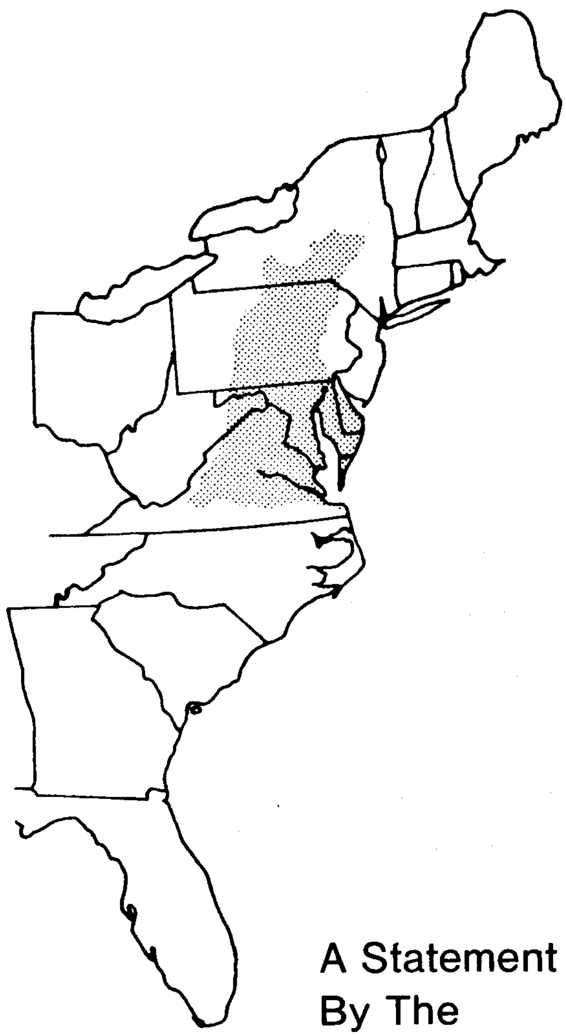
Nutrient Control In The Chesapeake Bay



A Statement Issued By The

Scientific And Technical Advisory Committee

Chesapeake Bay Program

NUTRIENT CONTROL IN THE CHESAPEAKE BAY

A Statement Issued

by the

SCIENTIFIC AND TECHNICAL ADVISORY COMMITTEE (STAC)

CHESAPEAKE BAY PROGRAM

Prepared by the

STAC Ad-Hoc Working Group on Nutrients

January 1986

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PREFACE

Nitrogen and phosphorus enter the Chesapeake Bay from both point and nonpoint sources. Appropriate agencies are addressing the policy of nonpoint source loadings of both nutrients and there has been development of a point source phosphorus policy as part of the Chesapeake Bay cleanup strategy. A comparable strategy regarding point source loadings of nitrogen, however, has not been articulated.

Many wastewater treatment plants in the Bay area discharge large quantities of ammonia-nitrogen, which has been associated with dissolved oxygen sags such as in the James River; algal growth stimulation such as in the Potomac River and the Patuxent River in the summer and; toxicity such as in Baltimore Harbor. In addition, the potential to transmit nitrogen loads from the upper tributaries to the Chesapeake Bay exists, and there is evidence that nitrogen may limit algal production in the downstream estuarine portions.

The Chesapeake Bay Program Scientific and Technical Advisory Committee strongly recommends that wastewater treatment strategy in the Chesapeake Bay environs address nitrogen control. The traditional belief that nitrogen removal in treatment plants is not economically feasible is no longer valid, given the evidence from operating treatment plants in 9 countries. Nitrogen removal should, therefore, be considered in all plans for plant upgradings, expansions, and constructions.

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GLOSSARY OF ACRONYMS

ADP	Adenosine disposphate
A/O, A^2O	Anaerobic/Oxic nutrient removal process systems
ATP	Adenosine triphosphate
BOD	Biological oxygen demand
COD	Chemical oxygen demand
DO	Dissolved oxygen
EPA	Environmental Protection Agency
N	Nitrogen
MGD	Million gallons per day
NOD	Nitrogenous oxygen demand
OP	Orthophosphate
ORP	Oxidation-reduction potential
P	Phosphorus
SOD	Sediment oxygen demand
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen
TP	Total phosphorus
UCT	University of Cape Town

ACKNOWLEDGEMENT

The initial drafting of this document was performed primarily by Clifford W. Randall, Louis Sage, and Maurice Lynch. Much of the information and data for the section on Nutrient Limitation of Growth in Aquatic Systems was taken from publications by Christopher D'Elia, and his contributions deserve special recognition. Patricia Furey and Karen McDonald merit expressions of gratitude for their efforts in the compilation, preparation and final editing of this paper. In addition, the contributions of all of those who attended the two workshop meetings are gratefully acknowledged.

TABLE OF CONTENTS

INTRODUCTION

A principle conclusion of the Environmental Protection Agency Chesapeake Bay Program Studies (U.S. EPA, 1982) was that deteriorating water quality in the Chesapeake was directly correlated with increased nutrient enrichment over the past few decades. Water quality deterioration was considered to have major detrimental effects on a number of living resources including submerged aquatic vegetation, shellfish and finfish. The technical synthesis concluded that:

> "management strategies to address these problem areas must take into account the seasonal patterns of nitrogen and phosphorus we have described and the degree to which each contributing source may be controlled..." (U.S. EPA, 1982).

The point source nutrient control strategies developed both prior to and subsequent to the Bay Program research findings have, however, only focused on phosphorus.

Nitrogen removal has not been considered, primarily for these reasons:

1. Most of the scientific information on the need for nutrient controls is based upon freshwater studies which have clearly demonstrated that phosphorus removal is an effective freshwater nutrient control strategy.

2. Conventional engineering wisdom has considered that nitrogen removal is infeasible from an economic and available technology perspective.

3. EPA policies since the 1977 Clean Water Act amendments have discouraged serious consideration of nitrogen removal by the states by failing to provide construction grant cost-sharing for multiple nutrient removal or by requiring extraordinary review and approval of nitrogen removal as a sole strategy.

As will be explained further in this paper, recent studies have shown that with the variation in salinity in Chesapeake Bay waters, both nitrogen and phosphorus play critical roles in nutrient enrichment. Both must be included in an effective nutrient control strategy for the Bay. The emphasis on the particular nutrient of primary concern will depend upon the time of the year of a release and the segment of the estuary most directly affected by the discharge and delivery. In addition, advances in nutrient removal practices in other countries have demonstrated that nitrogen removal can now be considered as a viable economic and technical strategy in nutrient control.

The total annual nutrient loading to various portions of Chesapeake Bay was estimated during the EPA study. On the basis of these estimates, overall point and nonpoint source nutrient strategies have been proposed. However, annual loadings alone are inadequate for the development of comprehensive strategies and short-term "seasonal" dynamics also should be considered. For example, there is little runoff during the primary algal growing season and, consequently, the principal inputs of nutrients during that period are from sewage treatment plant discharges and groundwater inflow. Because groundwater inflow is very difficult to control, although it eventually can be reduced by changes in agricultural practices, initial strategy must center on reducing nutrient inputs from wastewater treatment plants if significant reductions are to be accomplished.

Figure 1 provides an estimate of the loadings of nitrogen and phosphorus entering various segments of Chesapeake Bay from municipal sewage treatment plants during the eight month algal growing season. Examination of these estimates indicates the removal of nitrogen and phosphorus from these sources can have a significant impact on total nutrient loading to the Bay. For example, reasonably economical technology would reduce both the nitrogen and the phosphorus discharged by municipal treatment plants by about 75 percent. This assumes effluent limitations of 6 mg/L total nitrogen and 2 mg/L total phosphorus. This would very substantially reduce the amounts of nutrient entering the Bay during the height of the algal growing season.

As spokespersons for the research community, the Scientific and Technical Advisory Committee to the Chesapeake Bay Program, believes it is vital that the importance of nitrogen control be brought to the attention of the policy making and management segments of the Bay community for use in developing effective nutrient control strategies.

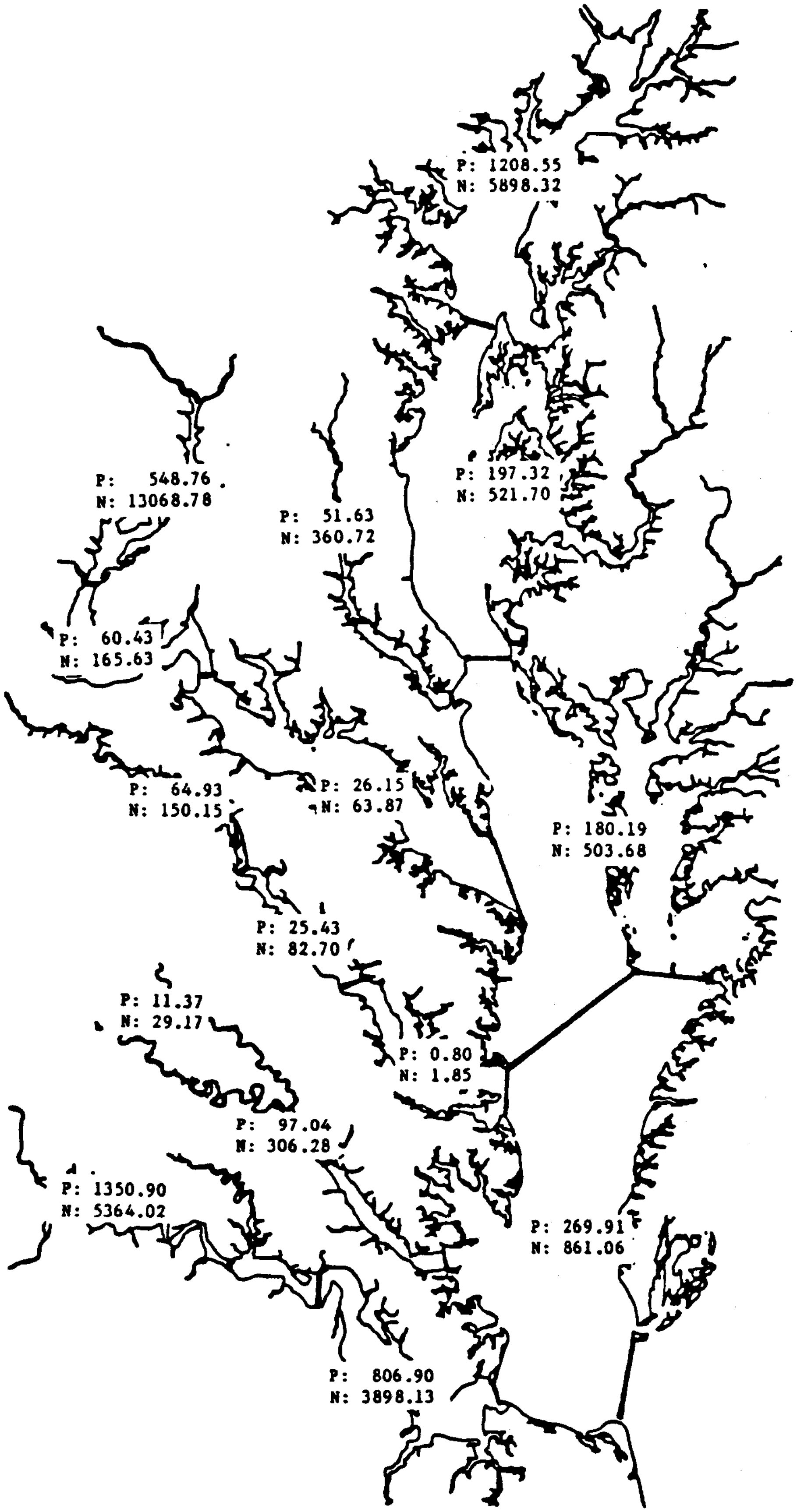

Figure 1: Total Phosphorus (P) and Nitrogen (N) Discharged From Major Municipal Wastewater Treatment Plants Below the Fall Line - 1983[1] (1000 lbs nutrient, 240 days/year).

[1]Combination of estimates and actual measurements of discharges. Data obtained from EPA Chesapeake Bay Program.

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NUTRIENT LIMITATION OF GROWTH IN AQUATIC SYSTEMS

Two nutrient elements (phosphorus and nitrogen) are considered to have the potential among the chemical parameters to limit primary productivity in aquatic systems. The paradigm regarding their roles as limiting nutrients is: in freshwaters, phosphorus (P) is typically the limiting nutrient, and in marine waters, nitrogen (N) is considered to be the limiting nutrient (D'Elia and Boynton, 1982). Those waters which range between the fresh and marine, namely the estuarine area, have not been fully categorized with experimental data as to the limiting nutrient (D'Elia and Bishop, 1985). Despite this, nutrient control strategies have traditionally favored P. Effective nutrient control is a prevalent concern in the regulatory bodies of many estuaries and coastal systems (Gray and Paasche, 1984; Price et al., 1985) and the Chesapeake Bay has not escaped the throes to strategize on the regulation of nutrient inputs (Heinle et al., 1980; Brush, 1984; Officer et al., 1984; Price et al., 1985).

The identification of a limiting nutrient within an estuary has been fraught with controversy in the absence of substantive data. The EPA Chesapeake Bay Program is now in the implementation phase to improve water quality and aquatic resources. An adequate nutrient control strategy that is based on "hard" data is critical at this juncture. The bulk of data used to discuss a nutrient control strategy in the past has been based on indirect data. This section is intended to review the literature for relevant information as well as to present site-specific data also relevant to formulating a nutrient control strategy.

When nutrients are introduced into an estuary in excessive amounts detrimental effects may result. The growth rate of phytoplankton may be enhanced which results in greater densities of plankton cells. These densities may be unesthetic and, more importantly, adversely affect the oxygen concentration within the water column, depleting it in the bottom waters to the point of hypoxia or in the extreme, anoxia. The overriding objective in developing a nutrient strategy is to control these algal densities to the point where they do not adversely affect the dissolved oxygen concentrations. The relationship between phytoplankton growth and oxygen depletion is fairly direct and well-documented. The area that needs better understanding is the response of phytoplankton to concentrations of the nutrients. High concentrations of a nutrient do not always result in higher productivity of phytoplankton and ambient nutrient levels do not always correlate with phytoplankton biomass as determined by chlorophyll-a levels. In addition, our understanding of the movements and transformations of nutrients in an estuary is poorly known and at best is based on indirect data. Direct measurement of phytoplankton response to varied nutrient concentrations and environmental conditions, is essential.

The overriding concern in formulating an effective nutrient control strategy is determining which nutrient to eliminate in which portions of the Chesapeake estuary during which seasons. The best available data prior to 1984 (derived from indirect sources) indicated that phosphorus is the limiting nutrient in the main stem of the bay, particularly during the spring and fall (U.S. EPA, 1982). Nitrogen was considered to be the limiting nutrient during most summer months. During those times when nutrients are not the limiting factor, light can be the dominant consideration regulating the phytoplankton standing crop. The summary document for EPA indicates that in winter, light or phosphorus can be the limiting factor. At the same time, conventional theory has suggested that primary productivity is typically P-limited in freshwaters (Schelske and Stoermer, 1972; Schindler, 1977, 1981) and N-limited in marine waters (Ryther and Dunstan, 1971; Thomas et al., 1974; Bishop et al., 1984). Since an estuary represents the transition zone between these two extremes great uncertainty and controversy has raged over whether to reduce P or N or some combination of both to achieve reduced algal production.

There are four categories of approaches which have been used to assess nutrient limitation of phytoplankton growth in natural waters:
(1) "physiological" in which some metabolic characteristic is assayed to indicate nutrient limitation; (2) "mathematical", in which a statistical (Vollenweider, 1976) or dynamic numerical model (O'Connor et al., 1977, 1981) is used to evaluate presumed response to nutrient loading: (3) "stoichiometric" in which the relative availability of inorganic nutrients in loadings (Jaworski et al., 1972; Jaworski, 1981), or in dissolved or particular standing stocks in the water are taken to indicate nutrient limitation or repletion, and (4) "growth bioassay", in which changes in plant biomass are evaluated when nutrients are added to phytoplankton cultures (Ryther and Dunstan, 1971; Maestrini et al., 1984a,b), enclosed natural phytoplankton assemblages (reviewed by Schelske, 1984), or entire ecosystems (Schindler and Fee, 1974).

Although most regulatory and resource management agencies have relied heavily on dynamic numerical water quality models, Maestrini et al., (1984a,b) and Schelske (1984) have emphasized that the growth-bioassay approach provides the best evidence for determining nutrient limitations of primary productivity in natural waters. Previous enrichment studies have been conducted in well-mixed estuaries (Type "A" by the classification of Pritchard (1955)) characterized by relatively low inputs of freshwater. In one of these type A estuaries, Narragansett Bay, Smayda (1974) found evidence that nitrogen, not phosphorus was limiting, as might be expected for coastal waters (Ryther and Dunstan, 1971).

Type "B" estuaries, (such as Chesapeake Bay) exhibit much stronger seasonal variations in freshwater flow and nutrient regimes, implying seasonal variations in nutrient limitations (Boynton et al., 1982). The Bay and its major tributaries are characterized by large winter/spring inputs of fresh water containing a surplus of N relative to P compared to that of typical phytoplankton needs (Redfield, 1934). During the late summer, low-flow season, sediment and water column recycling play a proportionately larger role in supplying nutrients (Kemp and Boynton, 1984). Nutrients derived from sediments are quantitatively most important during that season and have a surplus of P to N relative to typical phytoplanktonic needs (Kemp and Boynton, 1984).

Patuxent River Growth Bioassay Study

In an attempt to address the void of information on phytoplankton response to varying concentrations of the critical nutrients, a joint program was started with the Academy of Natural Sciences at the Benedict Estuarine Research facility and the University of Maryland at their Chesapeake Biological Laboratory. The Benedict facility was chosen as the experimental site primarily because of its proximity to that reach of the Patuxent River where the greatest oxygen sag was measured (D'Elia, Sanders and Boynton, 1985). Nutrient enrichment studies were conducted in large-scale continuous cultures exposed to natural sunlight. These cultures were fed a continuous supply of nitrogen (either as ammonium or as nitrate) and phosphorus. The cultures were inoculated with natural phytoplankton from the Patuxent River estuary. These experiments were conducted year-round starting in 1982 through the present. Approximately 15 experiments with treatments replicated in triplicate have been performed.

The results of this study (Figures 2, 3, and 4) indicate that the natural phytoplankton have greatly enhanced growth rates (4-10 fold above the control) by supplements of nitrogen, in either the ammonium or nitrate form, during the summer season. Especially enhanced growth was measured during the low-flow late summer season; a period when N:P ratios of dissolved inorganic nutrient standing stocks are characteristically below 5:1 (by atoms). Growth response to the nitrogen addition was very rapid, accelerating within one day after the start of the experiment. This implies that the phytoplankton in the bioassay were nitrogen limited when removed from the estuary.

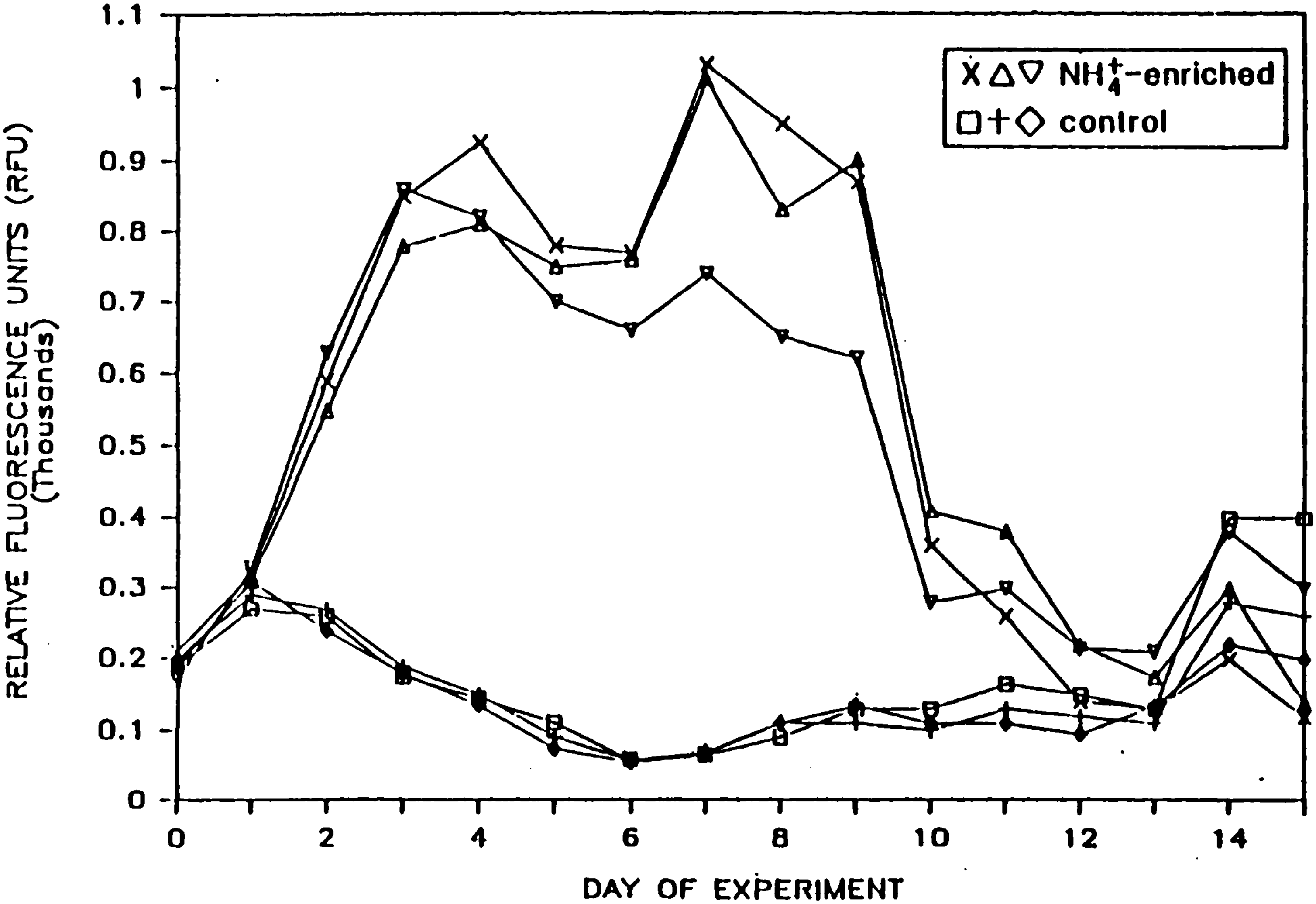
Phosphate (PO_4^{3-}) addition enhanced phytoplankton growth during the late winter, high flow season when the N:P ratios typically exceed 90:1 (Figure 5). The biomass increase was less than one-third that achieved in the nitrogen enriched cultures during the late summer. In addition the response time of phytoplankton to the P-enrichment lagged by at least 4 days.

The results of these studies have been consistent through 3 years despite a salinity range from 0-14 parts per thousand. Although the experimentation was at one site, the range of salinities was broad and the replicability of response seasonally through 3 years lends credence to the growing awareness in the scientific community that nitrogen control has to be a critical component of any nutrient control strategy formulated for Chesapeake Bay.

Summary of Rationale for Nitrogen Limitation

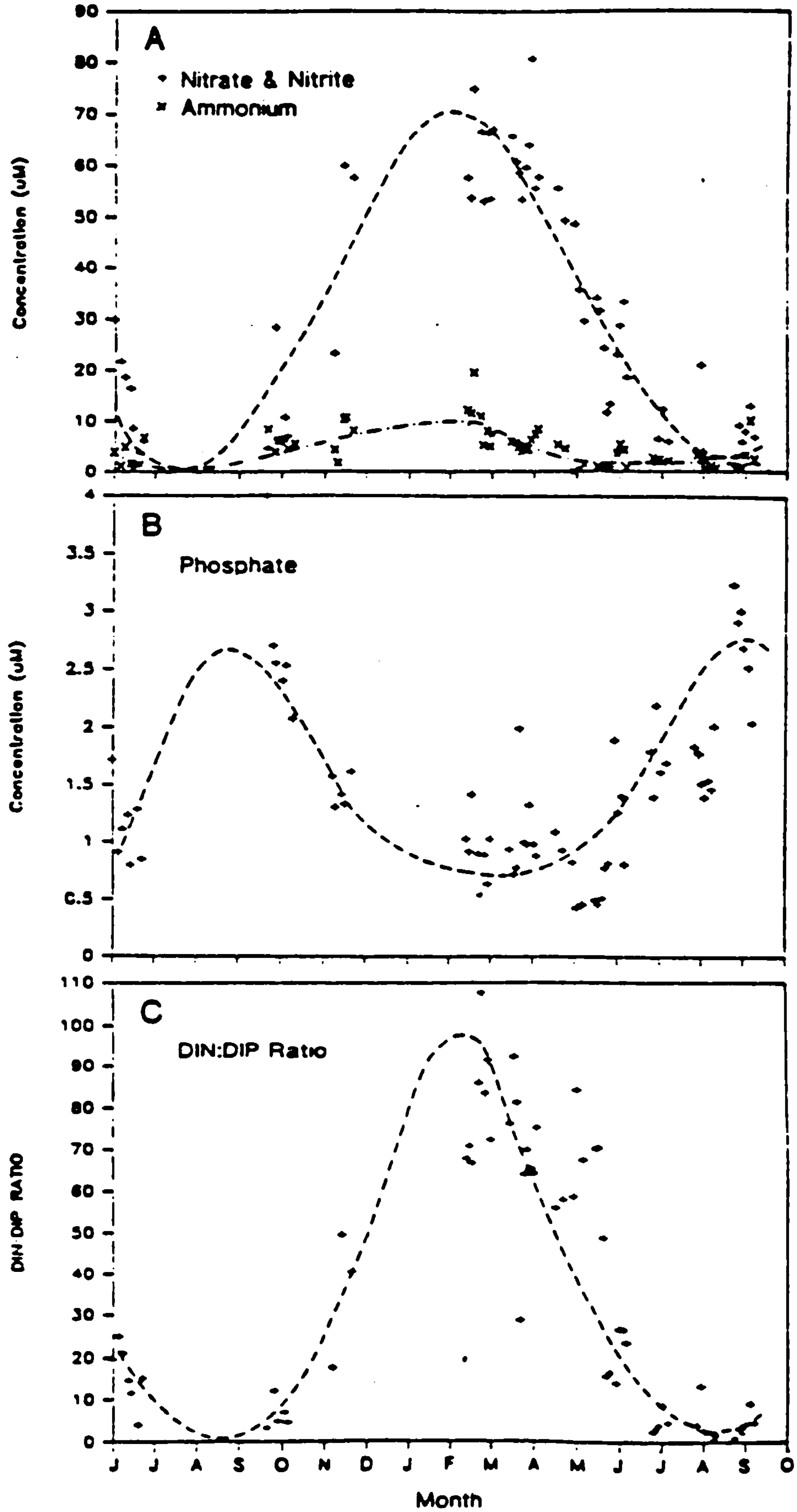
While so-called "conventional wisdom" may identify phosphorus as the most likely limiting nutrient in an estuarine environment, and this may appear to be true for the Chesapeake Bay based on estimated annual loadings of nutrients, a closer look at the available information reveals that the chemical properties of the nutrients plus seasonal conditions determine which nutrient is limiting at any given time. Analysis of the data

Figure 2: Experimental replicability--biomass (expressed as *in vivo* relative chlorophyll fluorescence, "RFU") changes during replicate NH_4^+-enriched and control (unenriched) treatments in an experiment of August 1984.



(D'Ella, Sanders and Boynton, 1985)

Figure 3: Nutrient concentrations and ratios in unenriched dilution water to the cultures vs. month, 1983-1984. A. Nitrate & nitrite (summed) and ammonium. B. Phosphate. C. Dissolved inorganic nitrogen (DIN): dissolved inorganic phosphorus (DIP) ratios. All curves were fitted by eye.



A
Nitrate & Nitrite
Ammonium
Concentration (uM)
B
Phosphate
Concentration (uM)
C
DIN:DIP Ratio
DIN:DIP RATIO
Month

Figure 4: Immediate and strong response to nitrogen--biomass (expressed as *in vivo* relative chlorophyll fluorescence, "RFU") changes during a "standard" experiment in August 1984. A. RFU (mean of triplicate tanks and unfiltered Patuxent River water) vs. day after start of experiment. B. Ratio of mean RFU of triplicate experimental tanks to mean of triplicate unenriched control tanks vs. day after start of experiment. C. Ratio of mean RFU of triplicate PO_4^{3-}- enriched experimental tanks to mean of triplicate NO_3^-- enriched and NH_4^+- enriched tanks on a given day. The greater the range in the x or y direction, the greater the nutrient enrichment potential for the nutrient. "River" = unfiltered Patuxent River water, and "control" = unenriched tanks.

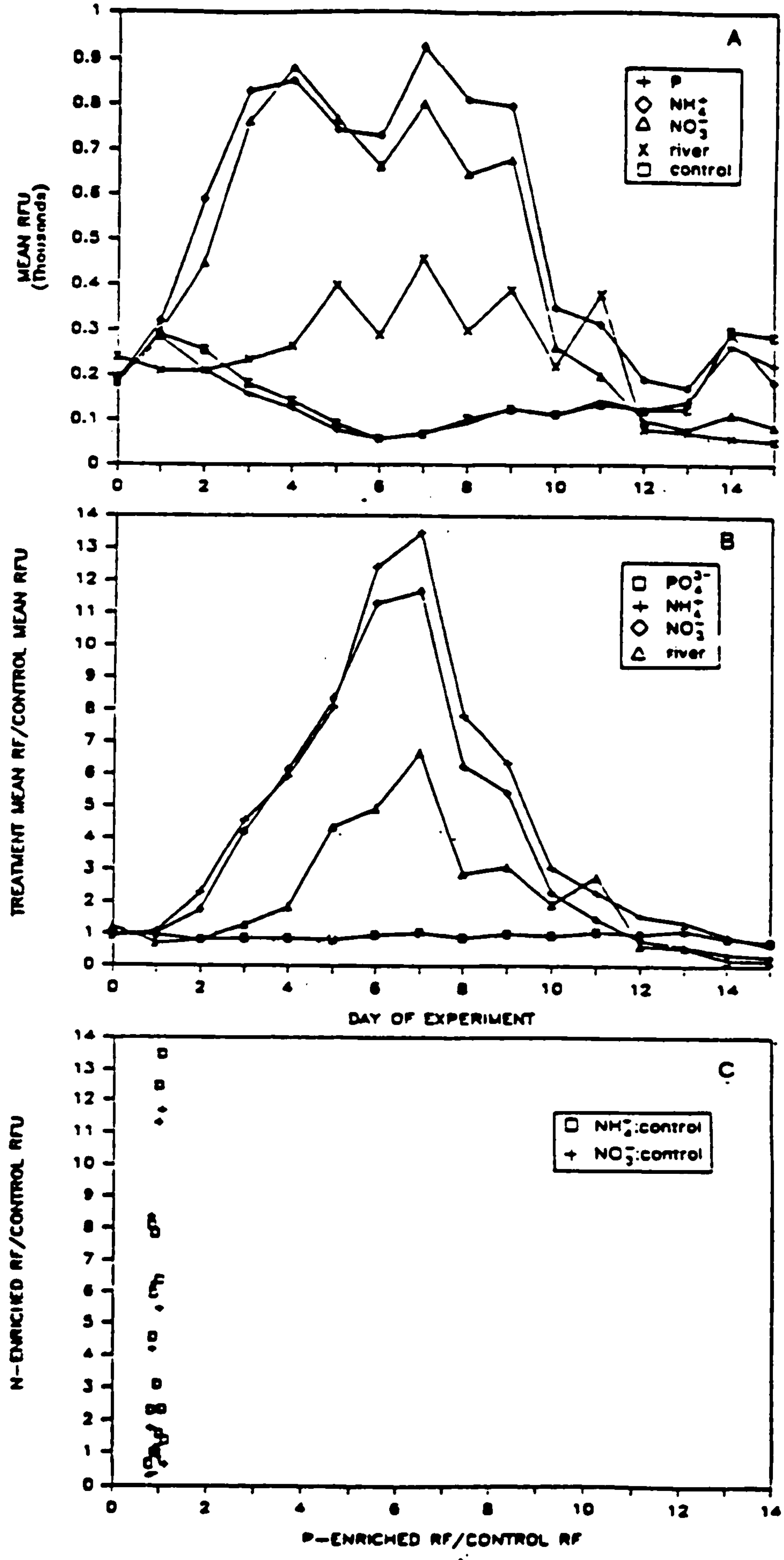
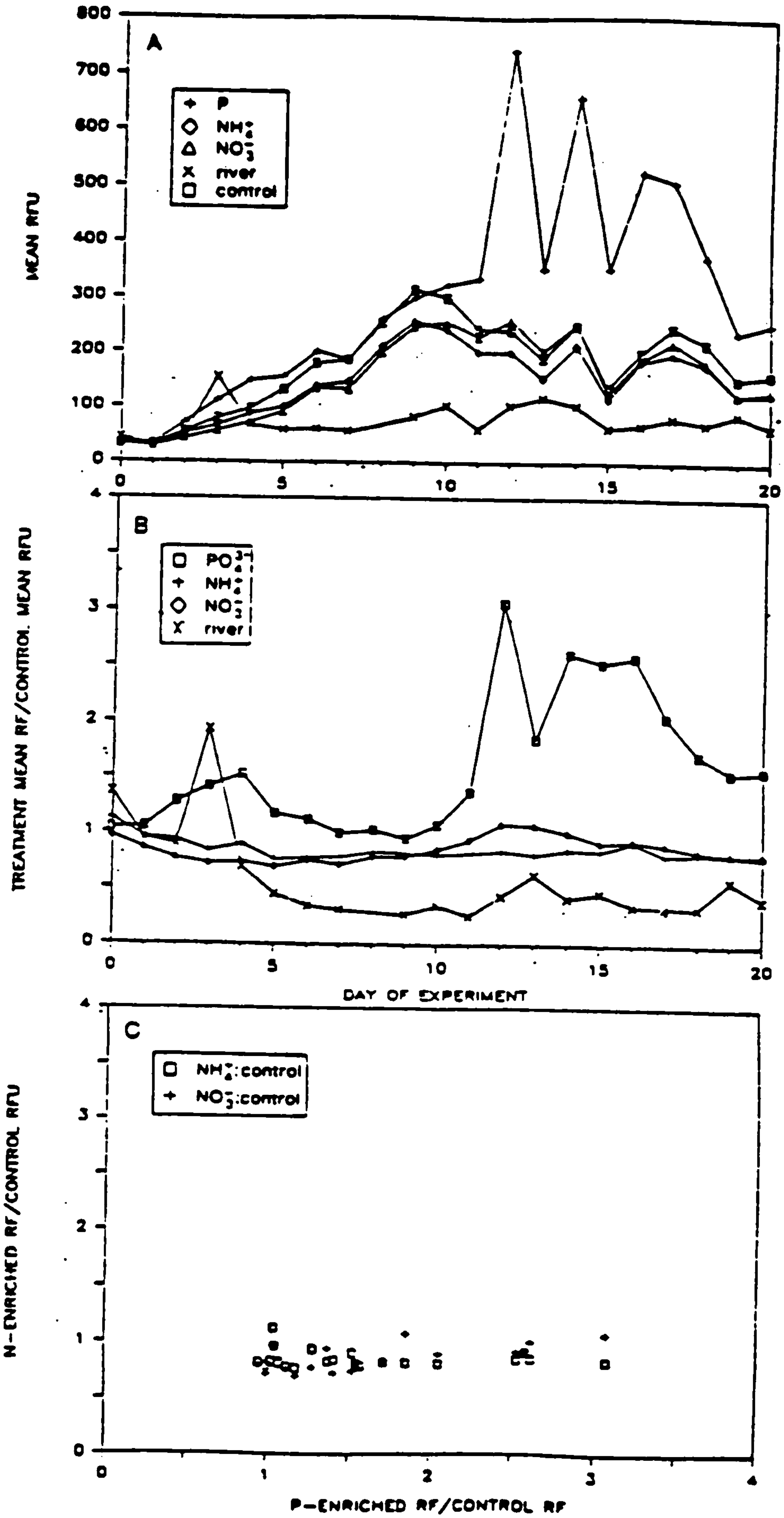

A
P
NH_4^+
NO_3^-
river
control
MEAN RFU (Thousands)
B
PO_4^{3-}
NH_4^+
NO_3^-
river
TREATMENT MEAN RF/CONTROL MEAN RFU
DAY OF EXPERIMENT
C
NH_4^+:control
NO_3^-:control
N-ENRICHED RF/CONTROL RFU
P-ENRICHED RF/CONTROL RF

Figure 5: Delayed and weak response to phosphorus for an experiment of February, 1984. Treatments and abbreviations as in Figure 4.



A
P
NH4+
NO3-
river
control
MEAN RFU
B
PO4 3-
NH4+
NO3-
river
TREATMENT MEAN RF/CONTROL MEAN RFU
DAY OF EXPERIMENT
C
NH4+:control
NO3-:control
N-ENRICHED RF/CONTROL RFU
P-ENRICHED RF/CONTROL RF

indicates that algal productivity during the primary growing season is controlled by short-term dynamics, rather than yearly nutrient mass balances. While very large masses of nitrate nitrogen enter the Chesapeake Bay system during the winter months, the nitrate is consumed by the sediment oxygen demand (SOD) with the onset of warm weather (SOD first depletes the oxygen, then the nitrates), and is converted to nitrogen gas (a process called denitrification) in which form it leaves the water and becomes part of the atmosphere. The depletion of nitrate causes a decrease in the oxidation-reduction potential (ORP) of the top sediment layers which results in the solubilization of the iron-bound phosphorus. The phosphorus is released to the overlying waters and mixing causes phosphorus enrichment of the entire water column. Thus, the nitrogen content of the water column has been depleted by denitrification by the peak growing season. Although some ammonia is released from the sediments when low ORP occurs, the quantity is small relative to the amount of phosphorus released, and the water column becomes very nitrogen sensitive with respect to algal growth requirements. Therefore, control strategy during the primary growing season should emphasize nitrogen control. Because there is little runoff during the growing season, the principal inputs of nitrogen are from sewage treatment plant discharges and groundwater inflow. The groundwater inflow of nitrogen can be adequately controlled only through widespread changes in agricultural practices for a long period of time. Therefore, immediate control strategy must be centered on reducing nitrogen inputs from wastewater treatment plants.

POINT-SOURCE POLLUTION CONTROL STRATEGY

The available information reviewed in the preceding section indicates that the best way to achieve water quality improvements in the saline portion of the Chesapeake Bay in the immediate future is to reduce the amounts of nitrogen entering the Bay system during the growing season. It seems especially prudent to reduce the quantities of ammonia-nitrogen entering the system because of the multiple effects they have on water quality, i.e., oxygen depletion, potential toxicity, and growth enhancement.

Wastewater treatment plant discharges, the primary source of ammonia-nitrogen at all times, and of total nitrogen during the algal growing season, are the most easily controllable sources of nitrogen entering the Bay. Control strategy, then, seemingly dictates that the reduction of point sources of nitrogen be given top priority.

The role of phosphorus in promoting algal growth, particularly in the tidal freshwater portions of the Bay, should not be overlooked, however. Also, the release of phosphorus from the Bay sediments under anaerobic conditions accentuates the nitrogen limitation effects during the growing season and adds to the overall enrichment of the system. Comprehensive long-term control strategy should include reduction of both point and non-point sources of phosphorus.

Historically, wastewater treatment engineers in the USA have relied on inorganic chemical precipitation for phosphorus removal, and "two sludge" systems incorporating methanol addition for nitrogen removal. Chemical precipitation of phosphorus is very reliable and can achieve the desired effluent quality, but it increases the cost of wastewater treatment by adding the cost of the chemicals and substantially increasing waste sludge disposal costs. The capital cost of "two sludge" systems is exceptionally high and the cost of methanol greatly increases operating costs. Consequently, regulatory agencies historically have been reluctant to impose phosphorus limitations, and nitrogen limits have been imposed under only the most extreme conditions.

Fortuitously for the Chesapeake Bay situation, recent developments in activated sludge wastewater treatment technology have provided sufficient information for the design and operation of treatment systems that utilize biological nutrient removal processes to achieve both nitrogen and phosphorus removal simultaneously with Biological Oxygen Demand (BOD) removal (See Appendix A). Furthermore, these processes can be incorporated into new plants for very little, if any, increase in cost over that required for BOD removal alone, and can be added to existing plants for a small fraction of the cost of the original plants. For example, preliminary engineering design evaluation for the upgrading of the 40 MGD Lambert's Point Primary Treatment Plant in Norfolk, Virginia to secondary treatment, concluded that the construction of a system that would remove nitrogen and phosphorus in addition to BOD would be within 10% of the costs of a system removing only BOD. Also, the Pontiac, Michigan 3.5 MGD East

Boulevard Plant was converted from a facility that removed only BOD biologically and phosphorus by iron salts addition, to a facility that removes both BOD and phosphorus biologically and nitrifies seasonally, for a cost of only $50,000. The new facility consistently removes phosphorus to concentrations well below 1.0 mg/L. A proposal was recently submitted to the Virginia State Water Control Board by the Hampton Roads Sanitation District for the retrofit and operation of their 7 MGD York River Plant for nitrogen, phosphorus, and BOD removal, and the projected conversion cost was only $137,000.

Biological nutrient removal processes are inherently more energy efficient than purely aerobic BOD removal processes, in terms of aeration requirements, and systems incorporating biological nitrogen and phosphorus removal can potentially be operated with greater energy efficiency than systems removing BOD alone. In fact, systems that accomplish nitrification in addition to BOD removal can be operated for 20 to 40% less energy by conversion to nitrogen and phosphorus removal. For example, Best, et al. (1984) converted a nitrifying activated sludge plant operated by the Thames Water Authority to nitrogen removal by denitrification and reduced the total energy costs of the plant by 17% over a 12 month period. Randall et al. (1985) have shown that biological phosphorus removal can reasonably reduce the operating energy costs by an additional 20%.

Thus, technology is available for the reduction of nitrogen and phosphorus discharges simultaneously with BOD reduction at little or no increase in cost compared to plants that remove BOD only, and do not nitrify. It would be reasonable to take advantage of such treatment improvements under any circumstances, but it is particularly appropriate to implement this technology throughout the Chesapeake Bay area considering the critical needs and the potential benefits.

Although there are more than 50 full-scale wastewater treatment plants in nine different countries around the world where nitrogen and phosphorus removal has been, and continues to be, accomplished simultaneously with BOD removal, it is recognized that there has been a lack of reliable information about the processes, economics, and performances of the plants, and that historical circumstances have predisposed the engineering community to skepticism, particularly in regard to excess biological phosphorus removal. Appendix B presents the background necessary for an understanding of the biological nutrient removal processes and includes discussions of observed performances at the full-scale plants.

Other treatment techniques are also available for the removal of nitrogen and phosphorus, and may be more economical to implement in some areas. For example, where land area is available spray irrigation can be used with considerable success, particularly for phosphorus removal. The disposal of municipal sewage by spray irrigation on forested and pasture lands has been extensively studied at Pennsylvannia State University. Design specifics are available from their Institute on Land and Water Resources. Also, an extensive land disposal project at Muskegon, Michigan has been in operation for more than 10 years and now serves 18 municipalities and five large industries. Extensive monitoring has demonstrated that land disposal projects of this magnitude can be safely accomplished.

Land disposal of wastewater is excellent for phosphorus removal if runoff and erosion are controlled, but nitrogen control is more difficult. The phosphorus is adsorbed and held by the soil, but nitrogen is usually oxidized to nitrate, which is very soluble, and eventually migrates to the groundwater, and ultimately reaches a stream, lake, or estuary. To be removed, the nitrogen must be incorporated into a growing plant which is harvested, or converted to nitrogen gas by denitrification. High groundwater tables can result in zones of denitrification, and these conditions commonly occur in wetlands. Thus, artificial wetlands have been suggested for the denitrification of both wastewater effluents and stormwater runoff. Further investigation of this approach is needed before widespread implementation can take place.

The principal purpose of this discussion is to point out that economical methods exist for the removal of nitrogen and phosphorus from wastewaters. The relatively new technology of biological nutrient removal simultaneous with BOD removal has been emphasized because most point sources in the Bay area are currently being treated by wastewater treatment plants. However, present treatment is inadequate for the water quality requirements of the Bay, and must be improved. It is believed that implementation of recently developed biological nutrient removal wastewater treatment systems provides a way of economically achieving the desired environmental benefits.

The wastewater treatment plant design options for nutrient removal are compared in Table 1. The relative costs and energy needs are based on information provided by the references given for this paper and the evaluations mentioned in the text. The comparison is clearly very favorable to a complete biological nutrient removal system when the environmental benefits are considered.

It requires specific amounts of BOD to obtain unit removals of either nitrogen or phosphorus. Consequently, it may not be possible to meet the desired effluent limits for both nutrients when the wastewater is low in organic strength relative to its contents of nitrogen and phosphorus. However, the system can be operated to maximize the removal of one nutrient, and chemical addition can be used to complete removal of the other. Regardless, prior biological nutrient removal would substantially reduce any chemical costs and/or waste sludge production.

TABLE 1. COMPARISON OF WASTEWATER TREATMENT OPTIONS FOR NUTRIENT REMOVAL

Process	Relative Capital Cost	Relative Energy Needs	Relative Chemical Needs	Relative Waste Sludge Production	Removals Achieved BOD	NOD	Nitrogen	Phosphorus
Conventional	1.0	1.0	none	1.0	X			
w/Nitrification	1.2	1.55	0 or L*	<1.0	X	X		
W/Two Stage N Removal(Methanol)	2.1	1.75	H	1.5	X	X	X	
w/One Stage N Removal (Influent BOD)	1.1	1.25	none	<1.0	X	X	X	
w/Chemical P Removal								
Simultaneous	1.2	1.0 to 1.55**	M	1.2 to 1.7***	X	V		X
Tertiary	1.8	"	H	2.0	X	V		X
w/Biological P Removal	1.0	0.8	none	1.0	X	V		X
w/Biological Nutrient Removal (N & P)	1.1****	1.0	none	<1.0	X	X	X	X

Compares activated sludge treatment options, effluent filtration not considered
*If alkalinity is low, may require chemical addition for pH control and/or nitrification completion
**Variation depends upon extent of nitrification accomplished, i.e., operating sludge age chosen
***Variation depends upon whether or not primary sedimentation is practiced, 1.2 to 1.3 with primary sedimentation
****Preliminary Engineering Estimate for Lambert's Point Plant

L = Low M = Medium H = High X = Yes V = Variable

RECOMMENDATIONS

The amounts of nitrogen and phosphorus discharged in the effluents of the wastewater treatment plants in the Chesapeake Bay environs should be reduced. This would assist the effort to restore the quality of the Bay and would have beneficial effects on the aquatic habitats.

Wastewater treatment strategy in the Chesapeake Bay environs should implement nitrogen control in addition to phosphorus control. Nitrogen removal should be incorporated in all plans for plant upgradings, expansions, and new construction. Evidence indicates that removal is technically and economically feasible, based on the operation of more than 50 biological nutrient removal wastewater treatment plants in nine countries.

The potential for transmission of nutrient loads from the tidal freshwater portions to the saline main body of the Bay exists. There is strong evidence that nitrogen contributes significantly to algal production in the downstream estuarine portions. Future modelling efforts should focus on these transport processes and incorporate nutrient kinetics appropriate for each salinity regime. While these efforts can provide useful information regarding transport of nutrients, they are not adequate to distinguish between the relative importance of nitrogen and phosphorus.

It is believed that more measurements of rates and fluxes need to be made if we are to improve our understanding of nutrient transformations. The adequacy of the existing data base used in the modeling effort should be examined.

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Appendix A

A BRIEF HISTORY OF THE DEVELOPMENT OF BIOLOGICAL NUTRIENT REMOVAL SYSTEMS IN THE USA

The biological processes for the removal of nitrogen and phosphorus are generally considered to be too expensive and/or technically unfeasible by USA wastewater treatment professionals because of historical developments and disputes. An appreciation for the historical events is essential to an understanding of this resistance.

Nitrogen removal technology was studied in Europe during the early and mid-1960's but there was little activity in the United States until Professor McCarty of Stanford University began a series of definitive studies in the late sixties. However, he was concerned with the removal of nitrates from agricultural irrigation water, which contained almost no biodegradable organics, rather than sewage, and, therefore, he had to supply the biodegradable organics to achieve denitrification. He experimented with a wide variety of organics and concluded that methanol was the compound of choice based on economic and biochemical consideration.

Shortly after McCarty's studies, nitrogen removal from sewage became a matter of concern in some areas, and systems were devised to accomplish it biologically. Because McCarty's data were the most readily available, irrigation water treatment technology was applied to the treatment of sewage, resulting in plants that were unnecessarily complicated and very expensive to build and operate. Rather than using the sewage organics for denitrification, two separate plants including clarifiers were built in series, the first for the removal of sewage organics and nitrification, and the second, to accomplish denitrification with the addition of methanol. This design approach became accepted practice with the result that in a short period of time, biological nitrogen removal was considered to be too expensive to utilize except for extreme circumstances.

This series of developments overlooked the fact that denitrification using the sewage organics would actually reduce the total aeration requirements of a nitrifying activated sludge plant. Consequently, a technology that could have increased the environmental benefits of wastewater treatment while simultaneously reducing the operating costs was never implemented in the USA. However, this technology was widely implemented in both Europe and South Africa, and there is no technical or economic reason why it could not be widely implemented in the USA.

Excess biological phosphorus removal was first studied by Shapiro and Levin at The Johns Hopkins University in the early 1960's. This led to the development of the Phostrip process, which has been marketed since that time, but has had a series of only partially successful applications. Reasons for the partial failures have been both economic and technical. It is important to recognize, however, that Phostrip is a sidestream rather than a mainstream process, and that it utilizes a chemical addition step for the actual phosphorus removal. By sidestream is meant that the activated sludge is separated from the wastewater after organic removal and pumped to separate tanks for the phosphorus removal steps, then returned to the organic removal tank.

During the later 1960's, a few conventional activated sludge plants were observed to be removing phosphorus without chemical addition. The most celebrated of these, and the most thoroughly studied, was the Rilling Road Plant in San Antonio, Texas. Efforts to identify the mechanisms responsible were made and numerous laboratory studies by a variety of investigators were stimulated. However, very few of the studies were successful and those that were did not yield sufficient information for design and operation control.

Studies by Menar and Jenkins at the University of California, Berkeley, obtained high phosphorus removal, but the responsible mechanisms involved were identified as being chemical, not biological. Using their experimental results, Jenkins was able to explain the phosphorus removal at the Rilling Road plant on the basis of the chemical composition of the San Antonio water supply, which is entirely groundwater and high in calcium. His explanation was disputed and rejected by the San Antonio investigators, but Jenkins' argument was persuasive to the wastewater treatment profession. Consequently, research into biological phosphorus removal ceased in the USA and an entire generation of professionals were taught that excess biological phosphorus removal was biochemically impossible.

Ironically, biological nitrogen removal studies in the early seventies by James Barnard, a South African native doing doctoral work at the University of Texas, led to the ultimate revival of excess biological phosphorus removal research. In a mainstream activated sludge system specifically designed to remove nitrogen using the incoming sewage for denitrification, he observed that he was also removing excess phosphorus. He dubbed the system "Bardenpho", and upon his return to South Africa worked with the South African government to develop the system on a full-scale basis. He subsequently developed the Phoredox modification of the Bardenpho, which consisted of five reactors in series which had a hydraulic retention time of approximately 21 hours. It was capable of 90% BOD, nitrogen, and phosphorus removal without any chemical addition if properly operated.

The key to excess biological phosphorus removal proved to be anaerobic-aerobic sequencing of reactors. This provided the conditions under which bacteria that could remove, store, and utilize excess amounts of phosphorus could flourish. The importance of true anaerobic conditions in the first reactor, i.e., a redox potential of at least minus 400 mv, was not initially recognized and this led to unstable performance at a majority of the full-scale plants designed for nitrogen and phosphorus removal. The need for true anaerobic conditions was first discovered by Marais and his co-workers at the University of Cape Town, and led to modifications of the Bardenpho-Phoredox system, which are called the UCT and the modified UCT processes. Marais also simplified the system from five to three reactors. Subsequent experience has shown that systems designed to prevent the feedback of nitrates to the anaerobic reactor will operate consistently with high phosphorus removal.

Today the principle of excess biological phosphorus removal is widely accepted worldwide. Ironically, Professor Jenkins of U. C. Berkeley is the current Chairman of an international group formed specifically for the purpose of coordinating and disseminating research results on biological phosphorus removal.

A somewhat parallel development of an excess biological phosphorus removal system also occurred in the USA. The Anaerobic/Oxic (A/O) system was patented by Air Products, Inc., based on research led by S. N. Hong. This system was originally designed to remove only phosphorus and was operated at a low sludge age and short (6 to 8 hours) hydraulic retention time. A later modification to remove nitrogen as well is known as the A^2/O process (Anaerobic-Anoxic/oxic).

More recently, it has been shown in England and France that existing conventional activated sludge plants can be easily and economically modified to achieve both nitrogen and phosphorus removal. It is particularly important to note that the removals can be accomplished with wastewater hydraulic retention times of 6 to 10 hours, which makes the systems economical from a capital cost standpoint, and that conversion results in about a 20% savings in aeration energy costs. These same principles can be used to design and operate new plants.

When it is considered that the environmental benefits of nitrogen and phosphorus removal along with BOD removal can be obtained at approximately the same cost as BOD removal alone, and that waste sludge production will not be increased as it is with chemical phosphorus removal, it is clear that biological nutrient removal systems should be used for wastewater treatment under all but the most unusual circumstances. It is likely that additional operator training will be required for successful implementation, but the price is a small one to pay for the potential environmental benefits.

Appendix B

BIOLOGICAL NUTRIENT REMOVAL PROCESSES

AND OBSERVED PERFORMANCES

Biological Nitrogen Removal

Biological nitrogen removal consists of the separate reactions of nitrification and denitrification. These processes are well known and thoroughly documented in the literature. Nitrification is a two step autotrophic bacterial reaction that is generally described by the following equations:

$$2NH_4^+ + 3O_2 \xrightarrow{\text{Nitrosomonas}} 2NO_2^- + 4H^+ + 2H_2O \qquad [1]$$

$$2NO_2^- + O_2 \xrightarrow{\text{Nitrobacter}} 2NO_3 \qquad [2]$$

$$\text{Overall:} \quad NH_4^+ + 2O_2 \xrightarrow{\text{Nitrifiers}} NO_3^- + 2H^+ + H_2O \qquad [3]$$

During the oxidation of ammonium to nitrate by the bacteria, a total of eight electrons are transferred and accepted by oxygen. This requires 4.57 pounds of oxygen for each pound of ammonium oxidized (the nitrogenous oxygen demand - NOD).

Except for very high rate activated sludge systems, some nitrification nearly always takes place during the biological treatment of municipal wastewaters. The extent of nitrification will vary considerably throughout the year depending on the temperature with minimum activity during the winter months unless operations are adjusted to maintain it. In recent years, effluent standards in many areas have been amended to include limits for the quantity of unoxidized nitrogen that can be discharged in an effort to protect the oxygen resources of the receiving body of water. Complete nitrification is now standard treatment at many facilities although it is not in most areas of the Chesapeake Bay, particularly Virginia. This has resulted in substantially increased oxygen requirements, and therefore energy costs, of biological wastewater treatment. Specifically, complete nitrification will generally increase the costs of aeration by 50 to 60%.

The ability of many bacteria to use the end products of nitrification, i.e., nitrite and nitrate, as electron acceptors in place of dissolved oxygen during the metabolism of organic compounds is also well-known and thoroughly documented in the literature. The process is known as denitrification and can be described by the following equations, which were developed by McCarty (1969) using methanol (CH_3OH) as the organic substrate:

$$6NO_3^- + 2CH_3OH \longrightarrow 6NO_2^- + 2CO_2 + 4H_2O \quad [4]$$

$$6NO_2^- + 3CH_3OH \longrightarrow 3N_2 + 3CO_2 + 3H_2O + 6OH^- \quad [5]$$

$$\text{Overall:} \quad 6NO_3^- + 5CH_3OH \longrightarrow 3N_2 + 5CO_2 + 7H_2O + 6OH^- \quad [6]$$

As the equations show, complete denitrification reduces nitrate to nitrogen gas, which becomes part of the atmosphere and is no longer a pollutant, and adds alkalinity (OH^-) to the water, thereby replacing part of that destroyed during nitrification.

The total number of electrons transferred during the reduction of nitrate to elemental nitrogen gas is five. Considering that eight were transferred during the oxidation of ammonia to nitrate using dissolved oxygen, the oxygen equivalence recovered from the use of nitrate as an electron acceptor for the stabilization of organic matter is 5/8(100) = 62.5%. That is to say, whereas oxygen had to be supplied for nitrification, the nitrate formed can be used to stabilize organic compounds during denitrification and reduce the amount of oxygen needed for subsequent organic (BOD) stabilization if the BOD of the influent wastewater is used instead of methanol. Although 62.5% of the NOD can theoretically be recovered, cellular nitrogen requirements by the denitrifiers during growth reduces the actual recovery to approximately 50% (van Handel, et al., 1981).

Considering oxygen "recovery" through denitrification (when influent wastewater is used as the organic carbon energy source instead of methanol) the economics shift sharply in favor of denitrification. Not only is the cost of methanol eliminated, but a substantial fraction of the influent wastewater is stabilized which reduces the amount of oxygen that must be supplied for BOD stabilization. The difficulty encountered in attempting to apply denitrification in conventional wastewater treatment systems is that with standard flow patterns, unless organic compounds needed for denitrification are present, nitrate and nitrite are absent. When nitrate is present, the necessary organic compounds are present in insufficient quantity.

Fortunately, the economies of denitrification can be obtained by recycling nitrates to an anoxic (without oxygen) reactor that precedes the aerated reactor of an activated sludge system. The Bardenpho system (Figure 1) was specifically designed for this purpose and has been thoroughly demonstrated on a full-scale basis. An oxidation ditch (race-track) configuration is even more ideal if the influent wastewater enters at the proper location because it eliminates the need for recycle pumping (Figure 2). Utilization of denitrification systems for the treatment of municipal wastewater will typically decrease the overall energy costs by 15 to 25%.

Stated another way, once nitrification has been accomplished, if denitrification is <u>not</u> implemented, a 15 to 25% reduction in energy costs

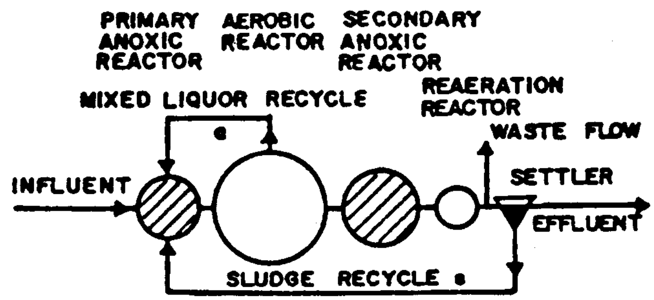

Figure 1. The Bardenpho process for biological nitrogen removal.

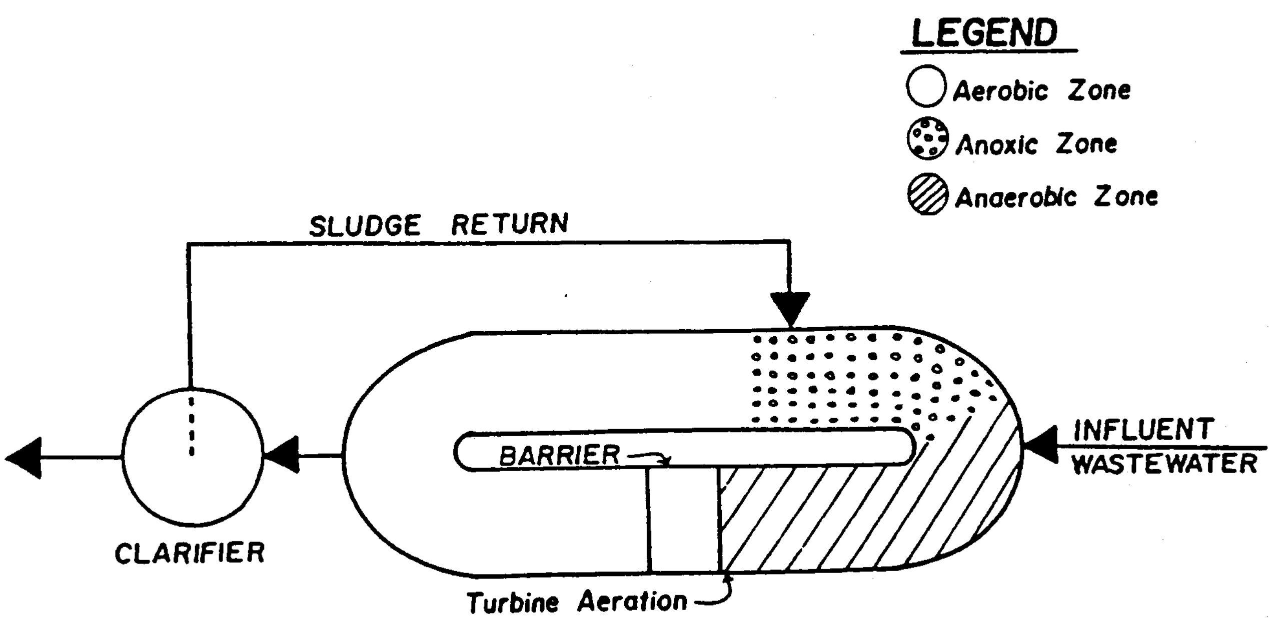

Figure 2. Oxidation Ditch design for nitrogen removal.

is wasted and the system is unnecessarily costly to operate. For many industrial wastewaters the potential savings resulting from denitrification would be much greater.

Biological Phosphorus Removal

Compared to denitrification, the potential reduction in aeration costs that can be accomplished by biological phosphorus removal reactions is less-well known, but experimental results at Virginia Polytechnic Institute and State University (Randall, et al., 1985) indicate that the savings may be substantial (on the order of 20 to 30%).

The key to biological phosphorus removal is the linkage of anaerobic and aerobic units in the same activated sludge system. The anaerobic unit must receive the influent wastewater flow and the activated sludge must be exposed to true anaerobic conditions, i.e., negative oxidation reduction potential (ORP) of less than -200 mv as measured by a silver chloride electrode, for a significant period of time prior to exposure to highly aerobic conditions. This shifts the growth advantage to the phosphorus removing (poly-P accumulating) bacteria. When these conditions are met, fermentation of the influent BOD occurs in the anaerobic unit and the fermentation products are immediately complexed and stored by the poly-P bacteria using energy from adenosine triphosphate (ATP) bonds previously formed under aerobic conditions. Phosphates are released to solution during this reaction as the ATP is reduced to adenosine diphosphate (ADP).

The ability to anaerobically store BOD gives the poly-P bacteria a substantial advantage over the other aerobic organisms in an anaerobic-aerobic system, because they remove the most readily biodegradable BOD during passage through the anaerobic unit, which makes it unavailable to the other aerobes. Upon entering the aerobic unit, the poly-P bacteria use the stored BOD for growth, excess energy is produced, and ADP is oxidized to ATP to store the energy. This results in the uptake of phosphorus in the aerobic reactor. Uptake is in excess of that previously released to compensate for that lost through sludge wasting. The sequence is remarkably efficient for the poly-P bacteria, which accumulate eleven ATP's for each one expended during substrate storage. Thus, they proliferate at the expense of the other bacteria producing an activated sludge that has the ability to remove large amounts of phosphorus.

The reactions describing the sequence, assuming acetate as the organic available for storage, are (Seibritz, et al., 1983):

Anaerobic Reactor

$$\text{acetate} + 2\text{ATP} + \text{Co-A} \longrightarrow \text{acetyl Co-A} + 2\text{ADP} + 2\text{Pi} \qquad [7]$$

$$\underline{2\text{acetyl Co-A} + 2\text{ADP} + 2\text{Pi} \longrightarrow \text{acetoacetate} + 2\text{ADP} + 2\text{Co-A}} \qquad [8]$$

$$\text{Overall: } 2\ \text{acetate} + 2\text{ATP} \longrightarrow \text{acetoacetate} + 2\text{ADP} + 2\text{Pi} \qquad [9]$$

Aerobic Reactor

$$\text{acetoacetate} + 4O_2 + 8NAD_{RED} + 22ADP \longrightarrow 4CO_2 + 8H_2O + 8NAD_{OX} + 22ATP \qquad [10]$$

For municipal wastewaters the acetate must be formed by anaerobic fermentation. When it occurs, energy is obtained by the anaerobic bacteria during fermentation, cell mass is produced, and the organic loading to the subsequent reactors is reduced. When the loading is reduced, the oxygen requirements are also reduced and this results in an energy savings.

Summary

In summary, while the addition of nitrification will typically increase the oxygen requirements (NOD) of a municipal activated sludge system by about 55%, the utilization of the influent BOD to accomplish denitrification will result in a recovery of 50% or more of the NOD, and biological phosphorus removal will reduce the BOD stabilization oxygen requirements by 20 to 30%. Thus, the final balance of oxygen requirements for an activated sludge system removing BOD, nitrogen and phosphorus is approximately the same as for a system removing only BOD. The oxygen requirements of a biological nutrient removal system are substantially less (55%) than those of an activated sludge system removing BOD and accomplishing complete nitrification.

Biological Nutrient Removal Process Configurations and Full-Scale Experience

Several mainstream process configurations are available which combine biological nitrogen and phosphorus removal processes. The most prominent are the modified Bardenpho (Phoredox) (Barnard, 1975), the University of Cape Town (UCT) and modified UCT (Siebritz et al., 1982), and the A/O and A^2/O processes (Hong et al., 1979). Representative flow schemes are given in Figures 3 through 7. The Bardenpho and A/O - A^2/O processes are proprietary but the UCT systems are not. The greatest amount of full-scale operating experience has been obtained with the Modified Bardenpho configuration, mostly in southern Africa, but other full-scale installations of each are currently in operation. Operating results have shown that correct design and operation can accomplish nitrogen removal to less than 3 mg/L and phosphorus removal to less than 1 mg/L in the same system.

The nine countries known to have full-scale, operating mainstream biological nutrient removal plants, and the known numbers of plants, are: South Africa (31+), Zimbabwe (8), USA (6), Japan (?), France (2+), England (1), Denmark (1), Namibia (1), and Canada (1). The plants range in size from less than 1 million gallons per day (MGD) to about 40 MGD. The most significant observations and developments are:

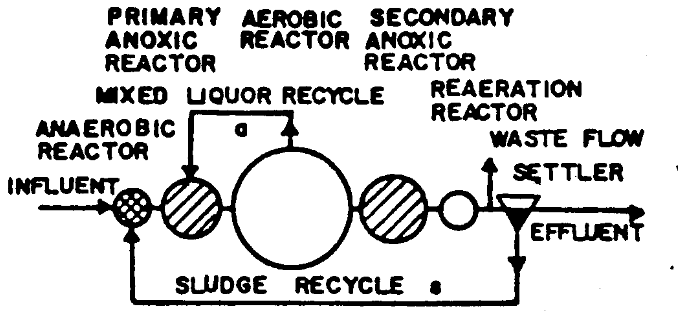

Figure 3. The Phoredox process for biological nitrogen and phosphorus removal, also called the Modified Bardenpho process.

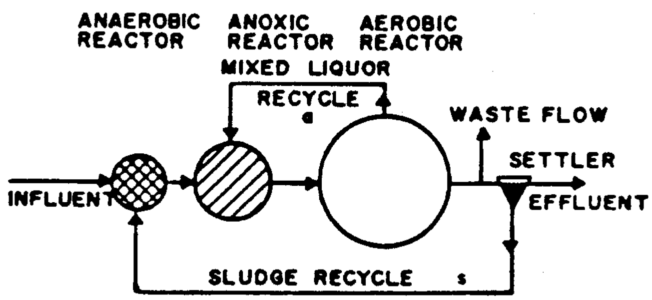

Figure 4. The 3-stage Phoredox process for biological nitrogen and phosphorus removal.

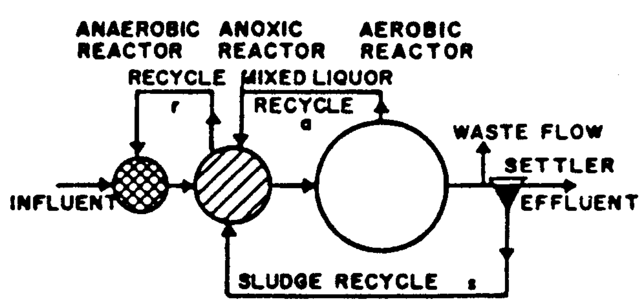

Figure 5. The UCT process for biological nitrogen and phosphorus removal.

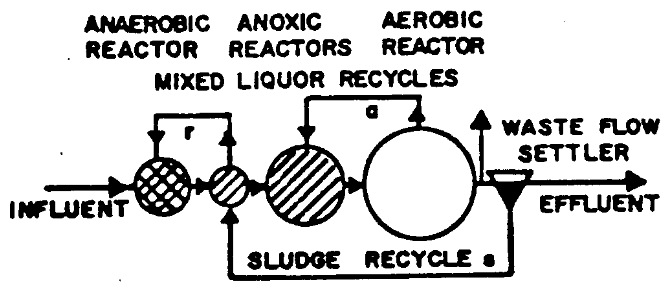

Figure 6. The modified UCT process for biological nitrogen and phosphorus removal.

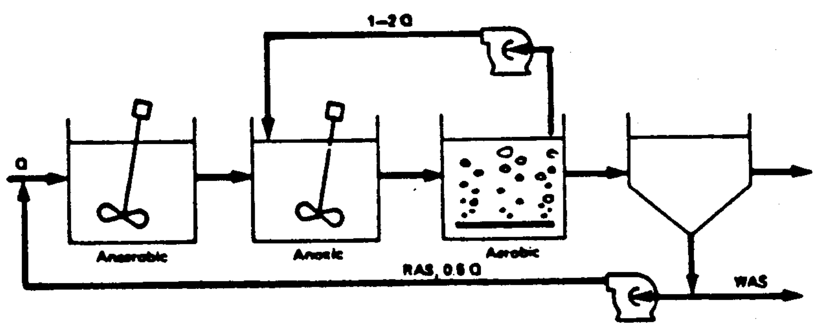

Figure 7. A^2/O Process.

··The 40 MGD modified Bardenpho Goudkoppies plant at Johannesburg, South Africa, has been successfully operated for several years. For example, from January 10 through July 11, 1982, it discharged average effluent concentrations of 0.66 mg/L total phosphorus (TP), 0.36 mg/L orthophosphate (OP), 2.78 mg/L total Kjeldahl nitrogen (TKN), and 1.6 mg/L nitrate nitrogen. The mean design value for 95% reliability for the same period was 0.45 mg/L OP, 2.7 mg/L mg/L TKN, and 3.1 mg/L nitrate. Freezing temperatures are frequently experienced in Johannesburg during June and July.

··Kerdachi and Roberts (1982) have shown that very simple process configurations can achieve the same results as the multi-stage modified Bardenpho system if they are properly operated. By controlling the air input to the fixed-platform turbine aerators in the square, completely-mixed, single reactor of a 1 MGD plant operated by the City of Pinetown, a suburb of Durban, South Africa, they have consistently achieved the following reductions and effluent qualities for several years:

PO_4-P	Influent	10.5 mg/L	Effluent	0.8 mg/L
NH_3-N	"	30.0 "	"	<0.5 "
TKN	"	56.0 "	"	1.5 "
NO_3-N	"	0 "	"	<0.5 "
BOD_5	"	390 "	"	<10 "
COD	"	700 "	"	35 "

··A 0.37 MGD sequencing batch reactor plant serving the city of Culver, Indiana, produces average monthly effluent concentrations of 0.3 - 1.7 mg/L NH_3-N, 0.4 - 1.7 mg/L NO_3-N, and 0.3 - 1.0 mg/L TP on a year-round basis.

··Best, et al. (1984) have shown that large-scale conventional activated sludge systems can be simply and economically modified for simultaneous nitrogen and phosphorus removal. They accomplished the necessary configuration by first converting adjacent plug flow tanks to an oxidation ditch (race-track) flow pattern for nitrogen removal, and then added baffling in the influent zone for phosphorus removal. During the first full year of operation with both modifications, they achieved 95% nitrification, 66% denitrification, and 47% TP removal on a year-round average. The utilization of denitrification reduced energy consumption by 17%.

··A 6 MGD modified Bardenpho plant at Kelowna, British Columbia, Canada, has been further modified for the treatment of low strength sewage after primary sedimentation, and is currently removing TP to less than

1.0 mg/L and total nitrogen (TN) to less than 5.0 mg/L before effluent filtration. This plant incorporates continuous ORP and dissolved oxygen (DO) monitoring, is computer-controlled, and has been used experimentally to generate information for the improved design of biological nutrient removal plants. It has also demonstrated excellent performance in a very cold climate.