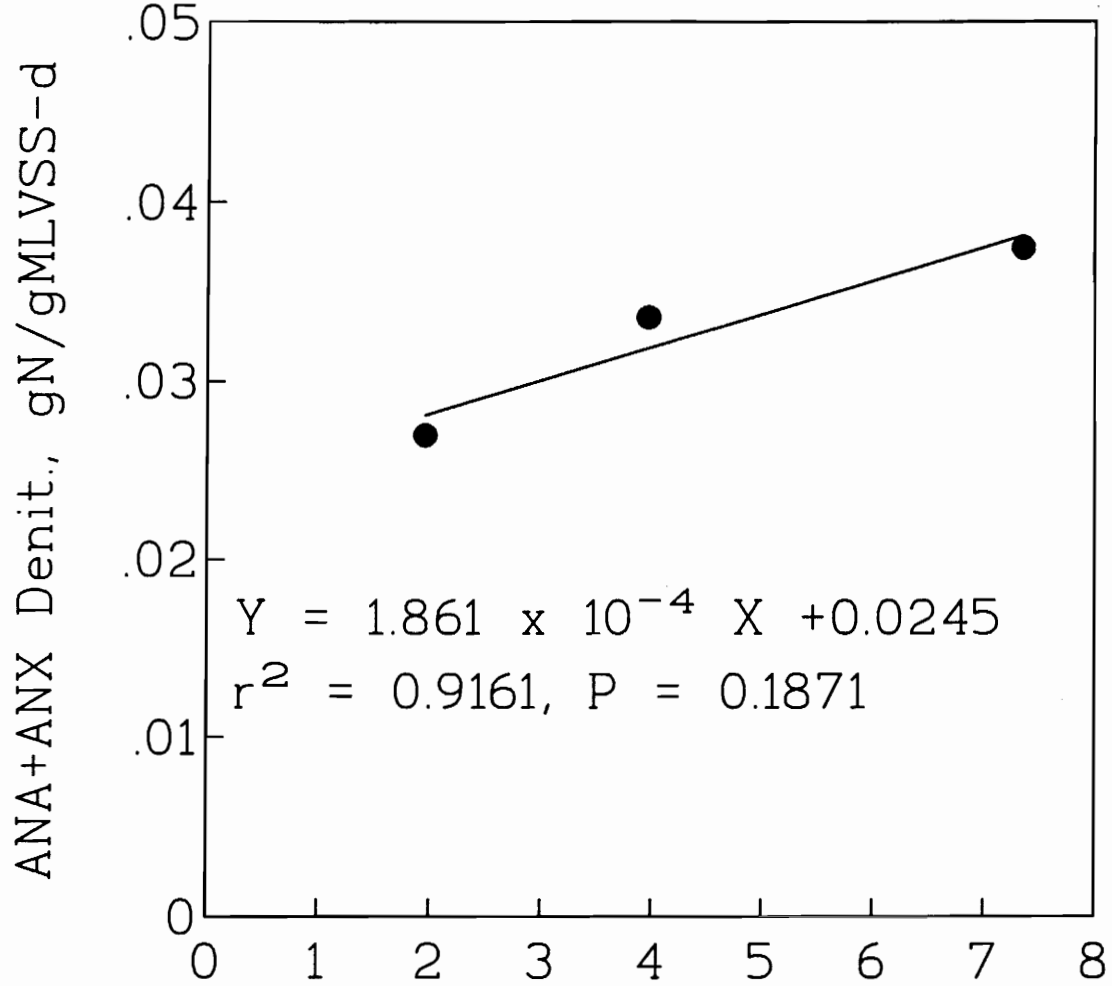


Figure 4.10. VIP ANA + ANX Denitrification vs. NRCY

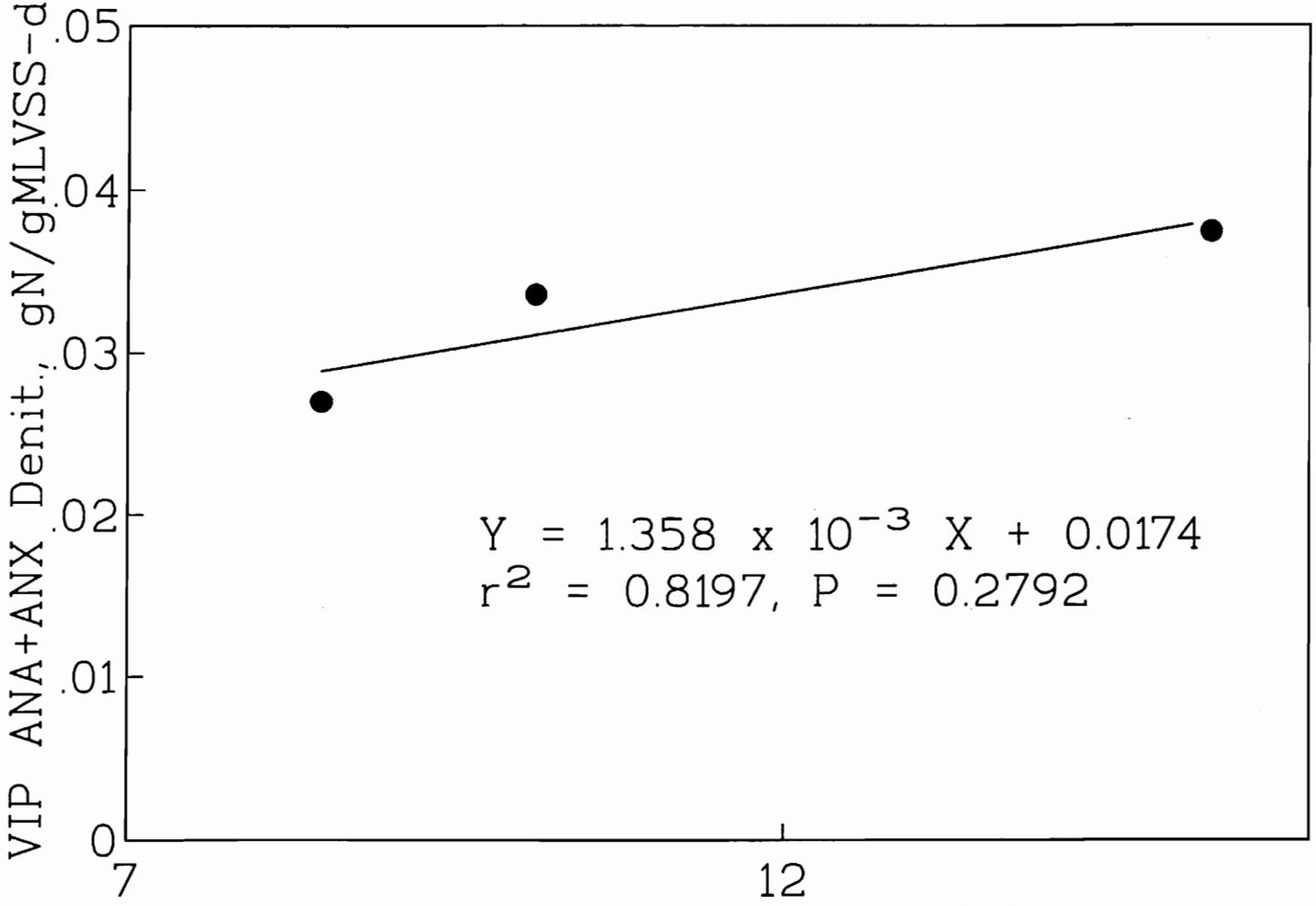


Figure 4.11. VIP ANA+ANX Denitrification vs. Total Nitrate Recycle

considering both systems of operation, from 23311 to 87264 gN/day or 0.0038 to 0.1520 gN/gMLVSS-d. The A^2/O mean secondary clarifier denitrification was 8 percent of the total system denitrification rate. The VIP mean secondary clarifier denitrification was 12 percent. Twenty three percent more secondary clarifier denitrification took place under the VIP mode of operation compared to the A^2/O mode. The results suggest that the denitrification that would have occurred in the aeration basin under the A^2/O process mode was taking place in the secondary clarifier under the VIP process mode. This could be due to decreased sludge ages found in the VIP mode. The mean A^2/O secondary clarifier denitrification was 42528 gN/day or 0.0050 gN/gMLVSS-d; the mean VIP secondary clarifier denitrification was 55079 gN/day or 0.0065 gN/gMLVSS-d.

The secondary clarifier nitrate (lb/day) was found to vary inversely with the NRCY (r^2 = 0.62)(Figure 4.12)(Table 4.9). This relationship was found to be significant (P= 0.02). The correlation coefficients found for both the A^2/O mode and the VIP mode were similar at 0.67 and 0.60, respectively. The significance values were also similar (0.18, 0.22, respectively) (Figures 4.13, 4.14). Since the data for the overall relationship produced a significant P value and neither analysis of the separate process mode datasets were significant, it suggests that more data for both process modes would be significant. Both the VIP and A^2/O data plots are traced on Figure 4.15. The slope of the line corresponding to the VIP data is steeper than that of the line for the A^2/O data.

The secondary clarifier nitrate (lb/day) was found to vary inversely with the total nitrate recycle (r^2 = 0.37), though less positively than with the NRCY (Table 4.9). This relationship was not found to be significant (P= 0.11). The correlation coefficients found for both the A^2/O mode and the VIP mode were found to be 0.18 and 0.56, respectively. The

Table 4.9

York River Demonstration Project

Secondary Clarifier and Recycle Rates Regression Analyses

Relationship	Overall	A^2/O	VIP
Secondary Clarifier Nitrate (lb/day) vs. Total Nitrate Recycle (MGD)[1]			
r^2=	0.37	0.18	0.56
P =	0.11	0.58	0.25
Y =	-16.96 X + 519.5	-14.24 X + 488.9	-18.57 X + 535.2
Secondary Clarifier Nitrate (lb/day) vs. NRCY Recycle (MGD)[2]			
r^2=	0.62	0.67	0.60
P =	0.02	0.18	0.22
Y =	-21.01 X + 419.1	-18.81 X +410.2	-25.35 X + 438.3

[1]Includes April, 1988 data in overall and VIP statistics. Excluding April, 1988 data in the overall dataset yielded an r^2 = 0.26 and a P = 0.24. Excluding April, 1988 data in the VIP dataset yielded an r^2 = 0.44 and a P = 0.54. These are not significantly different from including the data, so the data were included to increase the number of datapoints.

[2]Includes April, 1988 data in overall and VIP statistics. Excluding April, 1988 data in the overall dataset yielded an r^2 = 0.65 and a P = 0.03. Excluding April, 1988 data in the VIP dataset yielded an r^2 = 0.58 and a P = 0.44. These are not significantly different from including the data, so the data were included to increase the number of datapoints.

Secondary Clarifier Nitrate York River STP 1987 and 1988 Table 4.10								
Mode	July 1987 Nitrification without Nitrate Recycle	August-October A_2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Average monthly nitrate, mg/l	6.72	5.47	4.45	3.92	2.46	3.71	5.70	4.82
Process Average, mg/l		4.61				4.74		
Average monthly nitrate, mg/l	410	337	263	222	195	319	404	316
Process Average, lb/day		274				346		

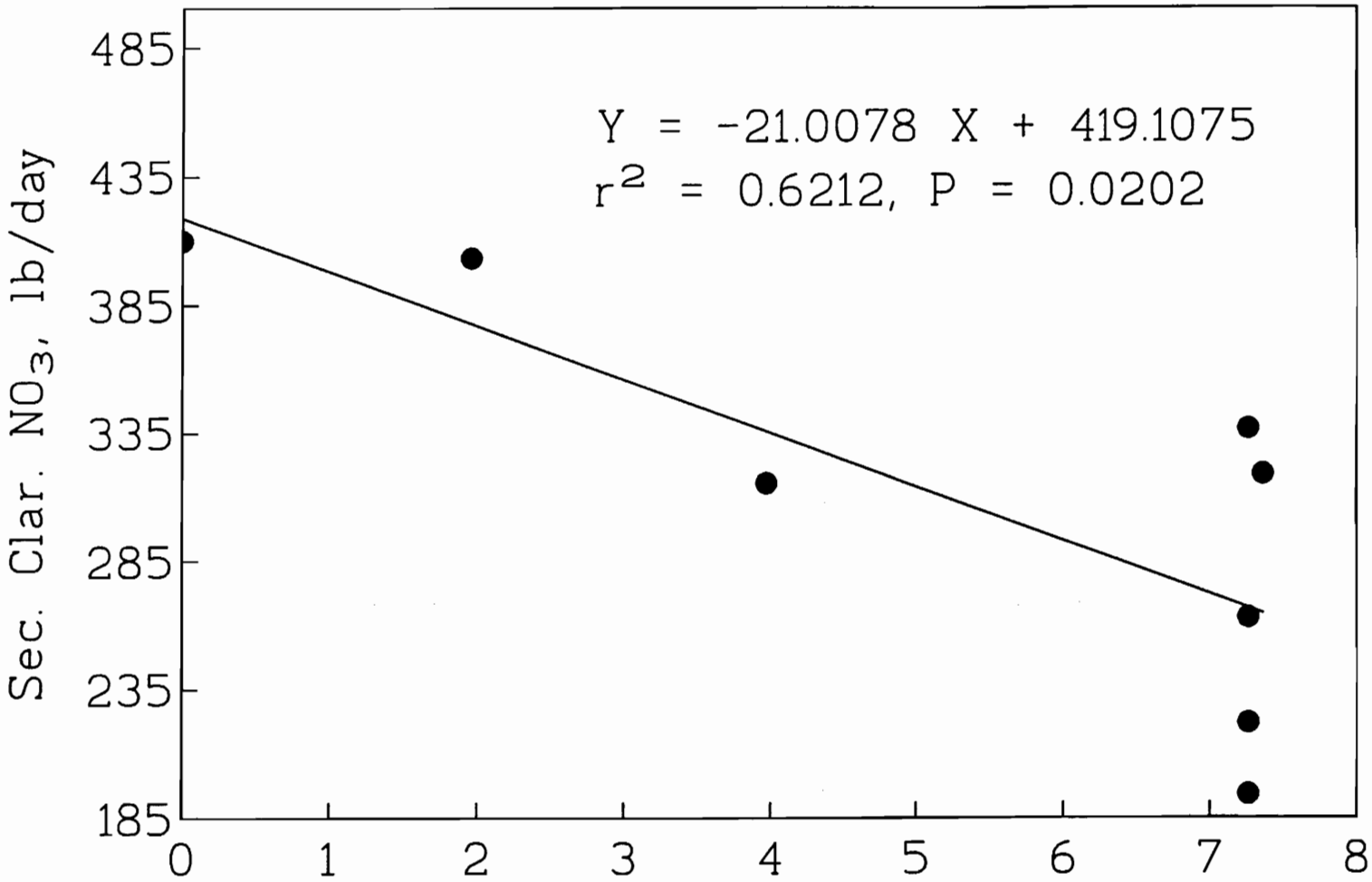


Figure 4.12. Secondary Clarifier Nitrate vs. NRCY Recycle

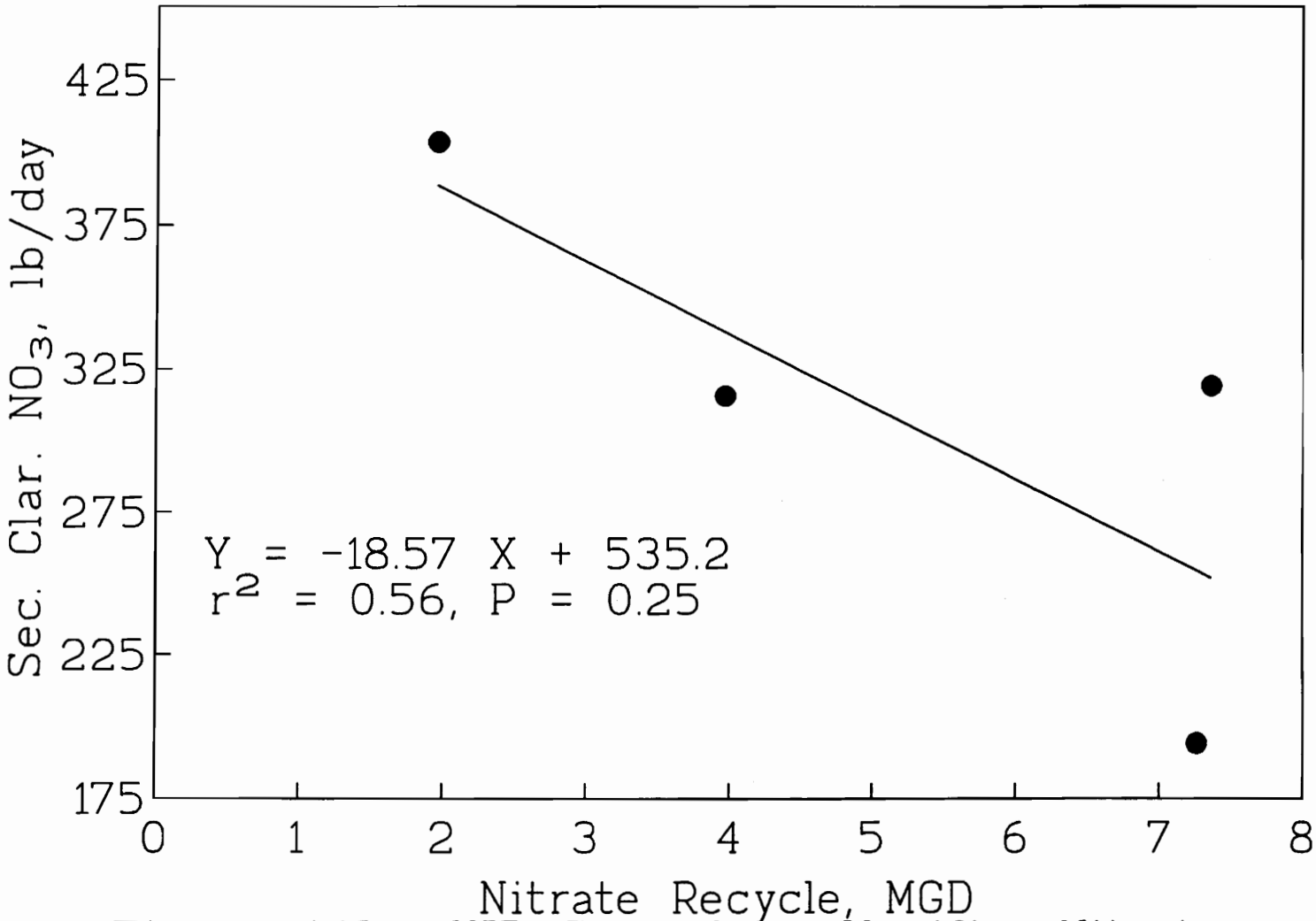


Figure 4.13. VIP Secondary Clarifier Nitrate vs. Nitrate Recycle

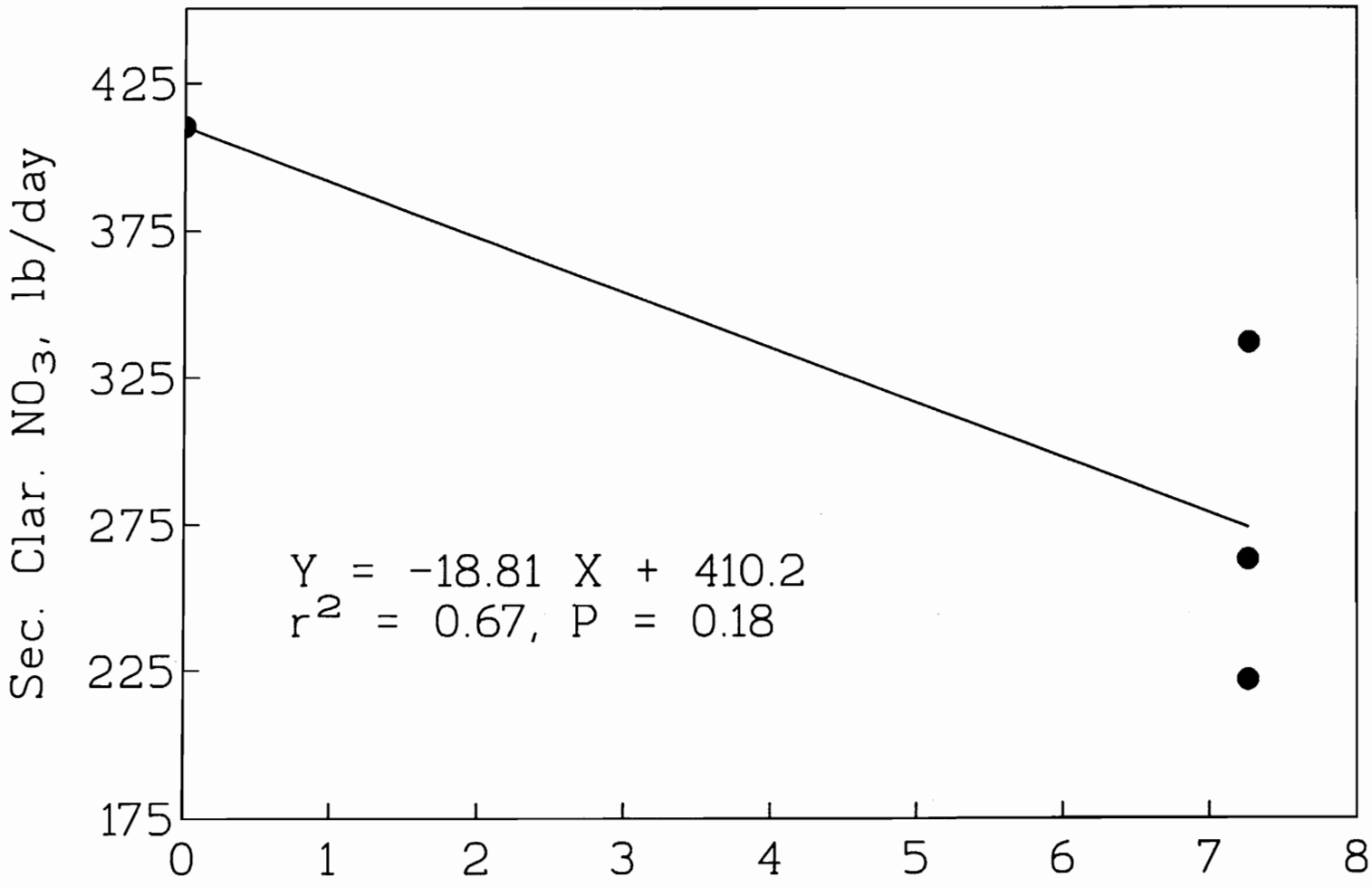


Figure 4.14. A^2/O Secondary Clarifier Nitrate vs. Nitrate Recycle

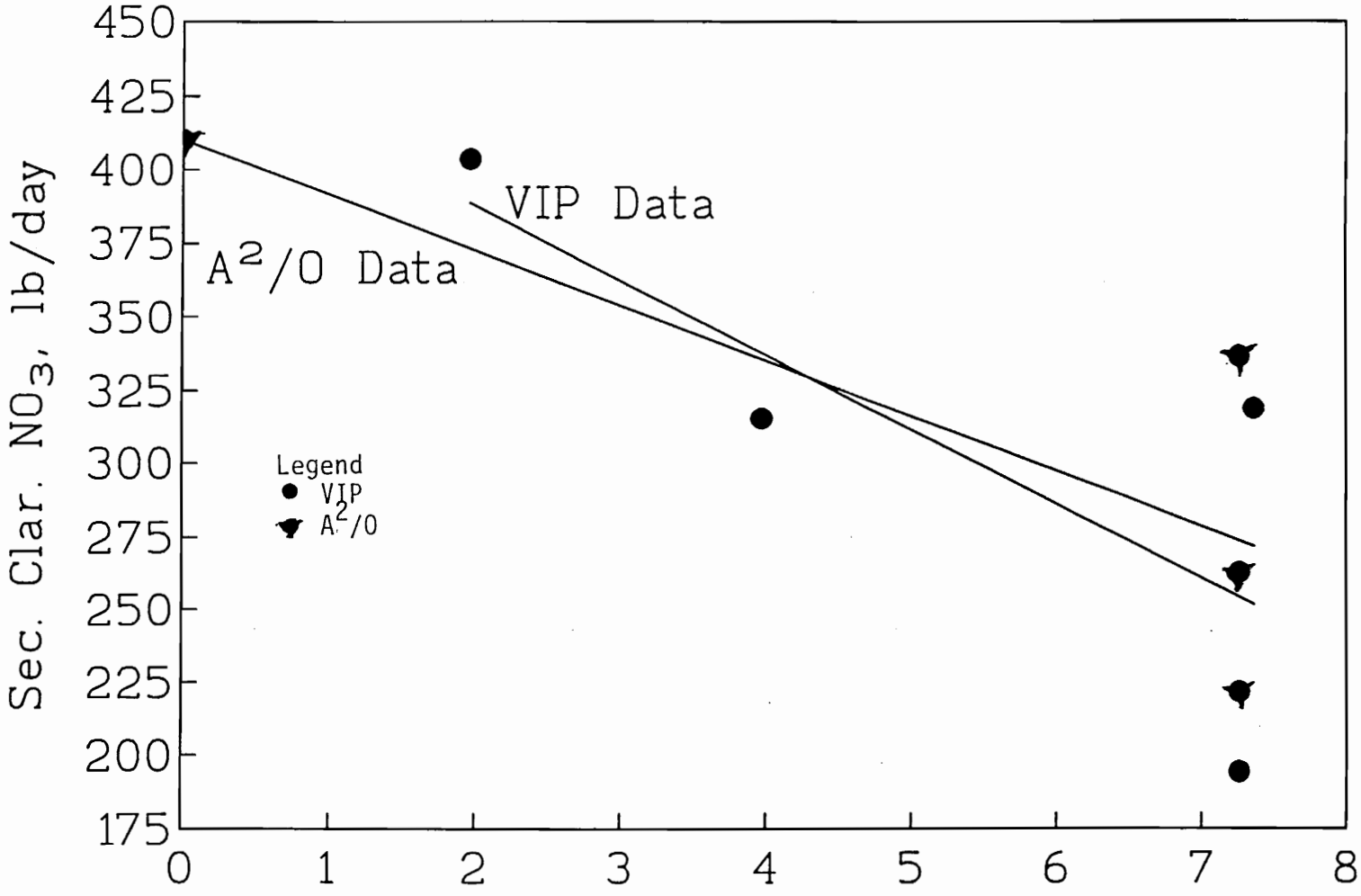


Figure 4.15. VIP and A^2/O Sec. Clarifier NO_3 vs. Nitrate Recycle

significance values were 0.58 and 0.25, respectively. The VIP plot of the secondary clarifier nitrate vs. the total nitrate recycle (Figure 4.14) was very similar to the VIP secondary clarifier nitrate vs. the NRCY. Two of the three VIP NRCY values were low, and the total nitrate recycle for those months would reflect the return sludge recycle more strongly. The relationship between the secondary clarifier nitrate and the total nitrate recycle was strongest for the VIP mode due to the configuration, which maximized nitrate to the anoxic zone.

Sludge Production

Heterotrophic observed yields were calculated in order to examine the sludge production for the two modes of treatment operation at the York River Sewage Treatment Plant BNR Demonstration Project. Associated calculations such as the specific growth rate, the sludge age, the fractional substrate utilization rate, biomass in the reactors, substrate consumed, food : microorganism ratio (F:M) and waste sludge production were also performed.

Heterotrophic yields, with the exclusion of the high flow period of February 10 through March 31, 1987, averaged 0.34 $days^{-1}$ in the A/O and A^2/O periods. The VIP heterotrophic yields in 1988 averaged 0.49 $days^{-1}$. The yields for the A/O and A^2/O periods varied from 0.18 $days^{-1}$ in September 1 through October 31, 1987 to the second highest value overall of 0.56 $days^{-1}$ for May 1 through May 31, 1987. In the VIP mode, the heterotrophic yield was 0.64 $days^{-1}$ in April, 1988. The yield may have stabilized subsequent to that time to a value of 0.40 $days^{-1}$ for the period July 1 through August 31, 1988 and a similar value of 0.42 $days^{-1}$ for July 1 through 31, 1989. The yields varied inversely and significantly with increasing sludge age (Figure 4.16). With increased sludge ages, endogenous metabolism becomes a greater factor and the net result is less sludge.

Table 4.11

York River Demonstration Project
Hampton Roads Sanitation District

Heterotrophic Sludge Parameters

Operational Mode	A/O Mode			No Anoxic Zone	A^2/O Mode	VIP Mode		
Time Period	11/1-11/30/86	2/10-3/31/87	5/1-5/31/87	7/1-7/31/87	9/1-10/31/87	4/1-4/30/88	7/1-8/31/88	7/1-7/31/89
Mass Fraction	50-0-50	33-0-67	33-0-67 from 5/1-5/14 25-0-75 from 5/15-5/31	25-0-75	25-25-50	25-25-50	25-25-50	25-25-50
q (1/Day)	0.87	0.47	0.48	0.62	0.80	0.36	0.61	0.58
μ (1/Day)	0.33	0.33	0.27	0.13	0.14	0.23	0.24	0.24
Biomass XV_a lb VSS	7698	17000	12470 5/1-5/14 14400 5/15-5/31 13400 avg	18450	16290	21720	13400	17600
Substrate Consumed lb/day BOD_5	6695	8009	6418	11420	13090	7843	8121	10230
Waste Sludge lb/Day	2550	5589	3605	2469	2347	5056	3210	4274
F:M BOD_5 basis	0.95	0.52	0.53	0.63	0.82	0.41	0.62	0.62
Sludge Age, days (Eqn 3.20)	3.0	3.0	3.7	7.5	6.9	4.3	4.2	4.1
Y_{obs} BOD_5 basis	0.38	0.70	0.56	0.22	0.18	0.64	0.40	0.42

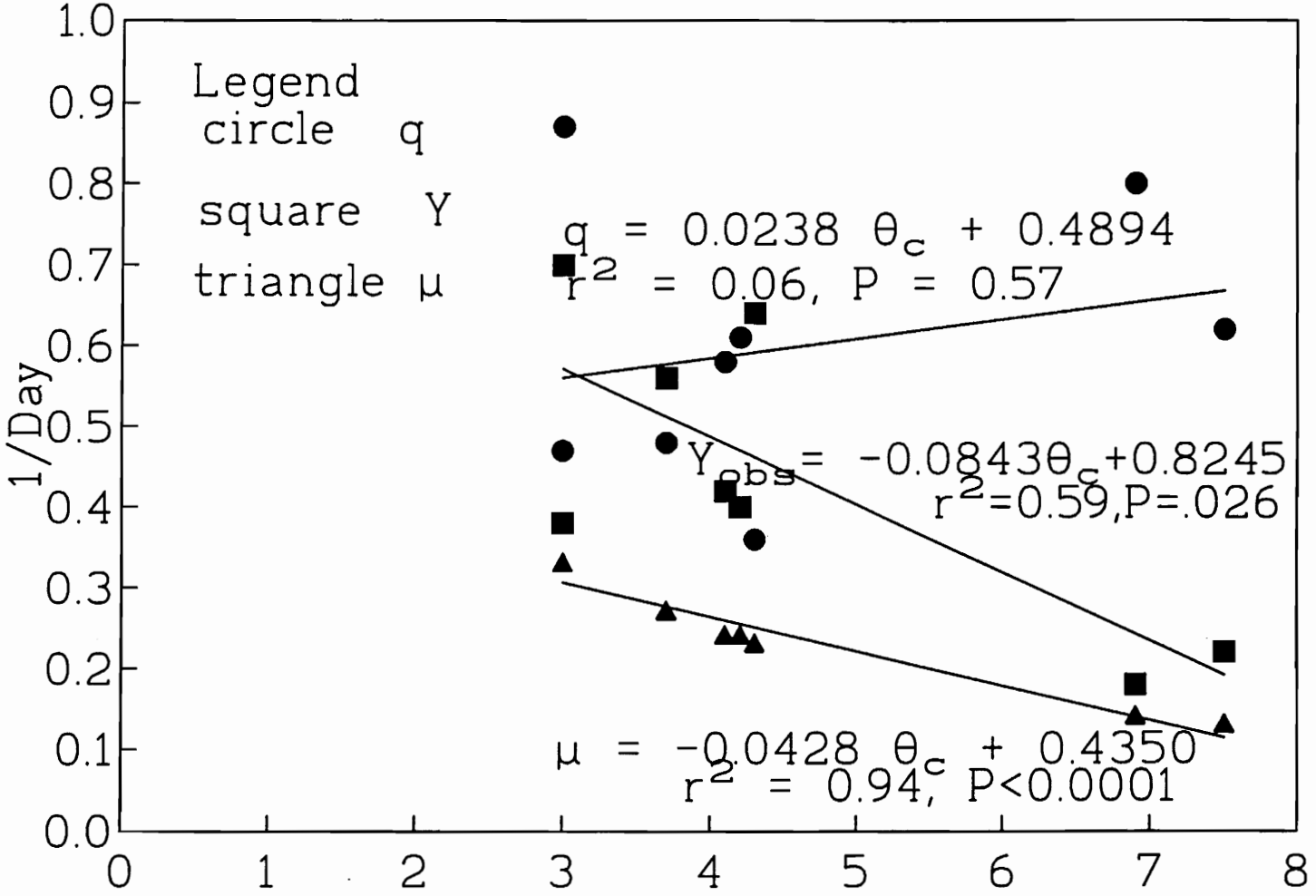


Figure 4.16. q, Y and mu vs. Sludge age

During the high flow period discussed, raw influent flows averaged 10.43 to 11.36 MGD. During the other months of 1987 and 1986 discussed, flows ranged from 5 to 7 MGD, except for May, 1987 which ranged from 5 to 8 MGD. For the months of 1988 and 1989 considered, flows ranged from 5 to 9 MGD.

Heterotrophic observed yields were highest at 0.70, 0.56 and 0.64 during the three highest flow periods under study: February 10 through March 31, May 1 through May 31, 1987 and April 1 through 30, 1988.

The specific growth rate function, μ, is the rate of growth per unit of biomass and has units of time^{-1} (Benefield and Randall, 1980). It is derived from the understanding that the rate of growth of the biomass is proportional to the amount of biomass present. The specific growth rate had two levels of values during the A/O and A^2/O periods examined. The first tier of values of the specific growth rate consisted of 0.33 days^{-1} for the periods November 1 through November 30, 1986 and February 10 through March 31, 1987 and 0.27 days^{-1} for May 1, through May 31, 1987. February 10 through March 31, 1987 was the high flow period and the sludge from that time would understandably have a high specific growth rate, since the growth rate is the reciprocal of the sludge age (the amount of time the sludge remains in the system). This is why the graph of the growth function vs. the sludge age varies inversely, linearly and significantly (Figure 4.16). However, flow was another factor to consider in the high growth rate for in November, 1986 since the mean raw influent flow was 6.09 MGD. During November, 1986, the biomass inventory was the lowest of all the time periods under study. Since the growth rate is calculated by dividing the amount of sludge wasted by the amount of biomass in the system and the system biomass is a quantity located in the denominator of the growth rate proportion, a low value leads to a high growth rate. For May, 1987, flows were slightly higher

at 7.38 MGD; both the waste sludge and the biomass were in the lower range of values, resulting in a higher proportion.

The second level of growth rate seen in 1987 is represented by the values 0.13 days for July 1 through July 31, 1987 and 0.14 days for September 1 through October 31, 1987. Values for the waste sludge or the biomass in the reactors, components of the specific growth rate relationship, do not show this type of tiered variation. However, the rate of substrate utilization does show lower values for the periods November 1 through November 30, 1986, February 10 through March 31, 1987 and May 1, through May 31, 1987. The substrate consumed makes a large increase during the period July 1 through July 31, 1987 and increases to September 1 through October 31, 1987. The heterotrophic observed yield shows this relationship in that the periods November 1 through November 30, 1986, February 10 through March 31, 1987 and May 1, through May 31, 1987 show an average yield of 0.55, whereas the average yield for the remaining periods of A/O and A^2/O operation is 0.20.

Specific growth rate levels were remarkably consistent for the VIP months examined. The rate was calculated as 0.23 $days^{-1}$ and 0.24 $days^{-1}$, respectively, for the periods of April 1 through April 30, 1988 and the two periods July 1 through August 31, 1988 and July 1 through July 31, 1989. Since neither the rate of substrate utilization nor the amount of biomass present stayed constant during this period, the consistent results suggest a strong correlation between the two for the VIP study period.

The sludge age was calculated for Table 4.11 by dividing the mean amount of biomass by the amount of sludge wasted. Sludge ages for the A/O and A^2/O modes of operation examined ranged from 3.0 to 7.5 days, with a mean of 4.8 days. The phases of VIP operation examined exhibited sludge ages which ranged from 4.1 to 4.4 days. The mean VIP sludge age

was 4.2 days.

The mean fractional substrate utilization rate, q, or $(dS/dt)_u/X$ was 0.65 $days^{-1}$ during the A/O and A^2/O periods and 0.52 $days^{-1}$ during the VIP months under study. The range for the A/O and A^2/O periods was 0.47 $days^{-1}$ for the high flow time frame from February 10 through March 31, 1987 to 0.87 $days^{-1}$ for November 1 through November 30, 1986. The range for the VIP months was 0.36 $days^{-1}$ for April 1 through April 30, 1988 to 0.61 $days^{-1}$ for July 1 through August 31, 1988. The fractional substrate utilization rate for the time period July 1 through July 31, 1988 at 0.58 $days^{-1}$, like the specific growth rate, was also similar to that from July 1 through August 31, 1988. According to Figure 4.16, the mean fractional substrate utilization rate, q, increased with increasing sludge age. More time spent in the system allowed more BOD to break down and become more usable to the biomass.

The mean amount of biomass in the reactors was 14583 lb VSS for the A/O and A^2/O periods and 17572 lb VSS for the VIP months. The range for A/O and A^2/O periods was 7698 lb VSS for November 1 through November 30, 1986 and 18453 lb VSS for July 1 through July 31, 1987. The VIP range was 13406 lb VSS for July 1 through August 31, 1988 to 21719 lb VSS for April 1 through April 30, 1988.

The mean amount of substrate consumed (as BOD_5) was 9127 lb/day for the A/O and A^2/O periods and 8731 lb/day for the VIP months. The range for A/O and A^2/O periods was 6418 lb/day for May 1 through May 31, 1987 and 11420 lb/day for July 1 through July 31, 1987. The VIP range was 7943 lb/day for April 1 through April 30, 1988 to 10230 lb/day for July 1 through July 31, 1989. Substrate consumed, like the mean fractional substrate utilization rate, q, increased with increasing sludge age (Figure 4.17). Unlike q, the BOD_5 consumed vs. the sludge age graph was linear and significant. The mean fractional substrate utilization rate is

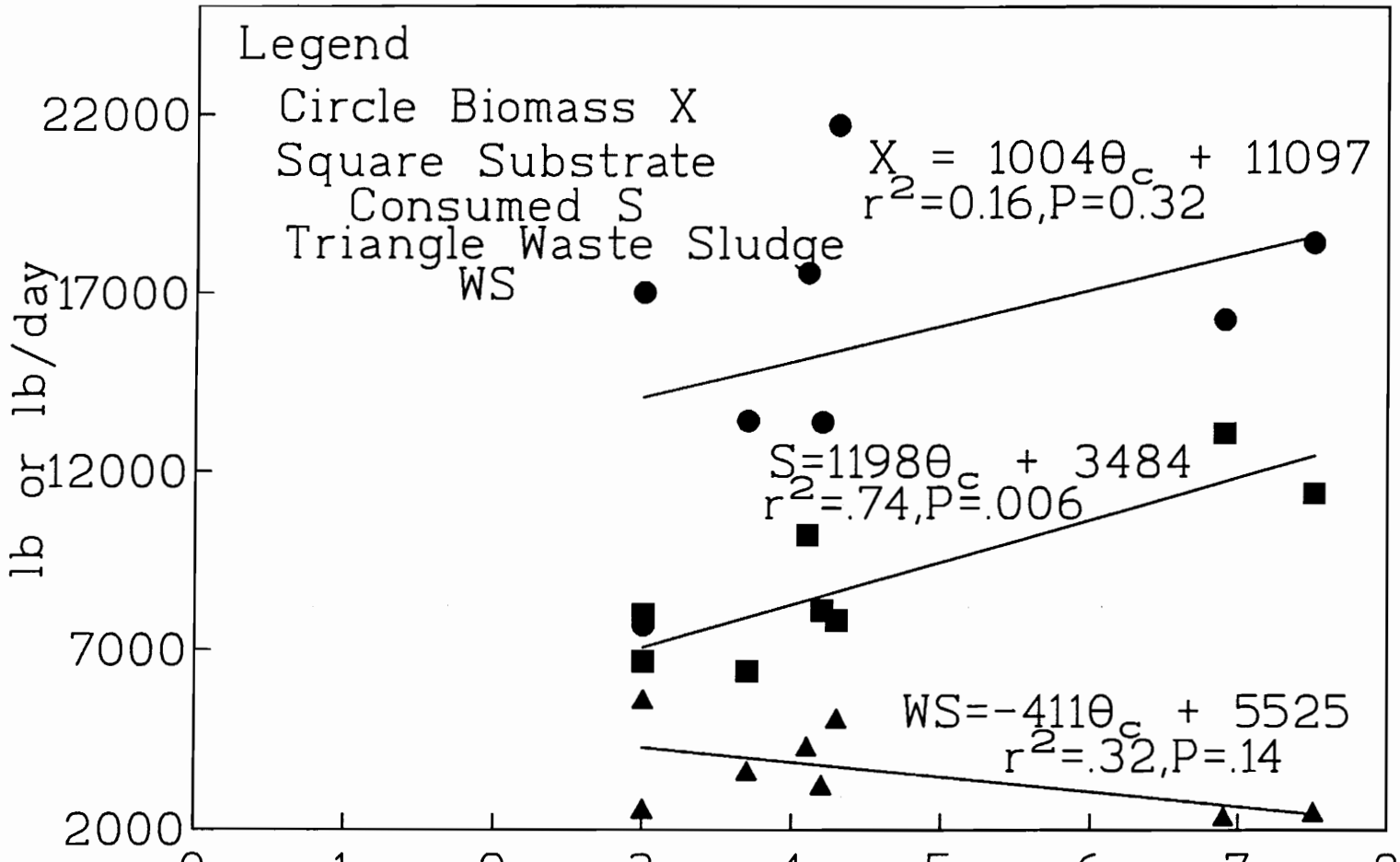


Figure 4.17. X, S and WS vs. Sludge Age

calculated by division by the biomass value, X, in the denominator. From Figure 4.17, X did not vary linearly with increasing sludge age and for this reason, the mean fractional substrate utilization rate vs. sludge age graph was not linear either.

The mean F:M ratio was 0.69 for the A/O and A^2/O periods and 0.55 for the VIP months. The range for A/O and A^2/O periods was 0.95 for November 1 through November 30, 1986 to 0.52 for February 10 through March 31, 1987. The VIP range was 0.42 for April 1 through April 30, 1988 to 0.62 for both July 1 through August 31, 1988 and July 1 through July 31, 1989.

Waste sludge averaged 2743 lb/day for the A/O and A^2/O periods and 4180 for the VIP months. The high flow period yielded a high amount of sludge at 5589 lb/day, but the April, 1988 time frame yielded almost as much (5056 lb/day). The total pounds of biomass were also high for April, 1988, probably to induce nitrification. The range for A/O and A^2/O periods was 2347 lb/day for September 1 through October 31, 1987 and 3605 lb/day for May 1 through May 31, 1987, not counting the high flow period. The VIP range was 3210 lb/day for July 1 through August 31, 1988 to 5056 lb/day for April 1 through April 30, 1988. As seen with the sludge yields, the amount of waste sludge decreased with increased sludge age (Figure 4.17).

BOD to TKN ratios were calculated for the months under study. This ratio ranges from a low of 4.66 in May 1988 to a high of 7.96 seen in October 1987. With the exception of the August 1987 ratio, ratios for 1988 were lower than those for 1987. COD to TKN ratios were very stable throughout the study period. Results were 0.08 for every month except July, 1988, when the ratio was 0.07.

Table 4.12

BOD/TKN Ratios York River STP 1987 and 1988

Mode	July 1987 A/O Mode Nitrifi-cation without Nitrate Recycle	August-October A_2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Average PCE BOD/ PCE TKN Ratio[1]	192/ 29.0= 6.62	162/ 29.2= 5.54	207/ 26.9= 7.69	229/ 29.0= 7.89	117/ 19.8= 5.89	109/ 23.5= 4.64	188/ 24.9= 7.55	184/ 29.7= 6.20
Inverse (for compar-ison)	0.15	0.18	0.13	0.13	0.17	0.22	0.13	0.16
Average Raw TKN/Raw COD Ratio[2]	29.1/ 377= 0.08	29.5/ 383= 0.08	30.4/ 395= 0.08	32.3/ 413= 0.08	29.2/ 345= 0.08	26.0/ 346= 0.08	27.7/ 364= 0.08	29.2/ 437 =0.07
Inverse (for compar-ison)	13.0	13.0	13.0	12.8	11.8	13.3	13.1	15.0

[1]As discussed in EPA (1975)
[2]As discussed in Ekama et al. (1984)

CHAPTER 5
DISCUSSION

The York River Biological Nutrient Removal (BNR) Demonstration Project was conducted at the Hampton Roads Sanitation District (HRSD) York River plant to develop greater understanding of BNR outside of a laboratory or pilot plant setting. The processes of nitrate formation and nitrogen removal at the York River Demonstration Process were studied to compare full scale rates to pilot plant results, and other full scale experiments reported in the literature. The process of sludge production was examined from the perspective of determining differences between modes of operation.

The month of July, 1987, is excluded from discussions and totals of the A^2/O processes unless noted even though nitrification was complete because there was no nitrate recycle in operation. The month is not included in anaerobic and anoxic denitrification totals since during this month a combined anaerobic and anoxic basin denitrification was calculated. The month of April, 1988 experienced incomplete nitrification, and thus provided low values for nitrification and denitrification processes. Only the values representative of the processes will be included in the discussion.

Nitrification Rates

York River Demonstration Project system nitrification results were found to be equal when determined by ammonia appearance and nitrate disappearance. There were no unknown losses by denitrification. The equality of the mass balance equations are shown in Appendix II. Hall and Murphy (1980) determined that reaction rates for *Nitrosomonas* determined as either the rate of disappearance of ammonia or the appearance of nitrate were found to be statistically comparable.

York River Demonstration Project
Hampton Roads Sanitation District

Table 5.1

Nitrification Rates

	mg/l-hr	Range	gN/gMLVSS-d	Monthly Range
York River				
A/O	10.36		0.1129	
A^2/O	10.01	9.22-10.91	0.1168	0.1153-0.1192
VIP	9.36	7.65-11.08	0.1157	0.1116-0.1230
Hockenbury (1977)		7.5-12.0		
Gee (1990a)		7.16-25.86		
McClintock (1993)			0.1373	
Wild (1971)				0.08-0.18

York River BNR Demonstration Project aeration basin nitrification rates ranged from 9.22 to 11.08 mg/L-hr and 0.1116 to 0.1230 gN/gMLVSS-d for the eight months of study. Averages of A^2/O mean cell residence times ranged from 6.9 days for September and October, 1987 to 4.3 days in April, 1988 and 4.2 days in July and August, 1988 for VIP average mean cell residence times. Hockenbury *et al.* (1977) found that laboratory reactor cultured nitrifiers (at unknown temperature and simulated 24 day sludge age) produced nitrification rates between 7.5 and 12 mg NO_3-N/L-hr in the presence of active heterotrophic populations and domestic sewage sludge. Gee *et al.* (1990a) determined ammonia oxidation rates in a laboratory reactor at a room temperature of 23 plus or minus 2 degrees C and a hydraulic retention time of about 4 days for *Nitrosomonas* to be 7.16 to 25.86 mg-N/L-h. McClintock *et al.* (1993) found an ammonia oxidation rate of 5.720 mg N/g MLVSS - h for a BNR system operating at a sludge age of 5 days and a temperature of 20 degrees. If this figure is multiplied by 24 hr - grams/1000 mg-day to convert to units of gN/gMLVSS-d, the result is 0.1373 gN/gMLVSS-d. Wild *et al.* (1971) cultured nitrifiers in a laboratory reactor at 20 degrees C and pH of 7-8 and found nitrification rates with increasing pH of 0.08 to 0.18 g NH_3nitrified/day/g MLVSS. The York River rates are similar to those reported in the literature (Table 5.1).

System nitrification rates, in gN/day, averaged 9 percent lower for the VIP mode than they did for the A^2/O operating mode. Lambert's Point VIP Pilot Study reported comparable nitrification results between VIP and A^2/O operating modes (CH2M Hill, 1987). Influent flows were not as consistent during the York River Demonstration Project VIP operating mode for the months under study as they were during the A^2/O operating mode. The data indicate that the aerobic zone was D. O. limited during July, 1988, *i.e.*, during VIP operation. More sludge was wasted during the VIP mode of operation, which decreased the ammonia availability for

nitrification. Four percent less ammonia was available for nitrification for the VIP mode of operation than was available for A^2/O. The highest nitrification rate of the study observed with the VIP process was during May, 1988.

Nitrification outside the Aeration Basin

In the months when nitrification took place outside of the aerobic basin, it occurred in the return lines and the secondary clarifier. This occurred during the months of April, 1988, September, 1987 and October, 1987. These were the cooler months of those examined. These months were lowest in system nitrification rates (with the exception of June and July, 1988, during which effects from decreased aeration basin dissolved oxygen were seen, as will be explained below). During cooler weather, nitrification takes longer to accomplish. One would expect that some nitrification would therefore be occurring in the oxygenated portion of the secondary clarifier further along in the treatment train.

Temperature Effects

The effects of temperature are seen in the highest 1987 overall nitrification and denitrification rates (in gN/day), which occurred in August, 1987. August, 1987 shared with July, 1987, the high raw influent monthly average temperature of 24.5 degrees C. The month of August, 1987, differed from July, 1987, in that the nitrate recycle was active in August and not in July.

Temperature effects are not seen in the VIP rates. In May, 1988, the highest overall nitrification and denitrification rates occurred, with continued decreases seen through June and July, 1988. Since one would expect the overall rates to increase from May to July as

temperatures increased, some other factor besides temperature must have affected these rates. Denitrification dropped during June, 1988 because the nitrate recycle was decreased by half as much as during the previous month. This decreased rate was continued during the month of July, 1988. D. O. decreased in the aeration basin during June, 1988 and became limiting during July, 1988. Nitrification decreased during those months and, because less nitrate was formed, denitrification decreased also. The effect of temperature may be seen in 1987 more than in 1988 because flows during the 1987 months were more consistent. Flows peaked in May, 1988 and fell off during June and July, following the nitrification and denitrification trend.

The aeration basin specific nitrification rates (expressed per MLVSS) for the year 1987 do not seem to vary as much as those calculated in units of gN/day. The percent difference between the largest and the smallest values divided by the sum of the three values was 8 percent for the gN/day values compared to 1 percent for the specific rate. Temperature effects were not seen in the rate expressed per MLVSS expressions since both the rates and the biomass production are biological activities affected by temperature according to the van't Hoff relationship. Thus, the impact is seen both in the numerator and denominator, and cancel.

Temperature effects were not seen in the specific denitrification rates calculated in this study because seasonal effects were not seen. The specific denitrification rate calculated in the Lambert's Point VIP pilot did not vary with the primary effluent COD or BOD concentration but with the temperature in the first anoxic reactor (Daigger *et al.*, 1988).

Denitrification

York River Demonstration Project specific rates of denitrification (0.1249 gN/gMLVSS-d, A^2/O average, and 0.0815 gN/gMLVSS-d, VIP average) were lower than those of Turk and

York River Demonstration Project
Hampton Roads Sanitation District

Table 5.2

System Denitrification Rates

	gN/gMLVSS-d	Monthly Range
York River		
A/O	0.0921	
A^2/O	0.1249	0.1095-0.1520
VIP	0.0815	0.0624-0.0971
Lambert's Point VIP Pilot		0.13-0.19
Randall et al. (1991)	0.14	
Turk and Mavinic (1987)		0.7-1.17
Engberg and Schroeder (1975)		0.13-0.35
Abufayed and Schroeder (1986b)		0.03-0.24 MCRT 6 days: 0.07-0.12 MCRT 12 days: 0.02-0.14

Mavinic (1987) in a bench scale batch test using glucose and lactose as a substrate, ranging from 0.7 to 1.17 gN/gMLVSS-d. Engberg and Schroeder (1975) determined the specific nitrate removal rate to range from 0.13 to 0.35 $days^{-1}$ for a continuous feed methanol substrate laboratory experiment with mean cell residence times ranging from 4 to 14 days. Abufayed and Schroeder (1986b) found overall nitrate and nitrite denitrification rates ranging from 0.03 to 0.24 mg/l NO_x removed per mg/l MLSS-day in laboratory experimentation with various carbon sources. The range for the mean cell residence times of six days was 0.07 to 0.12 mg/l NO_x removed per mg/l MLSS-day (similar to York River VIP operation) and the range for the mean cell residence times of 12 days was 0.02 to 0.14 mg/l NO_x removed per mg/l MLSS-day (similar to York River A^2/O operation). It is not surprising that rates found in a laboratory setting under controlled circumstances using specific organic substrates would be higher than those found in the field using a substrate consisting of the multiple compounds found in municipal wastewater. Organic substrates typically used in the laboratory are readily biodegradable and thus produce higher reaction rates whereas a portion of municipal wastewater is not and must first be broken down by bacterial action. An additional consideration is that generally in a laboratory the time for a process to take place may be known. In a field setting it is known that the biological process, if complete, took place at a time equal to or less than the HRT, but this time is not known precisely. Instead it is assumed that the process took the same amount of time as the HRT, which is an overestimation. Since the overestimation is in the denominator of the rate expression, the expression as a whole is underestimated in the field (Randall, 1995).

With the HRSD Lambert's Point VIP Pilot Plant, denitrification rates were 4 to 6 mg N/g MLSS-hr at temperatures greater than about 17 degrees C. Multiplying this figure by g-24 hr/1000 mg-day and an approximate conversion of gMLSS/0.74 g MLVSS (to be able to

compare these data to those generated in units of gN/gMLVSS-d) yields results of 0.13 to 0.19 gN/gMLVSS-d. At Lambert' Point, 0.74 g MLVSS/g MLSS was found on average over the entire VIP study (CH2M Hill, 1987). Randall, *et al.* (1991) found a rate of 5.7 mg N/g MLVSS-h for a pilot plant BNR system at 20 degrees C and a MCRT of 5 days. This converts to 0.14 gN/gMLVSS-d., which compares to the upper range of the York River Demonstration Plant results. The York River Demonstration Plant results, ranging from 0.02 to 0.15 gN/gMLVSS-d are somewhat less than the Lambert's Point VIP pilot plant results.

Operational Comparisons: Denitrification

The average specific denitrification rate for A^2/O (calculated using aerobic denitrification equation 3.14) of 0.1249 g N per g MLVSS-d is one and one half times as high as the 0.0815 gN per g MLVSS-d rate for VIP. Both process modes had a nitrified recycle with the exception for A^2/O of July 1987. During VIP operation, the nitrate rich return sludge was returned to the anoxic basin instead of the anaerobic basin while the denitrified recycle originated from the anoxic basin and went to the anaerobic basin. This maximized the nitrates to the anoxic basin for removal through denitrification, at the same time minimizing nitrate levels in the anaerobic basin to reduce interference with phosphorus removal. However, in June and July, 1988, when one would expect the highest denitrification rates due to temperature effects, the rates were lowest. This was due to a 62% decrease in nitrate loading to the anoxic zone for VIP compared to A^2/O (Table 4.6). The nitrate recycle rate was reduced from 7.36 MGD to 3.68 MGD in June, 1988 and the rate of 4.03 MGD was maintained in July, 1988, while it remained constant at 7.26 MGD for A^2/O operation. Thus the denitrification rate was found to vary with the anoxic zone nitrate loading (Table 4.7, Figure 4.1).

Effects of Recycle on Denitrification

A significant relationship (r^2 = 0.65, P = 0.03) was found between the total denitrification rate and total nitrate recycle (the sum of the return sludge and nitrate recycle rates) (Table 4.7, Figure 4.3). Since there is a relationship between the denitrification rate and the anoxic zone nitrate loading, it follows that the rate at which nitrates are delivered to the anoxic zone would also be correlated to denitrification. Both the A^2/O and VIP configuration data by themselves (Figures 4.4 and 4.5) show a higher correlation coefficient than all of the data considered together. This indicates that the process denitrification rates were different as illustrated by differing slopes of the line on the graphs.

The relationship between the total denitrification rate and the nitrate recycle alone followed a trend similar to the total denitrification rate versus the total nitrate recycle, but less strongly (Figure 4.6). The correlation coefficient was 0.42 and the significance factor was 0.12. Again, better regressions were achieved by the data from the separate process modes (0.65 for A^2/O, 0.90 for VIP) (Figures 4.7 and 4.8). None of these three regressions were significant. The regression produced by the VIP data was much better than that for the A^2/O data since the nitrate recycle varied much more during the VIP mode of operation.

Aerobic Denitrification

Denitrification losses during aeration were larger than expected. High loading and high temperatures encourage losses of this nature (Ludzack and Ettinger, 1962). Aerobic denitrification rates ranged from 50 to 75 percent of the system denitrification rate, when it was expected to be insignificant. Denitrification, or nitrate dissimilation, by the *Pseudomonas* heterotrophic bacteria, is inhibited by oxygen. When oxygen is present, oxygen molecules are used (in heterotrophic metabolism) preferentially by the organism for the stabilization of organic

matter instead of the bound oxygen in the nitrate form (denitrification), because denitrification produces less energy for the organism (U. S. EPA, 1975). The use of nitrate or oxygen as the final electron acceptor is the only difference in the organic carbon stabilization reaction for these organisms (Sedlak, 1991). However, since the aerobic zone selects for these facultative organisms and the location of some of these organisms will be beyond the reach of oxygen, or at least surrounded by an oxygen level of less than 1.0 mg/l in the floc, it makes sense to expect nitrate to be used opportunistically. The denitrification rate has been shown to be influenced by the readily biodegradable influent nitrate in that the denitrification rate in the second anoxic cell is smaller than in the first (Barnard, 1979). Despite the oxygen inhibition of denitrification in the aeration basin, since the majority of the system's nitrate is formed there, more nitrate will be present in this basin for denitrification than any other basin. If low D.O. occurs, either because of poorly mixed sections within the basin, or the high rate activated sludge flocs develop anoxic centers, denitrification will rapidly follow. The nitrate formed was being denitrified as soon as it was nitrified. It is thought that the organic substrate used for the aerobic denitrification at York River was the poly-β-hydroxybutyrate formed in the phosphorus removal process.

During the Lambert's Point VIP pilot study, observed nitrogen removals exceeded predicted removals by about 35 percent. It was speculated that denitrification was occurring in other areas of the system besides the anoxic basin (CH2M Hill, 1987). Based on the observations of the York River basin denitrification study, it appears that this denitrification may have been occurring in the aeration basin. The aeration basin was the location of 66 percent of the A^2/O and 69 percent of the VIP specific system denitrification rates.

Anaerobic Denitrification

Because nitrates were returned from the aeration basin to the anaerobic zone by the return sludge in the A^2/O mode of operation, and were returned to the anoxic zone in the VIP mode, the average specific anaerobic rate of denitrification for the A^2/O mode (0.0099 gN/gMLVSS-d) was three times the VIP rate (0.0033gN/gMLVSS-d). The mean A^2/O anaerobic denitrification expressed in gN/day was 23997; the VIP mean was 9060 gN/day.

The months of September and October, 1987 could be considered as representative months of A^2/O anaerobic basin operation, due to the presence of the anoxic basin and a decreased return sludge rate which produced a more lower and more consistent nitrate loading. The A^2/O anaerobic denitrification for these two months alone (in gN/day) was extremely low: 0.65% of the total denitrification. This could be due to anaerobic stabilization of BOD. Because no anaerobic effluent nitrate data was collected, the assumption was made that no nitrate persists after travel through the anaerobic zone (Randall, 1993a). Since no nitrate from the return sludge recycle was left to carry over to the anoxic zone and denitrification in this zone was extremely low, the stabilization of BOD with the oxygenated nitrogen by the heterotrophic facultative bacteria is an explanation for the nitrate disappearance with so low a denitrification rate. The anaerobic denitrification data produced by the VIP operation, by comparison, averaged 2.23% of the total denitrification (in gN/day).

Anoxic Denitrification

The VIP configuration optimized the presence of nitrate in the anoxic zone through the relocation of the return line to the anoxic zone, in addition to the location of the nitrate recycle in that basin. Despite process differences to optimize nitrates present for denitrification, VIP

specific anoxic denitrification rates averaged 61 percent of the A^2/O rates. The VIP average was 0.0292 gN/gMLVSS-d and the A^2/O average was 0.0475 gN/gMLVSS-d. These differences were proportional to the differences in the amounts of total denitrification because the VIP rate was 36 and the A^2/O rate was 38 percent of the respective total specific denitrification rates. Average VIP nitrate loading to the anoxic zone was 61 percent of the average A^2/O loading due to the differences in the nitrate recycle rates (Table 5.3). The reason for the decreased anoxic denitrification rates found during VIP operation was the decrease in the nitrate recycle rate, as previously noted. The anoxic plus anaerobic denitrification rate was also found to vary with the anoxic zone nitrate loading (Table 4.8, Figure 4.9). The anoxic zone was overloaded with nitrate for each month under study, except for June and July, 1988 because the nitrate loading to the zone exceeded the anoxic denitrification for all but the two months noted.

Effects of Recycle on ANA+ANX Denitrification

No particular relationship ($r^2 = 0.31$, $P = 0.19$) was found between the sum of the anaerobic and anoxic denitrification rates and the total nitrate recycle (the sum of the return sludge and nitrate recycle rates) (Table 4.7). Since there is a relationship between the sum of the anaerobic and anoxic denitrification rates and the anoxic zone nitrate loading, it should follow that the rate at which nitrates are delivered to the anaerobic and anoxic zones would also be correlated to the denitrification rates in the zones. This statement is shown only by the VIP configuration data ($r^2 = 0.82$, $P = 0.28$) (Figure 4.11). In the VIP configuration, both the nitrate recycle and the return sludge recycle feed into the anoxic zone. In the A^2/O configuration, the return sludge recycle feeds into the anaerobic zone, while the nitrate recycle feeds into the anoxic zone. The VIP configuration maximizes the nitrate to the anoxic zone,

where conditions favoring denitrification exist. A similar relationship in which only the VIP data seem to support a relationship was seen in the regressions between the anaerobic and anoxic denitrification and the nitrate recycle (Figure 4.10).

Secondary Clarifier Denitrification

Due to shorter mean cell retention times of the VIP mode, denitrification continued closer to the end of the treatment train. Thirteen percent more aerobic denitrification, on average, took place in the A^2/O mode of operation evaluated as compared to the VIP mode. The average secondary clarifier denitrification was 23 percent higher in the VIP months than in the A^2/O mode of operation.

Secondary Clarifier Nitrate and Recycle Rates

Nitrate loading in the secondary clarifier was found to vary significantly and inversely with the nitrate recycle ($r^2 = 0.65$, $P = 0.03$) (Table 4.9, Figure 4.12). Relationships for both process modes were similar, but not significant (Figures 4.13, 4.14). Since the lines on Figure 4.15 traced by the VIP and the A^2/O data intersect at 330 lb/day, 4.3 MGD, the data may predict that for nitrate recycles greater than 4.3 MGD, the VIP configuration will produce less secondary clarifier nitrate than the A^2/O mode. For nitrate recycles less than 4.3 MGD, the A^2/O mode of operation may produce less secondary clarifier nitrate.

BOD/TKN Ratio

The average BOD to TKN ratio of 5.69 encountered in 1988 and 7.06 seen in 1987 classifies the York River STP as a potential combined sludge removal plant according to the U.S. EPA (1975) criteria for a ratio greater than 5. Since a more concentrated waste is necessary to support a combined system to maintain an adequate nitrifier population (as

opposed to a separate sludge system), the U.S. EPA found the lower range of the BOD/TKN ratios to be 5 for existing combined sludge plants. Due to the variation in flows during the 1988 months, the York River Demonstration project data did not illustrate increased nitrification rates with a lower BOD to TKN ratio (U.S. EPA, 1975).

Ekama *et al.*, (1988) developed a TKN/COD ratio to determine the likelihood of denitrifying all the nitrogen available for a particular wastewater. The ratio relates the nitrate generated to the denitrification potential. With the understanding that the ratio discussion applied to wastewaters similar in characteristics and mean cell residence times to South African wastewaters, if the influent TKN/COD is greater than 0.1, the denitrification potential will not allow for all the nitrogen to be denitrified and a modified Ludzack-Ettinger process configuration will provide the most efficient removal. For the York River Demonstration Project, in every month examined, the ratio was 0.08, except July, 1988 when it was 0.07.

Heterotrophic Substrate Utilization Rate

The van't Hoff relationship states that for every ten degree increase in temperature, the rate of a biological reaction will double (Benefield and Randall, 1980). Increased reaction rates were seen in the York River Demonstration Project heterotrophic substrate utilization rate data with increased seasonal temperatures. Aside from the period of November 1 through November 30, 1986, the cooler winter-spring study periods showed a substrate utilization rate of 0.36 to 0.48 days.$^{-1}$ The warmer summer/fall periods had a substrate utilization rate that ranged from 0.58 to 0.80 days^{-1}. The larger value of 0.87 days^{-1} for the substrate utilization rate for the month of November, 1986 was probably due to the small biomass under aeration. A small amount in the denominator of the expression will increase the proportion. The relationship q =

μ/Y (where q is the heterotrophic substrate utilization rate, μ is the heterotrophic growth rate, and Y is the heterotrophic yield coefficient) is seen in the York River Demonstration Project data. The heterotrophic substrate utilization rate and the heterotrophic growth rate show corresponding increases, while the heterotrophic yield coefficient decreases accordingly. The substrate utilization rate increased from 0.48 to 0.80 $days^{-1}$ from May, 1987 through the end of October, 1987. The heterotrophic observed yield data also show the same two-tiered trend for 1987, described below for the growth rate data. After the sludge age increase during May, 1987 (and corresponding decrease for the growth rate), the observed yield decreased to 0.22 lb VSS/lb BOD_5 from an average yield of 0.56 lb VSS/lb BOD_5 for the prior periods studied.

Growth Rate

The York River Demonstration Project data show that the growth of biomass is proportional to the amount of biomass. The data show the same relative increases and decreases for these two parameters. The amount of the proportion is constant and is equal to the specific growth rate, μ (Benefield and Randall, 1980).

The heterotrophic specific growth rate, μ, exhibited two tiers of values during the A/O and A^2/O modes of operation. During the time periods analyzed during late 1986 and early 1987, the specific growth rate was 0.33 $days^{-1}$ and decreased to 0.27 $days^{-1}$ in May, 1987. The growth rate then stabilized at a value of about 0.13 $days^{-1}$ to 0.14 $days^{-1}$ through the end of October, 1987. During May, 1987, the sludge age was increased from 3.7 until it reached 7.5 days in July 1987. The sludge age decreased slightly to 6.9 days in September and October, 1987. The inverse relationship between the specific growth rate and the sludge age is apparent from these data. The target sludge age was kept much more stable during the VIP phase of

operation and this is reflected in the consistent growth rates for this period.

An additional factor in the decrease of the growth rate over the 1987 study period is the increase in the aerobic sludge mass fraction. Beginning in the latter part of May, 1987, the anaerobic/anoxic/aerobic mass fraction changed from 33-0-67 to 25-0-75. This action increased the aerobic mass fraction. During this process change, the first part of the anaerobic section functioned as an anoxic section since the nitrate rich return aeration sludge recycled to the head of the anaerobic section (Randall, 1994). The effect of the process modification most probably was to have a sludge mass with the configuration of 25 (anoxic)-25 (anaerobic)-50 (aerobic). In August, 1987, through October, 1987, the mass fraction was changed to 25 (anaerobic)-25 (anoxic)-50 (aerobic). The aerobic mass fraction was therefore effectively the same size during the time period from May, 1987 through October, 1987. The latter mass fraction configuration was then maintained through the end of the study period in 1988. Since the growth rate relationship is expressed as the proportion of the increase in biomass divided by the amount of biomass in the system, an increase in the denominator, the biomass in the system, would decrease the value of the proportion. The growth rate began to decrease in May, 1987, when the aerobic mass fraction was first increased. It was stabilized at its lower 1987 level by July, 1987.

Sludge Age

The sludge ages calculated as the reciprocal of the specific growth rate (Table 4.11) yielded smaller results than those calculated by HRSD (Table 5.3). HRSD's sludge ages for the time periods examined ranged from 4.5 to 11.8 days for the A/O and A^2/O modes of operation and 5.1 to 5.5 days for VIP operation. The reciprocal of the growth rate yielded ranges of 3.0 to 7.5 days for the A/O and A^2/O modes of operation and 4.1 to 4.4 days for VIP operation. The

A Comparsion of Average Sludge Age Calculation Results
for Selected Time Periods

Table 5.3

York River Demonstration Project
Hampton Roads Sanitation District

Operational Mode	A/O Mode			No Anoxic Zone, Complete Nitrification	A^2/O Mode	VIP Mode		
Time Period	11/1-11/30/86	2/10-3/31/87	5/1-5/31/87	7/1-7/31/87	9/1-10/31/87	4/1-4/30/88	7/1-8/31/88	7/1-7/31/89
Sludge Age, from HRSD's nutrient process control data, days (Equation 3.4)	5.7	4.5	6.4	11.0	11.8	5.4	5.1	5.5
Sludge Age, calculated from biomass and waste sludge data in this paper, days (Eqn. 3.20)	3.0	3.0	3.7	7.5	6.9	4.3	4.2	4.1

sludge ages calculated with HRSD's process control calculation for the nutrient demonstration project (equation 3.4 and 3.4a) included the volume of the anaerobic basin multiplied by the total suspended solids result of the anaerobic basin in the numerator for VIP and included the volume of the anaerobic basin in the total basin volume in the A^2/O calculation. The sludge calculations for this thesis did not. It was assumed for this work that any biomass growth that occurred in the anaerobic section was negligible, since no oxygen was present. The presence of any nitrate in the anaerobic basin that could be used less efficiently for respiration by heterotrophic organisms for the purpose of growth was minimized by the VIP configuration. The A/O and A^2/O modes of operation had the return sludge recycle placement to the anaerobic basin, so some nitrate was present during this operation. The difference in the means of the HRSD and growth rate reciprocal for the A/O and A^2/O modes of operation was 3.1 days, while the corresponding difference for VIP operation was 1.1 day. Since this difference was greater for the A/O and A^2/O modes of operation, it indicates that more nitrate was available for biomass growth in the anaerobic zone during this time and that anaerobic biomass growth was likely to have occurred.

Heterotrophic Observed Yields

Heterotrophic observed yields were calculated for the York River Demonstration Project. The average of the monthly values found for the A/O and A^2/O modes of operation (excluding the high flow period of February 10 through March 31, 1987) was 0.34 lb VSS/lb BOD_5 and 0.49 lb VSS/lb BOD_5 for the VIP mode. The observed yield must be lower than the true yield because the observed yield reflects the negative effect of endogenous metabolism. However, at the low mean cell retention times seen at the York River Demonstration project, one would not

York River Demonstration Project
Hampton Roads Sanitation District

Table 5.4

Heterotrophic Observed Yields

	lb VSS/lb BOD_5	wt. cells/wt. energy substrate oxidized, Range
York River		
A/O and A^2/O	0.34	
VIP	0.49	
Lambert's Point VIP Pilot	0.55	
Keyes and Asano (1975)	0.7	
Sharma and Ahlert (1977)		0.37-0.79

expect that endogenous respiration would be significant. The endogenous decay coefficient, K_d, is typically seen to be 0.05 to 0.1 (Benefield and Randall, 1980). For BNR systems operating at the same sludge age as a conventional system, endogenous decay was found to be significantly less (McClintock *et al.*, 1993). Keyes and Asano (1975) graphed net growth rate vs. substrate (BOD) removal rates for three steady state periods with sludge ages ranging from 6 to 15 days. They calculated a heterotrophic true yield of 0.7 MLVSS/BOD_5 for the Bozeman, Montana WWTP. Sharma and Ahlert (1977) listed a range of heterotrophic cell yields (from activated sludge with glucose substrate) for a variety of sludge ages from the literature of 0.37 to 0.79 wt. cells/wt. energy substrate oxidized. The average heterotrophic yield of all process phases of the VIP Lambert's Point pilot plant was found to be 0.74 mg TSS/mg BOD_5 removed. If the VIP Lambert's Point 0.74 mg TSS/mg BOD_5 removed figure is multiplied by the mean 0.74 g MLVSS/g MLTSS determined over the entire study and g MLTSS lb VSS/g MLVSS lb BOD_5 to compare to the York River Demonstration project data, a value of 0.55 lb VSS/lb BOD_5 is obtained. This value compares favorably with the mean 0.49 lb VSS/lb BOD_5 yield determined for the VIP mode of the York River Demonstration Project.

The low yields found overall at the York River Demonstration Project compared to the literature values indicate that sludge concentrations would be expected to be low. The yield concept is expressed as the increase in biomass (growth) per consumption of BOD. If the yield is low and the growth of biomass term in the numerator is large, then it follows that the denominator, the consumption of BOD, must also be large. BOD removal was excellent for the months under study; the monthly average BOD was 4 mg/L for the 1987 months and 7 mg/L for the 1988 months. High influent BODs were experienced during 1987. The strength of the influent BOD dropped approximately 15 per cent for the 1988 months under consideration, as

compared to the 1987 months.

Sludge Production

An increase in sludge production was noted under the VIP mode of operation in 1988 as compared to the A^2/O mode in 1987. The monthly average amount of sludge wasted in pounds (TSS and VSS) approximately doubled from 1987 to 1988. The monthly average amount of biomass was 17 percent less in the A/O and A^2/O modes of operation compared to 1988 and 1989. The average sludge age in the VIP mode (4.2 days) was not quite half that of the 1987 months (7.2 days). The net viable sludge production, being "younger" and thus experiencing less endogenous metabolism, would tend to be greater at lower sludge ages. According to Sherrard (1984), a high rate process which operates at a low sludge age or high food : microorganism ratio produces maximum sludge. There exists an inverse relationship between the observed yield and the sludge age and this is reflected in lower yields seen with the A/O and A^2/O modes of operation compared to the 1988 and 1989 VIP results. Higher sludge production is seen during the A^2/O mode of operation during the period of high influent flows, during February and March, 1987.

Waltrip (1990) documented that the presence of the denitrification (anoxic) basins at the York River Demonstration Project resulted in less sludge produced than expected due to the anoxic stabilization of BOD. This phenomena was discussed by McClintock *et al.* (1988). Because anoxic stabilization is less energy efficient than aerobic respiration, less sludge is produced. Also, the less highly oxidized the substrate during denitrification, the less sludge is produced (Gaudy and Gaudy, 1988). This is likely seen in the low yields of the York River project as well since the waste was well fermented by the time it reached the treatment plant. If

project as well since the waste was well fermented by the time it reached the treatment plant. If aeration follows the anaerobic basin, greater endogenous respiration is produced than just what may be attributed to the anaerobic reaction alone (Gaudy and Gaudy, 1988). The changing conditions may cause cell lysis; the organic cellular material is rapidly oxidized so that it does not appear in the effluent (Gaudy and Gaudy, 1988). Since mixed liquor aeration follows the anaerobic basin at the York River Demonstration project, this phenomena may have been partially responsible for less sludge production than expected, though McClintock *et al.* (1993) found endogenous decay to be a less significant phenomena at a BNR plant.

Summary

Nitrification rates observed during the York River Demonstration Project compare with published literature values for laboratory scale and pilot plant determinations. Mass balance equations were based upon both ammonia disappearance and nitrate appearance. Both equations gave the same results, indicating no unaccounted for losses due to denitrification. The system nitrification ranged from 473442 to 705247 gN/day over both modes of operation. The mean VIP mode nitrification was 8 percent less than that of the A^2/O mean nitrification, but probably was the result of D.O. limitation and decreased ammonia availability due to increased sludge wasting during VIP. The aerobic basin nitrification rates ranged from 0.1153 to 0.1192 gN/gMLVSS-d for the A^2/O configuration and 0.1116 to 0.1230 gN/gMLVSS-d for the VIP mode. During three of the cooler of the eight months examined, nitrification took place outside of the aeration basin. During September, 1987, October, 1987 and April, 1988, 4, 0.1 and 8 percent, respectively, of the nitrification occurred in the secondary clarifier. Denitrification rates observed during the York River Demonstration Project compare to published literature values

for laboratory scale and pilot plant determinations and varied with the amount of nitrate loading to the anoxic zone and the rate of total nitrate recycle. The relationships were linear and significant. The range for the A^2/O mode of operation was 460051 to 571772 gN/day, or 0.1095 to 0.1520 gN/gMLVSS-d, with a mean of 0.2249 gN/gMLVSS-d. The range for the VIP data was 335630 to 540882 gN/day or 0.0624 to 0.0971 gN/gMLVSS-d, with a mean of 0.0815 gN/gMLVSS-d. Mass balance equation 3.10 was the superior method of calculation of the total denitrification due to greater recoveries.

Most of the system denitrification took place in the aerobic basin. Fifty to 75 percent of the system denitrification took place in the aerobic basin under both process modes and this proportion was not different for the process modes. Thirteen percent more aerobic denitrification took place during the A^2/O study period than during the VIP study period. Most of the increased denitrification seen during the A^2/O mode was due to the greater nitrification experienced during that mode. The mean A^2/O aerobic denitrification was 331961 gN/day or 0.0669 gN/gMLVSS-d. The mean VIP aerobic denitrification was 289283 gN/day or 0.0584 gN/gMLVSS-d. Even though the presence of oxygen is inhibiting to denitrifying bacteria, they are facultative and sufficiently low dissolved oxygen levels apparently existed at the center of the microbial flocs for denitrification to occur.

Mean anaerobic basin denitrification, (4.1% of the total denitrification, 23997 gN/day or 0.0099 gN/gMLVSS-d for the A^2/O process mode, 2.2% of the total denitrification, 9060 gN/day or 0.0035 gN/gMLVSS-d for the VIP data), indicated that the anaerobic denitrification was three times as high under the A^2/O mode of operation than during the VIP process mode. However, during August, 1987, the anaerobic and anoxic basin system was still achieving steady state due to the establishment of the anoxic zone during this month. If August, 1987 is omitted

from the A^2/O data, the mean A^2/O mode anaerobic denitrification drops to 3166 gN/day (or, 0.65% of the total denitrification). Since the A^2/O configuration returned the return sludge to the anaerobic basin, providing nitrates to that zone, perhaps the stabilization of BOD is decreasing the nitrates available in the zone for denitrification. Return sludge was returned to the anoxic basin under the VIP mode. A total range of 0.5 percent to 11 percent of system denitrification for both process modes took place in the anaerobic basin.

Average VIP nitrate loading to the anoxic zone was 61 percent of the average A^2/O loading due to the difference in the nitrate recycle rates. Anoxic basin denitrification was 13 percent to 25 percent of the system denitrification under both process modes. Anoxic denitrification data was 22 percent of the mean percentage of system denitrification for the A^2/O mode; 17 percent was the comparable value for the VIP mode. The range of values over both process modes was 50429 to 144975 gN/day, or 0.0214 to 0.0601 gMLVSS-d.

The anoxic and anaerobic denitrification was a linear function of the amount of the mass of nitrate recycled to the anoxic zone. The anoxic zone was overloaded with nitrate for each month examined because anoxic zone nitrate loading exceeded the anoxic denitrification. Nitrate loading to the anoxic zone was greater under A^2/O operation than for the VIP mode. This difference was due to the rate of nitrate recycle. Only for the VIP mode of operation was a linear relationship able to be illustrated between the anaerobic and anoxic denitrification and the nitrate recycle and the anaerobic and anoxic denitrification and the total nitrate recycle. Under the A^2/O mode of operation, the months examined showed only two values for the nitrate recycle (0 and 7.26 MGD). These were not sufficient to establish a relationship between the anoxic denitrification and the NRCY. Additionally, the return sludge recycle feeds into the anaerobic zone in the A^2/O configuration where less denitrification is taking place. The VIP

mode of operation maximized the nitrate to the zone where the most denitrification was taking place.

Some of the denitrification that took place in the aerobic basin during the A^2/O study period occurred further along in the treatment train, i.e., in the secondary clarifier, during the VIP months. Twenty three percent more denitrification took place in the secondary clarifier during the VIP mode of operation than during the A^2/O mode. This is thought to be due to the decreased VIP sludge ages. The means for the A^2/O secondary clarifier denitrification was 42528 gN/day or 0.0050 gN/gMLVSS-d. The means for the VIP secondary clarifier denitrification was 55079 gN/day or 0.0065 gN/gMLVSS-d.

Nitrate loading in the secondary clarifier was found to vary inversely and significantly with the nitrate recycle. Both of the process modes showed a similar relationship. The data predict that for nitrate recycles greater than 4.3 MGD, the VIP configuration will produce less secondary clarifier nitrate than the A^2/O mode. For nitrate recycles less than 4.3 MGD, the A^2/O mode will produce less secondary clarifier nitrate than the VIP mode.

Less sludge was produced than expected at the York River Demonstration project. The reason for this is thought to be anoxic stabilization of BOD by denitrification (Waltrip, 1990) and the lack of oxidation of the organic carbon source (fermented wastewater) provided for the denitrification reaction. A highly oxidized organic carbon source produces more sludge than one that is less oxidized (Gaudy and Gaudy, 1988). Seventeen percent less biomass growth was encountered during the A^2/O study period than the VIP months because of the longer sludge ages used during A^2/O operation.

CHAPTER 6
CONCLUSIONS

1. Nitrogen mass balance equations based upon ammonia removal and nitrate production, respectively, gave the same results for the York River BNR Plant data. This indicated that there were no unaccounted for losses due to denitrification.

2. The York River nitrification rates compare with published literature values for laboratory scale and pilot plant determinations. The aerobic basin nitrification rates ranged from 0.1153 to 0.1192 gN/gMLVSS-d for the A^2/O configuration and 0.1116 to 0.1230 gN/gMLVSS-d for the VIP mode.

3. The mean VIP mode nitrification was 8 percent less than the mean A^2/O nitrification, but was probably the result of D. O. limitation and decreased ammonia availability due to increased sludge wasting during VIP operation.

4. During three of the cooler of the eight months examined, nitrification continued in the secondary clarifiers. During September, 1987, October, 1987 and April, 1988, 4, 0.1 and 8 percent, respectively, of the nitrification occurred in the secondary clarifier.

5. The York River Demonstration Project denitrification rates observed compare with published literature values. The range for the A^2/O mode of operation was 0.1095 to 0.1520 gN/gMLVSS-d, with a mean of 0.2249 gN/gMLVSS-d. The range for the VIP data was 0.0624 to 0.0971 gN/gMLVSS-d, with a mean of 0.0815 gN/gMLVSS-d.

6. There was a significant, linear relationship between total denitrification and both the total nitrate recycle rate (RAS + NRCY) and the nitrate loading to the anoxic zone.

7. The total system mass balance equation for denitrification gave superior results than the equation based on summing calculated denitrification in the three zones and the secondary clarifier. This implies that significant denitrification occurred in the activated sludge collection well and recycle lines.

8. Fifty to 75 percent of the system denitrification took place in the aerobic basin under both process modes and there was no significant difference in distribution between the two modes. However, 13 percent more aerobic denitrification took place during the A^2/O study period than during the VIP study period. This was because of the greater nitrification during A^2/O operation.

9. Anaerobic zone denitrification in the A^2/O mode was significant only when there was no nitrate recycle to the anoxic zone or before the anoxic zone was well established.

10. The total anoxic and anaerobic denitrification was a linear function of the mass of nitrate recycled to the anoxic zone.

11. Average VIP nitrate loading to the anoxic zone was 61 percent of the average A^2/O loading due to the difference in the nitrate recycle rates. Anoxic basin denitrification was 13 percent to 25 percent of the system denitrification under both process modes. Anoxic denitrification data was 22 percent of the mean percentage of system denitrification for the A^2/O mode; 17 percent was the comparable value for the VIP mode.

12. Some of the denitrification that took place in the aerobic basin during the A^2/O study period occurred further along in the treatment train, i.e., in the secondary clarifier, during the VIP months. Twenty three percent more denitrification took place in the secondary clarifier during the VIP mode of operation than during the A^2/O mode.

13. Nitrate loading in the secondary clarifier was found to vary inversely and significantly with the nitrate recycle. Both of the process modes showed a similar relationship.

14. The differences in sludge production between the VIP and A^2/O process could be explained by the differences in mean cell residence time.

CHAPTER 7
THESIS SUMMARY

Nitrification and denitrification rates for two modes of operation were examined for the York River Biological Nutrient Removal Demonstration Project of the Hampton Roads Sanitation District. The A^2/O mode (1987) and the VIP mode (1988) were each examined for four selected months of steady state operation. The overall nitrification rate was calculated according to two mass balances for the ammonia converted to nitrate-nitrite and the nitrate being formed. These two equations were found to be equal. For the cooler months where nitrification extended into the secondary clarifier, nitrification was broken down into aeration basin and secondary clarifier nitrification.

System denitrification and nitrification were each determined by two different equations. The two nitrification equations produced equal results indicating that no losses from denitrification occurred. System denitrification was broken down according to the basin in which the denitrification occurred. System denitrification was calculated according to a mass balance equation and a component equation in which the individual basin sums are added up. The mass balance equation was superior. Nitrification and denitrification rates were found to be comparable to literature values for laboratory and pilot scale operations. The mean VIP nitrification result (574448 gN/day) was found to be 8 percent less than the mean A^2/O mode value (629243) gN/day. Denitrification rates were found to be 35 percent lower for VIP compared to A^2/O. The mean total system denitrification for the A^2/O was 0.1095 gN/gMLVSS-d; the mean for the VIP data was 0.0815gN/gMLVSS-d. The total system denitrification was found to vary linearly and significantly with the amount of nitrate to the anoxic zone and the total nitrate recycle.

The aerobic basin is where the bulk of the system denitrification took place. Facultative bacteria may denitrify in the center of microbial flocs where low levels of dissolved oxygen occur. Aerobic denitrification was calculated with two different equations. The component equation provided a better recovery. In the component equation, the remaining individual basin denitrification totals are subtracted from the total system nitrification which was multiplied by a proportion consisting of the monthly average system denitrification divided by the monthly average system nitrification. Thirteen percent more aerobic denitrification took place during the A^2/O study period than during the VIP study period.

The mean A^2/O anaerobic basin rate (0.0099 gN/gMLVSS-d) was three times as high as the mean rate determined from the VIP mode (0.0035 gN/gMLVSS-d). The anaerobic denitrification was 0.4 to 11 percent of the total system denitrification. The anoxic basin denitrified 22 percent of the total system denitrification for the A^2/O mode of operation (0.0475 gN/gMLVSS-d) and 17 percent of the VIP denitrification (0.0292 gN/gMLVSS-d). The average VIP nitrate loading to the anoxic zone was 61 percent of the average A^2/O loading due to a decreased nitrate recycle rate.

The sum of the anoxic and anaerobic denitrification was a linear function of the mass of nitrate recycled to the anoxic zone. The decreased nitrate loading rate during VIP operation caused decreased anaerobic and anoxic denitrification. The nitrate and total nitrate recycle rates were only linearly related to the anoxic and anaerobic denitrification for the VIP mode of operation due to the recycle configurations which maximized nitrates to the anoxic zone.

Shorter detention times during the VIP process extended some of the aerobic denitrification into the secondary clarifier. Twenty three percent more denitrification took place

in the secondary clarifier during the VIP mode of operation as compared to the A^2/O mode. The mean A^2/O secondary clarifier denitrification rate was 0.0050 gN/gMLVSS-d. The mean VIP secondary clarifier denitrification rate was 0.0065 gN/gMLVSS-d.

The nitrate recycle varied inversely and significantly with the nitrate loading in the secondary clarifier for both process modes similarly. The data may predict that for nitrate recycles greater than 4.3 MGD, the VIP configuration will produce less secondary clarifier nitrate than the A^2/O mode. The A^2/O mode may produce less secondary clarifier nitrate for nitrate recycles less than 4.3 MGD.

Heterotrophic sludge characteristics were calculated and examined for process mode differences. The average sludge ages for the A^2/O and A/O modes study periods was 4.8 days; that for the VIP mode was 4.2 days. Sixteen percent less biomass was seen for the A^2/O and A/O periods than the VIP process mode. Twenty one percent less waste sludge was seen for the A^2/O and A/O periods than the VIP process mode. The 1988 VIP sludge was younger and undergoing less endogenous utilization of the biomass since the viable fraction of the sludge was larger. More is wasted. Additional explanation is provided through the anoxic stabilization of BOD in the anoxic basin (Waltrip, 1990). For a small increase in yield seen from 1987 to 1988, (which may be explained through the increase in the viable fraction of the sludge), the BOD efficiency term in the denominator must be increasing also. The less oxidized the organic carbon source (the fermented wastewater) provided for denitrification, the less sludge will be produced (Gaudy and Gaudy, 1988). The York River treatment plant's wastewater is well fermented. This would tend to affect both process modes.

CHAPTER 8
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Appendix I Explanation of Terms	
Symbol	Explanation
HRSD	Hampton Roads Sanitation District
EPA	United States Environmental Protection Agency
VPDES	Virginia Pollutant Discharge Elimination System
BOD_5, or S	5-Day Biochemical Oxygen Demand or Substrate
TSS, TVSS	Total Suspended Solids, Total Volatile Suspended Solids
TKN	Total Kjeldahl Nitrogen
NH_3	Ammonia
NO_x	Nitrate/Nitrite
NO_3	Nitrate
A^2/O	Air Products proprietary treatment mode: Anaerobic, Anoxic, Oxic
A/O	Air Products proprietary treatment mode: Anaerobic, Oxic
VIP	HRSD's patented treatment mode, named after the Virginia Initiative Plant where it was first studied
Q	Flow
RWI	Raw Influent, used with or as a subscript, refers to that location of a flow or sample
FNE	Final Effluent, used with or as a subscript, refers to that location of a flow or sample
PCE	Primary Clarifier Effluent, used with or as a subscript, refers to that location of a flow or sample
ANA	Anaerobic Basin, Cells 1-3, used with or as a subscript, refers to that location of a flow or sample
ANX	Anoxic Basin, Cells 4-5, used with or as a subscript, refers to that location of a flow or sample
ARE	Aeration Effluent, the Effluent from Cell 7, used with or as a subscript, refers to that location of a flow or sample

Appendix I Explanation of Terms, cont.	
Symbol	Explanation
SCE	Secondary Clarifier Effluent, used with or as a subscript, refers to that location of a flow or sample
WS or W	Waste Sludge, used with or as a subscript, refers to that location of a flow or sample
RASVSS	Return Sludge Volatile Suspended Solids
WSVSS	Waste Sludge Volatile Suspended Solids
RAS or R	Return Aeration Sludge, used with or as a subscript, refers to that location of a flow or sample
ARCY	Denitrified Recycle (VIP Mode), used with or as a subscript, refers to that location of a flow or sample
NRCY	Nitrified Recycle, used with or as a subscript, refers to that location of a flow or sample
X, X_{PCE}, X_e, X_w	Biomass (X): Biomass present in the PCE, the Effluent, the Waste Sludge, respectively
N, N_{PCE}, N_e, N_w	Nitrogen (N): Nitrogen present in the PCE, the Effluent, the Waste Sludge, respectively
V_a or V_{AER}	Volume of the Aeration Basin
$MLVSS_a$	Mixed Liquor Concentration of the Aeration Basin
$(dX/dt)_g$	Growth of Biomass in the System (equal to the sludge wasted each day)
$(dNO_3\text{-}N/dt)_{denit}$	The loss of nitrate in the system due to denitrification
$(dNH_3\text{-}N/dt)_{net\ growth}$	The uptake of nitrogen due to assimilation by heterotrophs and nitrifiers
g, mg, kg	Grams, Milligrams, Kilograms
l	Liters
g, MG	Gallons, Million Gallons
MGD	Million Gallons per Day
θ_c	Sludge Age, Biological Solids Retention Time BSRT, Mean Cell Residence Time MCRT

Appendix II

The calculated results for the nitrification rates calculated in terms of nitrate appearance in gN/day (equation 3.7) were determined to be the same as those calculated in terms of ammonia disappearance (equation 3.5). Equations 3.5 and 3.7 were found to be equal as shown below:

The nitrification expressions from equations 3.5 and 3.7 were set to be equal.

$$\left((PCE_{TKN})(Q_{PCE})\left(3.785\frac{gL1x10^6gal}{1000mg\ MG}\right)\right) - \left((SCE_{TKN})(Q_{PCE}-Q_w)\left(3.785\frac{g\ L\ 1x10^6gal}{1000mg\ MG}\right)\right)$$
$$- \left(\text{Uptake of N by biomass}\right) =$$
$$\left(SCE_{NO_x}\times(Q_{PCE}-Q_w)\times\left(3.785\frac{gL1x10^6gal}{1000mgMG}\right)\right) - \left(PCE_{NO_x}\times Q_{PCE}\times\left(3.785\frac{g\ L\ 1x10^6gal}{mg\ MG}\right)\right)$$
$$+ \left((PCE_{NO_x} + PCE_{TKN},\ mg/L)(\ Q_{PCE}\ )\left(3.785\frac{g\ L\ 1x10^6gal}{10\ mg\ MG}\right)\right) -$$
$$\left((SCE_{NO_x} + SCE_{TKN},\ mg/L)(Q_{PCE} - Q_w)\left(3.785\frac{g\ L\ 1x^6gal}{10\ mg\ MG}\right)\right) -$$
$$\left(\text{Uptake of N by biomass}\right).$$

The biomass uptake of N was removed from both sides of the equality. The conversion factor $\left(3.785\frac{g\ L\ 1x10^6gal}{1000\ mg\ MG}\right)$ may be factored from both sides, leaving the equality:

$$\left(3.785\frac{g\ L\ 1x1^6gal}{mg\ MG}\right)\left((PCE_{TKN})(Q_{PCE})\right) - \left((SCE_{TKN})(Q_{PCE}-Q_w)\right) = \left(3.785\frac{g\ L\ 1x10^6gal}{1000\ mg\ MG}\right)$$
$$\left(SCE_{NO_x}\times(Q_{PCE}-Q_w)\right) - \left(PCE_{NO_x}\times Q_{PCE}\right) + \left((PCE_{NO_x} + PCE_{TKN},\ mg/L)(\ Q_{PCE})\right)$$
$$-\left((SCE_{NO_x} + SCE_{TKN},\ mg/L)(Q_{PCE} - Q_w)\right).$$

The conversion factor was then eliminated from both sides. The terms were then distributed through multiplication:

$$\left((PCE_{TKN})(Q_{PCE})\right) - \left((SCE_{TKN})(Q_{PCE})\right) + \left((SCE_{TKN})\,(Q_w)\right) = \left(SCE_{NO_x}\,(Q_{PCE})\right)$$

$$-\left((SCE_{NO_x})(Q_w)\right) - \left((PCE_{NO_x})(Q_{PCE})\right) + \left((PCE_{NO_x})(Q_{PCE})\right) + \left((PCE_{TKN})(Q_{PCE})\right) -$$

$$\left((SCE_{TKN})(Q_{PCE})\right) + \left((SCE_{TKN})(Q_w)\right) - \left((SCE_{NO_x})\,(Q_{PCE})\right) + \left((SCE_{NO_x})(Q_w)\right).$$

Once the terms on the left side that cancel were eliminated, the same quantities on either side of the equals sign remain, indicating that the two expressions have been shown to be equal:

$$\left((PCE_{TKN})(Q_{PCE})\right) - \left((SCE_{TKN})(Q_{PCE})\right) + \left((SCE_{TKN})(Q_w)\right) = \left((PCE_{TKN})(Q_{PCE})\right) -$$

$$\left((SCE_{TKN})(\,Q_{PCE}\,)\right) + \left((SCE_{TKN})(\,Q_w)\right).$$

Appendix III

Two different aerobic denitrification equations (equation 3.14 and 3.15a) were substituted into the component equation, along with the appropriate equations for the other basins.

When aerobic denitrification equation 3.15a is substituted into equation 3.11, the following is seen:

$$\text{System Denitrification} = \text{ANA}_{denit} + \text{ANX}_{denit} + SC_{denit} +$$

$$\left(N_{nitrified} \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n} \right.$$

$$\left. -\text{ANA}_{denit} - \text{ANX}_{denit} - SC_{denit} \right).$$

When the ANA, ANX and SC denit terms on both sides of the equals sign are cancelled,

$$\text{System Denitrification} = \text{N}_{\text{nitrified}} \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n}.$$

When the algebraic manipulation between the $\text{N}_{\text{nitrified}}$ and the proportion of the system denitrification to the system nitrification is examined when the means of the daily values are computed:

$$\sum\Big(N_{nitrified} \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n} \Big)/n =$$

$$\Big(\sum N_{nitrified}/n \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n} \Big) .$$

When the above expression is substituted into the system denitrification equation left after cancelling the ANA, ANX and SC denit terms (below), we see that $\sum N_{nitrified}/n$ and $\frac{\sum \text{Total System Nitrification Daily Values}}{n}\Big)$ are equivalent. When these terms cancel, in the overall equation, system denitrification is left to equal system denitrification. Since these equations are equal due to correlation, only the independent system denitrification equation 3.10 and the component equation 3.11 and aerobic equation 3.14 were expressed in Table 4.3.

$$\text{System Denitrification} = \sum N_{nitrified}/n \times \frac{\sum \text{Total System Denitrification Daily Values}}{n} \Big/ \frac{\sum \text{Total System Nitrification Daily Values}}{n} \Big) .$$

$$\text{System Denitrification} = \frac{\sum \text{Total System Denitrification Daily Values}}{n} .$$

Appendix IV

Table A.IV.1.

November 1986 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qrwi	Qcpr	Qpce	Qe
1	141	5	3750	0.67	2512.5	1600	3	0.066	6.24	1.44	7.68	7.614
2	114	6	4300	0.64	2752	1800	6	0.073	6.09	0.8	6.89	6.817
3	122	5	4000	0.63	2520	1550	6	0.028	5.93	1.48	7.41	7.382
4	113	6	4200	0.6	2520	1750	12	0.030	6.06	1.53	7.59	7.56
5	109	4	4550	0.57	2593.5	1900	10	0.066	5.99	1.53	7.52	7.454
6	114	4	4550	0.63	2866.5	1700	10	0.090	5.94	1	6.94	6.85
7	99	8	4400	0.61	2684	1850	9	0.102	4.39	0.94	5.33	5.228
8	114	9	3550	0.67	2378.5	2250	6	0.096	6.17	0.82	6.99	6.894
9	107	6	4350	0.69	3001.5	1750	3	0.106	6.6	0.9	7.5	7.394
10	127	9	4750	0.69	3277.5	1950	6	0.109	6.25	1.24	7.49	7.381
11	113	8	4050	0.66	2673	1850	10	0.099	6.83	0.08	6.91	6.811
12	143	8	4550	0.69	3139.5	1750	2	0.106	6.21	0.09	6.3	6.194
13	107	10	4300	0.73	3139	1800	7	0.075	5.89	1.17	7.06	6.985
14	121	9	3950	0.71	2804.5	2100	9	0.115	5.58	0.87	6.45	6.335
15	99	10	3800	0.67	2546	1650		0.120	6.2	0.77	6.97	6.85
16	140	11	4150	0.66	2739	1950	8	0.127	6.44	0.77	7.21	7.083
17	112	11	4350	0.72	3132	2200	5	0.101	6.26	1.21	7.47	7.369
18	117	12	3900	0.66	2574	1950	5	0.125	6.01	1.65	7.66	7.535
19	106	8	4000	0.57	2280	1800	9	0.119	6.01	1.29	7.3	7.181
20	98	11	3250	0.7	2275	1450	9	0.055	5.96	1.58	7.54	7.485
21	88	9	4550	0.65	2957.5	1750	12	0.000	6.23	1.6	7.83	7.83
22	102	11	5900	0.67	3953	3850	4	0.000	6.36	1.66	8.02	8.02
23	129	9	6750	0.68	4590	3750	4	0.000	6.21	1.09	7.3	7.3
24	139	31	6600	0.61	4026	3650	12	0.054	6.24	1.17	7.41	7.356
25	152	7	6000	0.66	3960	2650	5	0.115	6.12	1	7.12	7.005
26	143	11	5650	0.68	3842	2550	1	0.123	5.96	1.63	7.59	7.467
27	149	11	5300	0.66	3498	2350	3	0.142	6.36	1.03	7.39	7.248
28	167	14	5200	0.8	4160	200	3	0.148	5.77	1.01	6.78	6.632
29	152	41	5300	0.68	3604	2700	8	0.142	5.93	0.98	6.91	6.768
30	123	13	5650	0.68	3842	2100	3	0.147	6.32	0.98	7.3	7.153

Table A.IV.2

February 10-28 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE %Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qrwi	Qcpr	Qpce	Qe
2/10/87	127	23	9150	0.7	6405	4050	21	0.169	9.36	2.39	11.75	11.581
11		7		0.7		3800		0.124	9.09			
12	107	11	8200	0.71	5822	4000	11	0.129	9.14	1.06	10.2	10.071
13	113	7	7750	0.73	5657.5	3400	4	0.119	8.97	1.16	10.13	10.011
14	103	12	7650	0.74	5661	3000	6	0.132	9.09	1.28	10.37	10.238
15	127	7						0.131	8.94	1.31	10.25	10.119
16			7500	0.76	5700	3100	13	0.135	8.82	2.04	10.86	10.725
17	117	6	7300	0.74	5402	3750	10	0.131	9.42	1.3	10.72	10.589
18	110	8	7450	0.75	5587.5	3400	7	0.13	9.78	1.54	11.32	11.19
19	130	7	7250	0.74	5365	3350	8	0.132	9.8	1.74	11.54	11.408
20	120	10	7400	0.74	5476	3350	5	0.104	9.86	1.73	11.59	11.486
21	110	13	7500	0.76	5700	3200	3	0.105	10.61	0.81	11.42	11.315
22	120	8	8350	0.73	6095.5	3450	7	0.123	10.78	1.54	12.32	12.197
23	90	11	6350	0.74	4699	3000	14	0.125	14.97	1.66	16.63	16.505
24	47	5	7550	0.72	5436	3300	5	0.12	12.5	1.88	14.38	14.26
25	83	7	7850	0.73	5730.5	3250	6	0.127	11.78	1.3	13.08	12.953
26	70	8	7200	0.72	5184	3400	4	0.121	11.15	1.11	12.26	12.139
27	127	6	7100	0.71	5041	3250	4	0.107	10.79	0.94	11.73	11.623
28	80	4	6800	0.71	4828	3100	4	0.104	10.98	0.92	11.9	11.796

Table A.IV.3.

March 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE% Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qrwi	Qcpr	Qpce	Qe
1	83	8	6700	0.73	4891	2900	5	0.112	13.82	1	14.82	14.708
2	97	5	7100	0.73	5183	3050	2	0.117	12.75	0.94	13.69	13.573
3	63	4	7050	0.74	5217	3050	3	0.126	11.55	0.87	12.42	12.294
4	60	4	7050	0.72	5076	3350	1	0.124	11.05	1.01	12.06	11.936
5	53	6	7050	0.72	5076	3350	5	0.12	10.65	0.96	11.61	11.49
6	80	5	6800	0.71	4828	2950	1	0.119	10.42	1.03	11.45	11.331
7	73	6	5750	0.73	4197.5	2550	8	0.131	10.48	0.81	11.29	11.159
8	87	5	6200	0.72	4464	2800	7	0.118	10.32	0.78	11.1	10.98
9	93	5	6500	0.73	4745	2950	4	0.127	11.03	0.98	12.01	11.883
10	83	8	4750	0.76	3610	2700	7	0.119	17.35	0.88	18.23	18.111
11	60	8	6050	0.73	4416.5	2600	7	0.111	14.36	0.74	15.1	14.989
12	57	6	6450	0.73	4708.5	3000	7	0.116	12.88	0.76	13.64	13.524
13			7550	0.72	5436	2950	1	0.119	13.56	0.73	14.29	14.171
14	70	9	5650	0.74	4181	2400	8	0.187	14.81	0.72	15.53	15.343
15			5550	0.73	4051.5	2600		0.118	14.27	0.66	14.93	14.812
16			7350	0.73	5365.5	4350	9	0.118	12.82	1.49	14.31	14.192
17			7000	0.75	5250	3250	6	0.117	11.63	1.41	13.04	12.923
18	50	9	6600	0.79	5214	2400	6	0.113	11.25	0.86	12.11	11.997
19	47	5	7750	0.76	5890	3700	7	0.134	11.08	0.84	11.92	11.786
20	70	7	6150	0.73	4489.5	2700	4	0.124	10.79	1.23	12.02	11.896
21	83	12	6400	0.82	5248	2350	5	0.123	10.9	1.1	12	11.877
22	83	8	6200	0.73	4526	2350	3	0.114	10.54	0.89	11.43	11.316
23	93	6	6656	0.74	4925.4	2800	9	0.13	10.25	0.88	11.13	11
24	63	7	6300	0.76	4788	3300	5	0.127	7.13	0.85	7.98	7.853
25	73	17	5800	0.76	4408	2350	9	0.073	6.21	1.37	7.58	7.507
26			7100	0.76	5396	2650	3	0.088	9.85	0.87	10.72	10.632
27	70	6	7100	0.76	5396	2850	5	0.117	9.77	1.57	11.34	11.223
28	93	9	5750	0.75	4312.5	2400	4	0.099	10.3	1.04	11.34	11.241
29	90	11	6250	0.75	4687.5	2650	7	0.117	10.06	1.04	11.1	10.983
30	97	11	6100	0.75	4575	3000	8	0.112	10.11	1.04	11.15	11.038

Table A.IV.4

May 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qrwi	Qcpr	Qpce	Qe
1	93	9	4800	0.71	3408	2250	5	0.129	8.42	1.9	10.32	10.191
2	73	8	4050	0.73	2956.5	2050	4	0.132	8.39	1.35	9.74	9.608
3	87	10	4750	0.75	3562.5	2050	4	0.137	8.27	1.44	9.71	9.573
4	90	9	5650	0.69	3898.5	2700	6	0.126	8.64	1.11	9.75	9.624
5	90	9	4400	0.64	2816	2050	5	0.118	8.74	1.01	9.75	9.632
6	80	8	4450	0.72	3204	1900	6	0.102	8.34	1.01	9.35	9.248
7	80	8	3800	0.68	2584	1950	7	0.089	8.14	0.72	8.86	8.771
8	87	10	4900	0.7	3430	1750	4	0.091	7.84	0.99	8.83	8.739
9	103	9	4300	0.71	3053	2050	3	0.102	7.88	0.94	8.82	8.718
10	123	10	4950	0.7	3465	2200	1	0.133	7.67	0.81	8.48	8.347
11	117	10	7550	0.7	5285	3850	10	0.097	7.89	0.99	8.88	8.783
12	113	9	7350	0.7	5145	3250	5	0.097	7.62	0.98	8.6	8.503
13	80	10	6050	0.66	3993	3000	9	0.098	7.41	1.02	8.43	8.332
14	87	6	5100	0.7	3570	2150	5	0.102	7.32	1.04	8.36	8.258
15	87	9	4750	0.69	3277.5	2100	5	0.086	7.37	0.88	8.25	8.164
16	107	19	5150	0.69	3553.5	2250	7	0.087	7.46	0.98	8.44	8.353
17	90	7	5250	0.65	3412.5	2850	1	0.088	7.29	0.35	7.64	7.552
18	120	10	5450	0.69	3760.5	2600	1	0.097	7.49	1.12	8.61	8.513
19	90	10	5100	0.67	3417	2300	11	0.107	7.4	2.24	9.64	9.533
20	107	11	4650	0.69	3208.5	2350	7	0.090	6.87	1.69	8.56	8.47
21	100	8	4850	0.67	3249.5	2450	7	0.094	6.36	1.71	8.07	7.976
22	100	12	4350	0.69	3001.5	2050	3	0.078	6.14	1.85	7.99	7.912
23	93	13	4500	0.69	3105	2550	5	0.062	5.96	1.41	7.37	7.308
24	93	25	4950	0.66	3267	3000	6	0.095	5.61	1.58	7.19	7.095
25	83	17		0.61		4200	9	0.101	7.93	2.17	10.1	9.999
26	73	16	6150	0.6	3690	2900	9	0.093	7.21	1.86	9.07	8.977
27	77	9	6500	0.63	4095	2700	6	0.105	6.86	1.62	8.48	8.375
28	53	4	5950	0.66	3927	2800	2	0.088	6.72	1.1	7.82	7.732
29	93	9	5400	0.64	3456	2200	3	0.105	6.61	1.25	7.86	7.755
30	77	11	5000	0.67	3350	2250	4	0.105	6.56	0.75	7.31	7.205
31	90	7	4150	0.64	2656	2600	1	0.074	6.33	0.6	6.93	6.856

Table A.IV.5

July 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

July 1987	PCE $TBOD_5$ mg/l	FNE $TBOD_5$ mg/l	RAS TSS mg/l	ARE % Vol	RAS/ WSVSS mg/l	ARE TSS mg/l	FNE TSS mg/l	Qw MGD	Q PCE MGD	Qe MGD	Q RWI MGD	Q CPR MGD
1	210	5	5900	0.55	3245	3750	5	0.08	7.27	7.19	6.5	1.09
2	177	4	6100	0.61	3721	3600	4	0.062	7.57	7.508	6.83	1.06
3	177	19	5900	0.6	3540	3600	19	0.069	7.17	7.101	6.75	0.74
4	143	5	6000	0.62	3720	3450	5	0.069	6.92	6.851	6.33	0.91
5	157	3	5900	0.59	3481	4300	3	0.07	7.33	7.26	6.56	1.09
6	183	6	6350	0.53	3365.5	3650	6	0.073	7.66	7.587	6.7	1.28
7	147	1	9900	0.6	5940	6550	1	0.071	7.92	7.849	6.67	1.57
8	200	4	5850	0.62	3627	3900	4	0.079	7.62	7.541	6.25	1.67
9	230	3	5800	0.52	3016	3950	3	0.079	7.5	7.421	6.36	1.46
10	177	6	4150	0.62	2573	2950	6	0.087	8.02	7.933	6.46	1.88
11	190	3	6200	0.6	3720	4450	3	0.086	7.27	7.184	6.39	1.19
12	210	5	5500	0.59	3245	4200	5	0.085	7.16	7.075	6.15	1.33
13	127	4	5400	0.61	3294	3050	4	0.089	8	7.911	6.81	1.54
14	153	4	5600	0.61	3416	3350	4	0.084	7.87	7.786	6.64	1.58
15	163	5	5800	0.53	3074	3300	5	0.089	7.84	7.751	6.52	1.63
16	153	4	5500	0.58	3190	3100	4	0.079	7.44	7.361	6.25	1.48
17	167	3	5200	0.61	3172	3000	4	0.059	7.45	7.391	6.12	1.67
18	140	5	5600	0.59	3304	4050	3	0.069	7.36	7.291	5.92	1.79
19	160	3	6800	0.51	3468	4950	5	0.07	7.63	7.56	6.2	1.76
20	157	3	7250	0.5	3625	6050	3	0.076	7.01	6.934	6.22	1.15
21	200	2	6150	0.63	3874.5	5050	3	0.081	6.93	6.849	6.17	1.07
22	180	3	5250	0.63	3307.5	3450	2	0.089	6.94	6.851	6.17	1.07
23	177	3	5550	0.6	3330	3650	3	0.086	7.21	7.124	6.14	1.41
24	180	3	3550	0.61	2165.5	3000	3	0.093	7.71	7.617	5.99	2.11
25	240	3	5650	0.61	3446.5	2950	3	0.088	6.74	6.652	6.04	1.09
26	230	3		0.63		3150	3	0.09	6.57	6.48	5.94	1.02
27	197	3	4900	0.67	3283	3000	3	0.095	7.2	7.105	5.99	1.58
28	230	8	11850	0.57	6754.5	3050	8	0.087	7	6.913	5.95	1.4
29	270	4	4800	0.63	3024	2950	4	0.074	6.71	6.636	5.5	1.26
30	173	4	4450	0.65	2892.5	2550	4	0.075	6.99	6.915	5.93	1.43
31	180	4	4300	0.69	2967	2600	4	0.069	7.22	7.151	6.31	1.27

Table A.IV.6

September 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Sept 1987	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qpce	Qe	Qrwi	Qcpr
1	177	4	6550	0.61	3995.5	2900	7	0.07	7.29	7.22	5.85	1.66
2	230	4	5850	0.66	3861	2800	7	0.07	7.96	7.89	6.5	1.77
3	193	2	6500	0.65	4225	3050	10	0.077	7.39	7.313	5.81	1.87
4	160	3	6050	0.61	3690.5	3050	6	0.06	7.22	7.16	5.84	1.64
5	260	3	6700	0.65	4355	3000	4	0.057	6.95	6.893	5.92	1.3
6	250	3	6550	0.65	4257.5	3150	3	0.07	7.22	7.15	6.32	1.23
7	210	3	6100	0.65	3965	2650	4	0.056	7.35	7.294	6.41	1.27
8	230	3	6400	0.61	3904	3100	3	0.064	7.61	7.546	6.27	1.64
9	187	4	6800	0.59	4012	3050	4	0.068	7.57	7.502	6.15	1.72
10	240	3	7000	0.66	4620	2900	3	0.055	7.55	7.495	6.02	1.73
11	190	4	7300	0.58	4234	3000	1	0.07	7.07	7	5.92	1.44
12	183	5	5250	0.66	3465	2850	8	0.066	6.77	6.704	6.05	0.95
13	190	4	5650	0.64	3616	2900	7	0.072	7.07	6.998	6.57	0.74
14	190	7	6300	0.65	4095	2950	5	0.047	7.26	7.213	6.26	1.24
15	183	4	7250	0.63	4567.5	3200	7	0.054	7.19	7.136	6.03	1.4
16	240	8	6350	0.64	4064	3100	8	0.059	6.56	6.501	5.97	0.83
17	240	10	6050	0.63	3811.5	3100	6	0.064	6.95	6.886	5.96	1.23
18	270	4	6900	0.56	3864	3200	4	0.056	6.93	6.874	5.99	1.2
19	260	3	6450	0.56	3612	3100	9	0.06	6.7	6.64	5.99	0.97
20	260	5	6500	0.61	3965	3200	10	0.059	7.1	7.041	6.41	0.95
21	187	6	7700	0.61	4697	3350	6	0.051	7.24	7.189	6.32	1.18
22	220	2	7750	0.55	4262.5	3450	6	0.059	6.94	6.881	6.01	1.19
23	197	3	7100	0.61	4331	3400	7	0.067	6.76	6.693	5.92	1.11
24	153	5	3250	0.54	1755	3550	6	0.036	6.68	6.644	5.87	1.06
25	370	2	7300	0.52	3796	3600	11	0.045	6.56	6.515	5.81	1
26	260	4	7150	0.67	4790.5	3200	10	0.059	6.34	6.281	5.61	0.98
27	183	3	7150	0.6	4290	3100	19	0.05	6.34	6.29	5.66	0.93
28	280	4	7350	0.6	4410	3500	7	0.054	6.64	6.586	5.67	1.23
29	167	3	7950	0.55	4372.5	2900	5	0.06	6.69	6.63	5.61	1.36
30	270	3	7000	0.57	3990	2650	7	0.06	6.52	6.46	5.7	1.1

Table A.IV.7.

October 1987 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE BOD_5	FNE BOD_5	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qpce	Qe	Qrwi	Qcpr
1	180	3	7650	0.55	4207.5	3600	3	0.064	6.81	6.746	5.62	1.47
2	133	3	7900	0.57	4503	3250	2	0.058	6.48	6.422	5.65	1.1
3	240	3	7900	0.53	4187	3350	9	0.073	6.48	6.407	5.81	0.94
4	250	3	7700	0.59	4543	3200	7	0.052	6.39	6.338	5.73	0.91
5	270	3	6900	0.54	3726	3050	8	0.054	6.99	6.936	5.66	1.59
6	167	3	6450	0.54	3483	2950	3	0.051	7.37	7.319	5.55	2.09
7	230	4	7900	0.56	4424	2850	5	0.054	8.04	7.986	5.83	2.3
8	310	4	7750	0.54	4185	3250	7	0.046	7.04	6.994	5.74	1.57
9	220	3	7550	0.53	4001.5	3150	5	0.046	6.62	6.574	5.5	1.39
10	360	4	6600	0.6	3960	3450	1	0.04	6.57	6.53	5.5	1.33
11	310	3	7750	0.58	4495	3650	2	0.05	6.43	6.38	5.35	1.34
12	300	4		0.56		3850	8	0.04	6.55	6.51	5.47	1.33
13	180	3	8000	0.6	4800	3750	5	0.051	7.04	6.989	5.7	1.6
14	230	2	6450	0.54	3483	3350	5	0.057	6.91	6.853	5.74	1.43
15	240	2	6750	0.54	3645	3400	1	0.036	6.73	6.694	5.5	1.49
16	200	4	6700	0.6	4020	2850	2	0.046	6.76	6.714	5.35	1.72
17	270	4	7550	0.53	4001.5	3650	7	0.044	6.57	6.526	5.49	1.34
18	270	4	8600	0.6	5160	3450	11	0.049	6.62	6.571	5.58	1.3
19	180	6	6700	0.61	4087	3250	6	0.052	8.07	8.018	5.61	2.77
20	240	4	6200	0.53	3286	3300	6	0.1	6.97	6.87	5.45	1.77
21	193	3	6200	0.56	3472	3450	7	0.12	6.4	6.28	5.52	1.14
22	197	7		0.62		3800	10	0.059	6.91	6.851	5.42	2.45
23		4	8450	0.55	4647.5	4050	10	0.35	6.42	6.07	5.16	1.52
24			8100	0.55	4455	3750	10	0.069	6.31	6.241	5.39	1.18
25	210	4	8100	0.55	4455	3850	12	0.023	6.06	6.037	5.6	0.73
26	153	5	6850	0.61	4178.5	3700	16	0.039	6.77	6.731	5.48	1.55
27	250	5	7900	0.58	4582	3750	13	0.052	6.71	6.658	5.54	1.43
28	177	6	7150	0.61	4361.5	3500	14	0.055	6.63	6.575	5.35	1.54
29	173	3	7200	0.57	4104	4000	9	0.053	7.11	7.057	5.45	1.92
30	177	5	7500	0.53	3975	4000	8	0.049	6.68	6.631	5.84	1.7
31			7500	0.53	3975	4000	8	0.045	6.75	6.705	5.41	1.5

Table A.IV.8

April 1988 York River Demonstration Project

BOD_5, Solids and Flow data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE %Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qpce	Qe	Qrwi	Qcpr
1	97	11	8650	0.72	6228	4650	11	0.11	8.45	8.34	7.28	1.42
2	93	9	8250	0.73	6022.5	4900	8	0.1	8.32	8.22	7.3	1.26
3	80	14	8900	0.73	6497	4650	9	0.101	7.89	7.789	6.75	1.38
4	90	10	10200	0.71	7242	4500	8	0.108	8.48	8.372	7.2	1.53
5	93	19	8300	0.71	5893	4200	10	0.118	8.33	8.212	7.05	1.54
6	107	11	8300	0.75	6225	2600	4	0.112	8.86	8.748	6.83	2.28
7	153	14	9350	0.72	6732	4150	2	0.109	8.53	8.421	7.15	1.63
8	110	12	7100	0.73	5183	3700	6	0.11	10.59	10.48	9.36	1.48
9	123	15	8250	0.7	5775	3550	7	0.11	9.13	9.02	8.21	1.18
10	77	12	7850	0.69	5416.5	4150	7	0.111	8.87	8.759	7.94	1.19
11	175	16	7950	0.71	5644.5	3600	7	0.109	9.62	9.511	7.76	2.11
12	97	19	8200	0.66	5412	4100	8	0.1	11	10.9	8.36	2.9
13	100	29	7350	0.71	5218.5	3100	9	0.105	12.31	12.205	10.66	1.9
14	87	19	8000	0.71	5680	3400	5	0.095	10.71	10.615	9.28	1.68
15	97	23	8150	0.72	5868	3500	15	0.095	9.31	9.215	8.11	1.45
16	97	10	8200	0.68	5576	4050	8	0.091	9.33	9.239	7.92	1.66
17	107	14	8250	0.69	5692.5	4050	13	0.092	8.74	8.648	7.73	1.26
18	130	21	7400	0.7	5180	3650	10	0.089	9.01	8.921	7.74	1.53
19	130	13	6700	0.71	4757	3550	10	0.09	9.78	9.69	8.26	1.77
20	117	26	7150	0.66	4719	3350	14	0.093	9.79	9.697	8.17	1.87
21	133	13	7850	0.71	5573.5	3650	4	0.083	9.39	9.307	7.89	1.76
22	107	11	7200	0.7	5040	3550	8	0.091	8.95	8.859	7.52	1.7
23	117	11	7950	0.69	5485.5	3750	9	0.091	8.72	8.629	7.56	1.41
24	113	8	6600	0.7	4620	3400	5	0.09	8.74	8.65	7.54	1.46
25	157	10	7000	0.74	5180	1500	8	0.091	8.89	8.799	7.42	1.72
26	117	7	2350	0.71		3200	2	0.096	8.98	8.884	7.23	1.98
27	110	7	6850	0.69	4726.5	3150	4	0.099	8.86	8.761	7.15	1.96
28	123	5	7400	0.7	5180	3050	2	0.109	8.95	8.841	7.68	1.53
29	133	8	6250	0.71	4437.5	3300	4	0.094	8.6	8.506	7.32	1.54
30	120	5	7350	0.69	5071.5	3600	9	0.13	8.14	8.01	7.17	1.23

Table A.IV.9

July 1988 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/WS VSS	ARE TSS	FNE TSS	Qw	Qe	Qrwi	Qcpr
1	147	4	5650	0.67	3785.5	2600	6	0.104	7.956	5.92	2.14
2	110	7	5700	0.61	3477	2650	9	0.105	7.235	5.79	1.55
3	117	5	5050	0.69	3484.5	2550	8	0.103	7.247	5.62	1.73
4	93	6	5650	0.65	3672.5	2700	5	0.111	7.639	5.81	1.94
5	135	7	8200	0.66	5412	2550	4	0.106	8.154	6.09	2.17
6	135	9	5350	0.66	3531	2550	7	0.132	7.858	6.07	1.92
7	230	7	4800	0.64	3072	2550	8	0.23	8.14	6	2.37
8	103	7	4650	0.66	3069	2450	10	0.086	7.904	6.03	1.96
9	103	7	5100	0.61	3111	2500	3	0	7.53	6.27	1.26
10	90	5	5250	0.66	3465	2600	5	0.091	7.649	6.14	1.6
11	140	6	5500	0.7	3850	2850	3	0.098	7.602	6.27	1.43
12	150	6	5400	0.69	3726	2600	7	0.106	8.384	6.22	2.27
13	123	7	5600	0.67	3752	2500	9	0.116	7.894	6.03	1.98
14	97	6	5550	0.72	3996	2550	11	0.104	8.696	6.59	2.21
15	144	5	5500	0.59	3245	2550	6	0.112	7.868	6.11	1.87
16	123	4	5050	0.64	3232	2500	9	0.113	7.427	6.07	1.47
17	115	5	5100	0.68	3468	2550	5	0.111	7.669	6.02	1.76
18	184	11	5050	0.68	3434	2400	7	0.104	8.306	6.24	2.17
19	135	6	5700	0.65	3705	2500	4	0.11	7.58	6.12	1.57
20	124	4	5050	0.7	3535	2600	7	0.1	8.14	6.69	1.55
21	90	4	5750	0.61	3507.5	2700	6	0.105	8.725	6.38	2.45
22	93	3	5700	0.66	3762	2850	5	0.112	8.188	6.21	2.09
23	87	3	5950	0.58	3451	2700	9	0.111	8.699	6.24	2.57
24	100	3	5550	0.67	3718.5	2700	5	0.114	8.076	6.2	1.99
25	270	3	5250	0.67	3517.5	2500	3	0.111	8.939	6.28	2.77
26	275	3	4350	0.69	3001.5	2050	3	0.104	7.286	6.04	1.35
27	107	4	5000	0.66	3300	2250	3	0.105	7.635	6.34	1.4
28	310	4	5900	0.63	3717	2700	4	0.105	7.795	6.47	1.43
29	145	4	6600	0.55	3630	2900	7	0.107	6.733	5.47	1.37
30	93	4	6250	0.62	3875	2950	11	0.1	7.57	6.14	1.53
31	70	3	5950	0.66	3927	2450	5	0.11	6.99	5.98	1.12

Table A.IV.10.

August 1988 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qe	Qrwi	Qcpr
1	129	3	5750	0.58	3335	2750	2	0.096	7.11	6.13	1.31
2	103	3	5700	0.65	3705	2700	3	0.114	7.10	6.07	1.16
3	103	4	5750	0.64	3680	2700	4	0.121	7.57	6.1	1.13
4	120	3	5950	0.59	3510.5	2550	8	0.099	7.88	5.54	2.13
5	90	5	5900	0.65	3835	2550	16	0.102	7.73	5.98	2.01
6	100	5	5650	0.66	3729	2700	6	0.104	7.57	6.15	1.69
7	113	4	5250	0.63	3307.5	2350	4	0.100	7.57	6.16	1.52
8	255	10	5750	0.67	3852.5	2450	5	0.096	7.68	6.1	1.57
9	103	27	5500	0.63	3465	2600	9	0.094	8.55	6.04	1.74
10	129	9	5650	0.62	3503	2550	5	0.106	8.78	7.19	1.47
11	122	9	5950	0.55	3272.5	2700	7	0.099	7.77	7.18	1.7
12	93	5	6200	0.56	3472	2800	5	0.095	7.33	6.14	1.73
13	93	4	5350	0.55	2942.5	2500	4	0.112	7.81	6.04	1.41
14	103	3	5850	0.57	3334.5	2450	6	0.111	7.53	6.06	1.87
15	132	7	6000	0.62	3720	2750	3	0.100	7.52	6.14	1.49
16	170	5	5950	0.62	3689	2600	4	0.101	7.51	6.03	1.6
17	239	4	5300				4	0.100	9.06	5.98	1.63
18	87	4	5400	0.56	3024	2500	5	0.113	8.50	6.26	2.92
19	93	5	5550	0.55	3052.5	2550	4	0.102	8.85	6.38	2.23
20	127	6	4800	0.53	2544	2300	5	0.107	8.25	7.02	1.94
21	83	2	5150	0.56	2884	2400	2	0.107	8.10	6.92	1.44
22	97	2	5050	0.65	3282.5	2400	3	0.100	8.44	6.67	1.53
23	103	3	5700	0.6	3420	2600	2	0.097	7.51	6.27	2.27
24	93	3	5750	0.64	3680	2600	8	0.108	8.19	6.33	1.29
25	97	3	5700	0.59	3363	2500	3	0.086	7.91	6.17	2.11
26	87	3	5750	0.58	3335	2750	3	0.088	7.23	6.22	1.78
27	93	5	6050	0.55	3327.5	2700	4	0.095	7.37	6.11	1.22
28	90	3	5700	0.57	3249	2700	3	0.090	7.45	6.01	1.46
29	83	3	5900	0.59	3481	2550	3	0.087	8.20	6	1.54
30	103	2	6300	0.56	3528	2800	2	0.088	7.16	6.77	1.52
31	90	3	5900	0.57	3363	2850	4	0.091	7.85	6.25	1.01

Table A.IV.11

July 1989 York River Demonstration Project

BOD_5, Solids and Flow Data, used in Sludge Calculations

Day	PCE $TBOD_5$	FNE $TBOD_5$	RAS TSS	ARE % Vol	RAS/ WSVSS	ARE TSS	FNE TSS	Qw	Qrwi	Qcpr	Qpce	Qe
1	132	12	5750	0.69	3967.5	2850	17	0.106	7.94	1.15	9.09	8.98
2	135	12	6200	0.68	4216	2750	12	0.091	7.5	0.99	8.49	8.40
3	120	12	6150	0.69	4243.5	2950	11	0.111	7.75	1.2	8.95	8.84
4	130	17	6000	0.65	3900	3100	8	0.110	7.73	1.23	8.96	8.85
5	210	8	5900	0.69	4071	3000	5	0.108	8.24	1.23	9.47	9.36
6	130	9	6250	0.71	4437.5	2600	3	0.096	8.32	1.7	10.02	9.92
7	144	6	6450	0.71	4579.5	3200	5	0.112	8.34	0.6	8.94	8.83
8	93	6	7000	0.67	4690	3100	4	0.099	8.14	1.82	9.96	9.86
9	110	3	6100	0.69	4209	3050	4	0.118	7.87	1.28	9.15	9.03
10	110	7	6800	0.68	4624	3150	4	0.139	8.07	1.5	9.57	9.43
11	189	9	6600	0.68	4488	3200	12	0.096	7.86	1.21	9.07	8.97
12	165	6	6950	0.72	5004	3250	3	0.091	7.67	1.53	9.2	9.11
13	122	8	7050	0.67	4723.5	3000	6	0.075	8	1.41	9.41	9.34
14	147	8	6600	0.7	4620	3250	2	0.093	8.77	1.23	10	9.91
15	113	5	6400	0.68	4352	3100	3	0.086	8.24	1.28	9.52	9.43
16	190	7	6600	0.68	4488	3250	4	0.076	8.67	1.1	9.77	9.69
17	113	12	6900	0.69	4761	3200	11	0.071	9	2.76	11.76	11.69
18	103	7	5850	0.71	4153.5	2700	4	0.090	8.75	3.1	11.85	11.76
19	80	8	6300	0.75	4725	2450	3	0.074	8.6	4.75	13.35	13.28
20	93	7	6350	0.74	4699	2650	4	0.102	8.5	4.09	12.59	12.49
21	90	7	5950	0.73	4343.5	2350	4	0.119	8.3	3.9	12.2	12.08
22	80	7	5900	0.66	3894	2700	5	0.106	8.25	3.5	11.75	11.64
23	152	8	5450	0.73	3978.5	2350	4	0.099	7.89	2.71	10.6	10.50
24	129	8	3750	0.73	2737.5	2850	4	0.301	7.94	2.88	10.82	10.52
25	137	7	6400	0.73	4672	2650	4	0.082	7.81	1.59	9.4	9.32
26	169	8	7300	0.69	5037	3100	6	0.089	7.87	1.61	9.48	9.39
27	165	14	7000	0.71	4970	3250	7	0.096	7.85	1.52	9.37	9.27
28	142	9	6800	0.79	5372	3200	6	0.130	7.89	2.21	10.1	9.97
29	107	7	7050	0.68	4794	3500	5	0.104	7.92	1.36	9.28	9.18
30	147		7100	0.71	5041	3350		0.100	7.71	1.33	9.04	8.94
31	120	16	6750	0.71	4792.5	3200	7	0.110	7.26	1.7	8.96	8.85

Table A.IV.12

July 1987 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

July 1987	QPCE MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE $TBOD_5$ mg/l	CELL 4 NO_x mg/l	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY Q MGD	MLTSS -cell 7 mg/l	%TVSS	sludge age-d
Jul 2 87	7.57	27.97	0.28	177	no data taken	no data taken	7.42	none	3600	0.61	12.1
Jul 5 87	7.33	36.08	0.07	157			8.84		4300	0.59	13.3
Jul 7 87	7.92	23.90	0.03	147			5.75		6550	0.67	12.5
Jul 9 87	7.5	21.88	0.12	230			6.33		3950	0.53	11.6
Jul 12 87	7.16	26.39	0.05	210			5.85		4200	0.59	11.9
Jul 14 87	7.87	26.17	0.04	153			5.94		3350	0.61	9.6
Jul 16 87	7.44	24.29	0.03	153			6.03		3100	0.58	9.0
Jul 19 87	7.63	34.40	0.05	160			9.15		4950	0.51	13.6
Jul 21 87	6.93	25.89	0.06	200			5.72		5050	0.63	13.6
Jul 23 87	7.21	26.74	0.05	177			6.85		3650	0.6	10.1
Jul 26 87	6.57	31.33	0.05	230			5.93		3150	0.63	0.20
Jul 28 87	7	40.68	0.09	230			8.06		3050	0.57	3.9
Jul 30 87	6.99	30.96	0.07	270			6.66		2550	0.65	10.0

Table A.IV.13

July 1987 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

July 1987	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS NH_3 (ARE) mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE $TBOD_5$ mg/l	FNE TSS mg/l
Jul 2 87	10.44	4.7	6100	0.062	0.29	4.70	1.50	0.17	7.75	4	4
Jul 5 87	10.03	5.69	5900	0.070	0.64	5.69	1.43	0.07	7.59	3	4
Jul 7 87	8.5	1.89	9900	0.071	0.50	1.89	1.28	0.50	6.99	1	2
Jul 9 87	9.55	3.06	5800	0.079	0.50	3.06	1.49	0.50	6.46	3	1
Jul 12 87	9.71	4.02	5500	0.085	0.33	4.02	2.44	0.12	5.72	5	2
Jul 14 87	9.65	2.83	5600	0.084	0.68	2.83	4.73	0.55	6.04	4	1
Jul 16 87	9.5	2.56	5500	0.079	0.50	2.56	1.33	0.50	5.57	4	1
Jul 19 87	9.87	6.17	6800	0.070	0.59	6.17	1.68	0.12	7.74	3	3
Jul 21 87	9.91	3.88	6150	0.081	0.37	3.88	1.28	0.10	5.32	2	1
Jul 23 87	8.36	4.96	5550	0.086	0.45	4.96	2.31	0.17	7.33	3	2
Jul 26 87	9.37	3.91		0.090	0.20	3.91	1.45	0.06	6.64	3	3
Jul 28 87	9.62	4.82	11850	0.087	0.51	4.82	4.03	0.18	7.39	8	6
Jul 30 87	9.8	4.78	4450	0.075	0.57	4.78	1.43	0.50	6.80	4	2

Table A.IV.14

August 1987 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

August 1987	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	CELL 4 NO_x mg/l	Cell 5 NO_x mg/l	Cell 7 NO_x mg/l	NRCY MGD	MLTSS mg/l	MLVSS, %	sludge age-d
Aug 2 87	6.77	26.10	0.07	177					3050	0.64	12.5
Aug 4 87	8.55	20.88	0.46	190					2550	0.67	12.5
Aug 6 87	7.88	24.98	0.26	180					2550	0.69	11.6
Aug 9 87	6.71	34.65	0.02	123	1.05	0.72	8.55	8.06	2600	0.70	12.8
Aug 11 87	7.25	26.18	0.01	117	0.28	0.1	4.31	8.06	2950	0.69	12.9
Aug 13 87	8.05	24.41	0.05	180	0.27	0.13	4.03	8.06	2800	0.64	10.3
Aug 16 87	7.04	28.57	0.22	163		0.09	5.59	7.26	2800	0.70	9.8
Aug 18 87	7.29	32.54	0.03	170	0.30	0.11	6.57	7.26	2850	0.65	11.2
Aug 20 87	7.67	30.84	0.03	153	0.24	0.12	7.64	7.26	3100	0.64	11.5
Aug 23 87	6.43	30.14	0.05	150	0.03	1.23	4.14	7.26	3300	0.61	10.6
Aug 25 87	7.08	28.23	0.03	150	0.10	0.08	4.1	7.26	2800	0.67	9.4
Aug 27 87	7.98	32.22	0.20	180	0.66	0.13	7.61	7.26	9000	0.60	22.6
Aug 30 87	7.22	40.21	0.05	173	4.92	0.5	8.27	7.26	3100	0.65	9.5

Table A.IV.15

August 1987 York River Demonstration Project Raw Data

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent Data

August 1987	RAS Q MGD	RAS NO_x mg/l	RAS TSS MG/L	WS Q mgd	WS (ARE) NH3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
Aug 2 87	9.52	3.53	4600	0.066	0.50	3.53	1.53	0.50	8.23	4	4
Aug 4 87	3	4.04	4450	0.154	0.50	4.04	1.33	0.50	6.56	4	4
Aug 6 87	9.74	4.19	4700	0.061	0.50	4.19	1.41	0.50	6.79	2	2
Aug 9 87	10.37	4.98	4100	0.063	0.50	4.98	1.42	0.50	5.88	3	3
Aug 11 87	8.06	0.84	4800	0.061	0.50	0.84	1.35	0.50	2.69	3	3
Aug 13 87	9.29	0.81	4850	0.068	0.50	0.81	1.30	0.50	3.58	6	6
Aug 16 87	9.74	1.99	4350	0.073	0.50	1.99	2.15	0.50	4.16	12	12
Aug 18 87	9.31	2.18	4650	0.071	0.82	2.18	1.41	1.12	4.06	3	3
Aug 20 87	8.71	2.53	4450	0.076	0.44	2.53	1.84	0.42	4.69	5	5
Aug 23 87	7.77	1.95	5300	0.078	0.74	1.95	1.97	0.62	4.86	2	2
Aug 25 87	5.56	0.24	6000	0.060	1.01	0.24	1.95	0.11	3.33	8	8
Aug 27 87	4.62	0.27	6350	0.078	5.16	0.27	2.63	1.14	7.64	8	8
Aug 30 87	5.25	4.1	5650	0.069	0.99	4.10	2.16	0.5	8.70	9	9

Table A.IV.16.

September 1987 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

September 1987	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	CELL 4 NO_x mg/l	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS -cell 7 mg/l	%MLVSS	sludge age-d
Sep 1 87	7.29	29.96	0.04	177	0.12	0.1	5.52	7.26	2900	0.61	7.9
Sep 3 87	7.39	25.75	0.01	193	0.92	0.2	6.08	7.26	3050	0.65	6.8
Sep 6 87	7.22	26.88	0.09	250	0.12	0.08	4.46	7.26	3150	0.65	7.4
Sep 8 87	7.61	28.32	0.14	230	0.45	0.24	5	7.26	3100	0.61	9.8
Sep 10 87	7.55	23.48	0.26	240	0.27	0.06	4.89	7.26	2900	0.66	9.8
Sep 13 87	7.07	26.86	0.07	190	0.05	0.08	5.65	7.26	2900	0.64	8.7
Sep 15 87	7.19	31.75	0.06	183	1.32	0.05	7.64	7.26	3200	0.63	10
Sep 17 87	6.95	28.20	0.07	240	0.12	1.66	0.14	7.26	3100	0.63	10
Sep 20 87	7.1	26.05	0.03	260	7.29	7.16	NS	7.26	3200	0.61	9.7
Sep 22 87	6.94	23.45	0.12	220	0.05	0.04	NS	7.26	3450	0.55	9.5
Sep 24 87	6.68	26.48	0.07	153	0.07	0.04	4.69	7.26	3550	0.54	ERR
Sep 27 87	6.34	28.20	0.05	183	0.1	0.05	3.62	7.26	3100	0.60	9.1
Sep 29 87	6.69	23.73	0.08	167	0.06	0.05	3.27	7.26	2900	0.55	7.8

Table A.IV.17.

September 1987 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

September 1987	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS (ARE) NH_3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD mg/l	FNE TSS mg/l
Sep 1 87	4.81	0.11	6550	0.070	0.50	0.11	2.04	0.50	4.55	7	7
Sep 3 87	5.55	0.27	6500	0.077	0.50	0.27	2.02	0.50	5.59	10	10
Sep 6 87	5.42	0.9	6550	0.070	0.50	0.90	1.68	0.50	5.11	3	3
Sep 8 87	5.16	0.44	6400	0.064	0.50	0.44	1.34	0.50	4.31	3	3
Sep 10 87	5.18	0.06	7000	0.055	0.52	0.06	1.38	0.50	5.89	3	3
Sep 13 87	6.46	1.1	5650	0.072	0.50	1.10	1.59	0.50	4.04	7	7
Sep 15 87	5.71	0.06	7250	0.054	0.50	0.06	1.39	0.50	5.15	7	7
Sep 17 87	6.27	0.02	6050	0.064	0.50	0.02	1.52	0.50	4.32	6	6
Sep 20 87	5.66	0.04	6500	0.059	2.97	0.04	2.19	0.67	2.24	10	10
Sep 22 87	4.92	0.02	7750	0.059	0.50	0.02	1.31	0.50	3.95	6	6
Sep 24 87	4.43	0.04	7200	0.036	0.50	0.04	1.76	0.50	5.08	6	6
Sep 27 87	5.13	0.14	7150	0.050	0.50	0.14	1.71	0.50	4.54	19	19
Sep 29 87	4.87	0.12	7950	0.060	1.27	0.12	1.46	0.50	3.08	5	5

Table A.IV.18.

October 1987 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

October 1987	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	CELL 4 NO_x mg/l	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS cell 7 mg/l	%TVSS	sludge age-d
Oct 1 87	6.81	28.06	0.04	180	0.04	0.03	4.8	7.26	3600	0.55	9.5
Oct 4 87	6.39	28.76	0.02	250	0.28	0.03	3.83	7.26	3200	0.59	9.9
Oct 6 87	7.37	24.96	0.14	167	0.05	0.02	6.29	7.26	2950	0.54	11.6
Oct 8 87	7.04	28.51	0.05	310	0.21	0.06	5.01	7.26	3250	0.54	12.6
Oct 11 87	6.43	28.00	0.04	310	0.17	0.03	4.55	7.26	3450	0.58	12.5
Oct 13 87	7.04	30.82	0.18	180	0.18	0.04	2.97	7.26	3750	0.60	11.7
Oct 15 87	6.73	26.83	0.04	240	0.09	0.01	3.37	7.26	3400	0.54	18.5
Oct 18 87	6.62	29.19	0.06	270	0.18	0.04	3.34	7.26	3450	0.60	9.7
Oct 20 87	6.97	25.06	0.06	240	0.18	0.04	3.98	7.26	3300	0.53	6.90
Oct 22 87	6.91	32.27	0.16	197	0.13	0.03	3.51	7.26	3800	0.62	10.96
Oct 25 87	6.06	32.45	0.05	210	0.05	0.03	2.05	7.26	3850	0.55	20.70
Oct 27 87	6.71	32.91	0.09	250	0.10	0.02	2.8	7.26	3750	0.58	10.4
Oct 29 87	7.11	29.26	0.07	173	0.12	0.04	0.99	7.26	4000	0.57	14.1

Table A.IV.19.

October 1987 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

October 1987	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS (ARE) NH_3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
Oct 1 87	4.51	0.06	7650	0.064	0.50	0.06	1.37	0.50	4.46	3	3
Oct 4 87	4.48	0.04	7700	0.052	0.50	0.04	1.46	0.50	4.94	3	7
Oct 6 87	5.54	0.23	6450	0.051	0.50	0.23	1.89	0.50	3.99	3	3
Oct 8 87	4.24	0.13	7750	0.046	0.50	0.13	1.85	0.50	5.23	4	7
Oct 11 87	4.89	0.02	7750	0.050	0.50	0.02	1.62	0.50	4.64	3	2
Oct 13 87	6.37	0.09	8000	0.051	0.50	0.09	1.92	0.50	4.27	3	5
Oct 15 87	6.23	0.05	6750	0.036	0.50	0.05	1.76	0.50	2.27	2	1
Oct 18 87	5.6	0.08	8600	0.049	0.50	0.08	1.58	0.50	3.94	4	11
Oct 20 87	6.98	0.11	6200	0.100	0.50	0.11	1.41	0.50	1.30	4	6
Oct 22 87	6.52	0.1	7000	0.059	1.21	0.10	2.61	0.50	4.49	7	10
Oct 25 87	5.57	0.15	8100	0.023	0.50	0.15	2.52	0.50	4.06	4	12
Oct 27 87	6.06	0.01	7900	0.052	2.64	0.01	3.61	2.24	3.98	5	13
Oct 29 87	7.38	0.1	7200	0.053	10.70	0.10	2.25	1.09	3.35	3	9

Table A.IV.20

April 1988 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

April 1988	QPCE MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	ARCY MGD	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS mg/l	%MLVSS	Sludge Age-d
Apr 5 88	8.33	20.27	0.04	93	7.49	0.09	0.12	7.26	3750	0.71	6
Apr 7 88	8.53	19.84	0.04	153	7.49	0.01	0.01	7.26	4300	0.72	5
Apr 12 88	11	21.03	0.06	97	7.49	0.03	2.01	7.26	6550	0.66	6
Apr 14 88	10.71	16.32	0.04	87	7.49	0.22	2.22	7.26	3950	0.71	6
Apr 19 88	9.78	24.38	0.14	130	7.49	0.41	3.62	7.26	4200	0.71	6
Apr 21 88	9.39	19.82	0.18	133	7.49	0.23	4.07	7.26	3350	0.71	6.5
Apr 26 88	8.96	18.37	0.07	117	7.49	0.35	0.21	7.26	3100	0.71	17
Apr 28 88	8.95	18.33	0.17	123	7.49	0.31	5.34	7.26	4950	0.7	4

Table A.IV.21.

April 1988 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

April 1988	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS (ARE) NH_3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
Apr 5 88	13.35	0.01	8300	0.118	10.70	0.01	13.05	11.45	0.58	19	10
Apr 7 88	7.08	0.01	9350	0.109	11.92	0.01	12.86	11.57	0.87	14	2
Apr 12 88	7.2	0.01	8200	0.100	10.72	0.01	8.87	8.62	1.29	19	8
Apr 14 88	9.01	0.01	8000	0.095	8.74	0.01	11.51	10.25	1.43	19	5
Apr 19 88	8.41	0.02	6700	0.090	11.02	0.02	8.93	8.74	2.55	13	10
Apr 21 88	7.88	0.04	7850	0.083	4.97	0.04	6.63	6.20	4.1	13	4
Apr 26 88	6.97	0.01	7000	0.096	25.64	0.01	2.91	0.70	4.28	7	2
Apr 28 88	7.31	0.22	7400	0.109	0.80	0.22	2.19	0.80	4.55	5	2

Table A.IV.22.

May 1988 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

May 1988	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	ARCY MGD	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS mg/L	%MLVSS	sludge age-d
May 2 88	8.25	27.41	0.17	154	7.36	0.35	5.18	7.36	2950	0.71	6
May 5 88	12.31	26.14	0.18	100	7.36	0.03	3.77	7.36	2700	0.71	5
May 10 88	10.42	20.29	0.14	83	7.36	0.36	4.34	7.36	3700	0.72	5
May 12 88	9.6	22.27	0.14	90	7.36	0.39	0.15	7.36	3100	0.69	7
May 17 88	9.65	28.37	0.21	122	7.36	0.34	5.87	7.36	3300	0.72	6
May 19 88	9.71	24.58	0.12	103	7.36	0.48	5.76	7.36	2900	0.68	5
May 24 88	12.45	20.88	0.14	130	7.36	0.17	5.13	7.36	3200	0.70	7
May 26 88	10.96	20.01	0.06	100	7.36	0.02	4.97	7.36	2850	0.71	6
May 31 88	9.48	21.55	0.09	100	7.36	0.39	5.07	7.36	3300	0.71	6

Table A.IV.23.

May 1988 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent Data

May 1988	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS (ARE) NH_3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
May 2 88	7.12	0.34	6700	0.097	0.50	0.34	1.44	0.50	4.13	11	10
May 5 88	8.33	0.05	5850	0.104	0.50	0.05	2.73	0.91	4.56	16	10
May 10 88	7.78	0.09	7200	0.096	0.50	0.09	1.20	0.50	2.85	3	1
May 12 88	7.62	0.03	6950	0.094	13.28	0.03	1.53	0.50	3.78	4	4
May 17 88	8.02	0.11	7000	0.091	2.31	0.11	1.44	0.72	4.34	4	8
May 19 88	7.52	0.03	6700	0.098	0.50	0.03	1.32	0.50	4.28	4	6
May 24 88	7.96	0.04	6050	0.090	0.50	0.04	1.44	0.50	2.77	5	5
May 26 88	8.28	0.01	6900	0.089	0.50	0.01	1.14	0.50	3.46	3	6
May 31 88	7.56	0.01	6700	0.092	0.50	0.01	0.88	0.50	3.18	4	6

Table A.IV.24.

June 1988 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

June 1988	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	ARCY MGD	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS mg/L	%MLVSS	sludge age-days
Jun 2 88	8.65	25.01	0.37	189	7.36	0.58	6.08	7.36	2950	0.74	5.0
Jun 7 88	8.80	25.92	0.33	100	7.36	0.54	6.72	0	3350	0.71	6.0
Jun 9 88	9.04	21.40	0.36	132	7.36	0.33	6.46	0	3350	0.68	10.0
Jun 14 88	9.18	28.49	0.68	280	7.36	0.01	6.38	0	3400	0.64	7.4
Jun 16 88	9.28	19.17	0.21	77	7.36	0.05	6.89	0	3000	0.69	5.3
Jun 21 88		27.22	0.17		7.36	0.02	6.6	3.68	2800	0.68	4.9
Jun 23 88	6.91	27.14	0.13	350	7.36	0.04	7.29	3.68	2600	0.64	5.5
Jun 28 88	7.76	24.02	0.16	127	7.36	0.17	6.33	3.68	2500	0.65	4.4
Jun 30 88	8.29	25.59	0.11	249	7.36	0.15	6.51	3.68	2800	0.68	5.2

Table A.IV.25.

June 1988 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

June 1988	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS NH_3 (ARE) mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
Jun 2 88	6.71	0.07	7550	0.085	0.50	0.07	1.78	0.50	4.35	6	6
Jun 7 88	6.89	0.01	7050	0.085	0.50	0.01	1.56	0.50	6.06	3	5
Jun 9 88	6.71	0.08	6650	0.100	0.50	0.08	1.52	0.50	5.17	4	12
Jun 14 88	6.73	0.02	7500	0.064	0.50	0.02	1.44	0.50	4.90	3	6
Jun 16 88	6.48	0.02	6800	0.091	0.50	0.02	1.42	0.50	6.28	3	8
Jun 21 88	6.63	0.02	6350	0.100	0.50	0.02	1.98	0.50	7.04		9
Jun 23 88	6.1	0.07	5400	0.100	0.50	0.07	2.31	0.50	6.09	2	4
Jun 28 88	5.98	0.42	5550	0.122	0.50	0.42	1.95	0.50	5.90	3	3
Jun 30 88	5.91	0.03	5800	0.108	0.50	0.03	2.18	0.50	5.53	3	5

Table A.IV.26.

July 1988 York River Demonstration Project

Nutrient and Related Data for Primary Clarifier and Biological Reactor

July 1988	PCE Q MGD	PCE TKN mg/l	PCE NO_x mg/l	PCE BOD_5 mg/l	ARCY MGD	CELL 5 NO_x mg/l	CELL 7 NO_x mg/l	NRCY MGD	MLTSS mg/L	%MLVSS	sludge age-d
Jul 5 88	7.65	27.84	0.04	135	7.36	0.03	3.73	3.68	2550	0.66	3.4
Jul 7 88	8.02	37.28	0.36	230	7.36	0.24	0.09	4.03	2550	0.64	2.7
Jul 12 88	8.04	26.37	0.03	150	7.36	0.03	4.86	4.03	2600	0.69	4.9
Jul 14 88	8.5	23.26	0.01	97	7.36	0.02	4.37	4.03	2550	0.72	4.7
Jul 19 88	7.37	27.85	0.33	135	7.36	0.12	6.11	4.03	2500	0.65	4.6
Jul 21 88	8.53	27.03	0.10	90	7.36	0.01	6.99	4.03	2700	0.61	5
Jul 26 88	7.08	32.31	0.17	275	7.36	0.01	7.28	4.03	2050	0.69	5.2
Jul 28 88	7.58	33.45	0.05	310	7.36	0.01	0.84	4.03	2700	0.63	4.9

Table A.IV.27.

July 1988 York River Demonstration Project

Nutrient and Related Data for Return Sludge Recycle, Waste Sludge and Final Effluent

July 1988	RAS Q MGD	RAS NO_x mg/l	RAS TSS mg/l	WS Q MGD	WS (ARE) NH_3 mg/l	WS NO_x mg/l	SCE TKN mg/l	SCE NH_3 mg/l	SCE NO_x mg/l	FNE BOD_5 mg/l	FNE TSS mg/l
Jul 5 88	6.17	0.04		0.106	3.54	0.04	13.05	3.29	4.75	7	4
Jul 7 88	6.38	0.19	5500	0.230	29.00	0.19	12.86	8.46	3.64	7	8
Jul 12 88	6.23	0.03	5600	0.106	0.50	0.03	8.87	0.5	4.99	6	7
Jul 14 88	7.04	0.03	5550	0.104	0.50	0.03	11.51	0.5	4.63	6	11
Jul 19 88	5.57	0.02	5700	0.110	0.50	0.02	8.93	0.5	5.84	6	4
Jul 21 88	6.25	0.02	5750	0.105	1.07	0.02	6.63	1.07	6.27	4	6
Jul 26 88	5.7	0.02	4350	0.104	0.65	0.02	2.91	0.5	4.67	3	3
Jul 28 88	5.97	0.68	5900	0.105	7.58	0.68	2.19	8.38	3.79	4	4

HRSD D.O. and Uptake Profile York River STP 1987 and 1988
Table A.IV.28

Mode	July 1987 Nitrification without Nitrate Recycle	August-October A^2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Cell 1 D.O. mg/l	0.2	0.1	0.2	0.3	0.2	0.1	0.1	0.1
Cell 2 D.O. mg/l	0.2	0.2	0.2	0.3	0.1	0.1	0.1	0.1
Cell 3 D.O. mg/l	0.2	0.2	0.2	0.3	0.1	0.1	0.1	0.1
Cell 4 D.O. mg/l	1.2	0.3	0.2	0.3	0.1	0.1	0.1	0.1
Cell 5 D.O. mg/l	2.1	0.4	0.2	0.3	0.1	0.1	0.1	0.1
Cell 6 D.O. mg/l	3.5	3.6	3.0	3.1	4.2	4.9	4.6	3.4
Cell 7 D.O. mg/l	4.7	3.8	2.5	2.4	4.7	5.7	5.0	3.8
Cell 4 Uptake mg O_2/l/hr	133.3							
Cell 5 Uptake mg O_2/l/hr			108.0					
Cell 6 Uptake mg O_2/l/hr		161.5		82.4	69.1	97.3	104.1	88.0
Cell 7 Uptake mg O_2/l/hr	60.2	49.4	80.1	64.7	54.2	83.2	87.8	75.8

Table A.IV.29.

COD/TKN Influent Data York River STP 1987 and 1988

Mode	July 1987 A/O Mode Nitrification without Nitrate Recycle	August-October A_2/O Mode Nitrification with NO_3 Recycle			April-July 1988 VIP Mode Nitrification with ARCY and NRCY			
Month	July 1987	August 1987	Sept 1987	October 1987	April 1988	May 1988	June 1988	July 1988
Average RWI COD	377	383	395	413	345	346	364	437
Average RWI TKN	29.1	29.5	30.4	32.3	29.2	26.0	27.7	29.2

Vita

Denise Michele Mosca was born on Long Island, New York on April 28, 1955 and moved to Gloucester, Virginia following graduation from Southampton College of Long Island University with a B. S. in Biology and Marine Science (Biology) in 1977. In Gloucester, at the Virginia Institute of Marine Science, she worked as a summer aide and laboratory technician. She continued her career as a laboratory technician at James R. Reed and Associates, and later, Hampton Roads Sanitation District at their North Shore Laboratory, both in Newport News, Virginia. At the time her masters work began, she worked as a chemist in the North Shore Laboratory and presently works as a permit engineer for the Department of Environmental Quality - Water Division in their Kilmarnock Office.