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NONPOINT NITRATE POLLUTION OF MUNICIPAL WATER SUPPLY SOURCES: ISSUES OF ANALYSIS AND CONTROL

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INTERNATIONAL INSTITUTE FOR APPLIED SYSTEMS ANALYSIS
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FOREWORD

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IIASA pursues this goal, not only by pursuing a research program at the Institute in collaboration with many other institutions, but also by holding a wide variety of scientific and technical meetings. Often the interest in these meetings extends beyond the concerns of the participants, and proceedings are issued. Carefully edited and reviewed proceedings occasionally appear in the International Series on Applied Systems Analysis (published by John Wiley and Sons Limited, Chichester, England); edited proceedings appear in the IIASA Proceedings Series (published by Pergamon Press Limited, Oxford, England).

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PREFACE

In many developed and developing regions throughout the world, water supply and management agencies are confronted by a steadily increasing demand for water. Water supply is usually constrained by natural, technological, and economic conditions. The limit on the quantity of water which can be tapped grows more severe, because deteriorating water quality necessitates more and more complex utilization constraints. Operating with this in mind, regional water managers attempt to satisfy different supply interests, especially when these interests conflict with each other. Therefore, the competing interests of agriculture, environment, and municipal water supply become increasingly important. For example, in recent years, water supply agencies have become progressively more concerned by high nitrate levels in municipal water supply sources.

In 1980, an exploratory study on "Analysis and Control of Nonpoint Nitrate Pollution of Municipal Water Supply Sources" was initiated at IIASA. The project, which formed part of the Research Tasks "Regional Water Management" and "Environmental Problems of Agriculture", was carried out as a collaborative effort between IIASA's Resources and Environment Area (REN) as well as scientific institutions in several National Member Organization (NMO) countries. In concluding the project, a Task Force Meeting was held at IIASA from 10 to 12 February 1981. At the meeting, which was attended by 16 scientists from 7 NMOs, the OECD and IIASA, the study results obtained by IIASA and the collaborative institutions were discussed in depth and topics for further research were suggested. At the closing session, the participants expressed their appreciation for the unique opportunity to discuss important practical problems in water and agricultural management from a broad systems point of view. It was particularly appreciated that the meeting brought together experts from different fields, as there were hydrologists, hydrogeologists, engineers, and agricultural economists dealing with the problems in question.

Janusz Kindler
Chairman
Resources and Environment Area

ACKNOWLEDGMENTS

As the editor of the proceedings I express my gratitude to all those who contributed to the Task Force. In particular, I would like to thank the chairmen of the working groups at the meeting, R.C. Ward and P.E. Rijtema, whose reports have been used to prepare the introduction to the proceedings. I also wish to thank Elisabeth Jaklitsch for helping with organizational arrangements for the meeting, Anna John for assisting with preparation of the proceedings, and Vicky Hsiung for typing portions of the manuscript.

Karl-Heinz Zwirnmann

CONTENTS

INTRODUCTION

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THE REN NITRATE STUDY

The Resources and Environment Area (REN) of IIASA deals with, among other problems, the pollution of water resources caused by intensive agricultural activities. In 1980, one of the major environmental problems of agriculture addressed was nitrate leaching due to the use of nitrogen fertilizers. Two levels of analysis were introduced; the field and the global level. In field level investigations, the CREAMS model of the U.S. Department of Agriculture (Knisel, 1978) was applied in several case study areas in IIASA member countries. Golubev (1980), conducted a global survey of nitrate leaching hazards. Both studies as well as findings of the regional water management projects revealed the nitrate problem to be of great practical relevance to agricultural and water management.

To ensure a safe drinking water supply while attaining agricultural production goals is a complex, interdisciplinary problem, which has to be attacked in its entirety by the adoption of some sort of systems approach. The exploratory study on "Analysis and Control of Nonpoint Nitrate Pollution of Municipal Water Supply Sources" carried out as a collaborative project between REN and several external institutions represents a preliminary attempt in this direction. While aiming at generating a methodological outline of a multidisciplinary approach to the problem, the study at the same time aimed at providing practical guidelines for dealing with the nitrate pollution of water resources. Its main focus was agricultural nonpoint pollution sources and the use of inorganic fertilizers in particular. An approach has now been formulated, which appears to be generally suitable for most other types of pollution control in water quality management (Zwirnmann, 1981). The principal issues which best illustrate the findings of the study are summarized below.

Nitrate Pollution of Water Resources: Sources and Control

The specific concern about nitrate pollution of municipal water supply sources stems from the hazard to public health caused by the toxic effects of nitrates in drinking water. Nitrate removal from water supplies cannot be accomplished by conventional treatment procedures, consequently, there is a clear necessity to clarify the potential extent and severity of the situation in order to understand the constraints imposed by nitrate pollution on water supply planning. There is a tremendous variety of nitrogen sources in the environment which contribute to water pollution. However, among the major sources of nitrogen pollution of water supplies, chemical fertilizers have been found to be the dominant cause of the recent, rapid increase in nitrate concentrations in water resources. Agricultural interests are also served by looking at the problem, because the amount of nitrogen which pollutes water resources constitutes waste of a valuable resource which must be prevented by better management practices in agriculture.

Nitrogen and Water Resources

The initial step in developing options for the control of nonpoint nitrate pollution of municipal water supply sources is the analysis of the physical system to be controlled. The interactions of various components of the system, such as the water resources of a region, or the input and output of nitrogen to and from the water resources system, need to be identified. When considering a regional water resource system, the amount of nitrate present in water supply abstraction is basically controlled by the various processes taking place in the nitrogen cycle, particularly by the interaction of water with the soil-plant system. Consequently, the system to be controlled can be divided into three generalized parts: surface water, groundwater, and the soil-plant system.

The relative importance of water supply sources (rivers, lakes, reservoirs, and aquifers) generally depends on the given conditions of a specific region. Golubev (1980), proved that the hazard of nitrate leaching is particularly high in certain countries because of their general climatic features. Often, the effect of this natural situation is compounded by the use of supplemental irrigation. It is important to note that for those countries identified as having a particularly high potential for nitrate pollution, groundwater resources play a key role in potable water supply. Groundwater resources then deserve special attention, especially because there is an important difference between groundwater and surface water pollution and their respective management strategies. While the decision to purify river water is made with the knowledge that water quality can be restored relatively quickly after removing the pollution source, the same does not apply to lakes, reservoirs, or particularly to aquifers, where pollutants may be retained for decades or even centuries. Nevertheless, examination of the effects of fertilizer nitrate water pollution in a regional context usually requires a conjunctive consideration of the groundwater and surface water resources of a region.

Outline of a Control System

The physical system considered so far is now ready to be fitted into a more general management system for the control of nitrate pollution in municipal water supply sources. As seen from the preceding analysis, the major concern in outlining such a system is controlling nonpoint pollution sources in agriculture, such as organic and inorganic fertilizer, with most importance given to the latter. Hence the system must provide a framework for the analysis of the various factors affecting regional water resources management, and consider the interests of the competing users of soils and waters. In order to understand how water supply and management is influenced by increasing nitrate concentrations in water resources and to ensure a safe drinking water supply, management must link land use and water supply development. In the study the framework for analysis therefore followed the concept of a decision-making process based on the control system shown in Figure 1. The major components considered are:

a) the system to be controlled, encompassing

-- the <u>municipalities</u> (representatives of the general public) which are supplied with water and agricultural commodities and govern the overall control system by setting the management objectives; they also contribute to nitrate pollution of municipal water supply sources through the disposal of human and industrial wastes;

-- the <u>environment</u>, especially the atmosphere, which provides the background load of nitrogen to the two environmental subsystems of interest, the soil-plant system, and the water supply sources;

-- the <u>water supply and management</u> agencies managing the <u>municipal water supply sources</u> and responsible for ensuring potable water supply;

-- the <u>agricultural production</u> sector which strives to achieve production goals, causing nitrate pollution of water supply sources as a side effect of the technologies used for crop production and waste dispersal in the soil-plant system;

b) the <u>management objectives</u> of the overall control system which should be accomplished through management measures appropriate to the specific system;

c) the <u>management subsystem</u>, where management objectives are achieved through <u>planning</u> and <u>implementation</u> of management measures not only in the field of water supply and management, but also in the agricultural sector.

The components of the system to be controlled (the municipalities and the environment, with its subcomponents, the water resource system and soil-plant system) are physically connected by mass flows (nitrate polluted water, drinking water, agricultural commodities) and constitute the basis and target for

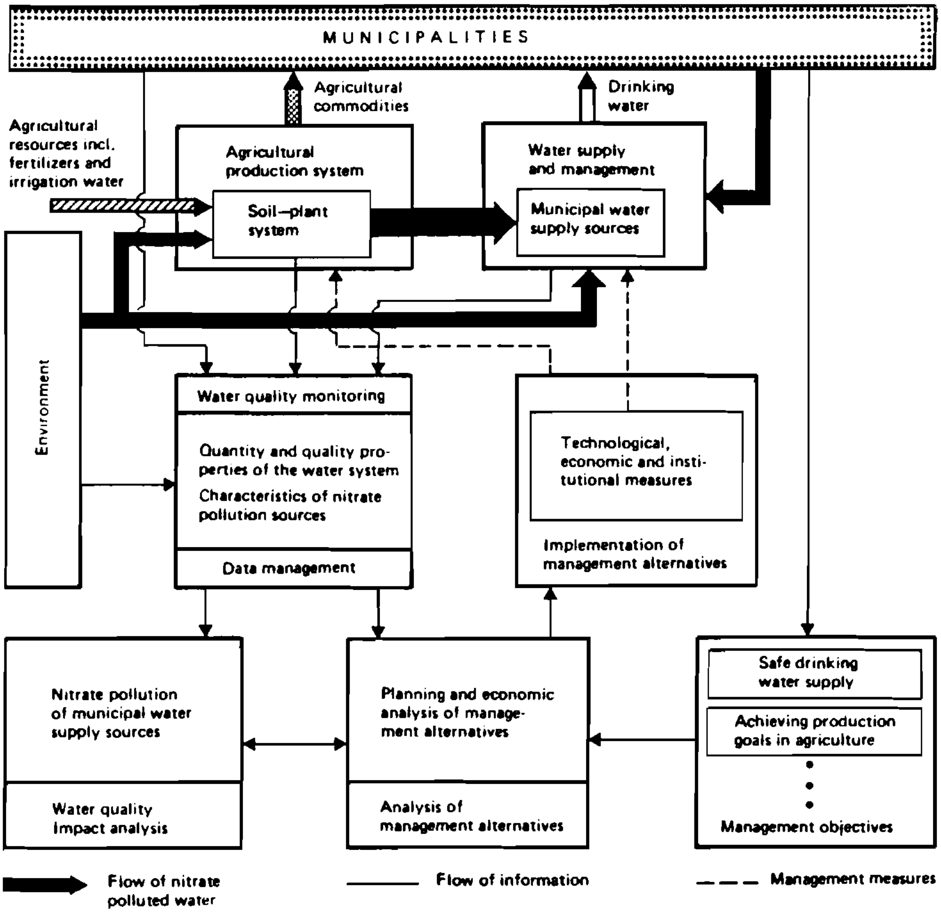

Figure 1. The System for Control of Nonpoint Nitrate Pollution of Municipal Water Supply Sources

decision making. In contrast, components of the management subsystem are linked by the flow of information. The conjunction between these two main parts of the overall control system is provided by the implementation of management measures.

Control Options

Water management objectives are generally achieved through an integrated implementation of technological, institutional, and economic measures. Depending on the system to which they refer, two general alternatives for water supply pollution control can be distinguished; first, controlling potential pollution sources and second, treating polluted water and taking special measures to ensure water supply.

Nonpoint nitrate pollution control measures used in the agricultural sector to manage fertilizer application, animal waste disposal, land use, runoff, erosion, and leaching, are generally preferred to the elimination of nitrates by water purification. The development and application of new kinds of fertilizers and inhibitors for controlling fertilizer release or transformation also must be taken into account. However, in reality, due to the advanced state of water pollution, one has to consider problems facing municipal water supply in the short run, for example, the need for alternative supply sources, new water treatment technology, or special supply measures. As practical implementation of pollution control strategies is largely based on institutional, legal, and economic actions,it strongly depends on the existence of regional authorities, and their capabilities. The management policies pursued by such authorities must recognize that water purification technologies for nitrate elimination cause a tremendous increase in expenditure in the water industry. Even when neglecting the long-term requirements of water supply protection, short-term social benefits can only be received through the overall control system when the benefits gained from agricultural production outweigh the additional costs of municipal water supply.

Moreover, when dealing with nonpoint source pollution control, it is important for researchers, practitioners, and policy makers to realize that only a beginning has been made. While the control of point source discharges of wastewater is based on over hundred years of research and testing, continued investigations into nonpoint source control are necessary to establish a comparable level of technology.

Issues of Analysis

To answer the question of how water supply and management are influenced by increasing nitrate concentrations in water resources and how a safe drinking water supply can be ensured, the goals of the agricultural production sector should be coordinated with public decisions regarding land use, water supply development, surface and groundwater systems. Therefore, water pollution control management must be based on an effective planning procedure. The tools applied in the analytical process must be capable of matching both the ability of data to yield information with confidence and matching the expectations of the decision makers. Hence, the study did not mainly focus on methods for detailed analysis of the physical, chemical, and biological processes constituting the behavior of nitrogen in water resources. It was intended rather, to show how such means as monitoring and modeling can support the decision-making process in nitrate water pollution control management with particular reference to nonpoint source pollution. To comprehensively analyze the nitrate problem obviously requires the integration of physical processes and decision oriented aspects. At least three aspects seem to be of general interest. First, there is a general methodological issue concerning problems in analyzing and modeling, such as those of scale as well as risk and uncertainty. Many questions about how to translate model results from field studies into regional terms, how to precisely consider long and short term effects of pollution and its control, how to deal more comprehensively with uncertainties in the control objectives and alternatives, as well as in the structures, parameters and input data of the applied models still remain unsatisfactorily answered. Second, the physical processes have to be given attention. A comprehensive evaluation of the spatial and temporal development of nitrate pollution has to be carried out by monitoring and modeling the processes (mechanisms, pathways, etc.) relating pollution sources to water resources in order to achieve a meaningful water quality impact analysis. Third, the decision aspect has to form an integrated part of the overall analysis. Planning and socioeconomic evaluation of control strategies require the analysis of tradeoffs in and between water supply and management, as well as agriculture. Moreover, an assessment has to be made of the potential or proved impacts of pollution and its control, on man, the environment, agricultural yields, etc., before control methods can be designed for implementation.

The issues mentioned above were discussed at the Task Force meeting and these proceedings summarize the results of the research activities which began in 1980.

THE PAPERS OF THE TASK FORCE MEETING

All the papers presented at the Task Force meeting appear in this volume. The IIASA Collaborative Paper (CP-80-34) is included although it was not presented at the meeting, as it was specifically prepared in the framework of the study.

The presentations given by the participants dealt with selected aspects of the problem thereby providing a meaningful overview of its complexity. In keeping with the issues discussed in the previous section of this introduction, the papers are divided into three groups. One group is mainly concerned with identifying the problem from the point of view of hydrology, hydrogeology and agricultural economy, while a second group deals with control options in the field of agriculture and water supply and management. A third group represents more methodology-oriented approaches ranging from water quality monitoring through modeling groundwater and complex water resource systems, to an integrated physical-economic analysis of water resource development and land use planning.

The paper by P.E. Rijtema on the effects of regional water management on nitrogen pollution, in areas with intensive agriculture, represents a typical hydrological view of the problem. It deals with a pilot region situated in the flat south-eastern part of the Netherlands where very intensive agriculture and livestock breeding are accompanied by an increased use of chemical fertilizers. Using hydrological water balance models, the author calculates the net subsurface inflow to the aquifer for the different soil types in the region. The inflow values are assumed to indicate the potential of the soils to pollute the aquifer by nitrate transport from the unsaturated zone. In continuing this approach, the improvement of local drainage as a strategy for reducing nitrate water pollution is investigated. Improved local drainage would cause a reduction in the amount of surface runoff and discharge by interflow through the soil system. This results in an increased residence time of the precipitation excess in the soil system, i.e. additional denitrification may take place, which would reduce the nitrogen load in surface water.

In his paper, L. Alföldi deals with hydrogeological aspects of groundwater nitrification. He states that nitrogen accumulation in subsurface waters is not the result of a simple infiltration process, but it is the result of complex and intermittent biochemical and chemical processes closely related to the periodical variation of the groundwater budget. The soil zone is taken to be an independent nitrogen budget in which the above mentioned processes take place. Following this line of discussion, the author proposes three basic hydrogeological types of soil, those with a self-contained water budget, soils where the water budget is affected by subsurface waters, and soils with an independent water budget. The characteristics of the different categories are discussed in detail. As groundwater nitrification is shown to be an irreversible process resulting in the accumulation of nitrate in aquifers, the conclusion in the paper is that more attention should be paid to controlling the soil-plant system by improving agricultural management practices.

The latter suggestion is, to a certain extent, also of interest to H. de Haen, who develops a cost-benefit analysis concept for dealing with the impact of nitrogen fertilizer application on the nitrate concentration of groundwater. He starts his discussion by raising the question: What would be the costs and the benefits of reduced nitrate concentration? To avoid the difficulties involved in finding a "socially optimal" degree of nitrate concentration, the author proposes assuming certain nitrate standards to be exogenously given and looks for least-cost alternatives to meet them. However, some technical difficulties remain in this case as well. In comparing the costs of alternatives, e.g. reduction of fertilizer levels or water treatment, not only direct costs but also external effects as well as administrative efforts and institutional implications have to be compared. Moreover, as de Haen says, the debate on the role of agriculture suffers from a lack of information, especially on the quantitative relationship between the amount of nitrate leached or runoff from a given field and the amount which finally reaches the drinking water source at a certain distance. He suggests that for further cost calculations a Nitrate Leaching Matrix should be employed, to assess the leaching for entire farming systems under alternative fertilizer levels and exogenous soil and climate factors. This concern of de Haen clearly links the agricultural economics problem to those discussed in the first two papers addressing hydrological and hydrogeological issues related to soil systems. Moreover, knowledge on nitrate behavior in the saturated zone of aquifers is required in order to answer the author's major question: is the nitrate pollution problem manageable within narrow bounds of catchment areas (or protected areas) of drinking water resources? The author concludes the answer to this question will eventually determine the borderline between local (control of fertilizer levels, prohibition of certain crops, etc.) and sectoral (nitrogen tax, quota, etc.) policies to be implemented in agriculture.

The three papers which appear next deal with pollution control options. As an agricultural control strategy chosen from a set of feasible alternatives must be appropriate for local conditions and acceptable to the farmers, nonpoint source pollution control programs also have to provide general information and education to assist farmers. The paper by K. Beer, H. Ansorge, and H. Görlitz discusses a computer-aided advisory system for fertilizer application as an example of such information and education programs. Two major objectives can be fulfilled by the advisory system. First, the system serves to plan the demand for mineral fertilizers (amount, type) on farms, in districts, and in regions, taking into account the availability of organic manure. Second, it determines the type of fertilizer used and timing, rate, splitting, and technological method of fertilizer application on specific fields. Although it has proved to be a useful tool for planning and control of fertilizer use in agricultural practice, the authors list some problems yet to be overcome. An important one, if not the most important one, is the need to consider more precisely, in advance, the meteorological conditions and impacts of irrigation because they have a particularly noticeable influence on the effects of fertilizer use.

As all the above papers show, in looking at agricultural systems, it is mainly the inputs of nature which remain uncontrolled and cause the stochastic features of the outputs, including fertilizer losses. Since the outputs can be controlled only by varying the inputs, or the system itself, the overall control problem is very complex. Perhaps we will never be able to completely control agricultural crop production systems in order to efficiently prevent water pollution. At least in the short run, the water supply industry therefore faces and will continue to face a nitrate problem which has to be solved by water treatment and management.

The papers by M. Roman as well as D. Lauterbach and H. Klapper recognize this fact. For example, Roman points out that the improvements in agricultural technology may, at least theoretically, prevent water contamination by nitrogen compounds, but in practice significant results cannot be expected, because the agricultural industry concentrates above all on increasing food production.

In dealing with the nitrate problem from a water treatment point of view, Roman stresses the fact that nitrogen compounds other than nitrates have to be taken into account. The various forms of nitrogen, such as organic nitrogen, ammonia, nitrite-, and nitrate-nitrogen could undergo changes and depending on conditions, could be converted into other compounds. Doubtlessly, in arguing in this way a phenomenon is uncovered which has to be clearly understood before water treatment technologies are designed.

The paper by Lauterbach and Klapper which was presented by R. Enderlein deals more explicitly with water treatment and management options. Three general options are distinguished, namely reducing or preventing nitrate input into water bodies through sewage treatment and water protection zones, hydraulically controlling or biologically treating surface water bodies with a high nitrate content, and raw water treatment by ion exchange. According to the authors, the additional costs of nitrate elimination by ion exchange almost equals the cost of the complete conventional treatment of medium polluted raw water. However, the authors consider it wrong to take decisions only because of current economic and technological conditions. Moreover, as nitrate pollution of water resources is a consequence of intensified industrial and agricultural production as well as urbanization, they conclude that assessments of benefits or costs must not be made one-sidedly with regard to any one of these sectors. Further, they point out that fresh water resources are limited and future generations also have a right to sufficient water supplies and a healthy environment. It should not be forgotten that nitrates which enter groundwater now cannot be eliminated at all or would require several decades to disappear. Therefore, measures should be taken to prevent or reduce pollution and for both ecological and economic reasons, these should be given priority.

After identifying some of the many faces of the nitrate pollution control problem, the need for effective tools and methods for analyzing them in their complex setting becomes obvious. The remaining four papers of the proceedings deal with exactly that.

The paper by R.C. Ward addresses an issue that is, of course, not unique to the nitrate problem but has become a key problem in designing water pollution control programs, namely water quality monitoring. Past approaches to that issue have considerably suffered from a system's perspective. Ward gives one of the very few known examples for overcoming this situation. His approach takes an integrated view on the activities involved in the design and operation of monitoring programs. Major activities such as data acquisition and data utilization are defined and discussed, as they are related to each other. Ward is much more interested in methodologically following the flow of data and information within a monitoring program than in the purely technical aspects of monitoring networks. But this seems exactly to be the point to be made where so much effort is spent on improving monitoring by installing modern electronic measuring devices. As the author points out:

> In many cases, the true monitoring objectives are extremely difficult to formulate.... How can management be made to understand the importance of clearly defining monitoring objectives? The system's perspective illustrates clearly the interaction between data collection and data use and notes how ultimate accountability of the monitoring programs rests on the data's use. Such a view of monitoring will, hopefully, assist designers in getting access to monitoring objectives.

In contrast to Ward's paper the other three analysis-oriented papers address nitrate water pollution control from different perspectives in mathematical modeling. D.B. Oakes deals with the mechanisms and modeling of nitrate pollution of groundwater resources. Field investigations on the impact of agricultural practices on the nitrate content of groundwater in the principal aquifers of the United Kingdom have been carried out. A strong correlation has been found between the high nitrate concentration of the water in the unsaturated zone and arable farming regimes. Transport models for the unsaturated and saturated zones have been developed. A vertical flow model describes nitrate leaching from the soil with a rate that depends on the infiltration and pore water content of the rock, taking into account the history of land use and fertilizer application rates. The nitrate movement in the saturated zone of the aquifer is simulated by a model referred to as a catchment model. In fact, this model is one of a fully mixed single-cell type model which uses as input the leached nitrate generated by the vertical flow model. The author reports that the model has been applied to chalk, limestone and sandstone catchments and has been able to accurately simulate nitrate concentrations in pumped abstractions.

J. Blake in her paper gives an overview on the nitrate project of the Thames Water Authority, U.K. According to the author a three-phase approach was adopted to devise a regional strategy for dealing with the nitrate problem. The first phase was aimed at assessing the potential severity of the problem, while in the second phase a technique was devised for evaluating options to be employed in the third phase for the selection of a preferred strategy. To accomplish the aim of the first phase, a time series analysis was used, based on Box-Jenkins' transfer function models. After establishing the fact that the Thames Water Authority is likely to have a serious nitrate problem, the agency is proceeding with a simulation approach to produce a regional nitrate model compatible with the water resource model already in existence. Major components of the model are: river reach, soil, aquifer, sewage treatment works, channel, reservoir and water denitrification plant. The model is intended for use in assessing potential new sources of nitrates, and evaluating options for nitrates management throughout the water resource system. Since cost estimates for each option configuration and mode of operation will be made in parallel with model runs, the final strategy can be selected on the basis of a performance-cost ranking.

D.J. Dudek and G.L. Horner state in their paper:

> Increasingly, an awareness of the complex interdependencies between bio-physical and socio-economic systems has stimulated the initiation of comprehensive resource planning methods and the abandonment of single-purpose approaches. However, irrespective of the recognition given to these concepts, examples of their practical application in resource planning efforts are scarce.

One of these examples is presented in the last paper of the proceedings and deals with an integrated physical-economic systems analysis of land use and water resource development planning in irrigated agriculture. To consider the four basic economic concepts such as commodity demand, commodity supply, resources demand, and resource supply, the whole analytical system, which is based on a linear programming approach, consists of four interrelated models. A projection model for specifying commodity demand is linked to a regional production model to estimate commodity supply as well as the demands for land and water resources. The latter are the basis of valuations in the land use model. Two location specific components form the water quality model. Again, a linear programming model derives optimal cropping patterns as well as use of water and fertilizer and is linked to a physical model encompassing three specific interdependent sub-models to analyze the hydrology, salinity balances, and nitrogen concentrations in the basin. In applying their model system to a real world planning problem, the authors found it to be sufficiently comprehensive and flexible, and felt it provided the opportunity for testing alternative water quality policies and evaluating their effects upon related resource uses.

In concluding the overview of the papers given above, one can doubtlessly agree with the following statement made by Dudek and Horner:

> Resource and environmental planning cannot be separated. Planning for resource use without recognition of the environmental goals or objectives of society may result in resource allocations which are socially suboptimal. Similarly, planning for environmental quality without assessing the suitability, availability, and productivity of the resource base may impair the efficiency and distribution of output.

CONCLUSIONS OF THE TASK FORCE MEETING

After the presentations two small working groups were formed, chaired by R.C. Ward and P.E. Rijtema. The discussions within the groups were mainly aimed at identifying topics for future research at IIASA or elsewhere. The reports of the working group chairmen provided an excellent basis for discussing such topics at the closing session.

It was agreed that pollution of municipal water supplies by nitrogen compounds is and will remain a real world problem in years to come. There are three topics of particular interest for further research:

- The health hazards of nitrogen compounds in water and food
- Agricultural activities as an important nitrogen pollution source
- The nitrate problem in municipal water supply and management.

The Health Hazards. The exact nature of the human health problems are not yet well understood nor documented, as witnessed by the variation of standard limits set in national regulations. There is a need to establish consistent criteria for safe (low risk) limits of nitrogen compounds, nitrates in particular, in drinking water and food. More toxicological and/or epidemiological studies are required to dispel uncertainties by accounting for factors such as the size and susceptibility of the population exposed, the number of water systems involved, the relative dose in water compared to the total burden, the positive response of nitrate in tests for carcinogens, teratogens, mutagens, etc.

Agriculture. There is a need to document the sources of nitrogen compounds from agriculture under various conditions (climate, soil type, cropping pattern, fertilizer type and application technology, etc.). The aim should be to derive cause-effect relationships between agricultural practices and the generation of nitrogen compounds leaving the agricultural system. Hence technological changes in agricultural production should be evaluated in terms of fertilizer use and environmental impacts. Based on this, policies to be used to encourage better agricultural management practices have to be identified and evaluated in terms of tradeoffs between agricultural production and pollution control.

Water Supply and Management. To deal with the nitrate problem from the point of view of water supply and management all sources of nitrogen (agriculture, industry, households, atmosphere, etc.) as well as all types of nitrogen compounds (organic nitrogen, ammonia, nitrite, nitrate, etc.) contributing to water pollution should be taken into consideration. Hence one has to deal with a "nitrogen problem". On account of the variety of municipal water supply sources (rivers, lakes, reservoirs, aquifers, etc.) the nitrogen problem eventually has multiple constituent pollutants, multi-source and resource aspects contributing to its complexity. As it can only be controlled by conjunctive water resource management, sufficient attention must be paid to the pollution of groundwater, which is of the greatest long-term concern.

Any pollution changes in time and space, as well as the total nitrogen balance need to be understood, i.e., there is a need to analyze water quality data in different settings to document trends and eventually correlate trends with activities which generate pollution. After proving that the limits set are likely to be exceeded, water treatment technologies (structural and nonstructural) have to be identified and evaluated. Policies for implementing them need to be designed after analysis of the tradeoffs between water treatment technologies and pollution source controls.

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EFFECTS OF REGIONAL WATER MANAGEMENT ON N-POLLUTION IN AREAS WITH INTENSIVE AGRICULTURE

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The effects of the improvement of local drainage and of deep well pumping on the hydrological conditions in the area are discussed. Local drainage does not affect the net subsurface inflow of the groundwater aquifer to a great extent. The main effect of local drainage appears to be the reduction in surface runoff. Due to this and to the increased residence time of the precipitation excess, a large reduction will be noticed in the nitrogen load in surface waters.

Deep well pumping for municipal water supply strongly affects regional water management. Low-lying areas, which did not originally feed the groundwater aquifer, do so when deep well pumping occurs. It was found that the N-flow to the groundwater aquifer in the intake area of deep well locations, greatly depended on the hydrological conditions in the area and land use. Some results of calculated N-inflow for 17 deep well locations are presented.

1. INTRODUCTION

Where in humid areas the ground surface has only relatively small differences in elevation and the transmissivity of the soil and subsoil is not very small, the excess precipitation is mainly carried-off by groundwater flow to a system of rather closely spaced drains of different size and level. The depth of the groundwater table and the rate of discharge by the drains are variable owing to the seasonal fluctuations of evaporation and the irregular variations of precipitation. Under these conditions river discharge will generally be large in winter and small in summer.

The landscape more or less prohibits the construction of large artificial reservoirs to store the winter discharge for municipal and industrial water supply. The needs for water supply in these areas force the water supply companies to explore the groundwater resources.

Deep well pumping of groundwater from thick phreatic aquifers or from semi-confined aquifers will cause a decline of the phreatic level, particularly in the case of phreatic aquifers. Primarily this results in a smaller discharge of water by surface drains. In those cases where formerly during summer – the period with main evaporation – the depth of the phreatic level was rather small a reduction in evaporation during periods of drought also has to be taken into account. As the growth of crops depends on the available soil moisture, which in its turn depends on the depth of the phreatic level; groundwater extraction can cause reductions in yield of agricultural crops in most regions.

Intensification in agriculture has led to an increased fertilizer use. The potential of livestock waste to pollute surface waters and groundwater is large, particularly when plant nutrients are supplied in excess of crop requirements. The extent of the problem of manure disposal was not discerned in the early stages of intensive farming. Difficult soil and drainage conditions on part of the farm limit the area suitable for winter application of slurry and may result in local overdosing. The consequences of fertilizing with regard to nitrogen pollution of the groundwater are dependent on land use, soil type, fertilization level and the hydrological situation. Leaching of nitrogen occurs for nearly 100 percent in the form of NO_3^-. When analyzing the consequences for water quality management one has to deal, apart from the transport of matter under influence of the hydrological situation, with aerobic and anaerobic biological degradation of organic matter, mineralization and denitrification, N-uptake by organisms, as well as growth and dying-off of organisms.

The conditions described above exist in the eastern and southern parts of the Netherlands, where the ground elevation is mainly between 5 and 30 m above sea level. Pleistocene sands form a thick and highly permeable aquifer. The discharge is provided by an irregular network of natural brooks and man-made ditches, of which only the major ones discharge throughout the year.

As the groundwater consists of fresh water, extensive pumping for municipal and industrial water supply is carried out or planned for the near future.

2. DESCRIPTION OF A PILOT REGION

Although the population density of the Netherlands' province East

Gelderland is below the country's average, the supply with high quality water may become a serious problem in this region of about 150,000 hectares. The region (fig. 1) is located between the frontier with Germany and the river IJssel, a branch of the river Rhine. In the South it is bounded by the river Rhine. East Gelderland is still predominantly rural and has a population density of about 500,000. The present water demand for municipal and industrial supply is about 50 million m^3 per year. At the end of this century the expected demand will increase to about 136 million m^3 per year. The Water Supply Company 'Oostelijk Gelderland' has estimated that of the total demand some 98 million m^3 can be withdrawn from the groundwater, whereas the remaining quantity have to come from other sources. The problems in this region can be used as a guide in the discussion of the nitrogen problems in regional water management.

2.1. Precipitation excess and drainage

The average yearly precipitation in the area of about 780 mm is fairly regularly distributed over the year. The yearly actual evaporation is about 450 mm. During the summer period the precipitation is, on the average, in balance with evapotranspiration, so the precipitation excess of about 330 mm is restricted to the winter period. About 80 percent of this precipitation excess comes to discharge during the winter half year leaving 20 percent as discharge in summer.

The groundwater table in the region is high, with an average value of 40 cm below the ground surface in winter and about 130 cm below the ground surface in the summer period.

Three classes of drainage channels can be distinguished in the region. Primary channels with a spacing of 1500 to 6000 m, such as

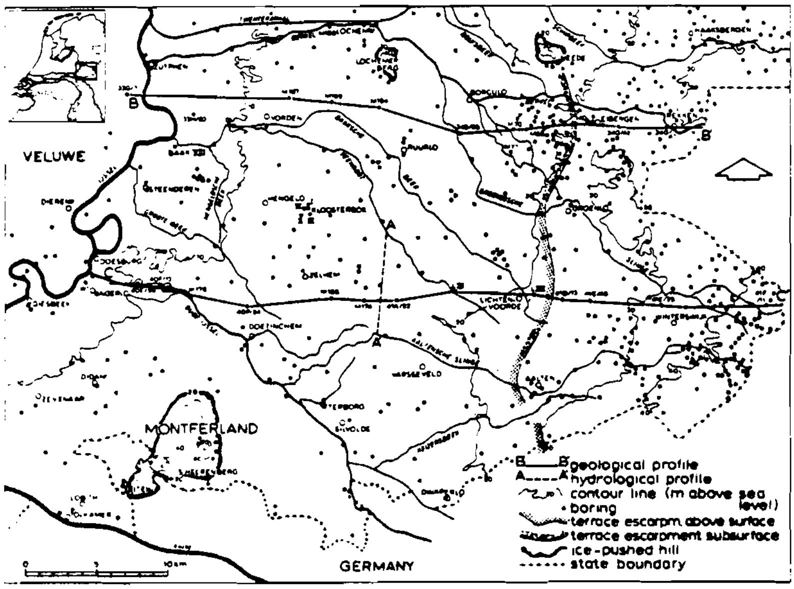

Fig. 1. Map of the eastern part of the province of Gelderland (Netherlands), showing the major topographic features and the locations of borings

larger rivers, brooks and other drainage channels of major importance. A secondary system with a spacing of about 250 to 1000 m, as large ditches and other medium sized drainage channels. A tertiary system, such as small ditches, trenches, furrows and subsurface tube drains, with a spacing of the open conduits of about 50 to 200 m and of the tube drains of about 10 to 40 m. This tertiary drainage system functions only during winter.

The insufficient drainage capacity of the drainage system, although narrowly spaced, may cause inundation and surface runoff in the lower parts of the area after excessive precipitation.

2. 2. Agricultural development

Agriculture in the region is very intensive, with a stocking rate of 4 to 6 standard livestock units per hectare. A standard livestock unit (SLU) is defined as the added value equivalent of different types of livestock expressed in that of a dairy cow. Such a standard livestock unit has a nitrogen production of 102. 6 kg N per year in slurry. About 60 percent of the agricultural soils are in use as grassland and 40 percent as arable land. About 75 percent of the arable land is used for the growth of fodder maize.

3. NITROGEN CONCENTRATION AND LAND USE

The nitrate concentration of the shallow groundwater in nature areas in sandy regions is on the average 0. 3 g. m^{-3} N, which also gives an indication of the natural nitrate in deep aquifers. The nitrate concentration in the shallow groundwater under forests seems to be dependent on both the age of the forest and the depth of the phreatic level. Data given by Oosterom and Van Schijndel (1979) vary from 0.2 to 22 g. m.$^{-3}$ N, with an average value of about 5 g. m^{-3}.

Many data for arable lands were given by the Curatorium Landbouw-emissie (1980). For sandy soils with a crop rotation of twice cereals, potatoes and sugarbeets, with a mean yearly application of 150 kg N per ha in the form of chemical fertilizer, an average concentration of 28 g. m^{-3} N in the shallow groundwater must be expected. When animal slurry was applied instead of fertilizer-N the concentration increased to 50 g. m^{-3} N. On soils where fodder maize was grown each additional overdosing of slurry with a quantity being equivalent to 50 kg N increased the concentration with 4.5 g. m.$^{-3}$. Sufficient evidence is present that about 50 percent of the nitrate in the groundwater system will disappear by denitrification.

A nitrogen emission model for grassland has been given in a previous study (Rijtema, 1980). The average data under mean meteorological conditions at a standard livestock density of 4 units per ha are 127, 104 and 41 kg N per ha for poorly, moderately and well-drained soils, respectively. Steenvoorden (1980) gives data of the nitrogen concentration (organic N and NH^{+}_{4}-N) of surface runoff water in the case of winter application of slurry. The data vary from 80 to 2 g. m^{-3} N, depending on the difference in time between slurry application and the occurence of surface runoff, as well on the quantity of surface runoff.

For the present discussion these average data will be used to give an idea of the order of magnitude of the contribution of diffuse nitrogen sources to the pollution of groundwater and surface waters.

4. WATER BALANCE EQUATION

For the investigation of groundwater flow problems of relatively large areas, the water balance as presented in fig. 2 can be given by the equation:

$$P + I_{ss}^{net} + \frac{Q_{inf}}{A} = E + \frac{Q_{dr} + Q_{dw}}{A} + \frac{\Delta W}{\Delta t} \qquad (1)$$

where:

A = horizontal surface area;

P = precipitation;

E = evaporation ;

I_{ss}^{net} = $I_{ss} - O_{ss}$ = net subsurface inflow per unit of time and per unit of horizontal surface area equals subsurface inflow (I_{ss}) minus subsurface outflow (O_{ss});

Q_{inf} = influent seepage from open channels with relatively high water levels;

Q_{dr} = effluent seepage to open channels with relatively low water levels;

Q_{dw} = discharge by deep wells;

$\Delta W/\Delta t$= change in soil water storage per unit of time and per unit of horizontal surface area.

When both Q_{inf} and Q_{dw} are that small that they can be neglected and when, moreover, only periods of long duration are considered so the change in storage ($\Delta W/\Delta t$) is very small, equation (1) can be replaced by:

$$\frac{Q_{dr}}{A} = I_{ss}^{net} + (P - E) \qquad (2)$$

This equation clearly shows the practical significance of knowing the net subsurfaces inflow, as I_{ss}^{net} represents the additional amount of water which on the average is to be drained above the average supply (P - E). It is clear that the value of I_{ss}^{net} can be positive or negative, depending on the values of I_{ss} and O_{ss}. In the undulating landscape of the sandy soils in the southern and eastern part of the country, areas with a negative value of I_{ss}^{net} generally coincide with the higher parts, whereas the river valleys receive a positive subsurface inflow.

The value of I_{ss}^{net} hardly changes with time as the major differences in water level generally are found over large distances, and these differences in water level are almost independent of time. Values of I_{ss}^{net} can be considered constant when compared with the large and frequent variation in drain discharge (Q_{dr}) caused by rainshowers.

The flow of groundwater can often be considered as taking place in a horizontal aquifer. This implies that the net subsurface inflow can be calculated as:

$$I_{ss}^{net} = v_{phr} + \frac{1}{A} (Q_{dr} - Q_{inf} + Q_{dw}) \qquad (3)$$

The values of Q_{dr} and Q_{inf} generally are small in summer, indicating that in areas without deep well pumping equation (3) reduces to:

$$I_{ss}^{net} = v_{phr} \qquad (4)$$

where v_{phr} is the intensity of vertical flow through the phreatic surface taken positive for upward flow.

Model calculations in regional studies of the water balance of the unsaturated zone, using soil physical data have been performed by Rijtema and Bon (1974) using the equation:

$$\int_{t_1}^{t_2} P dt - \int_{t_1}^{t_2} E dt + \int_{t_1}^{t_2} v_{phr} dt = \int_{t_1}^{t_2} \int_{0}^{Z} \frac{\delta\theta}{\delta t} dz dt \qquad (5)$$

where:

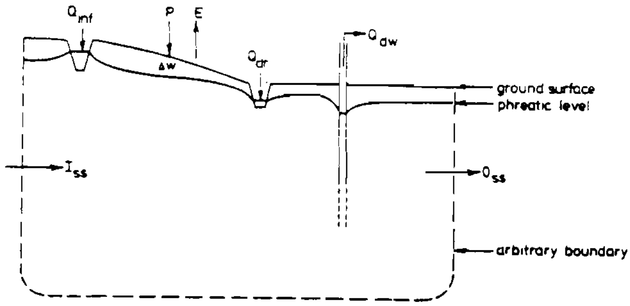

Fig. 2. Aquifer with the inflow and outflow components as given in eq. (1)

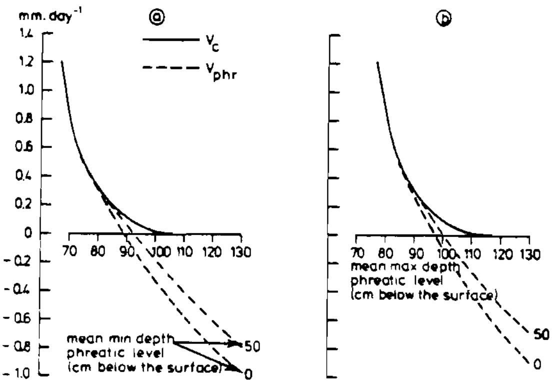

Fig. 3. Maximum capillary rise (v_c), mean maximum depth of the phreatic level and the value of v_{phr} in relation with the mean maximum and mean minimum depth of the phreatic level.
a: humic topsoil 20 cm; b humic topsoil 30 cm

z = the height above the deepest summer watertable;

θ = the volume fraction of soil water as a function of z and t.

During the summer period v_{phr} can be replaced by I_{ss}^{net}. With known soil physical properties and available climatological data the depth of the phreatic level can be calculated during this summer period.

Some data of the relation between the maximum depth of the phreatic level under mean meteorological conditions and the value of v_{phr} for pleistocene sand are given in fig. 3. The maximum values for capillary rise (v_c) also are given. The calculated values of v_{phr} approach these v_c-values when v_{phr} is positive. The curves are slightly dependent on the depth of the phreatic level in winter. When the maximum depth of the phreatic level under mean meteorological conditions is known from soil surveys, and used to describe the hydrological conditions, this information can be used to determine the regional distribution of v_{phr} during the summer, which in its turn quantifies I_{ss}^{net}. The values of Q_{dr}/A can now be calculated with equation (2) for the winter period.

Surface runoff is not only present when the infiltration rate of the topsoil is less than the intensity of the precipitation, but also when due to poor drainage conditions the phreatic level rises to the soil surface. It appears from data given by Steenvoorden and Buitendijk (1980) that surface runoff varies from year to year, depending on the distribution of the precipitation. Some results of their calculations are given in table 1.

Rough estimates of surface runoff can be made with the calculations performed by Rijtema and Bon (1974). This approach is useful to transfer a hydrological classification system, as used in soil surveys, into estimated values of real hydrological data. The results of this type

Table 1. Total surface runoff (mm) during the period 1 September 1961 to 1 June 1962 for soil with a drainage depth of 70 cm and a drainage intensity of 3, 5 and 8 $mm.day^{-1}$ at a phreatic level of 20 cm below the ground surface (Steenvoorden and Buitendijk, 1980)

	Drainage intensity ($mm.day^{-1}$)		
	3	5	8
Surface runoff (mm)	118	69	30

Table 2. Average hydrological data, as derived from a soil survey classification system for Pleistocene sand

Hydro-logical class	Humic topsoil cm.	Phreatic level mean min. cm	mean max. surface	E mm	I_{ss}^{net} $mm.day^{-1}$	Q_{dr}/A $mm.year^{-1}$	Q_{sr}/A
1	20	0	60	530	+ 0.8	542	163
2	20	0	70	510	+ 0.5	453	124
3	30	20	90	490	+ 0.3	400	86
4	30	40	90	500	+ 0.4	426	53
5	30	30	120	447	- 0.4	187	-
6	40	60	140	422	- 0.9	30	-
7	40	90	160	415	- 1.0	-	-

of calculations for pleistocene sand are given in table 2.

The table indicates that soils in the hydrological classes 1 through 4 have positive values of I_{ss}^{net}, indicating that these soils will not pollute the groundwater aquifer. The discharge of the precipitation excess occurs by interflow through the rootzone and by surface runoff.

The soils in the hydrological classes 5 through 7 have negative values of I_{ss}^{net}, which indicates that these soils have a large potential to pollute the groundwater aquifer. The hydrological classes 1 through 4 generally will have a great influence on surface water quality, whereas the classes 5 through 7 give only a very small contribution to pollution.

5. IMPRCVEMENT OF LOCAL DRAINAGE

The main objective of improvement of local drainage is to lower the phreatic level in winter and during extremely wet periods. The order of magnitude of I_{ss}^{net} generally is not very much affected by local drainage. By improvement of drainage conditions on a larger scale a slight reduction of I_{ss}^{net} might be present, whereas it increases somewhat in other parts of the area. Areas with a positive value of I_{ss}^{net} will keep a positive value. The main effect of the improvement of local drainage will be a reduction in the amount of surface runoff and an increase of the discharge by interflow through the soil system. The resulting increase in residence time of the precipitation excess in the soil system results in an additional denitrification, reducing the ultimate nitrogen load on the surface water.

Some results of model calculations are given in fig. 4. Surface runoff decreases with increasing depth of the drainage basis. Data of the total nitrogen load to the surface waters in relation to drainage depth are given in fig. 5. The figure shows that improvement of drainage conditions gives a large reduction in the nitrogen load to the surface waters.

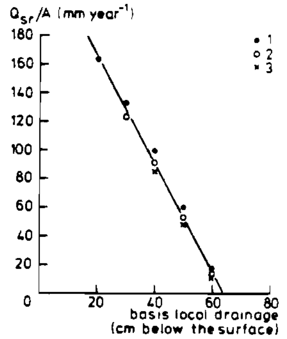

Fig. 4. Calculated relation between mean yearly surface runoff and depth of the basis of local drainage for the hydrological soil survey classes 1, 2 and 3

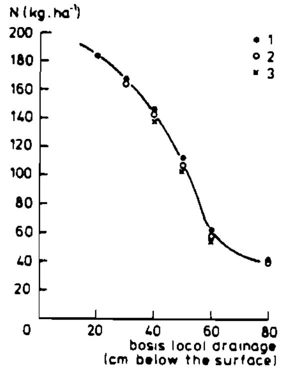

Fig. 5. Relation between calculated nitrogen load (kg. ha^{-1}) to surface waters and basis of local drainage for the hydrological soil surfey classes 1, 2 and 3. Assumed livestock density 3 SLU. ha^{-1}

6. EFFECTS OF DEEP WELL PUMPING ON GROUND AND SURFACE WATER POLLUTION

Each groundwater extraction results in a decline of the phreatic level. The pattern of drawdown of the phreatic level in the eastern and southern pleistocene sands in the Netherlands depends on the presence of different types of drainage systems. Near the center of the intake area of the well the drainage system does not function anymore, whereas at greater distances from the center the drainage system still operates.

Ernst (1971) gives a model for the calculation of the mean decline (ϕ) of the phreatic level in relation to the distance (r) from a deep well. The following relations have been used in the present calculations:

- For the area where the drainage channels are dry:

$$\phi(r) = \frac{\tilde{P}^+ (r_1^2 - r^2)}{4\,KH} + \frac{Q_{dw}}{2\pi KH} \ln \frac{r}{r_1} - \tilde{P}^+ (\gamma_e) \qquad r < r_1 \qquad (6)$$

- For the area, where the drainage channels contain water:

$$\phi(r) = -\frac{K_0(r/\xi)}{K_0(r_1/\xi)} \tilde{P}^+ (\gamma_e) \qquad r > r_1 \qquad (7)$$

The unknown quantity r_1 can be determined by the expression:

$$\frac{Q_{dw}}{\pi\xi^2 \tilde{P}^+} = \frac{r_1}{\xi} \left\{ \frac{r_1}{\xi} + 2 \frac{K_1(r_1/\xi)}{K_0(r_1/\xi)} \right\}$$

where:

$\phi(r)$ = decline in phreatic level;

Q_{dw} = constant extraction in $m^3 . day^{-1}$;

$\tilde{P}^+$ = mean precipitation excess in $m. day^{-1}$;

r = radius of the sphere of influence;

r_1 = maximum radius for the area with dry drainage channels;

K = horizontal permeability in m. day^{-1};

H = depth of the waterbearing in m. day^{-1};

KH = transmissivity in m^2. day^{-1};

γ_e = effective drainage resistance of the area in days;

ξ = $\sqrt{KH \gamma_e}$;

K_0, K_1 = modified zero and first order Bessel functions.

Transmissivity and deep well extraction data in East Gelderland were obtained from Water Company East Gelderland (W.O.G., 1973). Additional transmissivity and drainage resistance data were given by Ernst, De Ridder, and De Vries (1970). The data of 17 deep well locations have been used in the analysis of groundwater extraction effects on the hydrological regime. The basic hydrological data of the locations are given in table 3 and land use data in the intake area are given in table 4.

It is necessary to determine the environmental consequences when evaluating the possibilities of future extension of groundwater extraction for municipal water supply. The same holds for the choice of suitable locations. With the aid of the data of the existing deep wells the following questions can be answered:

a. the area of the sphere of influence to be expected;
b. the distribution of the decline of the phreatic level over the area;
c. the change in the net subsurface inflow over the area;
d. the nitrate inflow in the groundwater aquifer in relation to land use and hydrological conditions.

Sub a. The relation between the yearly capacity of the deep wells and the surface area of the sphere of influence with a decline of the

Table 3. Basic hydrological data of Q, Q_{dw}, KH, γ_e and r_1 for 17 deep well locations

Location	Capacity Q x $10^6 m^3$ per year	Q_{dw} m^3/day	KH m^2/day	γ_e days	$r_{1(one)}$ m
De Pol	3.5	9 600	1 000	675	1 068
Van Heek	3.7	10 200	1 500	600	996
Olden Eibergen	1.2	3 300	750	550	462
Lochem	3.0	8 200	2 000	550	682
Ruurlo	0.9	2 600	1 750	475	155
't Klooster	5.0	13 700	3 250	550	923
Harfsen	1.5	4 100	1 500	550	363
Vorden	5.0	13 700	3 750	550	804
Haarlo	1.7	4 700	1 250	525	486
Dinxperlo	2.0	5 500	3 000	575	236
Aalten	2.0	5 500	3 000	575	236
Baak	2.0	5 500	3 000	575	236
Stille Wold	4.0	11 000	3 000	550	719
Enghuizen	4.0	11 000	1 250	650	1 127
Gorssel	3.0	8 200	1 500	575	789
Noordwijkerveld	3.0	8 200	750	700	1 051
't Loohuis	2.0	5 500	1 000	575	652

Table 4. Distribution of land use in the intake area of the deep well locations

Location	Land use in percent								
	forest	urban area	across* rivers	hydrological classification					
				7	6	5	4	3	2
				arable land		grassland			
De Pol	13.5	12.0	19.4	18.2	10.8	13.6		10.8	1.7
Van Heek	100.0	-	-	-	-	-	-	-	-
Olden Eibergen	6.9	-	-	12.6	5.4	36.4	-	38.1	0.6
Lochem	21.8	25.9	8.0	14.9	10.7	11.0	-	7.7	-
Ruurlo	14.7	25.0	-	35.8	8.9	14.3	-	1.3	-
't Klooster	17.7	-	-	10.6	22.9	30.7	-	18.1	-
Harfsen	9.3	5.9	-	18.1	34.9	8.0	-	23.6	0.2
Vorden	48.1	0.6	-	13.0	12.0	13.5	-	38.8	2.9
Dinxperlo	1.8	1.5	6.5	17.9	28.7	33.6	-	9.9	0.1
Aalten	6.2	-	-	24.3	12.3	41.6	-	15.6	-
Baak	4.3	-	-	22.6	9.8	1.8	20.6	37.6	3.3
Stille Wold	9.8	4.8	-	35.2	18.9	9.6	-	20.2	1.5
Enghuizen	21.1	2.9	2.9	34.0	11.6	3.0	-	24.0	0.5
Gorssel	46.5	2.7	-	25.3	5.0	-	-	18.8	1.7
Noordwijkerveld	11.5	-	3.8	7.5	14.4	28.4	-	34.2	0.4
't Loohuis	0.7	5.5	-	24.3	12.3	41.6	-	15.6	-

* This column contains the area situated at the opposite site of rivers influenced by groundwater pumping

phreatic level of more than 0. 05 m is given in fig. 6 for 17 locations. The relation appears to be reasonably linear, with a slope of 6. 38 m^2 per m^3 groundwater extraction. This indicates that about 50 percent of the mean precipitation excess is used for feeding the deep well extraction.

Sub b. The distribution of the decline in the phreatic level over the area is not only dependent on the yearly capacity of the deep wells, but also on the transmissivity (KH) of the aquifer and on the drainage resistance (γ_e). Fig. 7 gives the relation between the percentage of the affected area exceeding a certain decline of the phreatic level and the value of $Q/KH\gamma_e$. The area with a decline of more than 0. 05 m is taken as 100 percent. Future locations can be chosen, with the aid of transmissivity and drainage resistance maps of the region.

Sub c. The effect of the mean decline of the phreatic level on the mean highest winter groundwater table and the mean deepest summer groundwater table had been analyzed by Rijtema and Bon (1974). These effects can also be described in terms of I_{ss}^{net}, Q_{dr} and Q_{sr}, giving a hydrological classification for the situation with groundwater extraction. The results of these calculations are given in table 5.

The data show that the amplitude of the groundwater table fluctuation grows with increasing decline of the mean phreatic level. It also appears that the value of the net subsurface inflow (I_{ss}^{net}) decreases and hydrological classes with positive values of I_{ss}^{net} in the original situation obtain negative values with increasing decline of the phreatic level. Consequently, the values of Q_{dr} and Q_{sr} sharply decrease. The data show that soils not contributing to the quality of the groundwater in the

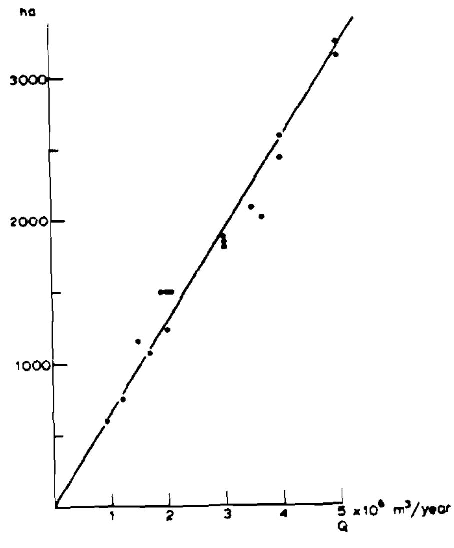

Fig. 6. Relation between extraction per year for 17 deep well locations and the surface area in which the decline of the phreatic level exceeds 0.05 m

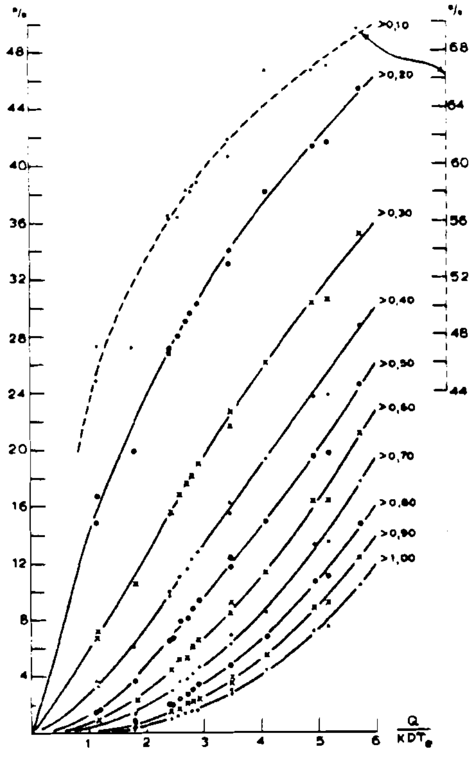

Fig. 7. Relation between percentage of the affected area with a decline of the phreatic level exceeding 0. 10, 0. 20, 0. 30, 0. 40, 0. 50, 0, 60, 0. 70, 0. 80, 0. 90 and 1. 00 m (area exceeding 0. 05 m = 100%) and the value of $Q/KH\gamma_e$ for 17 locations

Table 5. Influence of the decline in phreatic level by deep well pumping on the mean minimum and mean maximum depth of the groundwater table, I_{ss}^{net}, Q_{dr}/A and Q_{sr}/A for the different hydrological classes

Mean decline phreatic level cm	Mean minimum groundwater table cm-surface	Mean maximum groundwater table cm-surface	I_{ss}^{net} mm. day^{-1}	Q_{dr}/A mm. year^{-1}	Q_{sr}/A mm. year
Class 1					
0	0	60	+ 0.8	542	163
20	10	90	+ 0.1	350	136
40	22	118	- 0.5	127	27
60	36	144	- 1.0	-	-
80	50	170	- 1.1	-	-
100	64	196	- 1.1	-	-
120	80	220	- 1.1	-	-
Class 2					
0	0	70	+ 0.5	453	124
20	11	99	- 0.2	179	74
40	24	126	- 0.6	114	6
60	39	151	- 1.0	-	-
80	54	176	- 1.1	-	-
100	69	200	- 1.1	-	-
120	86	224	- 1.1	-	-
Class 3					
0	20	90	+ 0.3	400	86
20	35	115	- 0.3	188	32
40	49	141	- 0.7	85	-
60	65	165	- 0.9	30	-
80	81	189	- 1.0	-	-
100	98	211	- 1.0	-	-
120	107	233	- 1.1	-	-

Table 5. (sequel)

Mean decline phreatic level cm	Mean minimum groundwater table cm-surface	Mean maximum groundwater table cm-surface	I^{net}_{ss} mm. day^{-1}	Q_{dr}/A mm. year^{-1}	Q_{sr}/A mm. year
Class 4					
0	40	90	+ 0.4	426	53
20	54	116	- 0.2	232	21
40	68	142	- 0.6	120	-
60	83	167	- 0.8	30	-
80	98	192	- 1.0	-	-
100	115	215	- 1.0	-	-
120	133	237	- 1.1	-	-
Class 5					
0	30	120	- 0.4	187	-
20	41	144	- 0.8	55	-
40	63	167	- 1.0	-	-
60	81	189	- 1.0	-	-
80	100	210	- 1.0	-	-
100	119	231	- 1.0	-	-
120	138	252	- 1.0	-	-
Class 6					
0	60	140	- 0.9	30	-
20	78	162	- 1.0	-	-
40	97	183	- 1.0	-	-
60	116	204	- 1.0	-	-
80	135	225	- 1.0	-	-
100	154	246	- 1.0	-	-
120	174	266	- 1.0	-	-
Class 7					
0	90	160	- 1.0	-	-
20	110	180	- 1.0	-	-
40	130	200	- 1.0	-	-
60	150	220	- 1.0	-	-
80	170	240	- 1.0	-	-
100	190	260	- 1.0	-	-
120	220	280	- 1.0	-	-

aquifer of the original situation, are contributing in the situation with groundwater extraction. With the aid of soil survey classification data in the original situation, values of I_{ss}^{net} for the situation with groundwater extraction can be calculated.

Sub d. Estimates of the nitrate content of the precipitation excess feeding the groundwater aquifer can be made with the date derived under a, b and c. These data are to be combined with data on land use and agricultural intensity. The latter must be expressed in terms of fertilizer application and slurry production. The estimated N-inflow in the groudwater aquifer for the 17 existing deep wells is given in fig. 8. Based on the given discussion of some N concentration data in the shallow groundwater, it is assumed for the present calculations that the inflow concentration from grasslands equals 10.7 $g.m^{-3}$ N, arable land 36.0 $g.m^{-3}$ N, forests 2.5 $g.m^{-3}$ N and urban areas 10.0 $g.m^{-3}$ N. The mean nitrate concentration of the precipitation excess in the intake area is given in relation with the percentage of arable land. The total agricultural land use is also indicated. The scatter in the data is mainly caused by the variation in hydrological conditions in the intake areas.

The nitrate inflow will become a much more serious problem in regions with very high livestock densities, where dumping of slurry is more or less regularly practised on fields used for the production of fodder maize.

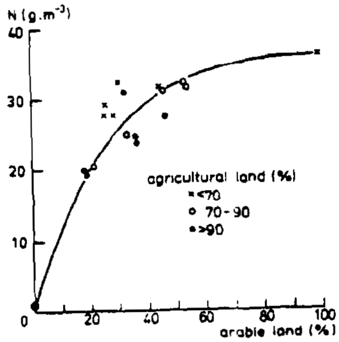

Fig. 8. Relation between nitrate inflow in g. m^{-3} in the groundwater aquifer and percentage of arable land in the intake area of 17 deep well locations

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List of used greek letters and other symbols

Δ	=	delta capital
δ	=	delta undercast
θ	=	thêta undercast
ϕ	=	phi undercast
γ	=	gamma undercast
π	=	pi undercast
ξ	=	xi undercast

r_1	=	r_{one}
$mm. day^{-1}$	=	$mm. day^{-one}$
K_0	=	K_{zero}
K_1	=	K_{one}

HYDROGEOLOGICAL ASPECTS OF GROUNDWATER NITRIFICATION

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With respect to their nitrogen budget, surface and subsurface waters constitute two separate subsystems, and the processes occurring in these subsystems can be described independently of each other, while their potential interconnections can be characterized by simple input-output relationships.

The accumulation of nitrogen in subsurface waters is the result of complex and intermittent biochemical and chemical processes closely related to the periodical variation of the groundwater flow and the hydrogeological conditions.

Independent of the form in which nitrogen enters the subsurface zones, the first, and determinative part of the nitrogen dynamics takes place in the soil profile. The soil zone constitutes an independent nitrogen system in which the processes occur in complex interaction.

Along the transport route between the soil and the aquifer, under the conditions of unsaturated flow, and while in contact with adsorbing surfaces, the nitrogenous material is exposed to further chemical reactions, finally resulting in the further oxidation of ammonia. The unsaturated transport zone cannot be considered an independent subsystem, but the saturated zone, i.e. the groundwater, can be. The accumulation processes described, are for basic hydrogeological types of soil in the near-surface zone.

INTRODUCTION

As is evident from the literature and also on the basis of our experiences, streams are mainly polluted by nitrates via nonpoint source runoff from watersheds. There is a close connection between the increased use of nitrogen fertilizers and the nitrogen monitored in streams. The water quality however can be restored relatively quickly after removing the sources of pollution (Figure 1). There is no doubt that the excess nitrate originates from inappropriate fertilizer use. The nitrate export from cultivated areas could be prevented by improving the agricultural technologies currently in use especially by paying attention to hydrological and biological cycles and by using organic fertilizers instead of inorganic ones, or at least by putting them directly into the soil. This problem can and should only be solved by improving fertilizer application technology. Quite often however, when one problem is solved another is created, therefore, interference with the hydrological cycle could cause further problems.

There is direct interaction between the water courses and the system of aquifers only if the stream bed cuts into the aquifer. The water transfer between these two systems is controlled by well known hydrological laws. The quality of the water exchanged will depend on the controlling effect of the filter system, which develops along the interface of the two systems. Owing to the biochemical and physicochemical microfiltration occurring in this filter system, water of essentially different quality from that of the surface water will enter the aquifer, resulting in practically nitrogen-free inflow, due to the removal of nitrate and nitrite by this filter system. Nitrogen influx can only occur in the case of excessive ammonia pollution or due to some flaw in the filter system. It is very advantageous therefore to draw water from wells drilled in the embankments, thus making use of this efficient filter. In Hungary, along the Danube, the bank-filtered aquifer yields water of good quality from the river trough via the natural filter systems but not from the aquifers further away from the bank.

It follows from the above that with respect to its nitrogen budget, surface and subsurface waters constitute two separate subsystems, and the processes occurring in these subsystems can be described independently of each other, while their potential interactions can be characterized by simple input-output relationships.

The nitrification of groundwater is a somewhat separate problem from that of surface waters. In an area with little or no runoff, a part of the nitrogenous materials applied on the land surface will infiltrate vertically into the soil. It will

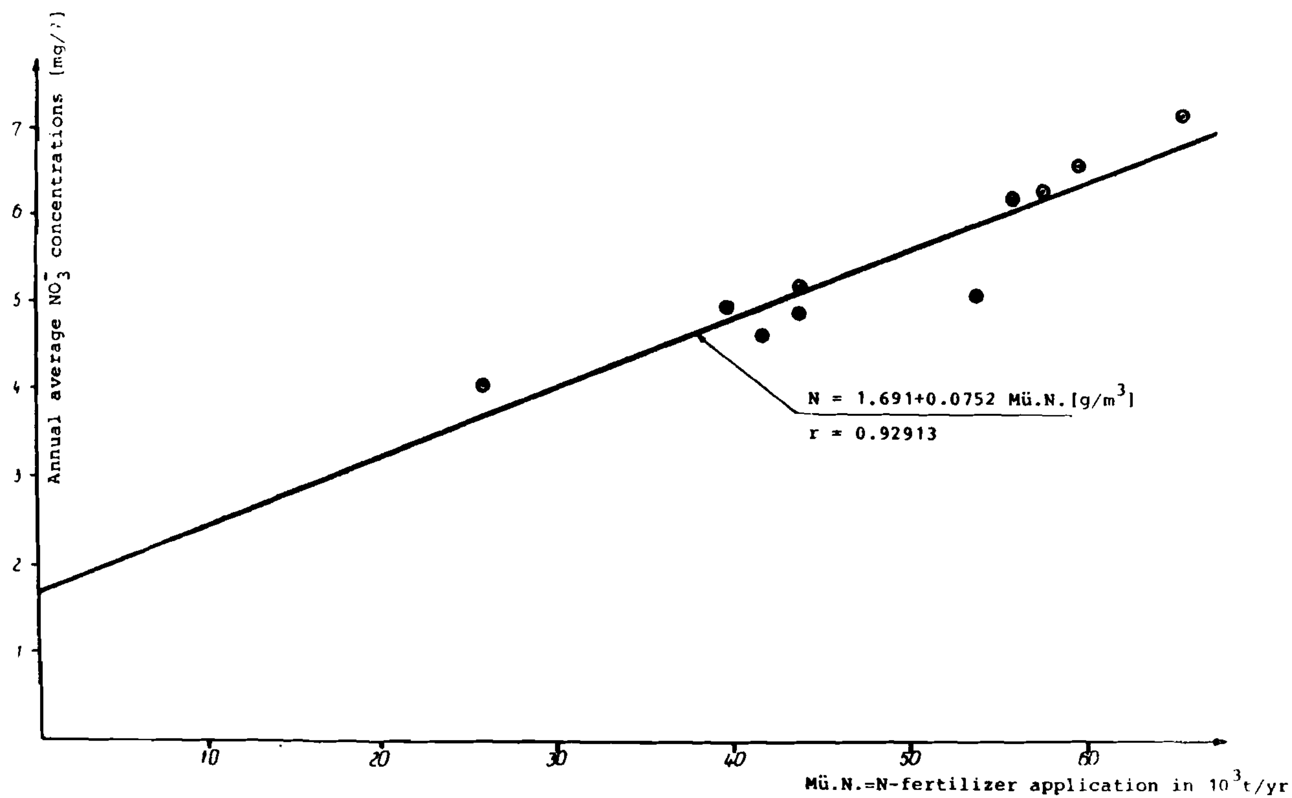

Figure 1. Relationship between annual average NO_3^- concentration values in the Tisza river, Hungary and the amount of N-fertilizers used on the corresponding watershed (Source: G. Jolankai, 1980)

partly be absorbed by the vegetation and partly accumulated in the subsurface waters, causing an increase in their nitrogen concentration.

The nitrogen accumulation in subsurface waters is not the result of a simple filtration process, but it is the result of complex and intermittent biochemical and chemical processes closely related to the periodical variations in the groundwater flow (Figure 2).

As a result of agricultural production technologies, organic materials and fertilizers with nitrogen content will be brought into the soil. The protein in the organic matter will decompose into carbamide which on further decomposition results in ammonia; the latter in the form of ammonia ions, dissolves in water; however, depending on temperature and pH values, free ammonia might also occur. Both forms of ammonia are well adsorbed by the soil and are readily utilized by plants, or they are oxidized by bacteria into nitrate.

Among the fertilizers most commonly used, ammonium nitrates have the highest nitrogen content. In water they dissociate into ammonium ion and nitrate ion, both with a nitrogen content. The nitrogen content in carbamide, ammonium phosphates and sulphates is only half that in ammonium nitrate. In addition to this, the decomposition of ammonia into nitrate largely depends on temperature conditions. In experiments (Literathy and Pintér, 1979) carried out with Danube water, the ammonia-nitrate process begins only above 15°C temperature (Figure 3). It can be assumed that in the soil, this process starts at temperature levels some degrees lower than that, however one may take it as certain, that during periods with a moderate climate and during the cool or cold periods, the ammonia oxidation process is interrupted, and the ammonia gets fixed on the surface of the generally highly absorbent soil particles, in the period when filtration across the soil profile is the most marked, due to the cessation of water utilization by plants. Biological investigations have unambiguously proved that ammonification, that is, the generation of ammonia from organic materials, continues at lower temperatures in the unfrozen soil zones, even during the winter, consequently, only as the result of excessive fertilizer use will the adsorption capacity of the soil become exhausted. This would then result in pollution of groundwater by ammonia. It is clear that the magnitude of nitrogen pollution depends also on the type of fertilizers used and the nitrogen-soil interactions.

NITROGEN-SOIL INTERACTIONS

Independent of the form in which nitrogen gets into the groundwater, the complex processes of nitrogen dynamics take place in the soil profile. The nitrogen dissolves in soil moisture, and is adsorbed on the surface of soil particles;

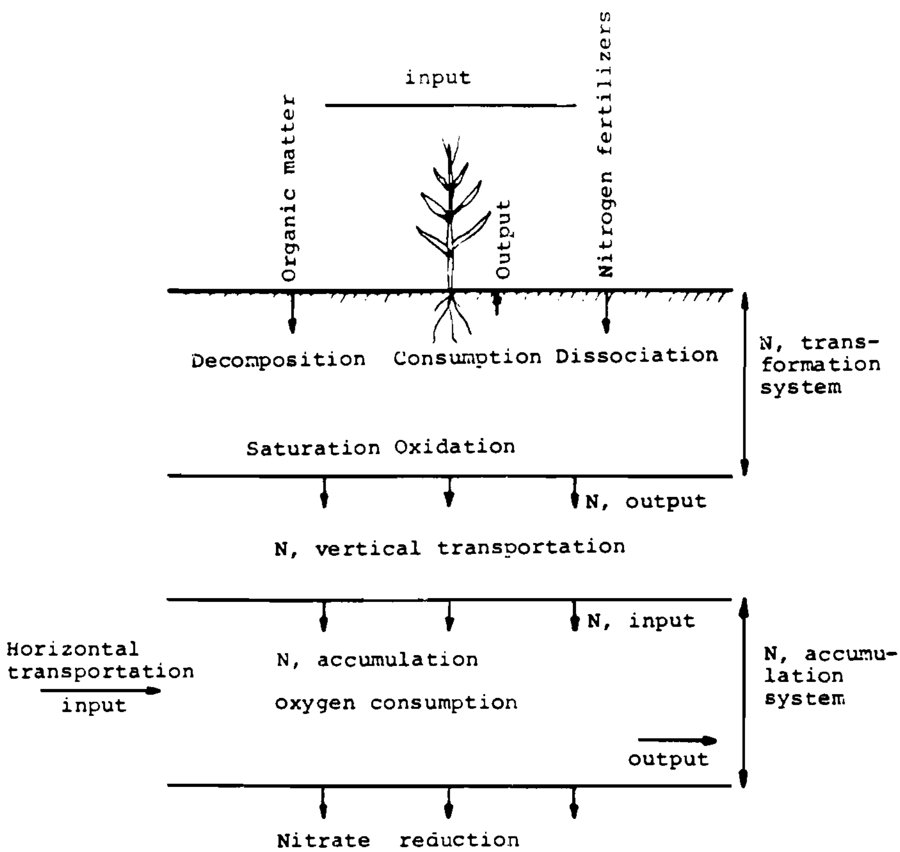

Figure 2 The process of nitrogen accumulation under the surface

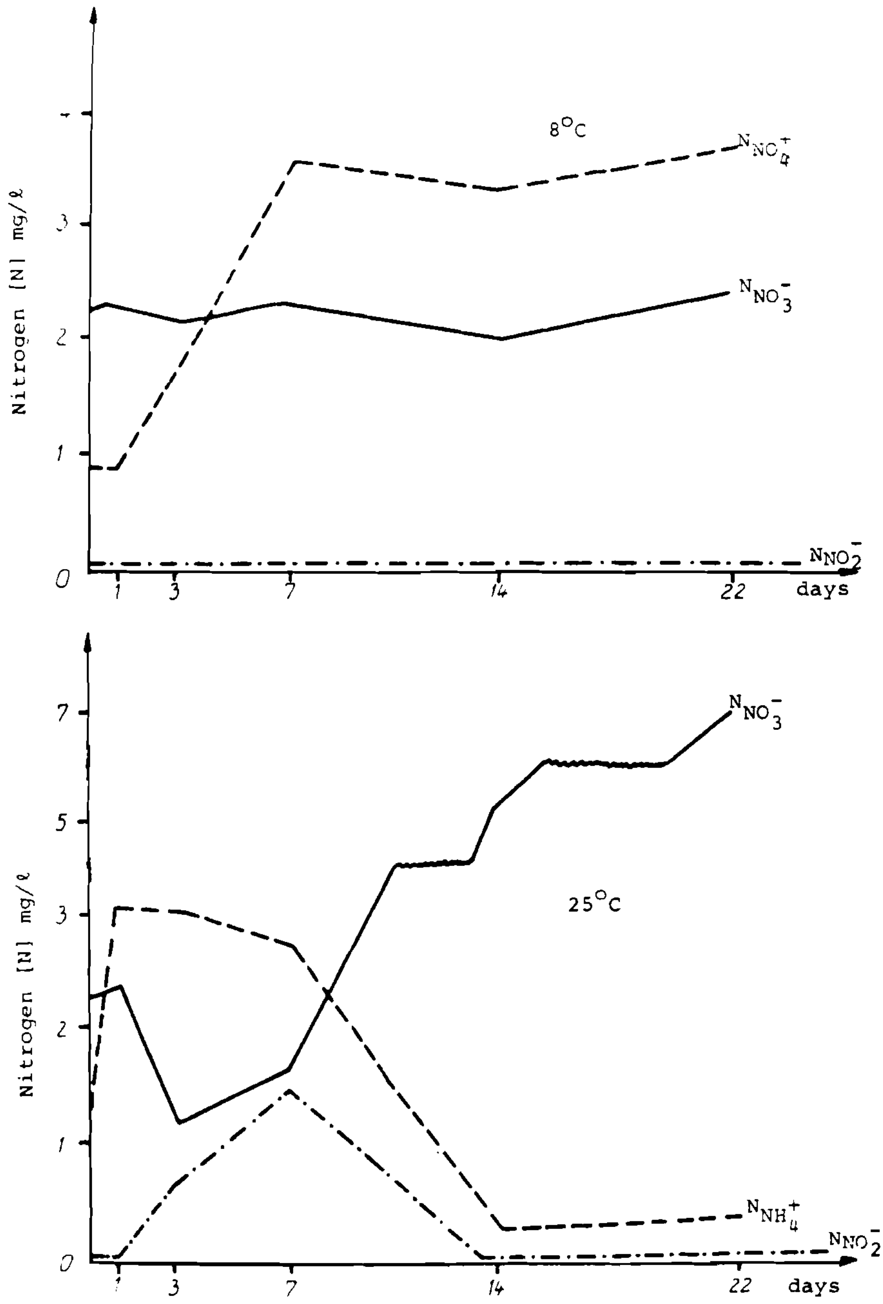

Figure 3. Influence of H_2O temperature on Nitrogen forms (Danube water samples), after Literathy, 1971

oxidation then takes place in this zone and finally plants utilize it. The soil zone constitutes an independent nitrogen budget in which the above processes occur in complex interaction. As is known, the nitrogen balance is determined by the meteorological conditions and the physical parameters of the soil, the biochemical conditions, and the type and form of the substances through which the nitrogen was brought into the soil.

Processes which affect the nitrogen dynamics are governed by the seasonal biological rhythm. It follows that the effects of meteorological factors are also significant. The occurrence of nitrogen below the ground surface is decisively determined by hydrogeological and geochemical conditions. In this context, the following basic hydrogeological types can be differentiated in the near-surface zone of influx:

(a) soils of self-contained water budget;

(b) soils with water budgets affected by subsurface waters;

(c) soils of independent water budgets.

Soil zones with self-contained water budgets are those which do not receive additional water from the subsurface aquifers by capillarity (Figure 4). Such soil zones are those covering gravel layers, karstic carbonate formations, or chalk rocks of irregular structure. Under arid and semi-arid conditions, downward water transfer rarely occurs in such self-contained soil zones, but it occurs along the erosion pathways only in the case of heavy rainfall. It follows from the above that under such conditions, the annual average of nitrogen transport is also of negligible quantity. Under humid conditions, systematic water transfer takes place, while mobilization of the unutilized, stored nitrogen in the soil may also be subsequently transported by the water which has infiltrated into the subsurface aquifers. This nitrogen output could be minimized by improved agricultural technology.

Along the transport route through the soil to the aquifer, in the conditions prevalent in the unsaturated zone, and while in contact with adsorbing surfaces, the nitrogen undergoes chemical changes resulting in the further oxidation of ammonia. The geochemical character of this thick unsaturated zone is the so-called open oxidant system (Champ et al., 1979) in which excess oxygen is present, and the additional supply of oxygen is assured.

In the upper part of the saturated zone with thick gravel layers and below the soils with self-contained water budgets, the water flows along the hydraulic gradient; thus the nitrate and other dissolved pollutants are transported in a lateral direction. In these zones nitrate reduction does not take place; this results in the continuous accumulation of nitrate.

In the saturated zone, i.e., in the groundwater, there is a closed oxidant system where dissolved organic carbon (DOC) is always present and which gradually exhausts the oxygen content.

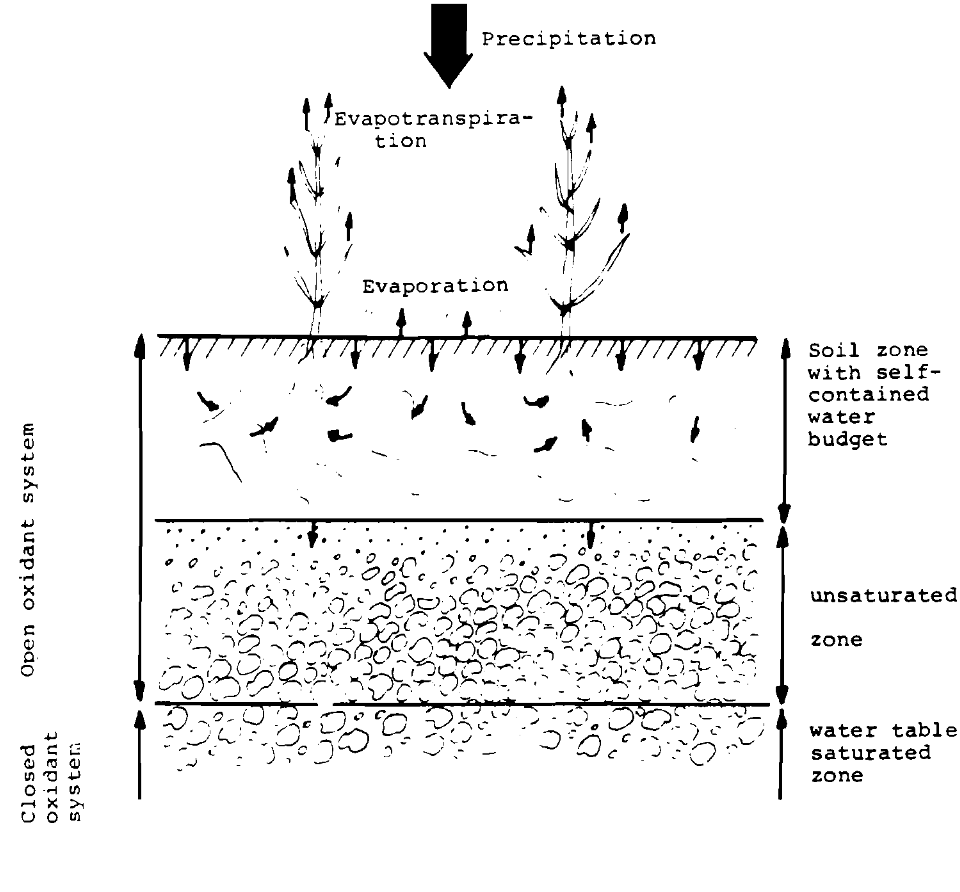

Figure 4. Soils with self-contained water budget (Type A)

Following a downward direction, the reduction system develops after the oxygen has been utilized, and according to Champ et al., (1979), the reductive decomposition of NO_3 will occur first. In the deep alluvial zones, the reduction process is further increased by the reduction of bivalent Fe^{++}, originating from organic decomposition, into a trivalent compound.

Processes taking place in the transport zone will not transform the quantity of nitrogen that has entered the unsaturated transport zone, and the nitrogen originating from the soil zone will be transported into the saturated zone with a rate that depends on the thickness and physical parameters of the transport zone; consequently the unsaturated transport zone can not be considered an independent subsystem.

The water budget of near surface soil zones is not self-contained in flat lands with fine sand, loess, chalk or other formations where the increase by capillarity is in the order of magnitude of one meter, as in periods without precipitation, the water leaving the system via evapotranspiration will be replaced from below due to the effects of capillarity. This situation can be considered the second basic hydrogeological group B (Figure 5).

In this system the excess water in the soil zone will infiltrate into the groundwater with a time lag, as the water filling the pores has to be displaced first. In periods of poor precipitation or without precipitation, the capillary soil water will replace the water already consumed, thus inducing a reverse flow.

In arid and semi-arid conditions, the dynamic balance of the soil zone water budget will be maintained by the deficit in the groundwater balance and consequently the pollutants dissolved by the flowing water will not get into the deeper saturated zones. In humid areas or in irrigated lands, this process is reversed and the annual net balance might show a downward movement of the water.

In this system, the transport zone is thin and saturated and therefore only a part of the soil zone operates as an open oxidant system and the transport zone forms a closed oxidant system where the dissolved organic carbon (DOC) reduces the excess oxygen, while the water which moves downward still contains unreduced oxides. Moving downward, reduction will become characteristic, soon turning the system into a reducing one.

Water seepage in the system is slow in all directions and due to the thickness of the open oxidant system, reduction processes are so significant that they, depending on the capacity for reduction, may prevent nitrate flux into the deeper zones. During the increased activity of the reducing agents--caused by accelerated or excess nitrogen pollution--the extent of the closed oxidant zone increases, thus depressing the upper level of the reducing zone, and allowing the nitrate flux to permeate the deeper zones.

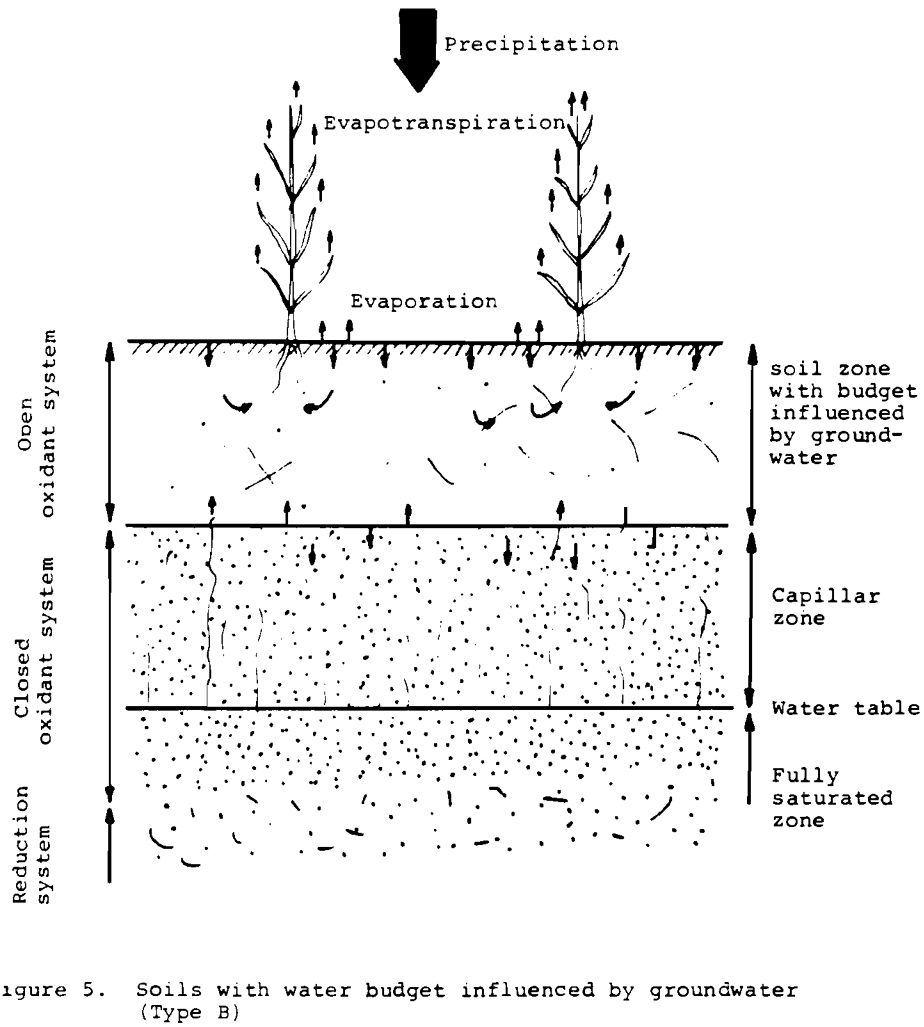

Figure 5. Soils with water budget influenced by groundwater (Type B)

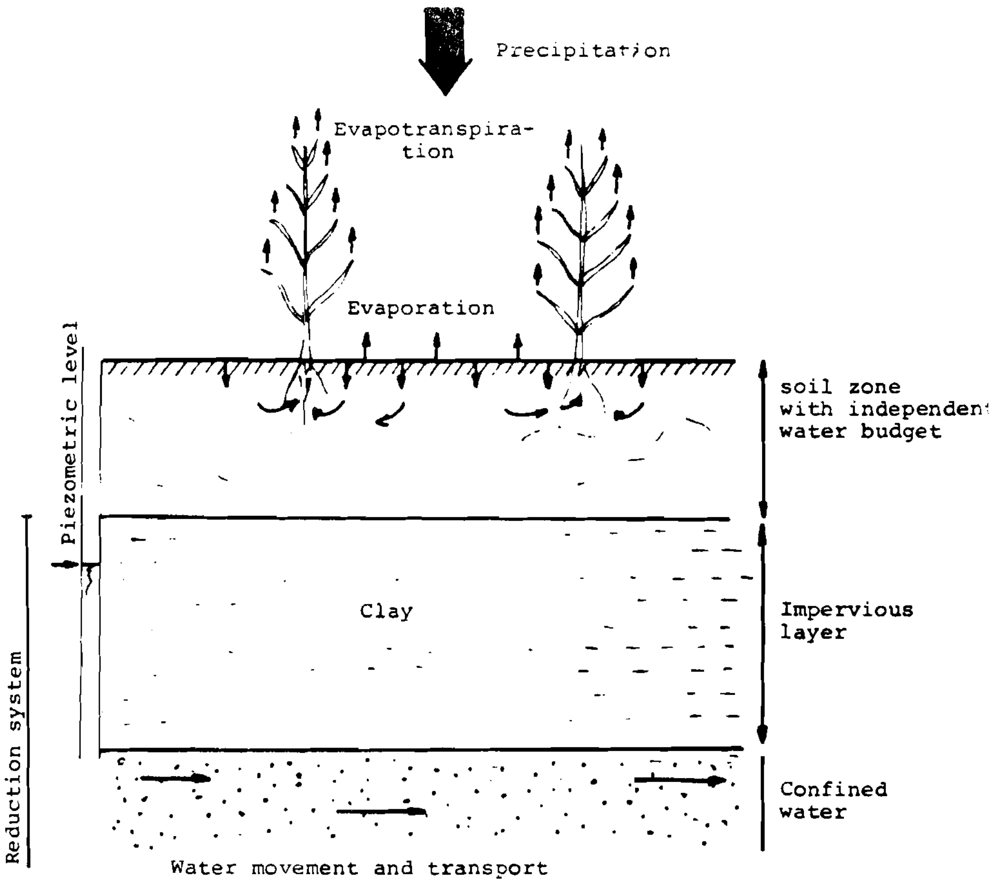

Figure 6. Soils with independent water budget

It readily follows from the above that in hydrogeological type "A" the transport zone is geochemically attached to the soil zone, while in type "B" it belongs to the saturated aquifer zone; this grouping is also justified hydrologically.

In the third hydrogeological group "C" (Figure 6), the first near-surface aquifer is covered by impervious rock formations and no rain water can get below the soil zone, i.e., the soil water budget is independent of the groundwater. It is thus obvious that under such conditions the aquifer cannot be polluted by dissolved pollutants via vertical transport, and only lateral transport might cause nitrogen pollution. The conditions of such systems are very similar to those of the confined aquifers.

Under natural conditions, no nitrogen oxides, originating from the atmosphere and from organic decomposition, accumulate in the near-surface aquifers. Nitrogen accumulation occurs, however, due to the excessive use of fertilizers and this might even be accelerated by other human activities. Under natural conditions, the downward movement of water towards the deeper aquifers is slow and the order of magnitude is 10 cms/yr., i.e., there is no danger of nitrate influx into the deeper layers. In developed countries, however, deep aquifers are also utilized, so the increased water production and resultant flow may also accelerate the downward flux of nitrate.

Actual hydrogeological and geochemical conditions are obviously much more complex than those illustrated by the three basic types described above. However, natural systems can be well represented with these basic structures and/or by their combinations.

CONCLUDING REMARKS

The nitrification of groundwater seems to be an irreversible process resulting in the accumulation of nitrate in groundwater, so the water polluted by nitrate will remain polluted for a long period of time, even after the nitrogen input has ceased.

The nitrification process could be or should be governed by control (optimization) of the nitrate input into the soil-plant system.

The key to the solution therefore lies in the improvement and control of agricultural practices.

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THE IMPACT OF NITROGEN FERTILIZER APPLICATION ON THE NITRATE CONCENTRATION IN GROUNDWATER: COST-BENEFIT ANALYSIS CONSIDERATIONS

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In this paper, a concept for cost-benefit analysis is developed for dealing with the impact of nitrogen fertilizer application on the nitrate concentration of groundwater. The discussion starts with some empirical evidence from data collected in the Federal Republic of Germany. After outlining basic issues of cost-benefit analyses in market economies, a proposal for empirical research is dealt with in greater detail. Major components of the concept are modeling nitrate flows in soils and aquifers. It is shown how those physical models have to be incorporated into economic approaches for estimating costs and optimal fertilizer adjustment. Finally, the paper makes some recommendations regarding the choice of policy instruments.

1. Nitrate Concentration in Groundwater - Some Empirical Evidence from the Federal Republic of Germany

As in many other countries[1] there is a growing concern in the Federal Republic of Germany (FRG) about the threats to human health of high nitrate concentration in drinking water (vgl. z.B. OBERMANN and BUNDERMANN, 1977). A recent study, based on water quality data for nearly 40 million people with samples taken between 1974 and 1977, indicates that 6.6 % of the population had access to water which exceeded the limit of 50 mg/l NO_3, recommended by the World Health Organization (AURAND et. al., 1978). Figure 1 indicates that 1 % of the population was supplied with water which even exceeded the high limit of 90 mg/l NO_3 which is currently effective in the FRG (Trinkwasserverordnung). Some countries maintain such low standards as 20 mg/l NO_3. According to the samples from FRG, nearly one third of the population receives water above this standard.

In so far as long run trends of the nitrate contents of drinking water have been measured, they indicate a steadily rising concentration (OBERMANN and BUNDERMANN, 1977; see also for the United Kingdom: OAKES, 1981).

Certainly, the contaminated water does not reach the consumers permanently. Yet, since one cannot exclude that even occasional or periodic high concentrations may threaten the health, especially of children, the data tend to give sufficient evidence for the urgency of measures against further nitrate concentrations.

Whether or not and to which extent the farm sector contributed to the observed nitrate pollution is a question which obviously requires further analysis. Nitrate concentrations may theoretically also come from the atmosphere or from reserves stored in the soil. Yet, a breakdown by size of water plants and by location does at least support the suspicion that the farm sector is a major cause of nitrate pollution since the limits are more frequently violated

1) Recent publications and their implications are summarized in GOLUBEV (1980) and in ZWIRNMANN (1981).

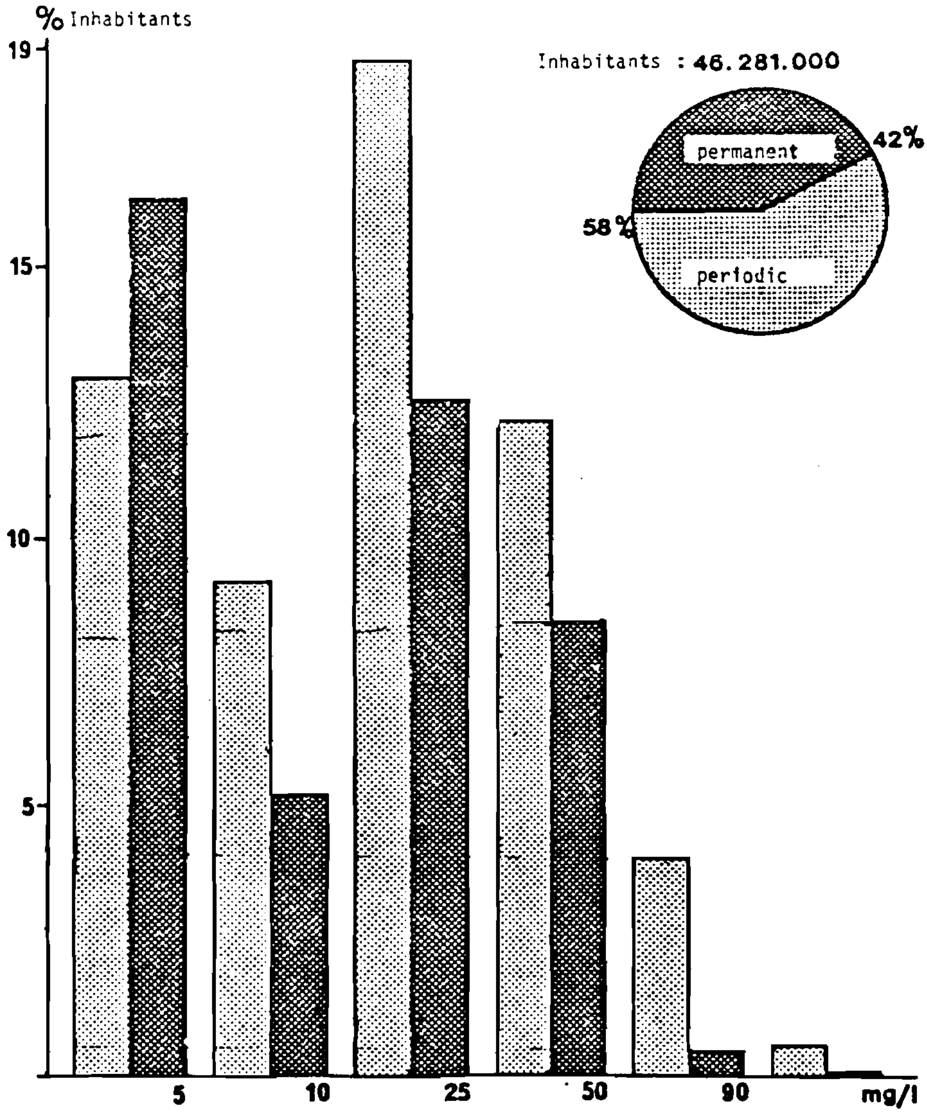

Figure 1: Distribution of Nitrate Content in Drinking Water

Source:Institut für Wasser-,Boden- und Lufthygiene (1979) in small water plants which are mostly located in rural areas (Table 1, Figure 2). Such plants are often supplied from agricultural areas rather than trans-regionally from mountainous forest covered watersheds.

Unfortunately, the debate about agriculture's role suffers from lack of information, especially about the quantitative relationship between the amount of nitrate leaching or runoff from a given field and the amount finally reaching the drinking water source at a certain distance. Those trying to keep agriculture out of the growing concern and the resulting call for nitrogen reducing measures emphasize that almost 1/3 of the drinking water supply comes from surface water with little connection to agricultural emissions and that groundwater supplies are taken from protected catchment areas ("Trinkwasserschutzgebiete") where intensive fertilizer application is often prohibited.

However, there are also reasons to be particularly concerned with agriculture's contribution to the nitrate pollution, namely in long-run perspective. This concern is substantiated by

- available empirical studies which demonstrate clearly a positive impact of fertilizer levels on nitrate concentration, even within protected water sheds (CZERATZKI, 1973; OBERMANN and BUNDERMANN, 1977).

- the expectation that nitrogen fertilizer application levels will continue to grow, in spite of declining real prices of farm products (as in the past). Increased fertilizer levels per hectare will - at standard application techniques and application frequencies - lead to increased proportions leaching into the groundwater.

- the perspective that the total size of catchment areas will have to be clearly expanded in order to meet the growth of water consumption as well as react to the rising share of drinking water taken from groundwater. According to available projections the total water utilization in the FRG may grow at an annual rate of 1.5 % from 30 to 44 billion m^3 between 1975 and 2000. The water used in private households is expected to grow even at a rate of 1.8 % per year[1].

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1) Projections by the Federal Ministry of the Interior (BATTELLE, 1972)

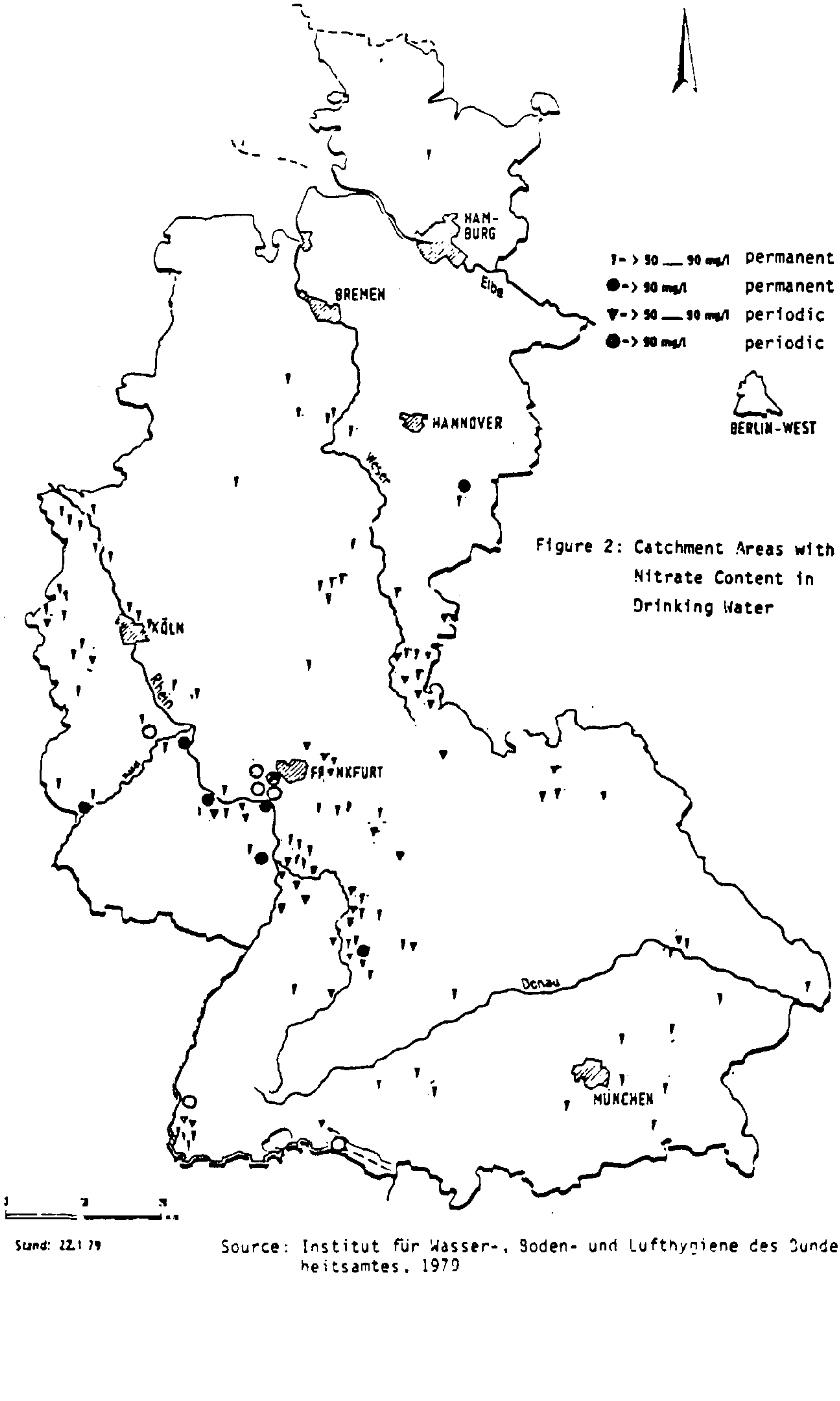

Figure 2: Catchment Areas with High Nitrate Content in Drinking Water

Source: Institut für Wasser-, Boden- und Lufthygiene des Bundesgesundheitsamtes, 1979

- the uncertainty about the underground water flows. It seems to be not at all evident that groundwater sources in catchment areas are not interconnected to more remote aquifers. Since some aquifers tend to recover very slowly from contamination one cannot exclude that nitrate is horizontally flowing from agricultural fields to water catchment areas over longer distances.

Table 1: Nitrate Content of Drinking Water by Size of Municipalities
- thousand inhabitants -

Size of Municipalities	Nitrate mg/l < 20	20 - 50	50 - 90	> 90
< 1	608	249	15	3
1 - 5	550	198	11	2
5 - 10	147	46	8	3
10 - 50	257	96	13	-
50 - 100	48	13	4	-
100 - 500	46	12	2	1
> 500	7	2	-	-

Source: Institut für Wasser-, Boden- und Lufthygiene, Bundesgesundheitsamt Berlin, 1979

Our conclusion from the available evidence is that nitrate pollution should be of concern to the farm sector, presently at least within the catchment areas, in future possibly in much more extended zones of agricultural production. Yet, the amount of nitrate leaching and nitrate flows cannot be known without specific empirical analyses. Moreover, recommendations concerning policies to reduce the nitrate concentration in drinking water require some fundamental welfare economic considerations.

2. Cost-Benefit Concept

2.1 The Basic Welfare Economic Problem

The basic welfare economic consideration can be expressed in the question: what would be the costs and the benefits of a reduced nitrate concentration? Benefits result from reduced dangers to human health, costs from measures to prevent a higher concentration. Since any reduction of the existing nitrate content is likely to create costs and since the marginal costs of nitrate reduction tend to increase while the marginal benefits would decrease with further nitrate reduction, standard welfare economics lead to suggest that there is something like a "socially optimal" degree of nitrate concentration, say P_0 in Figure 3. A nitrate concentration below P_0 would cause costs which exceed the benefits to the water consumer and vice versa. Hence, the fundamental

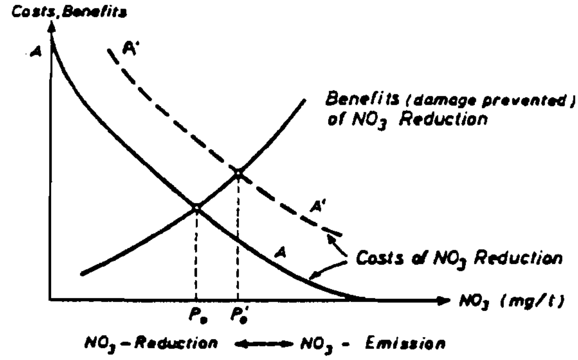

Figure 3: "Optimal" Nitrate Concentration in Drinking Water

economic rationale behind this simple consideration is the statement that pollution limits are not solely technical standards, but depend also on the social costs in terms of resources required and profits foregone to meet such limits. Situations with higher costs of installations for nitrate removals or higher agricultural prices and, hence, higher profits foregone from reduced fertilizer levels are represented by an upward shifted marginal cost curve say A' A' in Figure 3. A society in this situation might, compared to the marginal cost curve A A, consequently accept a higher nitrate limit in drinking water, for instance P_o'.

To quantify the benefits from a reduced nitrate content in drinking water is a rather difficult task involving at least two problems. One is the measurement of the effect on human health of alternative nitrate concentrations. This would not only require knowledge of the total daily nitrate/nitrite intake per person, which depends on the overall diet. It would also require a clear assessment of the adverse health effects of alternative nitrate concentrations, which are not sufficiently understood[1]. The other problem relates to the value

1) See the survey of related literature in ZWIRNMANN (1981).

of such effects. Even if the medical implications were known, their quantification in monetary terms touches the borderline of social science and is practically infeasible, unless some questionable value judgements (e.g. medical treatment costs saved) are used. In spite of these measurement difficulties there seems to be little disagreement among medical scientists that a reduced nitrate intake would reduce the threat to human health.

If one gives up the ambitious goal of finding something like an optimal pollution within the tradeoff between costs and benefits, then the economic consideration remains to take certain nitrate standards as exogenously given and to find least cost alternatives to meet them. Our analysis will be based on this approach. Once a certain nitrate limit is predetermined the question arises whether this limit should be met by avoiding the nitrate pollution at its origin, for instance by reducing fertilizer levels, or whether it should be met by removing the nitrate from the water at the plant. Technically, both ways are feasible[1]. The difficulty of cost comparisons is that not only direct costs but also external effects as well as administrative efforts and institutional implications have to be compared.

Moreover, the choice of the least cost alternative may depend on the level of the predetermined nitrate limit. Hypothetically, the profit foregone from fertilizer use (evaluated at appropriate shadow prices) may be higher than the costs of nitrate removal (or of blending with cleaner water) if the limit is set at 40 mg/l and it may be lower than the removal costs if the limit is set high, say 90 mg/l. This would result from a yield function with decreasing marginal product and an almost linear cost function of nitrate removal as demonstrated in the following Figure 4. If this hypothesis could be supported empirically the conclusion would be to reduce fertilizer levels if high limits are in effect, but to install technical equipment at the waterworks if the lower limit of 40 mg/l is to be implemented.

1) See the references given ZWIRNMANN (1981).

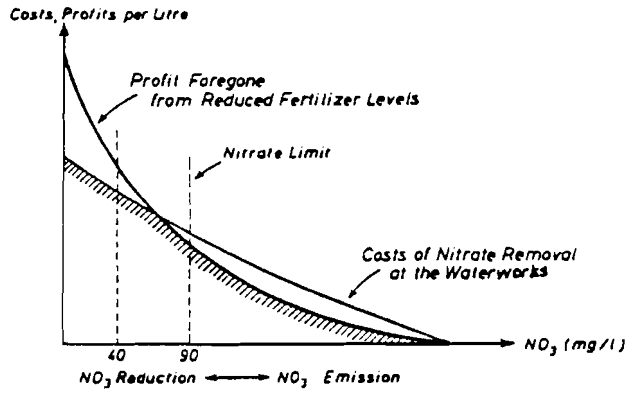

Figure 4: Least Cost Strategies to Reduce Nitrate in Drinking Water

2.2 Social Versus Private Costs

Welfare economic consideration are based on the concept of social as opposed to private costs and benefits. This distinction is in two ways relevant for the assessment of a cost comparison among alternatives. One relates to divergencies between market prices and economic values. In the European Community, for example, many farm prices clearly exceed the level at which the society could import farm products, hence farmers receive a price subsidy. As a consequence, social costs (income foregone) of a fertilizer reduction would be lower than the private income losses of the farmers. This is demonstrated in Figure 5. It is assumed that farmers are currently realizing a nitrogen fertilizer level N (α) with a related concentration, α, of leached nitrate. If they were to reduce this level to N (β) in order to lower the nitrate content of groundwater to (β; $\beta < \alpha$), they would suffer an income loss of Y_F. The loss of national income, however, would only amount to Y_S. In other words, the implementation of a nitrate reducing policy at the farm level may be much more in the interest of the society than of the farm population and possibly require additional compensating payments to avoid those adverse income effects.

The other aspect of the distinction between social and private costs is related to possible external effects of alternative nitrate reducing measures. For instance, a reduction of nitrogen fertilizer levels as opposed to nitrate

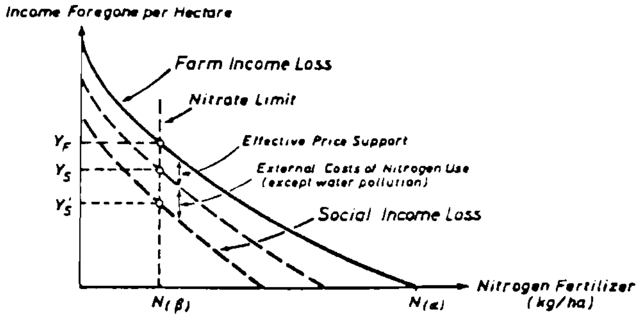

Figure 5: Private and Social Costs of Fertilizer Reduction[1)]

removal at the waterworks might avoid direct toxic effects of nitrates or nitrites in food or feed. Such effects through nitrosamines are increasingly considered as another threat to human health.
If quantification were possible this would again appear as a lowering of the curve of social costs of fertilizer reduction without affecting the private cost curve.

3. A Concept for Empirical Research

In order to assess cost functions of alternative ways to reduce the nitrate content of drinking water, both the empirical problem of measuring physical input - output as well as input - pollution relationships and the problem of assigning prices to those physical effects have to be solved. Among them, the estimation of physical relationships is most problematic.

It would be beyond the scope of this paper to discuss the technical alternatives of reducing the nitrate content of water in the waterworks. They include biochemical and physicochemical procedures on the one hand and the development of new water supplies or the blending of various water qualities on the other hand[2)]. In spite of remaining technical problems rough cost estimates of these alternatives should in principle be possible.

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1) This concept is based on the assumption that agricultural prices are not affected by the respective anti-pollution measures. Hence, the economic costs exclude losses of consumer surplus. This is a realistic simplification where either the region where such policies are implemented is relatively small or where the respective country is small compared to the world market.

2) ZWIRNMANN (1981) discusses alternative techniques and mentions related references.

The estimation of the impact of agricultural production on water pollution is much more complex, especially since it is a typical nonpoint pollution where any localization of sources emitting nitrate is rather difficult. Yet, if one wants to include a reduction of fertilizer levels in a cost comparison of alternatives to reduce nitrate concentrations in drinking water, one has to estimate the impact of nitrogen use in farm production on the nitrate content in drinking water. Methodologically, this requires a two-step procedure, namely the simulation of the vertical flow of nitrates from the field surface to the groundwater zones and the simulation of horizontal flows, possibly over far distances.

Both steps, but mainly the second, leave a wide range of open questions. The following is not an attempt to present a complete model or even results of such a model. It may rather serve as an outline for a research concept and framework for a detailed discussion of methodological procedures.

3.1 Modelling Vertical Nitrate Flows

The purpose of this submodel would be to simulate the amount of nitrates leached under alternative soil and climate conditions, fertilizer levels and cropping systems. Not all nitrates not accounted for in the crop yields contribute to the pollution of drinking water. Depending on the soil condition, on the reductive capacity of the ground as well as on the surface water system a considerable proportion is leaving the ground through runoff and through chemical and biological denitrification. OBERMANN and BUNDERMANN, 1977), for instance found that within their catchment study area, 50 % of the nitrate intake leaves the aquifer as N_2 gas. Much of this process takes place during the vertical water flow and, hence, has to be accounted for by the related submodel.

Various models seem to exist which simulate this vertical flow of nitrates. An overview is given in HAITH (1980) and ZWIRNMANN (1981). Not all of them were empirically tested against direct observation. Such tests require field level information about cropping patterns, fertilizer levels and frequencies, soil and climate data as well as nitrate concentrations in groundwater at various depths. What is finally needed for further cost calculations is a Nitrate

Leaching Matrix (Figure 6) allowing assessment of the nitrate leaching for whole farming systems under alternative fertilizer levels and exogenous soil and climate factors:

Nitrogen Fertilizer Intensity (kg/ha)	Nitrates Leached in kg/ha (mg/l) Soil and Climate Type 1				...	Soil and Climate Type n			
	Land Use Activity					Land Use Activity			
	Grains	Vege-tables	Grass-land	...		Grains	Vege-tables	Grass-land	...
0									
40									
80									
.									
.									
.									

Figure 6: Schematic Nitrate Leaching Matrix

In the FRG we are bound to experiments in a few catchment areas, where we are currently in the process of validating the CREAMS model[1], described elsewhere (USDA). The model gives results for intra-year seasonal quantities of leaching water and nitrates, depending on the respective crop, the leaf area index, weather and soil variables.

To exemplify results, simulations of nitrate leaching for grassland and vegetables, both for two levels of nitrogen fertilizer, are presented in Figure 7 and in Table 2. The data relate to a catchment area in Northwest Germany with a sandy and stony aquifer and mixed agricultural production. This first very preliminary test of the model was only possible for average yearly levels of nitrate concentration in groundwater underneath the root zone (Table 3). It is quite satisfactory for grassland, but does not match well the nitrate leaching under field crops.

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1) The model is being worked out by Mrs. Sabine STEMMLER, Institut für Agrarökonomie.

Table 2: Simulation of NO_3-leaching to groundwater with the CREAMS-model (preliminary results) - yearly averages in mg NO_3/l deep percolated water[1) -

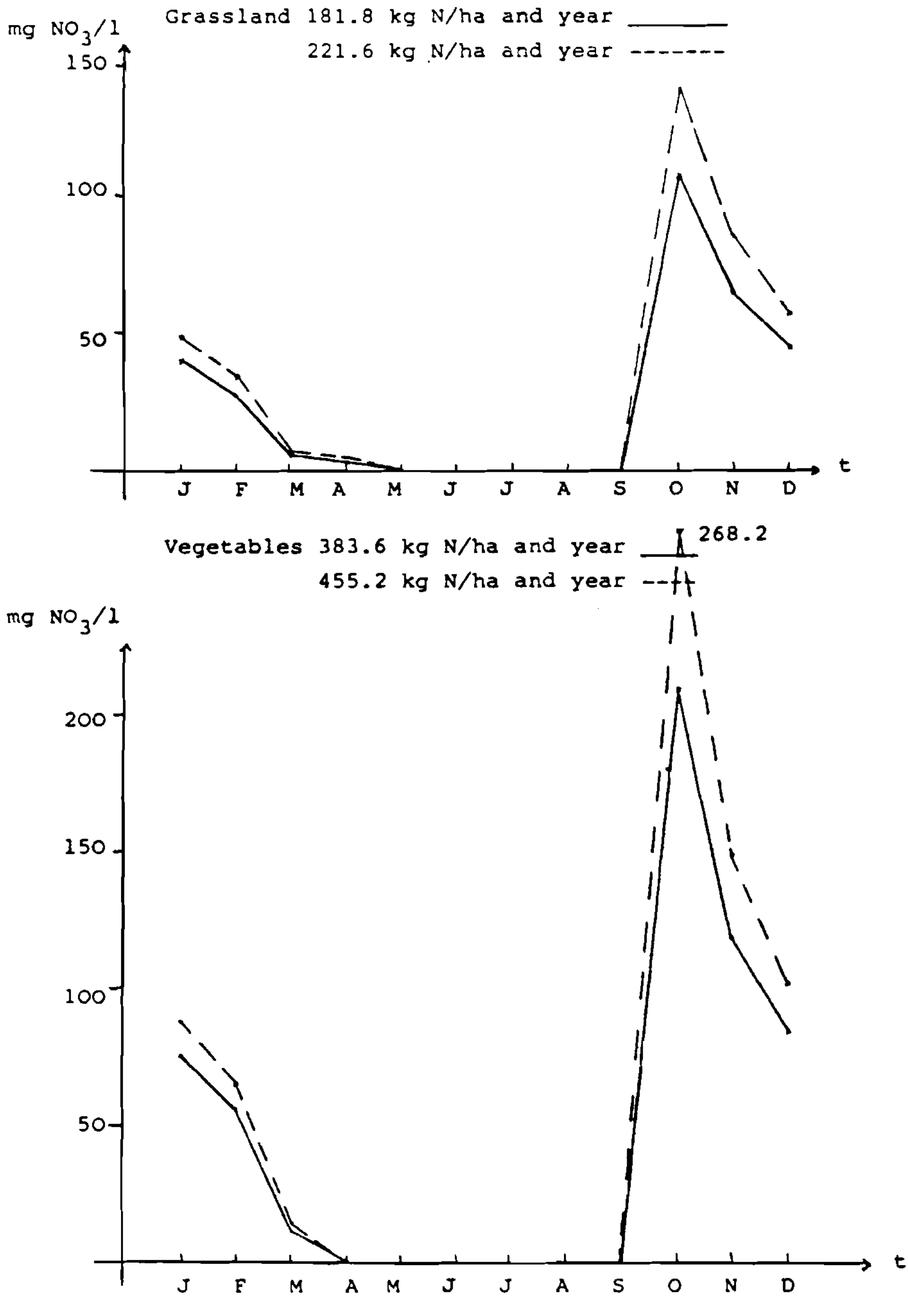
	1973	1974	1975	1976	1977
Grassland 200 kg N/ha	39.6	45.1	5.5	82.2	86.6
Vegetables 420 kg N/ha	60.1	91.9	11.3	192.1	160.8
Sugar beets 245 kg N/ha	19.6	49.9	20.5	68.6	93.8
Maize 220 kg N/ha	19.9	34.7	11.9	9.2	122.1
Grain 64 kg N/ha	8.2	14.9	6.1	9.9	14.8
Grain + second crop 180 kg N/ha	10.6	43.2	16.4	44.7	54.6

1) The results of the CREAMS-model in comparison with actual measuring of NO_3-leaching in a water command area (Mussum) are not yet satisfactory, except for grassland, as the following three-year-averages (mg NO_3/l) indicate. The original version of the model was used for the simulation. So far no adjustments of the model and no tuning of parameters were done.

Table 3: Measured and Simulated Nitrate Leaching at the Mussum Catchment Area (mg/l) Average 1973-1977

	Simulation (CREAMS-Model)	Measurement Average	Measurement Extrema
Grassland	44.3	46.2	max 130 min 9
Vegetables	98.4	236	max 570 min 180
Arable Land	33.5	90.9	max 157 min 24

Figure 7: Simulations of NO_3-leaching to groundwater with the CREAMS-model (preliminary results) - Monthly averages of a 5-year simulation (1973-1977) in mg NO_3/l deep percolated water



3.2 Modelling Nitrate Flows from Field to Pumping Station

In order to infer from a required drinking water nitrate standard on the permissible nitrogen fertilizer level it is necessary to simulate the flow of nitrate from the field to the pumping station. This is obviously a difficult task, especially in so far as the officially defined boundaries of current catchment areas do not correctly delineate the real bounds of the intake areas.

A model of horizontal nitrate flows should account for two processes. One is the flow of water from the field location to the related pumping stations, the other is the flow of nitrates leached from the entrance into the aquifers to the pumping station. Although less important in the deeper zones of the aquifers, denitrification has to be accounted for as in the vertical flow model.

Various model types are conceivable which simulate the flow of water from the field to the pumping station. Depending on the intended model output they either simulate the water flow through consecutive cells following hydraulic gradients or they represent average mass balance equations of water and nitrate movements for larger areas. ZWIRNMANN (1981) analyzes recent publications of three empirical studies and arrives at rather favorable conclusions with respect to some of the models' predictive capabilities.

Whether or not such models can be applied to extended agricultural areas or even - in appropriate regional disaggregation - to whole states, as the study of REEVES (1977) for England and Wales, remains to be investigated.

One interesting feature of several of the available studies is the long time lag of 10 to 30 years between nitrogen application and the resulting impact on the nitrate content of pumped water. The time lag seems to depend, ceteris paribus, on the thickness of the unsaturated zones, the speed of the horizontal water flow and the distance between field and pumping station (see e.g., YOUNG et. al., 1979). Hence, an apparent lack of correlation between farming activities and water pollution may be the result of taking a too short time horizon. Especially where the aquifers' capacity to recover is limited, one cannot exclude that intensive nitrogen application even outside the official catchment areas might pollute the drinking water resources of the next generation.

4. Issues for Model Applications

Various issues can be raised on the basis of the aforementioned models. One is the cost comparison of alternative instruments. Concerning the farm sector such cost estimates can either be based on current fertilizer technologies and production patterns (status quo analysis) or on appropriate adjustments (optimal adjustment analysis). Moreover, the models could be used to identify critical regions with high pollution potential.

4.1 Status-quo Cost Estimates

In order to estimate the social or farm income loss (depending on the respective prices) of a required reduction of nitrates in drinking water, one additional set of empirical information is necessary: the nitrogen-yield functions of various crops as well as production functions of those livestock categories which use fodder areas or provide manure to be spread on the fields.

A model to be used for cost estimates has essentially to be dynamic and account for the time lag between production activities and nitrate intake into drinking water. In other words, such a model should be able to indicate the tradeoff between current income losses resulting from anti-pollution measures and later ecological damage from not taking action today.

Such dynamic models are complex and require a large amount of time dependent information. Therefore, it might initially be advisable to exclude adjustment lags and discount factors and limit the analysis to comparative static cost estimates. The cost comparison would comprise the situation before implementation of anti-pollution measures and the new steady state situation after implementation.

The starting point of a comparative static cost analysis is the required nitrate concentration limit, $\bar{n}$. If, for simplification, one assumes that this limit, expressed in mg nitrate per litre, is to be met by the leaching water under each land use activity (and not just a rotation), then one can define the required reduction, dn_i, of the nitrate concentration as

$$dn_i = \alpha \; n \; (FN_i) - \bar{n}$$

where:

α = location specific denitrification parameter, indicating the proportion of leached nitrates which finally reaches the pumping station ($\alpha \leq 1$);

$n_i(FN_i)$ = the current concentration of nitrates in leached water and land use activity i, with n being a function of the nitrogen fertilizer level FN_i;

$\bar{n}$ = upper limit of nitrate concentration at the pumping station.

The costs of realizing this reduction through a lower fertilizer intensity are derived from the profit foregone :

$$K_i = \frac{\partial FN_i}{\partial n_i} \left(\frac{\partial Y_i}{\partial FN_i} \cdot P_{Y_i} - P_{FN}\right) dn_i \cdot \frac{1}{\alpha}$$

where:

K_i = income foregone per hectare of activity i

$\frac{\partial FN_i}{\partial n_i}$ = inverse incremental nitrate leaching (mg/l) per unit of marginal change of nitrogen fertilizer level. This function is to be generated by the vertical flow model

$\frac{\partial Y_i}{\partial FN_i}$ = marginal product of nitrogen fertilizer

P_{Y_i} = price per unit of output i

P_{FN_i} = price per unit of nitrogen, including the costs of other complementary crop specific inputs which would be varied parallel to the nitrogen level (e.g. other nutrients, chemicals, stem shortening treatment).

To give a rather hypothetical example for the grassland use for which a preliminary simulation of nitrate leaching was presented in the previous paragraph, it is assumed, that 50 % of the leached nitrate leaves the aquifer by denitrification ($\alpha = 0.5$), that the current nitrate concentration reaches

150 mg/l in October ($n(FN) = 150$), and that the required nitrate limit is 50 mg/l ($\bar{n} = 50$). Hence, the nitrate content in drinking water is to be reduced by $dn = 25$ mg/l.

If one further assumes that the marginal leaching effect is 1.33 mg/l nitrate for each additional kg/ha nitrogen fertilizer (tentative model result for month of October) ($\partial FN/\partial n = 0.75$); that the marginal product of nitrogen on grassland is 13 KSU (kilo-starch-units) per kg nitrogen ($\partial Y/\partial FN = 13$), that the revenue per unit of output is $P_Y = 0.50$ DM/KSU, and that the price of one kg of nitrogen plus other complementary inputs is $P_{FN} = 2.5$ DM/kg, one arrives at a farm income loss of $K_i = 150$ DM/ha. The social income loss would presumably be lower, depending on the shadow prices, P_Y and P_{FN}.

The average costs for a given region with homogeneous soil and climate conditions follow from aggregation over all land use activities

$$K = \sum_i K_i \cdot x_i$$

where x_i = relative share of i in total land use ($\Sigma x_i = 1$).

Finally, the income foregone per litre of leached water ($K' = \frac{K}{w}$), is to be compared to the costs of direct nitrate elimination at the waterworks and other related external effects of the alternative policies.

4.2 Optimal Fertilizer Adjustment Model

An economic analysis of nitrate pollution should not stop at this stage of status quo analysis. An important factor of adjustment has so far been left out, namely the farmers' possibilities

- to change the production structure (e.g. reduce the share of crops with high nitrate emission)
- to change the frequencies of fertilizer applications (e.g. apply more small portions), and
- to use alternative techniques and take better account of nitrate reserves in the soil at the beginning of the growing season.

Any of these measures will potentially reduce the costs of fertilizer reducing measures. They will enable farmers to apply less fertilizer without or with only small reductions of yields and income per hectare.

It has to be mentioned in this context that farmers will - in pursuit of even private profit maximization - have a rising interest in such adjustments, as real fertilizer prices rise so drastically as in the recent past. One outstanding example for this tendency is the rapid diffusion of the socalled N_{min}-method, where the nitrogen doses is related to the actual nitrogen stock in the soil at the field level.

One method of analyzing the so defined optimal adjustment is a linear programming model. Linear programming can not only be used to compute minimum "long-run" adjustment costs to meet certain nitrate limits (as opposed to the short-run costs at status quo conditions). It can also help to search for appropriate policy measures. Further it may be developed as an educational tool for farm extension. An example including external effects of nitrogen fertilizer was presented by SWANSON et al., (1978).

These long-run costs of optimal adjustment to certain nitrate limits will again have to be compared to the costs of nitrate elimination from drinking water. The conclusions with respect to an optimal policy may possibly differ from those of the status quo calculations, i.e. measures to enforce revised fertilizing strategies and cropping patterns may be recommended although the status quo costs of nitrogen fertilizer reduction exceed the costs of direct nitrate elimination.

4.3 Identification of Critical Regions

While the empirical basis of analyses at the watershed level seems to be quite sound, many uncertainties arise when one attempts to draw conclusions for the regional or sectoral level.

Basically, there is empirical evidence that intensive nitrogen fertilizer leads to considerable nitrate leaching, especially under light soils and where green cover can't be assured in winter time.

At the watershed level several studies support the hypothesis that this nitrate directly pollutes the drinking water sources. Policy measures can be implemented specifically for the respective zones of catchment areas.

But how can critical regions outside those watersheds be identified? How large are they? What measures should be taken here to prevent further pollution? What is the total damage of nitrate leaching in the overall sector and what costs would be involved if sector wide policies (e.g. nitrogen tax, quota etc.) were applied?

These are very important questions to which we don't have answers yet and where we can only attempt approximate calculations.

Lacking an operational water flow model we can try to estimate the upper bound of the nitrate pollution caused by agricultural production when we start from the hypothesis that all leaching water exceeding a certain nitrate limit, say of 50 mg/l, is adding to the potential pollution of drinking water at some point in time. Based on this assumption, the following procedure is conceivable to arrive at regional or sectoral estimates:

1. Classify a given region (or country) according to soil types and rainfall into a limited number of locational types.

2. Simulate the seasonal nitrate in leaching water per hectare for the typical cropping patterns and fertilizer levels.

3. Estimate the necessary fertilizer reduction for those crops where nitrate concentration limits are exceeded.

4. Compute yield and cost effects of the required fertilizer reduction, based on regional production function estimates and standard gross margins.

5. Compute aggregate cost and yield effects by summation over the locational types of the respective region.

6. Location types or regions with significant violations of nitrate limits and resulting high costs of fertilizer reducing policies are identified as potentially critical areas. They are candidates for more detailed studies of leaching problems and field to water source flow analyses. Moreover, recommendations for those regions have to be developed with respect to adjustments of cropping patterns and fertilizer frequencies.

Such a procedure certainly has the shortcoming, that it only approximates the maximum conceivable damage, that it has to rely on poor regional data for yield response functions and production cost estimates, and that it necessarily has to work at a high level of aggregation and neglect site specific conditions.

Yet, compared to the field level analysis the region-wide estimate of maximum damage has the advantage that it illustrates the relevance (or irrelevance) of more indepth investigations in well-defined regions, that it yields preliminary estimates of external costs of intensive agricultural production which may improve the evaluation of agricultural policies, and that it finally may help the decision maker to choose among site specific anti-pollution measures versus region or sector wide policies in order to achieve a desired reduction of nitrogen levels, if the latter should turn out to be preferable to a direct nitrate elimination from the water.

5. Conclusions for Choice of Policy Instruments

The purpose of this paper was to discuss open questions in the field of nitrate pollution through agricultural production and to present an analysis concept. It was not the purpose to present results and policy recommendations. We started from the widely accepted assumption, that intensive agricultural production causes considerable nitrate leaching into the groundwater and that this is already causing damages in many catchment areas.

We therefore propose an initial economic analysis at the level of catchment areas in order to define least cost alternatives to meet certain nitrate limits and in order to analyze economic feasible adjustments of production patterns and fertilizer application strategies. This analysis has to include cost estimates of nitrate elimination from the water without affecting agricultural production.

If the nitrate pollution problem turns out to be manageable within narrow bounds of catchment areas, location specific policies (control of fertilizer levels, prohibition of certain crops etc.) could be applied, and the major agricultural sector would hardly be affected. This perspective would drastically change if suspicions should be realistic that much larger groundwater sources contributing to human water supplies and that nitrate leached into the groundwater outside the official catchment areas may finally also pollute those supplies. The relevance of this hypothesis is actually the major open question requiring the advice of experts in hydrology and soil science. It will eventually determine the borderline between local and sectoral policies to reduce the nitrate problem.

In so far as it can be shown that more extended areas of agriculture contribute to the nitrate pollution - and this is not unlikely for certain locational types and farming systems -, one may have to consider sector or region wide policy instruments, e.g. taxation of fertilizer use, quota or prohibition of complementary inputs (e.g. stem shortening chemicals such as CCC). These policies would certainly have sizeable negative impacts on farm income, especially since quite drastic taxation may be required in order to stimulate a sensible reduction of fertilizer inputs.

This latter perspective indicates potential political pressure against more ecological considerations in agricultural policy. Compensations of income foregone may have to be included in policy analysis. Generally, there is an urgent need for more empirical evidence on which cost-benefit considerations can be based. They are necessary in order to provide more objective and less irrational arguments for the public debate in the domain of environmental implications of agricultural production.

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THE ADVISORY SYSTEM FOR FERTILIZER APPLICATION IN THE GERMAN DEMOCRATIC REPUBLIC EMPHASIZING THE MINIMIZATION OF NITROGEN POLLUTION*

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This paper describes one of three operational computer systems used to give advice to farm operators in the GDR. The system, employed nationwide, fulfills two objectives. First, it serves to plan the demand for mineral fertilizers on farms, in districts, and in regions, taking into account the availability of organic manure. Second, it determines the type of fertilizers used, timing, rate, splitting, and technological methods of fertilizer application in specific fields. The system has proved to be a useful basis for decisions to be taken in the management, planning, organization and control of fertilizer use.

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1. Introduction

Agriculture in the German Democratic Republic (GDR) faces the task to continuously supply high-quality food to the population and raw materials to the manufacturing industry. To fulfill this task, agricultural production has to be further intensified. Among the intensification factors, extensive chemicalization is of great importance. For this reason, special attention has to be given to the effective use of the available amounts of nitrogen fertilizers.

Whereas the share of nitrogen effectiveness in farm-scale dispersed trials on loamy sands and very sandy loams at a fertilization level of 100 kg N/ha under the condition in the GDR comes up to 25, 40, and 22 % in winter wheat, winter rye, and potato, respectively, it totals 66 % in field grass at a fertilization level of 160 kg N/ha (SCHNEE, M., 1976). Thus nitrogen takes an important part in yield formation. On light soils, however, it is not to be excluded that fertilizer nitrogen and nitrogen originating from mineralization of organic matter in the soil will penetrate into ground and surface waters. In unfavorable cases, the nitrate content in the water may exceed the limit of toxicological safeness. Hence, science is obliged to attend still more carefully to the elaboration of environmentally acceptable and protective solutions.

Consequently, the use of nitrogen fertilizer is aimed not only at reaching high yields per unit area but also at improving the utility value of the crop products and at minimizing nitrogen pollution. The realization of these aims will render the application of fertilizers, particularly of nitrogen fertilizers, highly effective.

To have the environment impaired as little as possible when carrying out fertilization operations, numerous measures to protect environment have been taken and will be taken in the GDR.

The state, society and each citizen in the GDR are obliged to protect nature and its resources. This obligation is laid down in the Constitution of this country. Therefore, state management and agricultural research bodies have been concerned with introducing, parallel to the application of increased amounts of mineral and organic fertilizers, a variety of measures contributing to a rise in soil fertility and, thus, to further improve the 'kidney function' the soil has in nature (RÜBENSAM, E., 1979) as well as continuously complete the scientific fundamentals of fertilizer use. Furthermore, it is not only a matter of balancing energy spent on the production, transport, handling, treatment, and spreading of mineral fertilizers by a high energy gain through plant yield, but of increasing this energy gain. It is started out from the fact that the use of large quantities of mineral fertilizers and organic manures constitutes an essential precondition for raising agricultural production and improving soil fertility. At the same time, the possible effects on environment, on the one hand, and the demands in the field of water management and environment protection, on the other hand, expand. Such effects have to be differentiated as direct and indirect ones.

The direct effects are insignificant after spring application of nitrogen fertilizers at the start of the growing season as, if done properly, the nitrogen generally will not be translocated to the subsoil. Experiments with small lysimeters on five sites between 1965 and 1972 revealed the N leaching to vary between 8.8 and 16.7 kg N/ha and year (below 1 m). Of these only 10 to 15 per cent accounted for fertilizer nitrogen. A certain exeption are sandy soils where nitrogen may be washed out eventually in spring if root and tuber crop are cultivated and high precipitation occurs in the period when the plants do not yet take up nutrients or do only to a small degree. The same holds true of high quantities of supplemental water from sprinkler irrigation. High rates of organic manure applied in autumn, however, may result in N translocation to deeper soil layers, particularly when slurry is used.

While properly applied mineral fertilizers generally do not have any direct influence on nutrient translocation, a direct effect may be caused at high fertilization level by 'nutrient residues' left after too high fertilizer applications or very low crop yields. This nitrogen is found in the soil in form of nitrate and, thus, is mobile.

The chance of nitrogen being built up in the soil is due to another indirect relationship with fertilization. The intensification of crop production and the increasing use of mineral fertilizers as well as of ever higher amounts of organic manures lead to a site-specific level of soil organic matter, which is an important characteristic of soil fertility. Organic manuring and soil organic matter are known to have a positive effect because they do exert a most favorable influence on the soil's physical, chemical, and biological properties. The same holds true of the cleaning efficiency, the 'kidney function' of the soil for contaminants to the environment.

As in 1977/78 129 kg N, 66.1 kg P_2O_5 and 63.4 kg K_2O were applied on average per 1 ha of farmland area in the GDR (according to Statistisches Jahrbuch der DDR 1979, economic year), fertilization has to be organized in such a way that crop yield and quality are strongly influenced and environment is impaired as little as possible. To meet these demands EDP programs have been established to an increasing extent since 1971 for the use of macronutrients (N, P, K, Mg, Ca) and micronutrients (B, Cu, Mn, Mo, Zn) and organic manures and have been put at the disposal of the GDR farms for crop production as decision aids for the planning and application of fertilizers and manures (Table 1 - Application range of the EDP-project 'Fertilization' and of plant analysis, in per cent related to the area attended to by the Agrochemical Analysis and Advisory Service of the GDR; BEER, K. and KOLBE, G., 1978).

The computation of recommendations is based on EDP programs including a variety of parameters from which decisions are derived by logical linkage of facts during the computing operation. It is possible by changing parameters and computing operations to have new scientific findings and experience of outstanding farms immediately introduced to a broad range of farms.

For mineral fertilizers and organic manures recommendations are given on quantity, splitting, time, fertilizer form, and application technique related to the respective crop in the field and the meadows and pastures, respectively. Furthermore, calculations of organic manure production and accounts of fertilizer requirements by quantity and assortment are made for planning purposes under consideration of the temporal demand of sections and departments of a farm as well as of the whole farm. Simultaneously these recommendations are further summarized and then serve the state management as fundamental material for planning the fertilizer requirements and the regional distribution of the total amount of mineral fertilizers to counties and districts as well as agrochemical

Table 1 Application range of the EDP project ' F e r t i l i z a t i o n ' and of plant analysis (in per cent related to the area attended to by the Agrochemical Analysis and Advisory Service of the GDR) (BEER, K. and KOLBE, G., 1978)

	1971	1972	1973	1974	1975	1976	1977	1978
EDP programme 'Macronutrients'	54	68	85	96	99	100	100	100
EDP programme 'Micronutrients'	-	4	5	7	14	19	28	33
EDP programme 'Organic Manuring'	-	-	-	18	46	96	90	94
Plant analysis								
Percentage of area under winter cereals	-	4	17	32	43	88	95	98

centers (ACC) that have been founded as inter-farm establishments by the farms for crop production and mostly perform the spreading of mineral fertilizers. The rapid growth of the GDR agricultural production, the results obtained in research work, and the evaluation of numerous proposals of experienced workers as well as the ever increasing information demand rendered it necessary to review and improve the computing techniques and parameters of the existing EDP fertilization programs along with the current program work.

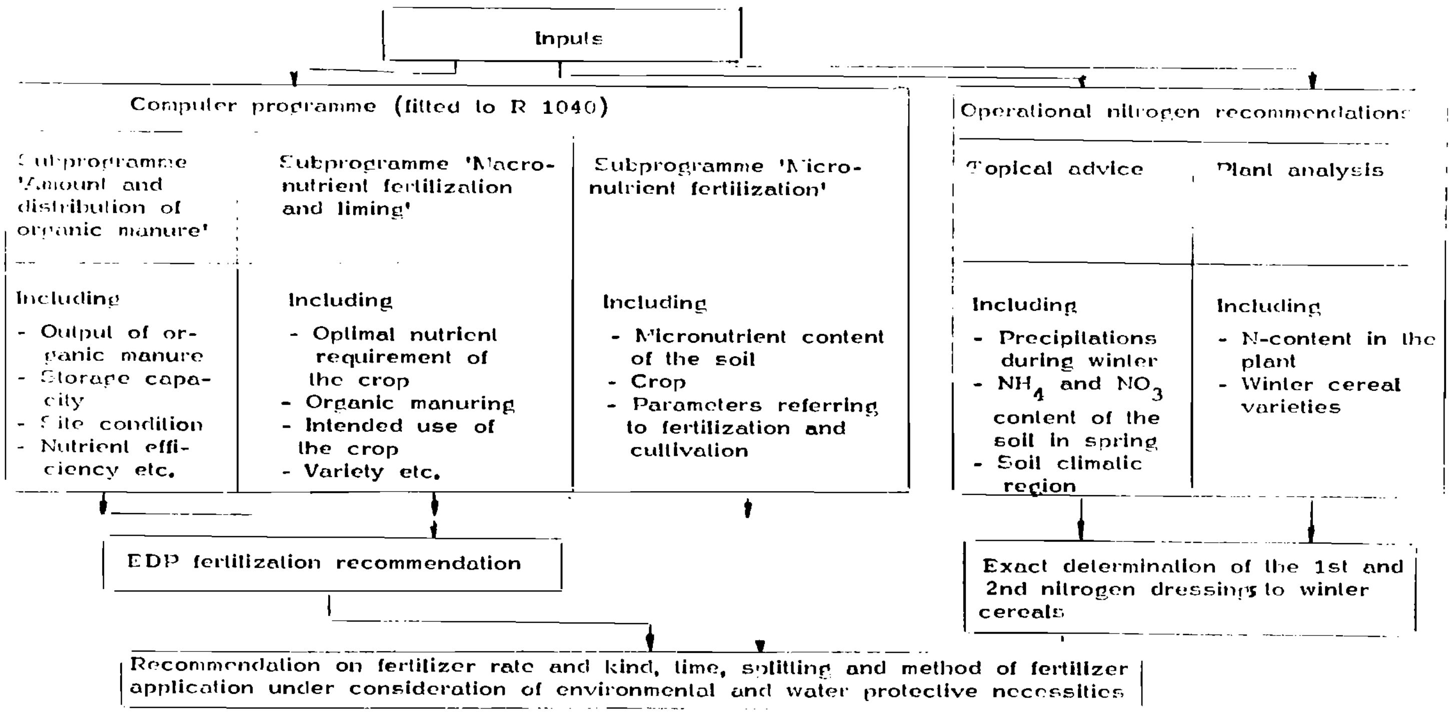
2. EDP fertilization project DS 79

The new fertilization project DS 79, programmed in PL 1 language for the EDP unit Robotron ES 1040 allowed to link the individual sub-programs (macronutrients, micronutrients, organic manuring). The latter may be computed in combination or separately. A survey of the structure of the EDP project 'Fertilization' is given in Figure 1 'Scheme of the fertilization system' (BEER, K. et al., 1978).

The fertilization project consists of the following linked sub-programmes:
- organic manuring
- mineral fertilization - macronutrients
- mineral fertilization - micronutrients

As for reasons of planning and as a basis for decisions on PK advance fertilization and liming, the fertilization recommendations must be computed already in the summer of the preceding year; the operational adaptation of nitrogen fertilization to the actual meteorological conditions forms another integral part of the fertilization project. The exact determination of the N fertilization is based on soil analyses for plant-available nitrogen content in early spring (Topical Advice on Fertilization) as to the first

Figure 1 Scheme of the fertilization system



N dressing, and on plant analysis as to the 2nd N dressing to winter cereals. The technological run of working out EDP fertilization recommendations is shown in Figure 2.

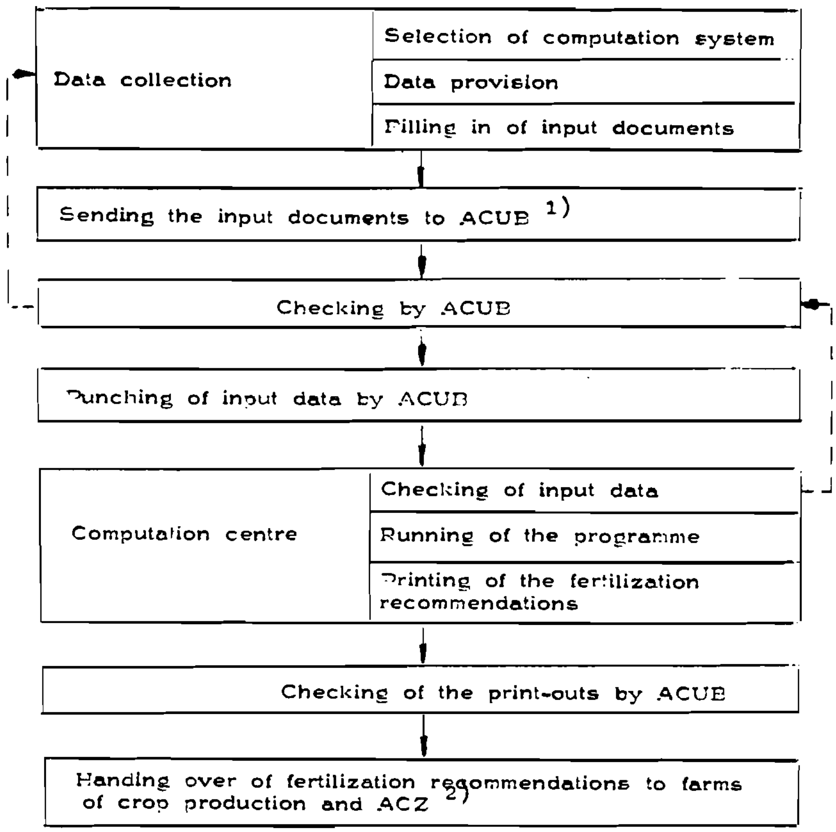
Data collection and provision as well as the filling in of the input documents are done under the supervision of cooperators of the Agrochemical Analysis and Advisory Service of the GDR (ACUB). ACUB belongs to the Institute of Plant Nutrition, Jena, of the Academy of Agricultural Sciences of the GDR. It has divisions in Jena, Halle, Bergholz-Rehbrücke, Dresden and Rostock. The staff members of these service divisions are responsible for advising the farms for crop production in their region. The samples of soil and/or plants as well as manures are analyzed in specialized laboratories in Jena, Rostock and Halle. The staff members of ACUB check the input documents filled in. All input data are punched in the Dresden division of ACUB.

Checking of the punched input data, running of the EDP program, and printing of the fertilization recommendations are carried out in the computer centre of the Ministry of Agriculture, Forestry and Food. Then the staff members of ACUB check the EDP fertilization recommendations, hand them to the farms and agrochemical centers, and give the necessary explanations.

To ensure a wide application range of the program, it was laid out for 207 crop species and utilization types. The crops are included in the computation as crop to be fertilized and for their properties as first and second preceding crops or catch crop. Table 2 gives a survey of the crop groups to be included.

Figure 2

Organization of data collection, computation and output of fertilization recommendations



1) ACUB - Agrochemical Analysis and Advisory Service

2) ACZ - agrochemical centre

Table 2 Groups of crop species and utilization covered by the EDP project 'F e r t i l i z a t i o n' DS 79 (ANSORGE, H., 1978)

	Number of the crop species or utilization types included
Cereal crops	10
Leguminous crops	10
Oil crops	10
Fibre plants	2
Field vegetables	32
Potatoes	5
Sugar beet	5
Root crops for fodder	10
Forage plants for fodder	88
Forage plants for seed	24
Tobacco	1
Grassland	10

The cropping form, and in several crops also the varietal type grown, as well as the intended use, are considered to further specify the fertilization recommendations.

The intended use of the harvested crops is required to consider in fertilizer application the influence of fertilization on the quality of the harvested crops.

The following uses are included:
- industrial processing of the harvested crops (e. g. malting barley, starch potato, manufacture of baby food from vegetables)
- feed and bread grain with increased crude protein content
- whole-plant harvest
- artificial drying of forage
- hay-making and feeding of fresh forage or pasturing, respectively
- ensilage
- immediate consumption of vegetables and potatoes
- storage of vegetables and potatoes
- multiplication

The effect and the dynamics of the nutrients supplied are strongly influenced by the site conditions. Therefore, the different soil properties and the climate are largely considered when computing fertilization recommendations within DS 79. In this context, it is not only the soil nutrient content being systematically determined in the GDR since 1952 on the basis of respective laws, that plays an important role. Table 3 shows the soil groups for arable land included in DS 79.

The first figure of the soil goes for the texture of the topsoil, the second figure for hydraulic conductivity and

Table 3 Soil groups DS 79 - arable land (ANSORGE et al., 1979)

Soil group	Fine particles < 6 µm %	Soil class	Characteristics of arable soils
1.1	≤ 7	Sand (S)	Low ground-water level
1.2	≤ 7	Sand (S)	Influenced by ground-water
2.1	8 - 15	Lightly loamed sand - loamy sand (Sl / lS)	Low ground-water level
2.2	8 - 15	Lightly loamed sand - loamy sand (Sl/lS)	Influenced by ground-water
3.1	16 - 25	Very loamy sand - sandy loam (SL/sL)	Without waterlogging
3.2	16 - 25	Very loamy sand - sandy loam (SL/sL)	With waterlogging
4.1	26 - 38	Loam (L)	Wihout waterlogging
4.2	26 - 38	Loam (L)	With waterlogging
4.4	26 - 38	Loam (L)	Chernozem soils
5.1	≥ 39	Clay (T)	Without waterlogging
5.2	≥ 39	Clay (T)	With waterlogging
6.1		Half-bog, shallow bogs	20 - 40 cm peat 15 - 30 % organic matter
6.2		Bog	40 cm peat > 30 % organic matter

waterlogging (water level). Excepted are chernozem soils (soil group 4.4), half-bog soils (soil group 6.1), and bog soils (soil group 6.2).

To be able to consider the different influences of the climate on the level and effect of nutrient supply, it was necessary to define four macroclimatic zones for the farm-land area in the GDR.

Climatic zone 1 = Lowlands under maritime influence in the north and the hilly country with humid, mild climate in the south of the GDR
Climatic zone 2 = Dry region and marginal areas in the middle and the southern parts of the GDR
Climatic zone 3 = Transitional region between the hilly country and the foothills up to the medium-altitude elevations of the low mountain range of the GDR
Climatic zone 4 = Elevations of the low mountains of the GDR, exceeding 500 m above sea level

The regional distribution of the climatic zones to the GDR counties is shown in Figure 3.

The delimitation of the climatic zones is based on meteorological limits (Table 4).

This macroclimatic approach necessarily includes influences of the micro- and local climates, which are the reasons for exceeding the limits quoted. These influences should be considered by assigning them to the respective climatic zone in accordance with the meteorological limits.

Table 4   Meteorological limits of the climatic zones - DS 79

Meteorological limits	Climatic zones			
	I	II	III	IV
(1) Height above sea level, m	0 - 350	50 - 250	250 - 500	500 - 700
(2) Precipitation, mm/year	540 - [illegible]00	4[illegible]0 - 5[illegible]0	600 - 950	900
(3) Precipitation in winter, mm/year	240 - 350	200 - 240	220 - 450	420 - 600
(4) Annual mean temperature, °C	6,0 - 8,0	7,0 - 9,0	5,8 - 7,8	5,0
(5) Number of frosty days	74 - 110	80 - 90	100 - 120	110 - 150
(6) Number of days above 10 °C	140 - 160	140 - 160	100 - 165	120
(7) Dryness Index	25 - 65	20 - 35	30 - 70	50 - 100

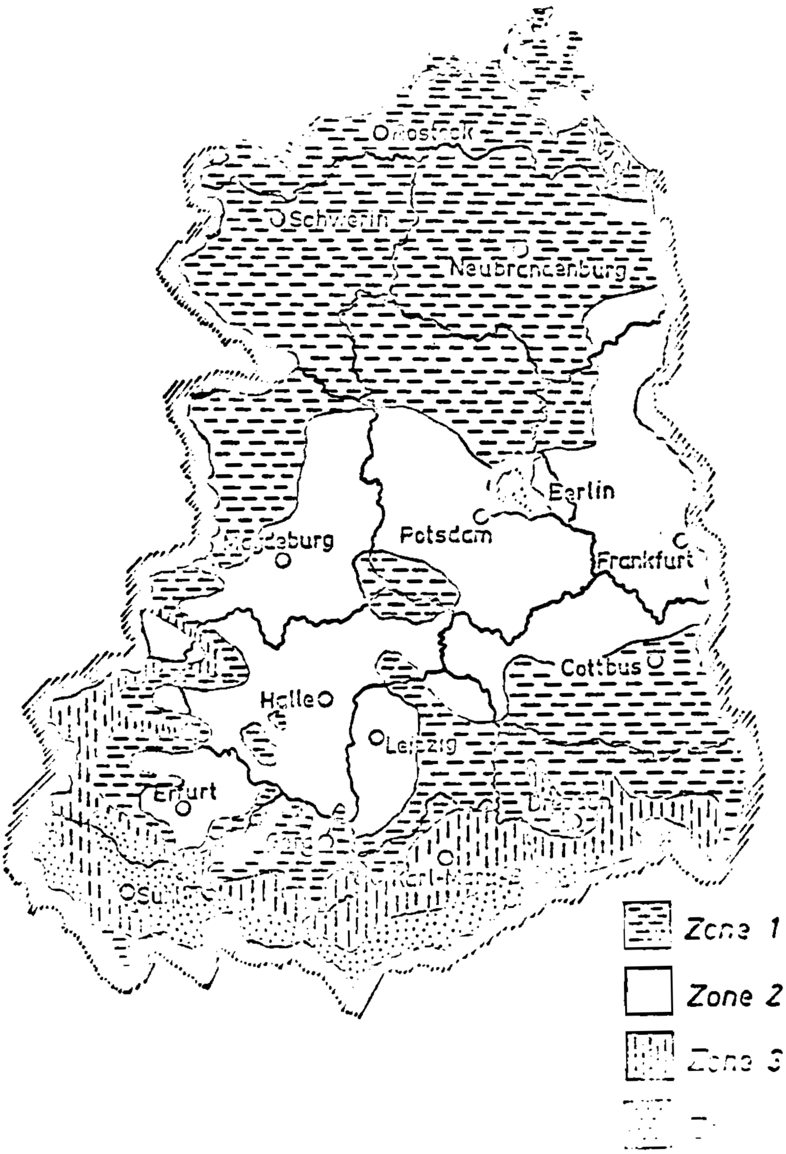


Figure 3. Climatic Zones DS 79

For the exact determination of the fertilization periods in the different GDR regions in dependence on climate and weather, four phenological zones are differentiated:

Phenological zone 1 = Region with normal start of vegetative period
Phenological zone 2 = Region with slightly late start of vegetative period
Phenological zone 3 = Region with normal start of vegetative period and very early grain harvest (early threshing)
Phenological zone 4 = Region with very late start of vegetative period

The structure of the flow chart of the sub-programs according to the unit assembly principle and an exact adaptation of the respective input and output information allow in DS 79 to compute separately and in combination the sub-programmes mentioned in Figure 1. Therefrom result the following possible computation systems (Table 5).

Computation systems 1 and 2 are those most frequently used by the farms for crop production. Computation system 1 'Mineral Fertilization (Macro- and Micronutrients)' serves to compute crop- and field-related recommendations for fertilization with macronutrients (N, P, K, Mg, Ca), and if results from soil analysis for micronutrients are available also for fertilization with micronutrients (B, Cu, Mn, Mo, Zn), in terms of quantity, splitting, time, fertilizer form, and fertilization technique. If micronutrients are required, feedback takes places to the use of macronutrient fertilizers containing micronutrients. Site conditions, economic factors, and the influence on the quality of the crop products are considered in the computation.

Table 5 Survey of the computation systems of the DS 79 programme (ANSORGE, H. et al., 1979)

Computation system	Information	Print-outs
(1) Mineral fertilization (macronutrients and micronutrients)	- N, P, K, Mg, Ca fertilization - B, Cu, Mn, Mo, Zn fertilization - Balance for reproduction of organic matter	- Fertilization to crop species and fields - Summary on field and farm level - Summary regarding labour organization on farm level - Summary by crop species - Balance for reproduction of organic matter
(2) Organic manuring	- Available organic manures - Straw balance - Use of organic manures - Balance for reproduction of organic matter	- Amount of organic manures produced, straw balance - Straw removal - Organic manuring to crop species and fields - Summary on field and farm level - Summary regarding labour organization on farm level - Summary by crop species
(3) Organic manuring and mineral fertilization	- As for systems 2 and 1 - Organic manuring is immedially included in mineral fertilization	- As for systems 2 and 1
(4) Fertilization with micronutrients	- B, Cu, Mn, Mo, Zn fertilization without any feedback to the use of macronutrient fertilizers containing micronutrients	- Application of micronutrients to crop species and fields
(5) Production of organic manure	- Amount and time of organic manure production	- Amount and time of organic manure production in the animal houses - N fertilization to crop species and fields
(6) Additional computation of nitrogen fertilization in the spring of the crop year	- N fertilization with special regard to the inorganic N content in soil	
(7) Straw balance	- Straw balance	- Straw balance

The farms have to provide data on the intended organic manuring to enable the integration of the nutrients supplied by the organic manuring and the computation of the balance for the reproduction of soil organic matter.

Computation system 2 'Organic manuring' is used to compute the availability of the different organic manures in terms of quantity and time, and their distribution to crops and fields under consideration of factors of agronomy and cultivation, demands of environmental ecology as well as aspects of labour organization and economy. These data may also serve as input data for computing mineral fertilization. Furthermore, balances are established for the quantity and use of straw and reproduction of soil organic matter.

Of the remaining computation systems, system 6 'Nitrogen Fertilization in Spring' is of special importance for considering the problems of environment. It is intended to additionally compute recommendations for nitrogen fertilization within a short time in spring. It serves to correct changes in the cropping plan and the use of organic manures, which particularly influence nitrogen fertilization of the fields. Moreover, nitrogen fertilization is adjusted to the conditions of the preceding year (crop yield, nitrogen extraction and nitrogen residues in soil) and to nitrogen dynamics in soil during the winter months. Thus the amounts of inorganic nitrogen compounds (N_{in} = NH_4^+-N and NO_3^--N) available in the soils in early spring are considered.

The fertilization recommendations are computed by fields for the respective crop species or utilization type. The fertilizer amounts required to reach the planned yields are printed out.

The level of N fertilization is computed by means of production functions (polynomials of 2nd order) representing the relations between N fertilization and crop yield. In a special program, the yield increments reached per kg N at the respective yield level are viewed against the additional expenditures for fertilization (technological costs for N, P, and K), taking the harvesting costs for the yield increment into account. The optimum is reached when the returns for the yield increment come up to the costs for the additional expenditures required for its production. The optimal N rate (N opt.) is calculated by means of the optimal yield (Y opt.) (RÜBENSAM, H.; KUNDLER, P.; WIENRICH, B., 1972).

$$\text{Y opt.} = \frac{PN^2}{(Py - My - \frac{Ep}{Ap} \cdot Pp \cdot \frac{Ek}{Ak} \cdot Pk)^2 \cdot 4c} - \frac{b^2}{4c} + a$$

$$\text{N opt.} = \frac{-b}{2c} - \sqrt{\frac{b}{2c}^2 - \frac{a}{c} + \frac{\text{Y opt}}{c}}$$

In this context, the symbols mean:

a, b, c - partial regression coefficients of the production function
PN - technological costs of N fertilization
Py - price of crop products in M/100 kg
My - harvesting costs in M/100 kg
Ep - P uptake by plant in kg/100 kg of yield
Ap - site-dependent P utilization coefficient
Pp - technological costs of P fertilization
Ek - K uptake by plant in kg/100 kg of yield
Ak - site-dependent K utilization coefficient
Pk - technological costs of K fertilization

By correcting the optimum N rates it is possible to systematically influence the crop yields and especially the quality of the crop products from certain species if there is the respective demand in national economy. The optimal N rates are not calculated from year to year, but are included as table values in the DS 79 computer program.

Crop species grown to a smaller extent are not optimized through production functions. In such cases, tables were worked out on the basis of experimental results.

Due to the strongly varying yields, the level of N fertilization to vegetables is investigated by way of balancing, the planned yields being included (GEISSLER, TH.; GEYER, B., 1976).

N kg/ha = E · a · A + b where

E	is	planned yield in 100 kg/ha
a	is	N uptake by plant in kg/100 kg
A	is	factor for crop- and site-specific assimilation capacity
b	is	addition depending on vegetables species and fertilization group

The basic value of optimal nitrogen fertilization relating to cereals as preceding crop, normal meteorological conditions, and absent organic manuring is further presented by additions and subtractions for:

- yield level
- preceding crop and position in the crop rotation
- use of the crop products
- cultivation form
- nutrient supply by organic manuring to the crop or residues from the fertilization of the first crop or preceding growth
- residual effect of organic manuring to the preceding crops
- variety in vegetables, cereal plants, potatoes, and seed growing crops
- application of culm stabilizers.

The additional computation of nitrogen fertilization in the spring of the crop year includes furthermore:

- nitrogen uptake by the preceding crop in dependence on the yield level
- level of N fertilization to the preceding crop
- nitrogen residues from the preceding year, and
- N translocation during the winter months.

Considering these factors of influence on the level of the overall nitrogen requirements allows an almost full adaptation to the respective production conditions so that it is possible in almost every case to avoid damages from overdressing and stronger N translocations into the ground-water even on low-sorption soils during the growing season.

A further measure for eliminating nitrogen overdressing and leaching consists in splitting N fertilization and observing optimal application dates. Thus, it is recommended to split N fertilization into two dressings for cereal crops and sugar beet and two to four dressings for almost all vegetables, perennial forage plants (including grassland) and seed growing crops. In this way, damages from overdressing will be avoided, the quality of the crop products will be improved and strong N translocations will not occur even after heavier precipitations.

The farms for crop production receive recommendations on the optimal application dates for all fertilizer dressings (including split dressings). They are given in form of print-outs on the time spans indicated in 10-day periods and months and the respective stage of plant development. Thus it is guaranteed that even in a year when the meteorological conditions deviate much from the standard, N fertilization will be carried out at the time when the nutrients are needed and due to a rapid uptake by the plant will not be washed out to a greater extent. Recommendations on autumn N dressings of 30 kg/ha are only given for the cultivation of winter rape grown after cereals in order to ensure a sufficient juvenile development of the plants before winter. Autumn N dressings are not recommended for other crop species, nor for winter cereals and after straw manuring, to prevent stronger N translocation to lower soil layers, which may become possible particularly during mild winter months with heavy precipitation.

Calculation of P, K, and Mg fertilization is done within nutrient balancing where nutrient uptakes by the crops are viewed against nutrient supply by mineral fertilization and organic manuring under consideration of nutrient utilization and nutrient status of the soils.

At an excessive nutrient content of soil, it is recommended not to apply mineral P or K fertilization. The field will even be excluded from organic manuring supplying a high amount of nutrients, when the range of toxic action (e. g. in potassium) is reached. Soil analysis thus has a regulating function in the calculation to eventually compensate for errors occurring in balancing (e. g. errors caused by several-year deviations of the actual yield from the planned yield).

On the basis of analytical results on the lime status of soil the level of liming is determined in dependence on the soil class, humus content and the crop species as well as utilization type. The lime quanities are fixed to reach optimal soil response.

The EDP program for computing recommendations on micronutrients is structured in such a way that all influencing factors hitherto known from fertilization trials and practice, such as level, technique, time, and form of fertilization with micronutrients, are considered.

In this context, a valuation is made of the individual factors exerting a differently strong influence. Special attention is paid to both the different micronutrient requirements of the individual crop species and the supply of the soil with the respective micronutrient.

The calculation of organic manuring indicates the available amounts of the organic manures and then defines their distribution to crop species and fields. The available amounts of organic manures are determined on the basis of the given quantities or the given livestock in dependence on the type and size of livestock management during the year, housing of the animals, and storage of the organic manures. The existing stocks, supply from other farms, and delivery to other farms are taken into account.

Requirements of environmental ecology have to be considered in planning the use of organic manures, particularly of slurry. For this reason, the conditions of environmental ecology and the specific farm conditions are taking a special position amongst the complex of factors considered for the recommendations on fertilizer use (Table 6).

To have them included for each field, the following limitations may be given:

- exclusion of any organic manuring,
- confinement to solid organic manures (exclusion of liquid organic manures),
- temporary prohibition of slurry spreading during the year,
- limitation of application rates in kg/ha of total nitrogen by solid and/or liquid organic manures in various levels (100 and 200 kg N/ha, respectively)

The combination of these possibilities allows to include all relevant conditions of water management with respect

Table 6 Factors for fertilization recommendation on organic manuring

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Factors of agronomy and cultivation

- Nutrient requirements and utilization by the crop species
- Organic matter requirements of soils
- Position in the crop rotation and organic manuring to preceding crops
- Soil class and nutrient losses
- Kind, nutrient content, and amount of organic manures

Demands of environmental ecology and water management

- Limitations regarding quantity
- Limitations regarding time

Factors of labour organization

- Ridability of the ground
- Special conditions for straw harvesting
- Application methods
- Transport distances
- Time of manure production, removal of preceding crop
- Storage capacity
- Suitability of crop species for fertilization
- Given times prohibiting the application of liquid organic manure
- Kind of organic manures

Specific farm conditions

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to water protection zones, ground water level, inclination of slope, management, and the like in fertilization recommendations.

Fertilization recommendations are given on the basis of the

- quantity and time of organic manure production
- storage capacity
- field data
- given transport distances, and
- parameters for the use of organic manures

The fields are selected by rank order (GÖRLITZ, H., 1978).

The recommendation of fertilizer use starts out from the optimal time spans for the individual crops. In how far it is necessary to recommend manure spreading beyond these time spans, depends, first of all, on the area available for application (cropping pattern and removal of the preceding crop) and the existing storage capacity. These problems are important in connection with factors of labour organization, particularly in case of slurry application, as partially it has not yet been possible to sufficiently coordinate cropping pattern, crop rotation, storage capacity, and spreading capacities to the quantity of slurry available.

Farmyard manuring is carried out according to the known crop-specific parameters. Due to the high proportion of soluble nitrogen content, the rate of slurry application must be in accordance with the nitrogen requirements of the plant stands. It, therefore, results from the N requirements of the plant stands, the N content of the slurry, and the mineral fertilizer equivalents. (The mineral fertilizer equivalents allow the comparison between the nutrients of slurry and those of mineral fertilizers.)

A special problem arises in that the slurry must be spread over the whole year, that means in the autumn and winter months, too, as for economic reasons the storage capacity cannot have any dimensions desired and spring spreading cannot be performed within a short time for reasons of labour organization in view of the large animal houses and, hence, the high quantity of slurry produced.

To keep the N leaching as low as possible, it is recommended to mainly apply slurry on the better soils of the farm in autumn. Furthermore, the application date in the autumn months is fixed as late as possible in dependence on the storage capacity. At the same time, it is attempted to combine it with catch cropping or straw manuring. The decline in N leaching due to straw manuring is caused by an immobilization of the soil nitrogen (Table 7).

Thus, straw manuring in general must be stressed as a measure to reduce N leaching attributable to nitrogen residues from fertilization or mineralization of soil organic matter. Catch cropping particularly effects a reduction of the percolation water rate through the water uptake by the plants and the N uptake from slurry application and soil (Table 7).

To reduce N leaching after autumn application of slurry, investigations have been made during the last years into the use of nitrificides. The addition of N-Serve or Cyanoguanidin to slurry inhibited the nitrification of the slurry nitrogen in incubation experiments (loamy sand, 20 °C, 50 per cent water capacity) and field trials.

3. Operational nitrogen fertilization recommendations

3.1. Topical advice on the 1st N dressing to winter cereals

As for reasons of planning and as a basis for deciding on PK advance fertilization and liming the fertilization recommendations must be computed already in the summer of the preceding year, it is only computation system 6

Table 7 Leaching losses after slurry manuring at various spreading dates on a Sand-Rost- Erde soil (S 5 D 20/1[illegible]), 1 September, 1971, until 31 August, 1972 (potato)

Manuring	Date of slurry spreading	Nutrient losses in kg/ha				
		N	K	Ca	Mg	P
320 kg N/ha as slurry	August	37	31	174	32	2
320 kg N/ha as slurry + 4 t of straw/ha	August	23	25	106	8	2
320 kg N/ha as slurry + manuring with green winter rape	August	12	[illegible]	46	7	1
320 kg N/ha as slurry	November	3[illegible]	23	144	19	2
320 kg N/ha as slurry + 4 t straw/ha	November	2[illegible]	22	77	15	1

'Nitrogen Fertilization in Spring' which includes the meteorological conditions of the preceding year and winter. But in this case, too, it is not possible to include into the calculations the actual meteorological conditions during the growing season and their influence on fertilization. It is, therefore, necessary to adapt the N fertilization to the actual meteorological conditions later on.

Starting out from the necessity of considering the different contents of inorganic nitrogen in the soils (N_{in} = NO_3^--N and NH_4^+-N) in N fertilization to winter cereals, comprehensive investigations in this field have been carried out in the GDR since 1972. It appears that an essential factor of influence on the yield is covered when including the N_{in} content in the dimensioning of the first N dressing to cereals, and hence the relations between N fertilization and yield are described in a better way.

The N contents in soils summarized for the soil groups of the DS 79 programme in Table 8 show a marked dependence of these contents on soil, weather and the crop yield in the preceding year.

To give an example it shall be stated that as compared with the average of the years, the N_{in} quantities were higher in 1977 due to the low crop yields of the preceding year, than in 1975 when heavy precipitations occurred in winter.

Since 1973, concrete comments have been made in 'Topical Advice on Fertilization' in early spring on the exact determination of the first N dressing to winter cereals under consideration of the inorganic N quantities in soil.

Table 8 Amounts of inorganic N (kg/ha) in the 0 - 60 cm layer of the 5 main soil groups in the DS 79 programme of the GDR between 1973 and 1979

Soil group	1973	1974	1975	1976	1977	1978	1979
Sand	62	64	50	44	64	53	66
Lightly loamed sand and loamy sand	70	66	52	118	112	67	58
Very sandy loam and sandy loam	109	80	82	78	124	69	93
Loam	94	61	70	77	138	92	83
Chernozem soil	126	144	92	138	137	127	148

Table 9 Annual differences in the soluble nitrogen content of the soil at the start of the growing season after autumn application of slurry on loamy sand

Year	Soluble N, kg/ha, at the start of the growing season Soil layer, cm		
	0 - 30	30 - 60	0 - 60
1975	35	35	70
1976	41	41	82
1977	49	82	131
1978	62	33	95

These comments on the correction of N fertilization are given in kg N/ha as additions to or subtractions from the first N dressings recommended in the print-outs on fertilization which are handed to the farms for crop production and the agrochemical centres. They are worked out for 21 climatic zones of the soil and all the soil groups occuring there (altogether 52 correction values). These correction values are based on the contents of nitrate and ammonium nitrogen determined every year in late autumn and early spring in a total of 1,600 soil samples from 0-30 and 31-60 cm (partially also 61-100 cm) depth in 5 fields per agrochemical centre.

As also the leaching losses after autumn application of slurry do not only depend on the soil class but decisively on the meteorological conditions in autumn and winter, these values have to be equally considered in the correction of mineral nitrogen fertilization. Thus the nutrient supply will be sufficient and the residues kept in limits. Greatest variations in N_{in} content are found on medium soils as on sandy soil the nitrate nitrogen is almost completely washed out by normal precipitations, whereas on heavy soils with high water capacity nitrogen translocation is low even after heavy precipitations. The annual differences may reach considerable values.

The evaluation of plot and farm-scale dispersed trials and data collection on the farm show that it is possible to reach better yield stability and to improve utilization of the existing yield potential by means of an aimed N supply to winter cereals.

3.2 Plant analysis

In the GDR, plant analysis as a measure for 'operational fertilization advice' is being carried out in winter cereals and shall be applied to other crop species (sugar beet, vegetables) in future. Under the conditions of the GDR it has been shown that the N rates required for reaching high crop yields and maximum utilization of the yield potentials, as a rule, should be split in two dressings because of the disposition of the cultivated cereal crops to lodging. This should also apply to future varieties with better resistance to lodging. Whereas the N_{in} content in soil, winter precipitation, and the soil climatic region are mainly used for exactly determining the first nitrogen dressing according to EDP fertilization recommendations, it is the plant's nitrogen content at the start of shooting that forms the basis for the exact determination of the second nitrogen dressing to winter cereals.

Sampling takes place when the cereal plants are 20 - 40 cm high and have reached FEEKES stages 4 to 7. It is jointly prepared and carried out by the farms for crop production and their agrochemical centres under the supervision of the ACUB staff members. Attention has to be paid that sufficient time is left between the first nitrogen dressing and plant sampling. The optimal time for plant analysis is given with the cereal plants reaching a height of between 30 and 35 cm.

Within 4 days the ACUB analyzes the samples for their nitrogen content and conveys the recommendation for a second nitrogen dressing to the farms for crop production.

The following standards are valid for the elaboration of the fertilization recommendation on the basis of the nutritional status of the plant:

Very high nitrogen content	0 kg N/ha
High nitrogen content	30 kg N/ha
Medium nitrogen content	N fertilization according to EDP recommendation
Low nitrogen content	"
Very low nitrogen content	"

The effectiveness of N fertilization is expected to be highest in the low to medium range of plant nutrition. Therefore the complete EDP rate is applied in this case. Very low N contents indicate N deficiency calling for immediate fertilization; an increase of the EDP rate is, however, not necessary in general, unless the EDP rate is very low (below 45 kg N/ha) or the first N dressing had been higher. In this case an addition of 10 to 20 kg N/ha may be advisable.

Very dry weather after the first N dressing may cause further corrections in the following stages of nutrition.

Deviations of the mentioned scheme for fertilization recommendations may also result from abnormal crop densities. If it is noted on the data sheet that high crude protein cereals are to be produced (without providing for a third dressing), the second N dressing is fully applied even at high N contents.

All the other factors of influence on the N regime (soil class, preceding crop, organic manuring to preceding crop and main crop, the use of culm stabilizers, sprinkler irrigation) are considered in the optimal level of the second dressing as is to be seen from the EDP programme, and generally need not be judged separately.

4. Final remarks

The introduction of the fertilization system allowed to plan fertilizer use for the whole farm for crop production as well as for each field of the farm and to further improve the effectiveness of nitrogen fertilization by means of the EDP recommendation and the operational recommendations 'Topical Advice' and 'Plant Analysis' on nitrogen fertilizer application to winter cereals. Further limitation of the possible leaching of fertilizer nitrogen is one objective of this fertilization system. A certain disadvantage of this system consists in that it is necessary to proceed from long-term mean values determined in field and farm-scale dispersed trials in elaborating EDP fertilization recommendations, and that it is not possible to exactly consider in advance the meteorological conditions which have a particularly strong influence on the effect of nitrogen fertilization. The 'operational' recommendations on the application of nitrogen fertilizers to winter cereals and in future to further crops, too, at the start of the growing season and in spring shall reduce this problem.

The EDP project 'Fertilization' and the 'operational' fertilization recommendations 'Topical Advice' and 'Plant Analysis' constitute a well proved basis for decisions to be taken by the heads of farms for crop production in the management, planning, organization and control of fertilizer use. Extensive research work has still to be done to complete the scientific character of these fundamental decision aids particularly with a view to overcome uncertainties resulting from meteorological conditions and further minimize the nitrogen losses, especially on light soils. It becomes necessary to care for the complex effect of field-related fertilization recommendations, EDP sprinkling advice, and the EDP system of pest control. To this end, it is advisable to set up optimization models for the complex interaction of the intensification factors to be able to make better use of the possible combining effects through mathematical and cybernetic methods including systems analysis, and to organize more effectively the production processes in their entirety (RODENBAM, 1979). This will contribute to make matter circulations in nature and the relations between man and environment more efficient for the benefit of the population and the protection of nature.

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POSSIBILITIES OF CONTROLLING NITRATE CONCENTRATIONS IN DRINKING WATER

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In this paper, it is emphasized that the analysis of the problem of nitrate concentration in tap or drinking water should also take account of other nitrogen forms (organic nitrogen, ammonia, nitrites), which under certain conditions may be transformed into nitrate nitrogen. Three integrated non-point sources of nitrogen entering tap water can be identified, including both natural and anthropogenic factors: (a) nitrogen substances contained in storm-water, (b) nitrogen substances leached out of soil and farmlands, entering the groundwater, and (c) nitrogen substances in surface runoff. Based on the results of various studies carried out in Poland, the three nitrogen sources are described in detail. Taking account of the allowable standards of nitrogen concentration in water resources for public supply and in drinking water, the current nitrogen hazards threatening potable water are presented. The risks occur mainly in small local water distribution systems which rely on shallow groundwaters. In large water supply systems utilizing surface water, the main problem is organic nitrogen and ammonia compounds.

In conclusion, the main directions of research on nitrates in tap water are presented. Operational experience has shown that the traditional methods of water treatment are quite ineffective as regards nitrate elimination. In view of this, attention is drawn to the significance of "non-treatment methods" involving, among others, implementation of a dual water supply system. Such a system of varied quality water distribution permits a rational utilization of good quality water resources for safe drinking water supply to the population.

VARIOUS NITROGEN FORMS IN WATER

Discussion of the problem of water nitrates should also take account of other nitrogen compounds occuring in water. As known, the practical evaluation of water quality involves assessment of the following nitrogen forms:

- organic nitrogen /N-org/ contained in organic nitrogen coumpounds /proteins, peptones, amino acids, pyridine, urea, amines, etc./,
- ammonia - nitrogen /$N-NH_4$/,
- nitrite - nitrogen /$N-NO_2^-$/,
- nitrate - nitrogen /$N-NO_3^-$/.

These various forms of nitrogen are to occur as interrelated and, depending on water conditions, the individual nitrogen substances may undergo changes, one being converted into another, as in the following diagram:

$$N_{org} \rightleftharpoons N_{NH_4} \rightleftharpoons N_{NO_2} \rightleftharpoons N_{NO_3}$$

The above diagram does not include the whole cycle of conversions and all the nitrogen forms occurring in this cycle. Therefore, the diagram cannot be a basis for the development of a dynamic model of nitrogen balance in water. Nevertheless, if the content of nitrogen compounds in water at a given time is being considered, it may be assumed that the content of the mentioned four forms of nitrogen makes up the so-called total nitrogen /total fixed nitrogen - N-Tot/ contained in water at a given time:

$$N_{Tot} = N_{org} + N_{NH_4} + N_{NO_2} + N_{NO_3}$$

The amount of organic nitrogen, ammonia-nitrogen and nitrite--nitrogen should be interpreted as potential amounts of nitrate--nitrogen, since under the conditions which favour mineralization or organic material and nitrification processes, organic nitrogen, ammonia and nitrites may be eventually converted into nitrates /at the same time a certain part of nitrogen may be lost, due to various causes/. Apart from that, from the point of view of tap water quality criteria,the ammonia nitrogen content itself is of significance, as it must not exceed certain values. The same applies also to nitrite-nitrogen but, in view of poor stability of nitrites, it is a realatively minor problem.

SOURCES OF NONPOINT WATER CONTAMINATION WITH NITROGEN COMPOUNDS

Theoretically, the sources of nonpoint nitrogen water contamination may be divided into natural and anthropogenic. The natural sources include: a/ substances carried out from natural soil and ground as a result of erosion and leaching, b/ precipitation with a slight content of ammonia and nitrates arising from electrical discharges in the atmosphere, c/ underground water coming up to the surface from soil layers rich with mineral nitrogen compounds, d/ biologic origin substances /water birds excrements, pollen, leaves, etc./. The anthropogenic sources of nitrogen water contamination include: a/ fertilizers and manure utilized in agriculture and forestry, b/ excrements of animals bred in the open space /pastures/, c/ industrial contaminants of free air picked-up by precipitation, d/ stormwater run-off from the surface of industrial and urban areas, e/ wastewaters filtrating into the soil from leaky septic tanks and other sewerage units, f/ wastewaters applied for farm and forest crop irrigation.

Theoretical classification of nonpoint contaminants into natural and anthropogenic is of low value in the practice of environment control, since in most cases the natural and anthropogenic sources of contamination are overlapping. This way, natural nitrogenic substances carried away from soil through erosion and leaching merge with nitrogenic substances coming from fertilizers and animal excrements. Natural content of ammonia and nitrates in the air merge with nitrogenic substances of industrial origin. Natural surface run-off and surface water flows are enriched with nitrogen compounds originating from industrial and urban areas. Under such circumstances it is advisable to consider integrated sources of dispersed nitrogen contaminants, i.e.:

- nitrogen substances contained in stormwater including the natural substances and air contaminants picked up from free air,
- nitrogen substances leached out from soil and transferred into ground water, including above all, nitrogen compounds coming from fertilizers and manure, as well as from sewage application for land irrigation or infiltrating into the ground from different sewerage units, and soil natural nitrogen compounds,
- nitrogen compounds contained in surface run-off, natural or coming from organic or mineral fertilizers and various contaminants which get into the ground surface.

Nitrogen substance concentration in stormwater run-off varies within a wide range and depends mainly on rainfall level. The total volume of nitrogen compounds in stormwater is also affected by air pollution level. Table 1 presents the average ammonia nitrogen and nitrate nitrogen concentration levels in stormwater run-off in agricultural, industrial and mountain areas, as reported by A. Chojnacki /1971/. Nitrogen load on the surface during one year, according to Chojnacki /1971/ is as follows:

- for agricultural areas: 4.1 - 10.5 kg/ha $N-NH_4$; 1.7 - 5.2 kg/ha $N-NO_3$; 6.7 - 16.1 kg/ha N-Tot.,
- for industrial areas: 4.1 - 20.6 kg/ha $N-NH_4$; 20.5 - 4.0 kg/ha $N-NO_3$; 6.6 - 24.5 kg/ha N-Tot.,

Table 1. Nitrogen compounds concentration in stormwater runoff, based on Chojnacki studies /1971/

Type of area	Average concentration in annual rainfall volume, g N/m³		
	N_{NH_4}, g/m^3	N_{NO_3}, g/m^3	N_{Tot}, g/m^3
Agricultural areas[1/]	0.71-2.27	0.36-0.61	1.29-2.77
Industrial areas[2/]	0.53-2.52	0.32-0.51	0.86-3.00
Mountain regions[3/]	0.88-1.18	0.23-0.45	1.1 -1.63

1/ Data from 14 meteorologic stations

2/ Data from 3 meteorologic stations

3/ Data from 2 meteorologic stations

-for mountain areas: approximately 19 kg/ha $N\text{-}NH_4$; 4.9 - 7.3 kg/ha $N\text{-}NO_3$; 24.0 - 26.4 kg/ha N-Tot.

These volumes of nitrogen load, compared with the amount of nitrogenous fertilizers used, may be considered significant.

Concentration of nitrogenous substances leached-out from the soil depends on numerous factors, the most important of which are the fertilizer dose, type of soil and technical design of the soil drainage system. Table 2 presents nitrogen compounds concentration in drainage water from an experimental corn field, according to the studies by Szymańska /1978/. Soil filtration coefficient on this field was 3.9 - 11.9 cm/d, drain pipe spacing was 9 - 21 cm, and drain depth was 1.2 m. It can be seen from Table 2 that the average concentration of total nitrogen in drainage water was approximately 15 g N-Tot/m^3 with the fertilizer application rate of 138 kg N/ha, and approximately 18.3 g N-Tot/m^3 with that of 276 kg N/ha. For another field studied, Szymańska /1978/ reports markedly higher average nitrogen concentration in drainage water, approximately 40.5gN-Tot/m^3 with the application rate of 125 kg N/ha, and approximately 52.4 g N-Tot/m^3 with that of 250 kg N/ha.

Table 3 presents the results of studies by Morgowski and Bartoszewicz /1977/ who found high variation of nitrogen compounds concentration in ground water, depending mainly on the type of land use. On foreset areas, where no mineral fertilizers were used, concentration of nitrogen compounds in ground water and drainage ditches was relatively low, amounting to approximately 4 g/m^3 N-Tot, while on agricultural areas concentration of nitrogen compounds in ground water was approximately 15 g N-Tot/m^3, and in drainage ditch water - approximately 10 g N-Tot/m^3. Meadows and pasture areas were characterized by a relatively high content of organic nitrogen in ground water /2.5 g N-org/m^3/.

The content of nitrogen compounds in run-off from the ground surface depends naturally on the type of land use. Table 4 presents the results of studies by Kostecki /1980/ on nitrogen compounds concentration in run-off from crop fields, meadows, pastures and forest areas. This author found that the annual load of nitrogen in surface water stream from crop fields was approximately 10 kg N/ha. Out of that, nitrate nitrogen content amounted to approximately 8.4 kg N/ha. A high load of nitrogen was also carried out from the surface of meadows and pastures - approximately 11.5 kg N/ha; in that case, however, the major part of it was organic nitrogen, coming probably from animal excrements. The studies by Kostecki /1980/ dealt with a small water basin with the surface of approximately 25 km^2. Januszkiewicz /1976/ analyzed nitrogen compounds balance in water run-off into the Vistula River from a very large basin. Its surface was approximately 170,000 km^3, that is about 55 % of the total area of Poland. He found that the annual load of nitrogen compounds washed out from the surface was approximately 4.1 kg N_{Tot}/ha /1.6 kg N-org/ha, 0.6 kg $N\text{-}NH_4$/ha, 1.9 kg $N\text{-}NO_3$/ha/.

Table 2. Nitrogen compounds concentration in drainage waters from crop fields, according to Szymańska /1974/

Nitrogen form	Fertilizer application rate 138 kg N/ha			Fertlizer application rate 138 kg N/ha		
	min.	max.	average	min.	max.	average
N_{org} g/m^3	0.14	4.76	1.18	0.21	4.16	1.30
N_{NH_4} g/m^3	0	0.68	0.13	0	1.55	0.15
N_{NO_2} g/m^3	0	0.012	0.005	0	0.036	0.01
N_{NO_3} g/m^3	4.0	19.2	13.7	10.5	32.4	16.9
N_{Tot} g/m^3	4.14	24.642	15.015	10.71	38.146	18.36

Table 3. Nitrogen compounds concentration in ground water and drainage ditches in areas of different land use /average values during 1973-74/

Land use	Type of water	Nitrogen concentration, $g\ N/m^3$			
		N_{org}	N_{NH_4}	N_{NO_3}	N_{Tot}
Agricultural fields crop soil medium sand on sandy loam. Fertilizer application rate 100 kg N/ha	a/ ground water	0.9	1.4	12.6	14.9
	b/ water in drain ditch	0.9	0.8	8.1	9.8
Meadows and pastures. Muck soils on loose sandy soils. Fertilizer application rate 80 kg/N/ha	a/ ground water	2.5	2.2	0.2	4.9
	b/ water in drain ditch	1.6	0.8	0.1	2.5
Coniferous forests. Loose sandy soil without fertilizers.	a/ ground water	1.5	2.7	0.3	4.5
	b/ water in drain ditch	2.4	1.3	0.1	3.8

Source: Margowski and Bartoszewicz /1977/.

While discussing the problem of nonpoint sources of water contamination with nitrogen compounds it should be emphasized that nitrogenous fertilizers are also an indirect cause of some point sources of water contamination. Fertilizer manufacturing plants discharge wastewaters with a high concentration of nitrogen compounds. Gromiec /1977/ reports the following concentration of nitrogen compounds in wastewaters discharged from a fertilizer manufacturing plant:

- organic nitrogen /N-org/	275 - 1579 g/m^3
- ammonia nitrogen /$N-NH_4$/	317 - 931 g/m^3
- nitrite nitrogen /$N-NO_2$/	3 -12.79 g/m^3
- nitrate nitrogen /$N-NO_3$/	73 - 642 g/m^3
- total nitrogen /N-Tot/	854 - 2459 g/m^3

Fertilizer plants also discharge large volumes of nitrogen compounds into the air which spread and are, in turn, a source of surface contaminants /through rainfall/.

HAZARDOUS EFFECTS OF NITROGEN COMPOUNDS ON DRINKING WATER QUALITY

The presence of nitrogen compounds in public water supply sources has two aspects: firstly, too high concentration of nitrogen compounds in tap water is dangerous for the population health, and secondly - the content of nitrogen compounds in river or stream water favours eutrophication and plankton growth which, in turn, cause technical problems in water treatment.

With reference to health risk arising from the content of nitrogen compounds in water, the following recommendations of "Water Quality Criteria" /1972/, a report prepared for the U.S. Environmental Protection Agency, may be quoted:

- "On the basis of adverse physiological effects on infants and because the defined treatment process has no effect on the removal of nitrate it is recommended that the nitrate-nitrogen /$N-NO_3$/ concentration in public water supply sources not exceed 10 mg/l."
- "On the basis of its high toxicity and more pronounced effect than nitrate, it is recommended that nitrite-nitrogen /$N-NO_2$/ concentration in public water supply sources not exceed 1 mg/l."
- "Because ammonia may be indicative of pollution and because of its significant effect on chlorination it is recommended that ammonia-nitrogen /$N-NH_4$/ concentration in public water supply sources not exceed 0.5 mg/l."

The Polish standards on the quality of water supply sources distinguish three classes of water purity:

Class 1 - water sources of public water supply for food industry and salmon-type fish breeding,

Class 2 - water supply sources utilized in fish breeding /except for salmon-type fish/, to water animals, for recreation and water sports,

Class 3 - water supply sources for industry /except for food industry/ and irrigation of crop fields.

Table 4. Average annual concentration of nitrogen compounds in run-off water and annual loading of nitrogen leached-out from different areas, according to Kostecki /1980/

Nitrogen concentration and load in run-off water	Run-off from crop fields	Run-off from meadows and pastures	Run-off from forest areas
Nitrogen concentration			
- N_{org}, g/m^3	0.38	2.87	0.15
- N_{NH_4}, g/m^3	0.50	0.42	0.75
- N_{NO_2}, g/m^3	0.01	0.06	0.01
- N_{NO_3}, g/m^3	3.08	0.77	0.33
- N_{Tot}, g/m^3	3.96	4.14	1.25
Leached-out nitrogen			
- N_{org}, kg N/ha	1.06	8.03	0.09
- N_{NH_4}, kg N/ha	1.33	1.19	0.47
- N_{NO_2}, kg N/ha	0.09	0.17	0.01
- N_{NO_3}, kg N/ha	8.40	2.15	0.02
- N_{Tot}, kg N/ha	10.88	11.54	0.77

Table 5 presents the quality standards for water supply sources as regards nitrogen compounds concentration, applying in Poland. Besides those standards, there are also drinking water quality standards which state that the concentration of ammonia-nitrogen /$N-NH_4$/ in drinking water must not exceed 0.5 mg/l, and that of nitrate-nitrogen /$N-NO_3$/ - 10 mg/l.

With regard to eutrophication it may be said that, for this reason, a lower concentration of nitrogen compounds would be desirable, since it has been generally accepted that the risk of eutroplhication and algae growth arises already with the nitrogen compounds concentration exceeding 0.3 mg/N-Tot/l.

In larger municipal water supply systems in Poland there is no problem of nitrates. Wiohrowska at al. /1979/ report that out of a total of 24 larger cities /over 100,000 thousand population/ studied, nitrate-nitrogen concentration did not exceed 2 mg $N-NO_3$/l in 14 cities, in 7 cities it remained within 2 - 5 mg $N-NO_3$/l, and in 3 cities - within 5 - 10 mg $N-NO_3$/l. Nitrite content in tap water of those cities did not exceed 0.5 mg $N-NO_2$/l, and in 15 cities it was lower than 0.01 g $N-NO_2$/l. The data quoted are the average annual values.

Table 6 presents fluctuations of nitrogen compounds content in the Vistula River water, the largest Polish river, during the recent few decades. The Vistula River waters have the 1st class of purity, and it may be seen that the concentration of different forms of nitrogen in river water approaches the permissible values /Tabel 5/. At the same time, ammonia-nitrogen concentration already exceeds the tolerable concentration levels for drinking water. Table 6 also contains data on the increased utilization of fertilizers in Poland during 1937-1980.

Smaller water supply systems and individual local water supply systems based on ground water are at a greater risk of high concentration of nitrogen compounds. Pilawska and Toruń /1971/, based on the studies carried out at the Szczecin region /North-Western part of Poland/ report that out of a total of 381 wells tested in 1970, 20 % had concentration of up to 10 mg $N-NO_3$/l, 30 % - 50 - 99 mg $N-NO_3$/l, and in 20 % the concentration exceeds 100 mg $N-NO_3$/l. The above data point out at a hazardous effects of nitrate on the quality of water supplied by systems based on ground water or distributing water to small rural estates or individual housings.

PROGRAMME OF INVESTIGATIONS ON THE PROBLEM OF NITRATES IN TAP WATER

The problem of nitrogen compounds in tap water is of a complex nature and, therefore, its solution requires multi-directional steps. The following actions may be mentioned as the most important:

I Improvement of crop technology aimed at reduced utilization of nitrogenous fertilizers, their more effective application and reduction of fertilizer leakage from soil.

II Identification of main sources of nitrogen entering water resources in the individual regions, field studies on spread-out and migration of nitrogen compounds in soil and water environment, mathematical modelling of those phenomena.

Table 5. Polish standards of water resources quality as regards nitrogen compounds content

Purity class of water resources	Permissible concentration of nitrogen compounds in g N/m^3		
	N_{org}	N_{NH_4}	N_{NO_3}
Class 1	≤ 1	≤ 1	≤ 1.5
Class 2	≤ 1	≤ 3	≤ 7
Class 3	≤ 10	≤ 6	≤ 15

Table 6. Utilization of mineral fertilizers and fluctuations of nitrogen compounds concentration in river water

Year	Utilization of mineral fertilizers kg N/ha	Concentration of nitrogen compounds in river water /Vistula River in the middle part/ Average annual values, g/m^3				
		N_{org}	N_{NH_4}	N_{NO_2}	N_{NO_3}	N_{Tot}
1934			0.08		0.16	
1937/38	2.0.					
1949/50	6.2	0.744	0.041	0.006	0.35	1.144
1959/60	16.4	0.828	0.203	0.014	0.71	1.755
1969/70	40.2	1.09	0.71	0.041	1.01	2.851
1979/80	69.0	1.52	0.73	0.054	1.3	3.604

III Improvement of water treatment technology, aimed at the development of effective methods of nitrogen compounds elimination.

IV Development of highly effective methods of wastewater treatment as regards denitrification and water regeneration.

V Development and improvement of water supply and sewerage systems
- replacement of local systems with regional systems,
- utilisation of different water sources permitting the reduction of nitrate concentration in tap water,
- application of water supply systems distributing different-quality water /dual water supply systems/,
- application of water closed-circulation systems.

VI Regional studies and economic analyses of the effectiveness of different technical solutions in order to select the optimum design for the conditions of a given region.

Here are some comments on the mentioned six directions of development and studies.

Improvement of agricultural technology mentioned in I. may theoretically have the fundamental meaning for prevention of water contamination with nitrogen compounds but, in practice, we cannot expect significant effects here because the agricultural industry will concentrate, above all, on the increase of food production. Szymańska /1978/ reports a method of reducing migration of nitrogen compounds into the soil, i.e. recirculation of drainage water with high nitrogen compounds concentration and its reuse for crop irrigation.

Field studies on the migration of nitrogen substances in the environment and mathematical modelling of those phenomena /item II/ should permit future forecasting of changes in water resources quality and form a basis for the proper regional water supply management.

Improvement of water technology aimed at the elimination of nitrogen compounds, mentioned in III. is necessary, since the present methods of water treatment are so far ineffective as regards nitrogen compounds, especially nitrates. Table 7 presents the results of studies on changes in nitrogen compounds content in the course of a traditional method of water treatment, i.e. sedimentation, rapid filtration /no coagulation/, and slow filtration. Table 8 gives similar results concerning another traditional method of water treatment using sedimentation, coagulation /in pulsators/ and rapid filtration. None of those methods reduces nitrate content, and in case of water treatment plant using slow filtration the resulting nitrate concentration is even slightly increased due to the nitrification process. One of the methods of nitrogen compounds elimination is ion exchange. Kosiński et al. /1974/ in their studies on the application of selective ion exchange in ammonia elimination from water have obtained the results confirming that the method is technically efficient and cost-effective.

Table 7. Fluctuations of nitrogen compounds concentration in the course of traditional water treatment processes sedimentation - rapid filtration without coagulation, and slow filtration /data refer to the average annual values in a large river-water treatment plant/

Consecutive processes	Concentration of nitrogen compounds in water $g\ N/m^3$				
	N_{org}	N_{NH_4}	N_{NO_3}	N_{NO_2}	N_{Tot}
Raw water	1.52	0,73	0.054	1.3	3.604
After sedimentation /12 h retention/	1.26	0.69	0.054	1.45	3.454
After rapid filtration	0.98	0.23	0.015	1.8	3.025
After slow filtration	0.78	0.17	0.004	1.9	2.854

Table 8. Fluctuations of nitrogen compounds concentration in the course of traditional water treatment processes: sedimentation, coagulation /in pulsators/ and rapid filtration /data refer to the average annual values in a large river-water treatment plant/

Consecutive processes	Concentration of nitrogen compounds in water $g\ N/m^3$				
	N_{org}	N_{NH_4}	N_{NO_2}	N_{NO_3}	N_{Tot}
Raw water	1.52	0.73	0.054	1.3	3.604
After sedimentation /12 h retention/	1.26	0.69	0.054	1.45	3.454
After coagulation /in pulsators/	0.85	0.47	0.007	1.35	2.677
After rapid filtration	0.44	0.44	0.009	1.4	2.680

Wastewater denitrification mentioned in IV. is at present considered mainly from the point of view of preventing river and lake water eutrophication. However, for that reason, the method has not been sufficiently implemented because it is believed that eutrophication may be controlled more effectively through phosphorous level control in wastewaters discharged into the receiving waters. In Poland denitrification has been considered mainly as regards industrial wastewaters discharged from nitrogenous fertilizer-manufacturing plants. Bożko et al. /1976/ and Wróbel et al./1979/ carried out pilot studies on nitrate elimination through algae culture. Using this method, the authors obtained 30 - 50 % elimination of nitrogen from wastewaters.

The steps in development and improvement of water supply and sewerage systems mentioned in V. include a number of methods aimed at the solution of the problem of tap water nitrogen; they may be defined as "non-treatment" methods. In general, they consist in elimination or reduced utilization of water sources with a high nitrate content. As a rule, these methods are associated with higher costs of transportation and sometimes also more expensive water distribution. One of the solutions of that type may be a dual water supply system in which higher-quality water /in this case - low nitrate concentration/ is distributed through a separate system for population and food industry, while lower-quality water /higher nitrate content/ is distributed through another network to the remaining users. From the economic point of view, the dual water supply systems is feasible when the increased costs of transportation and distribution of two types of water are compensated for by cost savings resulting from the elimination of expensive method of nitrogen removal. Technical, economic and sanitary aspects of dual water supply systems were analyzed by Roman /1978/.

Regional studies postulated in item VI. should permit the most effective solution of the problem of protecting drinking water from excessive nitrogen compounds content, with regard to the specific conditions of a given region. It seems that no universal solution to this problem will be found, as each region, depending on its land use and condition of water resources, will require different solutions to the problem of nitrogen in drinking water.

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POSSIBILITIES OF WATER MANAGEMENT FOR PROTECTING AND TREATING DRINKING WATER RESOURCES IN CASE OF NITRATE POLLUTION

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This paper gives an overview of some of the methods for managing the nitrate problem from the point of view of water management. The main types of methods distinguished are those for reducing or preventing nitrate inputs into water bodies, for controlling or treating water bodies with too high a nitrate content, and treatment in water works. Within the first group mentioned, particular attention is given to the establishment of drinking water protection zones, while the discussion of the second group addresses methods such as hydraulically controlling, or biologically treating surface water bodies. Ion exchange and artificial infiltration are recommended for implementation in waterworks.

*This paper was presented by R. Enderlein, Institute of Water Management, Berlin

1. Introduction

It is the objective of this Seminar to demonstrate how it is possible by methods of system analysis to thoroughly investigate the nitrate load of drinking water resources and which qualified decision bases for alternative solutions in the framework of regional water management have to be elaborated. With this in mind it is useful to give a survey on some possibilities for protecting and treating the drinking water resources, simultaneously showing their positive and negative aspects. The scientific as well as technical and technological fundamentals on which the processes are based shall only be mentioned inasmuch as they are required for proper understanding.

It is not the aim of this lecture to give a complete international survey. On the contrary, we want to speak about some experiences made in the GDR in this respect. According to Lauterbach, Tiemer, Busch and Luckner (1977) it shall be pointed out that the raw water for drinking water supply is coming from the following sources:

The first position on national scale is taken by the ground water since more than 70% of the drinking water demand are satisfied from this source. In many regions the supply is covered from relatively shallow ground water veins which are often not at all or insufficiently protected against contamination by impermeable covering layers.

Large regions of the highlands and their forelands are supplied with drinking water from drinking water reservoirs which have been specially built for this purpose; the lowland regions are partly supplied from lakes.

Finally, in overcrowded industrial and municipal regions there exists the necessity of taking surface water from rivers and lakes in order to enrich the ground water artificially; in isolated cases it is necessary, however, to directly treat this raw water which is generally of very poor quality in water works.

In the following we want to demonstrate possibilities of protection and treatment, however, it must be immediately accentuated that measures of protection are always given priority over measures of treatment. As in many other spheres of social life, the known proverb is also true in this field, i.e. "Prevention is better than cure".

Following this principle, the subsequent ideas shall be subdivided into 3 main sections:

1. Possibilities for reducing or preventing a nitrate input into water bodies
2. Possibilities of controlling or treating water bodies with too high a nitrate load

3. Possibilities of treating the nitrate-loaded raw water in the process of treatment to drinking water in water works.

2. Possibilities for reducing or preventing a nitrate input into water bodies

2.1. Drinking water protection areas

Since industrialization of the economy in general and of agriculture in particular is making great progress - the same is true of progressive urbanization in many areas - but, on the other hand, the protection of drinking water resources is the number one priority, all efforts are concentrated on supplementing and stipulating measures to be taken in special drinking water protection areas. To this end, all experience and knowledge has been comprised in a draft standard which in the near future will be approved by all authorities responsible. Now we want to present a few results as far as nitrate pollution is concerned.

Catchment areas of water bodies or parts of the latter which are used for obtaining drinking water or which are subject to special protection through prohibition or limited utilization are looked upon as drinking water protection areas.

The entire area to be protected will be subdivded into various zones; see Figure 1.

The protected zone I is the zone of direct water obtainment, and within this zone direct pollution of the withdrawal units would be possible. Adjacent to this zone to the outside comes protected

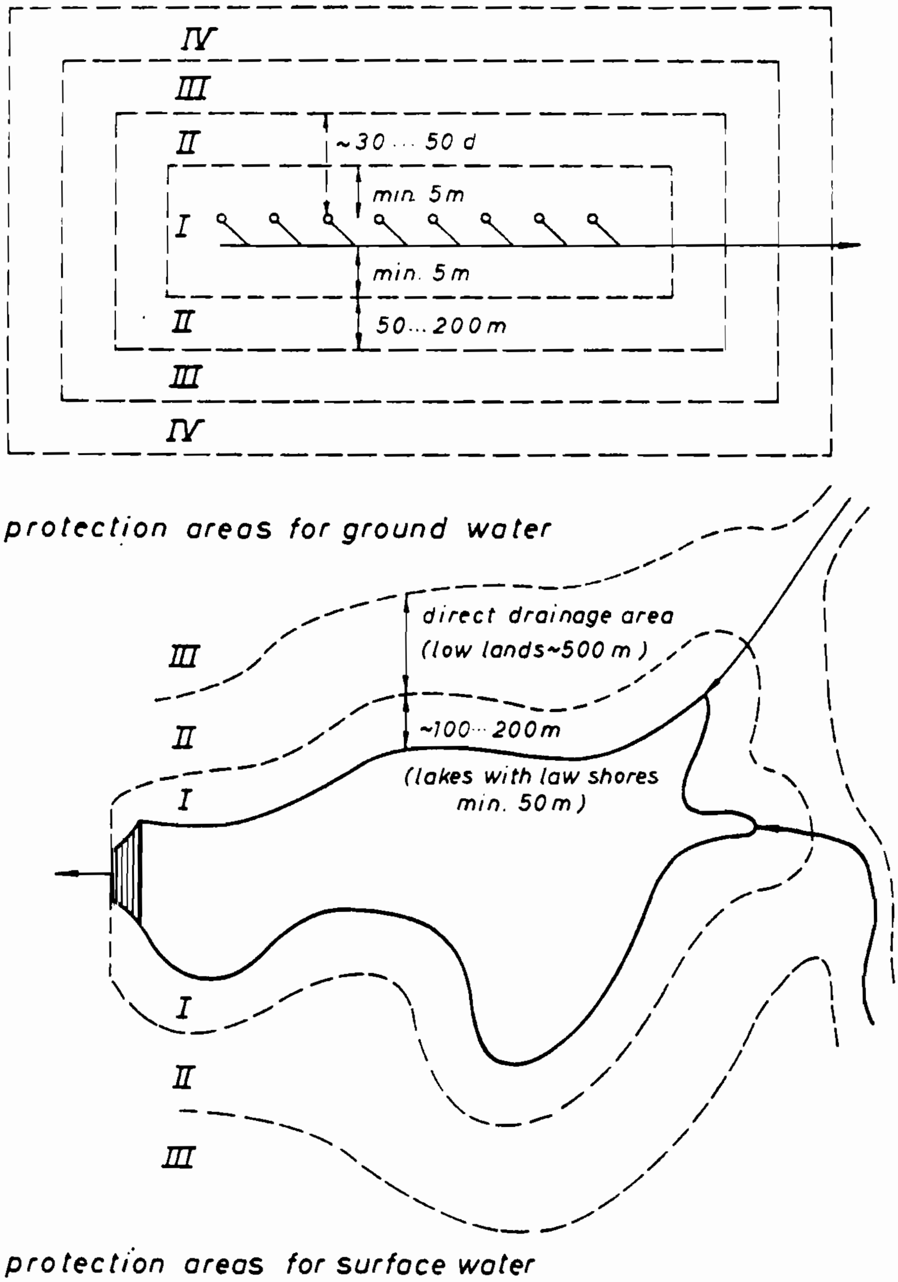

Fig. 1 Scheme of classification for drinking water protection areas

zone II, i.e. the closer protected zone, followed by protected zone III, i.e. the widest protected zone which comprises the underground or the above-ground catchment area. In case of surface water obtainment, zones III and IV fall together.

As shown in Table 1, zone I must give protection against all forms of direct pollution while zone II has the task of microbial and biological degradation, zone III has to exclude any pollution hard to eliminate and zone IV shall prevent any pollution which cannot be eliminated at all. It is also clear from Table 1 that it is possible already to give quite detailed information about the size of zone II on the basis of hydrogeological and geohydraulical stipulations; one of the reasons is the good state of knowledge in the field of mathematical modelling of geohydraulic processes. However, similarly precise information on the size of zone III which is of great importance with regard to the problems of nitrate pollution is still missing. The reason why this is so are insufficient investigations into the processes of degradation and transformation of material in the bottom and ground water including mathematical modelling. The same is true of nitrate pollution, even today, although great progress has been made recently. The draft standard mentioned above specifies in this respect in its "Basic Requirements" to dimension the size of protected zones depending on the location on the basis of scientific and technological investigations.

Tables 2 and 3 give an epitomized survey on some prohibitions and limitations inasmuch as they are of special relevance to

Table 1 Selected assessment bases for drinking water protected areas for ground water (special significance for nitrate)

Zone division	Distances	Retention time	Protection against pollution	Other stipulations
Zone I	min. 5 m from all sides around the well	-	no direct pollution	-
Zone II	favourable underground 50 to 100 m	30 to 50 days up to obtaining water in the caption installation	No microbial or biological degradable contaminants	Ground water vein with covering layer 2.5 m of clayish sand 3.5 m of fine sand 5.0 m of coarse sand
	medium underground conditions 100 to 300 m			No covering layers Partial or low thicknesses; but good cleaning performance
	unfavourable underground special stipulations			cleaning performance also insufficient
Zone III	-	-	No pollution by ... Chemical substances e.g. nitrate, difficult to eliminate	-
Zone IV	-,	-	No pollution by substances which cannot be eliminated	-

Table 2 Epitomized survey on prohibited utilizations (p) and limited utilizations (l) in water protection areas for ground water with special relevance of nitrate pollution

Type of utilization	Protected zone I	II	III	IV
1. Industry				
Sewage discharge	p	p	l	-
Sewage infiltration, subsoil irrigation	p	p	p	l
Waste product disposal	p	p	l	-
Camping grounds, bathing	p	p	-	-
2. Agriculture and forestry				
Used as farm land	p	-	-	-
Spray irrigation of agricultural acreage	p	l	l	-
Permanent pasturage	p	l	-	-
Use of solid inorganic fertilizer	l	l	l	-
Use of liquid inorganic fertilizer	p	p	l	-
Use of solid organic fertilizer	p	l	-	-
Use of liquid organic fertilizer	p	p	l	-
Storage of solid organic and inorganic fertilizer	p	p	l	-
Storage and transport of liquid organic fertilizer	p	l	-	-
Individual livestock breeding	p	l	-	-
Industrialized animal production plants (newly built)	p	p	l	-

Table 3 Epitomized survey on prohibited utilizations (p) and limited utilizations (l) in water protection areas for surface waters with special relevance of nitrate pollution

Type of utilization	Protected zone I	II	III
1. Any type of built-up area			
Enterprises with effluent discharge	p	p	l
Plants for sewage handling	p	l	-
Central sewage-treatment plants	p	p	-
Sewage introduction without sufficient treatment and nutrient elimination	p	p	l
Deep-well disposal of sewage and subsoil irrigation >50 EGW (population equivalent)	p	p	l
Sewage floor drain <50 EGW	p	l	-
Camping sites, holiday camps	p	p	-
Facilities and introduction of nutrients/ permitted limit value	p	p	p
2. Handling of water pollutants			
Waste product disposals	p	p	l
3. Utilizations by agriculture and forestry			
Industrialized animal production plants (newly built)	p	p	l
Used as farm land	p	l	-
Liquid fertilizer	p	l	l
Solid fertilizer, organic and inorganic	p	l	-
Sewage soil treatment	p	p	l
Aeroplane fertilization	p	p	-
4. Utilization for recreation			
Bathing	p	l	-

the nitrate problem. It is general policy in case of limited utilizations that special permissions to be given by authorized bodies or inspectorates are to be obtained. Prerequisites for obtaining such permissions are in the majority of cases specific studies or records, for example records which prove the adherence to EDP-recommendations on fertilizing given to agricultural cooperatives in the GDR.

2.2. Elimination of nitrogen in sewage treatment

Our Institute has gathered rich experience in eliminating nitrogen from artificial sewage and sewage from agricultural animal production plants, partly also from industrial waste water, by way of sewage soil treatment. It is not intended to give detailed information on this subject in the framework of this lecture since a wide range of technical literature on this subject is available, also internationally. Depending on the method applied, the conditions of soil and vegetation, the degree of load per area, given in kgN/ha·a, the operational regime and other factors of influence, elimination performances between 50% and as much as 90% and more may be achieved.

In sewage treatment plants, the processes of nutrient elimination (P, N) are primarily applied with the aim to oppose increasing eutrophication of the water bodies. On an international scale, phosphate elimination is given priority.

For nitrogen elimination in sewage treatment plants microbial denitrification is most economical. A series of process versions - mostly used in combination with the activated-sludge method - has been developed in this respect. After oxidation of the nitrogen contained in the sewage by intensive aeration up to the nitrate stage, the nitrate oxygen is used for oxydating organic substances by adding either internal (raw sewage) or external (molasses,methanol, organic acids) carbon sources (Manczak and Szymanska, 1976). The nitrogen is mainly escaping in molecular form. The hitherto biggest denitrification plant, i.e. the Blue Plains Plant, Washington D.C., denitrifies more than 1 million cubic metres of sewage per day using methanol as carbon donator (Barnard and Meiring, 1975). Special experience made in this respect in the GDR is not available.

3. Measures for nitrate load control and nitrate elimination in water bodies

3.1. Hydraulic control

The following example shows that it seems to be possible in individual cases to improve the quality of raw water along the discharge profile with a view to its nutrient content (N, P) by positive hydraulic control. It must be mentioned, however, that in the present case the absolute height of nitrate concentration is not the direct reason of problems arising in drinking water supply, but nitrogen and phosphorus present in the lake (point X in Figure 2) lead to an increased plankton development, as stated by Bauer, Röbisch and Warnke (1980). This, however, caused great difficulties in the filtration process in the water works (shorter service life of filters) and may result in impairments of odour and taste of drinking water.

Based on detailed investigations of the nutrient import into and export from the lake and the transformation processes taking place in the lake it was possible to set up season-dependent material balances for definite time intervals.

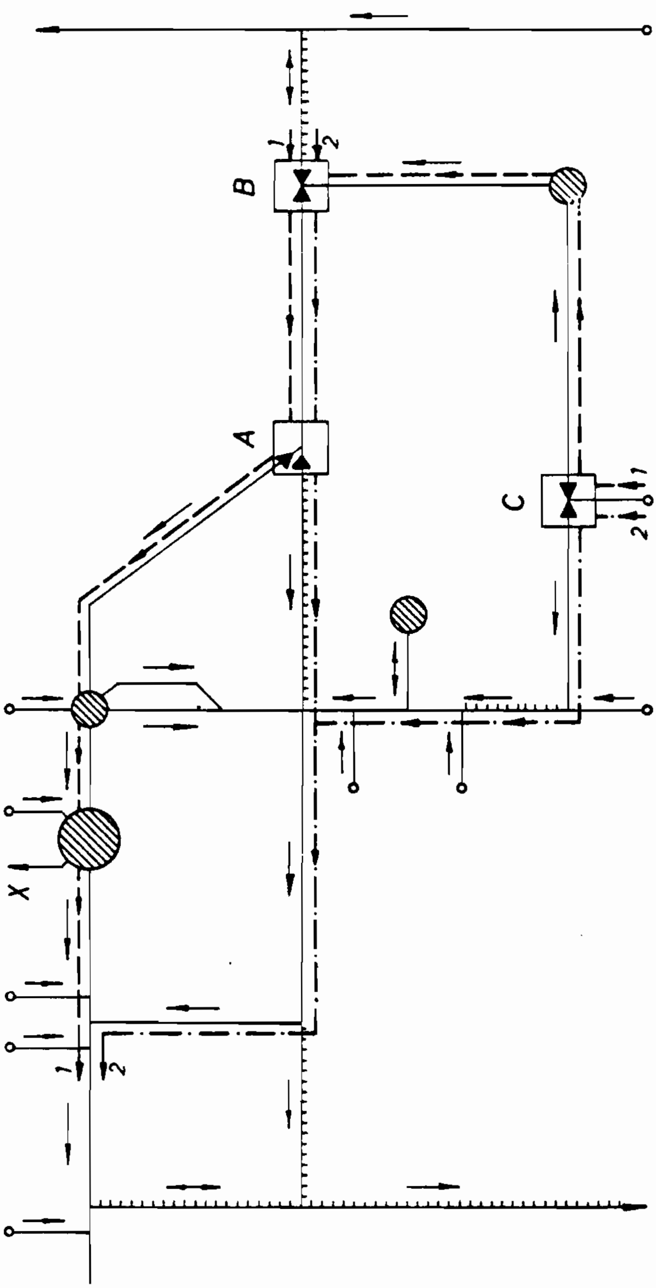

Fig. 2 Scheme of control of the hydrological system

The most important conclusions drawn are as follows:

1. Out of the total inorganic nitrogen imported, some 40% are retained in the lake in the period from August/September to January/February. This means that the true nitrogen import at the time of algea mass development during the main vegetation period is not the decisive factor, on the contrary it is the stock which had been stored in autumn and winter.

2. 15% of the phosphorus import taking place up to a value of 95% in the winter season are stored in the sediment of the lake. Remobilization of this phosphorus which is stored in the sediment occurs in the period from May/June to August/September at a rate of some 10 mg $P/m^2.d$.

3. The increased concentrations of phytoplankton in the lake inlet in spring time lead to a biomass input into the lake which corresponds to a net increase in biomass in the lake of about 30 per cent.

On the basis of these roughly summerized results and by means of control plants (weirs) A, B and C shown in Figure 2 the following control strategy had been established:

- The control plants A, B and C ensure the largest possible passage through the lake - Version: Primary flow - (in the Figure - flow way 1-1) in the period from June 1, to October 15.

  Aim: Export of the phosphate volume released and constant reduction of the phosphate share in the sediment.

In the period from October 16th to May 31st, the control plants A, B and C ensure passage on flow way 1-1 which is required for other utilizations only, while the remaining flow is passed around the lake on flow way 2-2. - Version: Secondary flow

Aim: Reduction of the nitrogen, phosphorus and biomass imports from autumn to spring.

The large-scale test which was started in January 1980, will be continued over a period of several years andit is expected that - in addition to an answer to many detailed problems - this test will throw light on the fact if the reduction of the nutrient load of the lake by an average of 20 to 30 per cent which was assessed by calculation may be really achieved.

It should also be mentioned here that tests of this nature have been made at a reservoir cascade, It was the aim of these tests to prevent by storage management measures exceeding of nitrate limit values in the reservoir from which the raw water for drinking water treatment is taken. Here are two of the results produced:

1. The temporal variations of nitrate concentration throughout the year, showing maxima in winter and minima at the end of summer stagnation, have been considered in the storage management scheme of the upper reservoir in addition to quantity problems, and on this basis a special discharge regime for the lower reservoir has been calculated. Control resulted in a scattering range of the nitrate concentration decreased by 19.4 per cent.
2. The vertical variations in cencentration in the water body

of the lowest reservoir showing somewhat higher NO_3-values in the near-surface layers can be utilized for improving the nitrate balance through the discharge of the natural bed from the epilimnion.

3.2. Incorporation by autotrophic assimilation

The objective is to supply nitrate contained in the water body as nutrient for the growth of algea and higher water plants. The tests made in this respect in drinking water reservoirs produced the following result:

Nitrate assimilation in the reservoir system when used as nutrient for plankton algae is rather low due to the very low phosphorus content as compared with nitrogen. An empirical assessment of nutrient relations, retention times and NO_3-elimination performances of reservoirs in the GDR is clear from the relations shown in Figure 3 (Pütz, 1978). In a reservoir which was studied in detail, elimination in case of a retention time of 113 days amounted to only 10 to 20 per cent per annum. The N:P-relation in water equals 150 : 1, thus being almost the tenfold value as compared with the medium composition of algea which was found to be 16 : 1. The N:P-relation may be harmonized by controlled P-fertilization while algea development and N-elimination can be increased by incorporation. However, such a solution to the nitrate problem would create another problem for drinking water treatment which would be similarly difficult to master.

Moreover, applicability of the nitrophytes method was tested (Niemann and Wegener, 1976). The basic idea is to grow such plants in the shore zone and the region filled up by sedimenta-

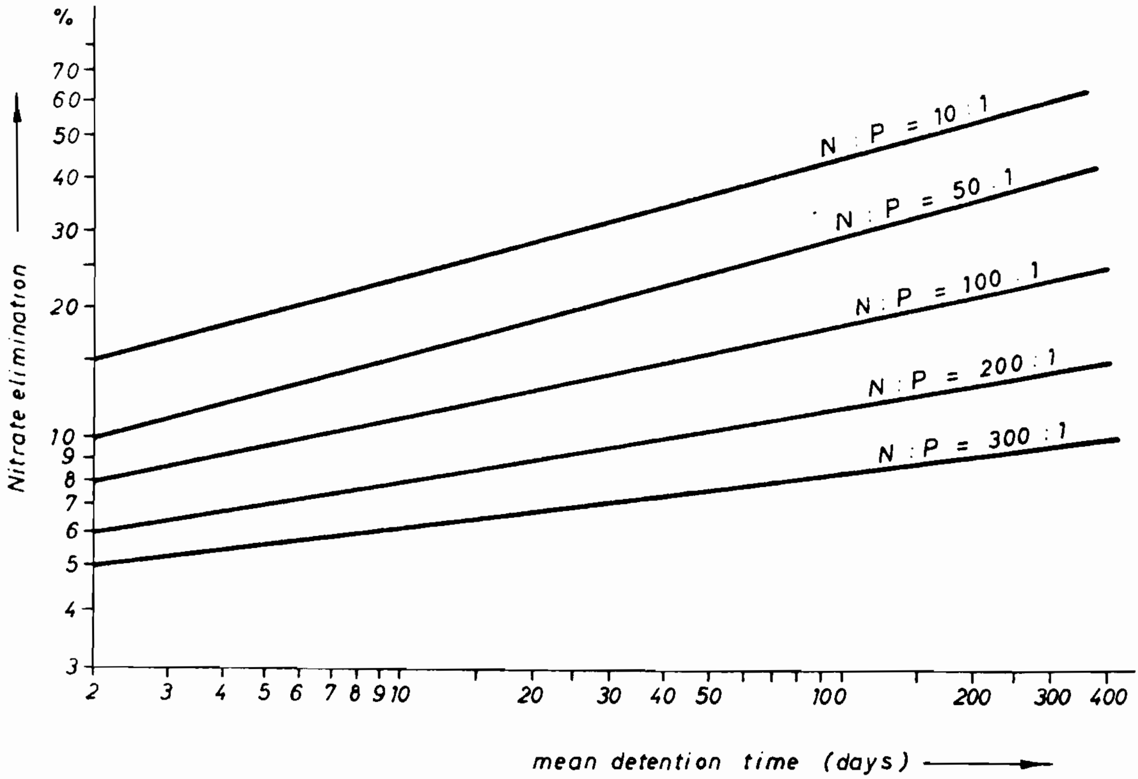

Fig . 3 Estimated relation between NO_3 -elimination and mean detention time depending on mass-relation of anorganic nitrogen and orthophosphate - phosphorus (N : P) tion which are known to have an excessive nitrogen consumption. The volume which could be eliminated by means of the nitrophytes method on an area of 40 hectares - as could be done in the region of the pre-impoudment basins of the object under investigation - was calculated to be some 12 t N/a. However, the problem of complete harvesting of the plants on difficult terrain where machines cannot be used is still far from being solved.

3.3. Nitrate elimination by anaerobic nitrate dissimilation

The partial process which is mostly suited for nitrogen elimination is the anaerobic nitrate dissimilation because nitrogen escapes in molecular form causing a genuine loss. The principle of reaction has been tested years ago in the framework of H_2S fighting in heavily loaded water bodies. In this case nitrate was added as oxygen carrier. The nitrogen balance revealed that by far the largest partwas relieved as N_2.

Successful heterotrophic denitrification requires simultaneous presence of a carbon source as hydrogen donator, an anaerobic environment and nitrate as hydrogen acceptor. The hypolimnion of the upper reservoir was selected to be the reaction zone. A large cage filled with 12,000 bales of straw and equipped with a water distribution system was built on the shore. The sinking straw unit sized 120 x 60 x 1,50 m was floated onto the water and sank after a few weeks to the bottom of the reservoir. The straw serves as cultivating ground for denitrifying bacteria and is partly used as hydrogen source. The principle of the procedure is shown in Figure 4.

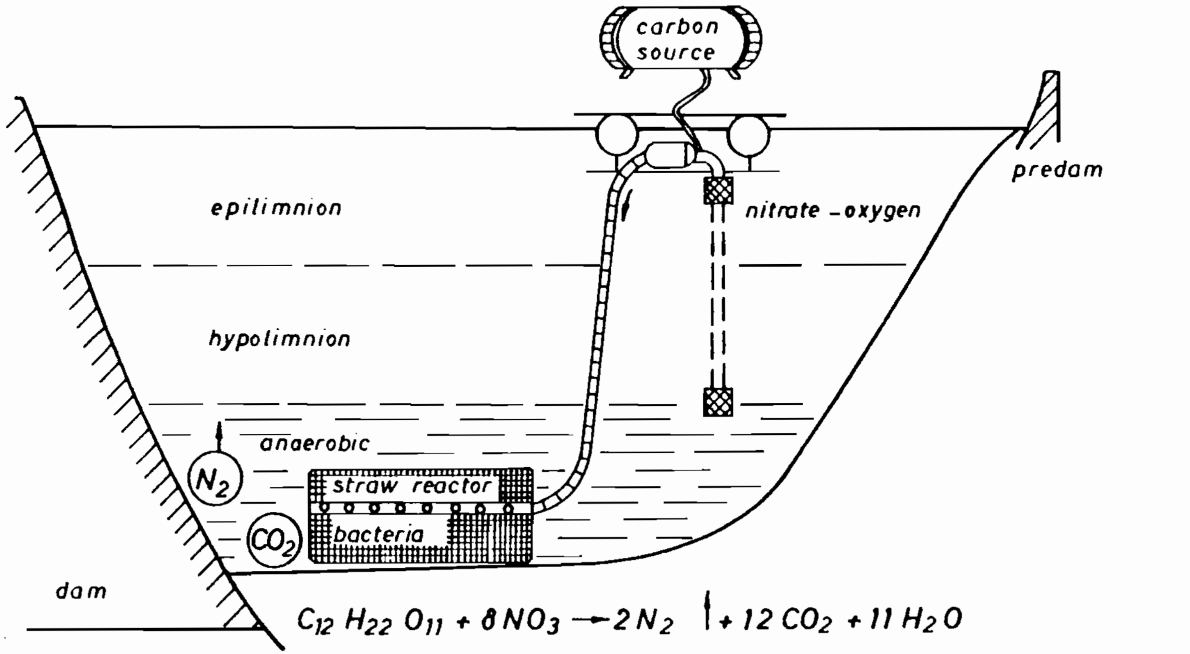

Fig. 4 Heterotrophic nitrate dissimilation within the hypolimnion of a reservior

Water which is rich in nitrate is pumped together with an easily degradable substrate used as hydrogen source through the sinking straw piece. A waste product containing lower fatty acids was used as substrate. 200 tons of this substrate were used to consume the hypolimnic oxygen, and subsequently it was possible to eliminate about 50 t of nitrate. Increased nitrite values occurred temporarily as intermediate product. Table 4 shows the inital state, the consumption of dissolved oxygen, the maximum nitrite values as well as complete nitrate and nitrite elimination over the reactor.

Table 4 Heterotrophic nitrate elimination in the hypolimnion of a reservoir
Selected service results of 1980

Date	Sampling place	NO_3^- mg/l	NO_2^- mg/l	O_2 mg/l
3rd of June	O	43.6	0.32	10.3
	A	40.5	0.60	6.2
	E	41.7	0.58	7.8
1st of July	O	42.5	0.36	9.1
	A	20.6	5.20	0
	E	35.6	5,80	0
17th of July	O	43.5	0.40	9.9
	A	13.2	12.30	0.3
	E	10.4	14.30	0.2
11th of August	O	39.5	0.40	9.3
	A		0.018	0
	E		0.009	0.6

Legend:
O Water surface
A directly over) the straw sinking unit
E 2 meters over)

A patent has been applied for the a.m. procedure.

Experiences have also been gathered in the GDR in the field of autotrophic denitrification which is independent of organic substrate. Thiobacillus denitrificans is a autrophic species of bacteria that oxydize sulphur and sulphur compounds by nitrate oxygen, simultaneously releasing N_2. Preconditions required include an anaerobic environment, a neutral pH-value and CO_2 as inorganic carbon source. The procedure is similar to the one described above, it produces approximately analogous results, but because of the cost of the thiosulphate used, it is more expensive.

4. Measures for nitrate elimination in drinking water treatment

4.1. Ion exchange method

The method is based on the principle that the nitrate-containing raw water is passed through reactors which are filled with strongly alkalinic anion exchange material. These artificial resins take up the nitrates which are dissolved in the water and release an adequate amount of chloride ions.

In the majority of cases, this method will be applied only to a partial stream of the total raw water. This partial stream is subsequently "diluted" with untreated raw water.

The artificial resins will be regenerated by chloride salt solution. The chloride content of the wastewater produced in that way is the limiting factor for its disposal (Figure 5). Extensive investigations made in the GDR, the results of which have been recently published, show that this method should be applied preferably for capacities between 50 and 5,000 m^3/d, partly up to 10,000 m^3/d, mainly because of economic reasons.

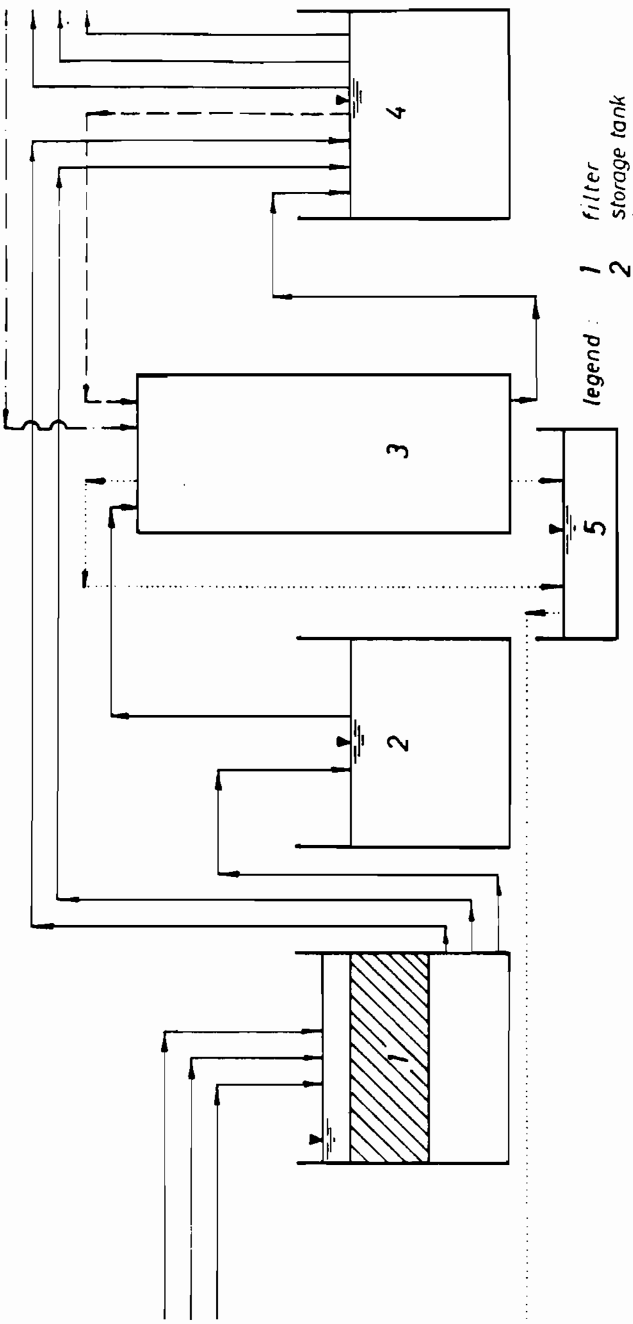

Fig. 5 Scheme of ion exchange process

Tests made with domestic artificial resins and their further developed products resulted in the clear suitability of Wofatit Y52 manufactured by VEB Chemiekombinat Bitterfeld. This artificial resin eliminates impairments of odour and taste in the outflow from the ion exchanger and ensures compliance with chemical and hygienic parameters for drinking water. The service life of this artificial resin lies between 5 and 8 years.

As to the detailed assessment bases, see Wiegleb (1980).

The advantages including safe treatment and high reliability of the plant as well as possibilities for short-term start and shut down are confronted by the following disadvantages: Use of 30-per-cent-magnesium chloride brine or NaCl as regenerant leads to a very high chloride content of the wastewater produced during regeneration, thus causing a new source of environmental pollution.

From the economic point of view, nitrate elimination alone causes additional costs of about the same hight as does the entire normal treatment process of a moderately polluted raw water. Depending on the quality of the raw water and the volume to be treated, specific prime costs of 0.08 to 0.60 Marks per cubic metre (M/m^3) are caused.

4.2. Artificial infiltration

As mentioned already at the beginning, this process is generally used in order to improve the quality of drinking water. Utilizing the purification ability of the soil has always a positive effect on water quality.

When preparing the artificial grass basin, we proceed from the fact that nitrate elimination performance results from nitrate assimilation of higher plants and microbial activity in the soil.

Investigations made to this end in the GDR are not yet completed since it is not only a matter of solving biochemical processes, but there is also a series of technical and technological questions which have to be answered. According to Scholze, Stolz, Wissel, and Wiegleb (1978) the latter complex of questions includes the following:

- at which position of the treatment process should the process stage "Infiltration grass basin" be included?
- Proposals as to the operational regime inclusive of harvesting and processing technology of the biomass in case of larger plants.

Although the costs of plant basins compared to the ion exchange process are relatively favourable according to the present state of knowledge (investment cost about 1 : 2 and specific prime cost per m^3/d approximately 1 : 2 to 1 :3), it cannot be overlooked that the space required is rather large, and at many densely populated locations this space is simply not available. The space requirement for plant basins is 5 to 7 times higher than that of sand infiltration basins.

5. Conclusions

Since the growing nitrate load of raw water is a consequence of intensified industrial and agricultural production and urbanization, assessments of benefit or damage must not be made one-sidedly with regard to only one of these aspects. If the problem of nitrate pollution is considered only from the standpoint of water management, there will always occur an increase in costs, no matter which process is used. However, an entire-economic analysis could answer the question of how far yields can be increased in agriculture in connection with higher nitrogen doses and, consequently, with a higher degree of leaching. Besides many other aspects, the fact should also be taken into consideration that (according to Schilling, 1980) production, handling and placing of 1 kg of fertilizer nitrogen require an energy expenditure of 75,000 kJ. Moreover, there is the energy expenditure required additionally in the field of water management for every kilogramme of leached fertilizer nitrogen by way of operating additional process stages for eliminating nitrate from drinking water.

Furthermore, all institutions dealing with those problems are surely fully aware of the fact that it would be wrong to take decisions only because of present-day economic considerations. From the standpoint of water management it should be rather considered that fresh water resources as a whole are limited and future generations will also have to be supplied sufficiently with water and they have a right to a healthy environment, and the nitrate load of the ground water can either not be eliminated at all or only over generations. Proceeding from this basic idea, measures of preventing or reducing pollution should be given priority because of both ecologic <u>and</u> economic reasons.

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Bl. 01 Allgemeine Grundsätze

Bl. 02 Wasserschutzgebiete für Grundwasser

Bl. 03 Wasserschutzgebiete für Oberflächenwasser

(Sh. 01 General principles

Sh. 02 Water protection areas for ground water

Sh. 03 Water protection areas for surface water)

WATER QUALITY MONITORING: A SYSTEM'S PERSPECTIVE

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Water quality monitoring is an effort to obtain information about the physical, chemical and biological characteristics of water via statistical sampling. The usefulness of the information is highly dependent upon a monitoring program, or system, being properly designed and operated. A monitoring system is defined as a sequence of operations: (1) sample collection; (2) laboratory analysis; (3) data handling; (4) data analysis; and (5) information utilization. The design of each component is discussed in the context of how it affects the overall usefulness of the resulting information.

The data collection/utilization balance, a scientific understanding of water quality, data utilization strategies and network design documentation are specific aspects of monitoring where improvements can be made, thus greatly enhancing our ability to obtain useful information on water quality.

With the system's perspective of monitoring and a discussion of specific areas for improvement, it is hoped that the important role water quality monitoring should play in enhancing management of water quality can be more readily visualized and that monitoring, with its better defined role, will ultimately provide a basis for a much more scientific understanding of water quality than that currently available.

Water quality monitoring is an effort to obtain information about the physical, chemical and biological characteristics of a water body (stream, lake, aquifer, etc.). Monitoring involves taking samples from a water body, analyzing the samples for the characteristics sought, and then using the sample results to make inferences about the characteristics of the entire water body. Since the entire water body is not analyzed, any attempt to extend sample results to the whole has uncertainty associated with it. This uncertainty, unless properly handled, can render a monitoring program useless or of little value.

Statistics is the science that permits the uncertainty of inductive inferences to be evaluated. Thus, water quality monitoring must be considered as statistical sampling if the uncertainties associated with the final conclusions are to be properly evaluated, reported, and understood.

The types of characteristics measured, and the inferences to be made, depend upon the purpose for monitoring in the first place. An intensive water quality survey may be designed to establish cause and effects of nonpoint pollution. A surveillance network or effluent monitoring program may be designed to check compliance with water quality or effluent standards. A hydrologic data collection program may be designed to simply establish a fundamental understanding of water quality in the hydrologic cycle. A state or national water quality monitoring network may be designed to determine trends in water quality. A local monitoring program may be established to inspect operation of on-site wastewater treatment systems and determine nitrate and coliform contributions to ground water quality.

Regardless of the specific purposes for a particular monitoring program, it is important, in all monitoring efforts, to recognize that water quality is a characteristic of water, and, as such, is greatly influenced by the hydrologic cycle. It is the uncertainty of the hydrologic cycle that introduces a large amount of uncertainty into inferences made about water quality from a monitoring program. About this point, Dumitrescu and Nemec (1974) note:

> "Without necessarily entering the technological aspects of water pollution abatement, hydrology has a major role to play in the studies of diffusion of pollutants in water media and of self-purification capacity of water systems, and in establishing the scientific basis for monitoring the quality of the water environment."

Of course, man's activities compound the uncertainty of water quality inferences from a monitoring program. In fact, a major goal of regulatory water quality monitoring is the separation of these effects—nature's and man's.

Given the uncertainties (whether man caused or hydrologically related), the chemical and biological interactions, and the need to make inferences, the design of water quality monitoring programs is very complex and difficult. These complexities and complications (discussed in more detail by Lettenmaier 1979) often overwhelm monitoring program designers and rasult in many, if not most, water quality monitoring programs being established on an ad hoc basis. However, if a water quality monitoring program is to supply scientifically sound information on the physical, chemical and biological characteristics of water in the hydrologic cycle, it must be established and operated within a well defined framework that: (1) helps place complexity in perspective and (2) accounts for the statistics, hydrology, chemistry, biology, etc.

The purpose of this paper is to present a framework, or system's view, of water quality monitoring which can be used to help organize a monitoring program and, consequently, assist program managers to better deal with the complexities of monitoring. A system's perspective of water quality monitoring will assist designers of monitoring programs to see a connection between the purpose of monitoring and all the activities associated with actually operating a monitoring program. A system's perspective also provides a framework within which: (1) the available literature on specific aspects of monitoring can be related to the whole and (2) a systematic approach to design of a total program may be formulated.

MONITORING SYSTEM

There are many ways to view a water quality monitoring program in its totality. Rodda (1974) used a schematic chart to illustrate the connection between the major elements of a hydrological data acquisition, transmission and processing systems. Rodda (1974) uses the flow of data and information to organize the chart. A similar approach will be used here to develop a system's view of water quality monitoring.

The activities involved in the operation of a water quality monitoring program will be categorized and ordered to follow the flow of data and information. Thus, the monitoring system, as herein defined begins with the collection of samples and ends with the resulting information being <u>used</u> to enhance our understanding of water quality. This understanding is then used for whatever purpose the monitoring program was initially established.

The activities involved with water quality monitoring are first divided into two major sections: (1) data acquisition, and (2) data utilization. The first half of a monitoring program concentrates on collecting or acquiring data while the second half concentrates on converting data into information which can be used.

Data Acquisition

Data acquisition consists of: (1) sample collection, and (2) laboratory analysis. Sample collection involves taking field measurements, collecting samples, processing the samples, and transporting them to the laboratory. Laboratory analysis involves analysis of the samples for the variables under study, handling and organizing the flow of samples through the laboratory, quality control and recording the data.

Data Utilization

Data utilization consists of: (1) data handling (storage and retrieval), (2) data analysis and (3) information utilization. Data handling involves taking data from the laboratory and entering it into a system (e.g., computer) for easy access. This activity also can involve the entering of data acquired from outside the monitoring program. Data handling also is the point in the system where data is screened and verified. Following verification, the raw data is ready for reporting and dissemination. Raw data reporting and dissemination may be extremely useful to others interested in water quality in the area being monitored. This activity also enhances the value (usefulness) of the data.

Data analysis is closely tied to data handling as many data analyses require the rapid manipulation of large amounts of data. Data analysis is the activity that converts the data into information. The exact analyses used will depend upon the type of knowledge sought on water quality and the confidence desired in the information (how the uncertainty of the inferences is to be handled). The knowledge gained at this point may be enhanced, beyond statistical analysis alone, by the use of water quality models and water quality indices. Again this depends upon the purposes of monitoring.

Utilization of the information, or understanding of water quality, constitutes the final activity of the monitoring system as herein defined. It is not enough to analyze the data and report the results. The results, be they from a model, statistics or an index, must be carefully interpreted and presented in a manner that transmits the understanding of water quality to laymen (i.e., the public, policy makers, regulators, etc.).

Since the ultimate success of a monitoring program rests on this last step, information utilization must be carefully evaluated on a periodic basis. This evaluation process is a part of the monitoring program.

A flow chart illustrating the monitoring system, as described above, is presented in Figure 1.

MONITORING SYSTEM DESIGN

The above developed concept of a monitoring system will now be used to discuss the design, or evaluation, of a water quality monitoring program. "Evaluation" is mentioned here since it is recognized that many fixed-station monitoring programs were established with a minimum amount of design and are now in need of evaluation and redesign.

The design of a monitoring system, or program, can be divided into two components: (1) network design, and (2) operational design. Network design is a term used to refer to the general logic and reasoning behind why a monitoring program is established and how it is to evaluate the inferences made. More specifically, network design includes: (1) identifying monitoring objectives; (2) translating general objectives into design criteria; (3) selecting and using design methodologies which convert criteria into station locations, sampling frequencies and variables to measure; and (4) specifying the data analysis procedures appropriate for the system.

Operational design refers to the mechanics of actually performing the monitoring on a day-to-day basis. Operational design normally follows network design. Operational design includes: (1) establishing

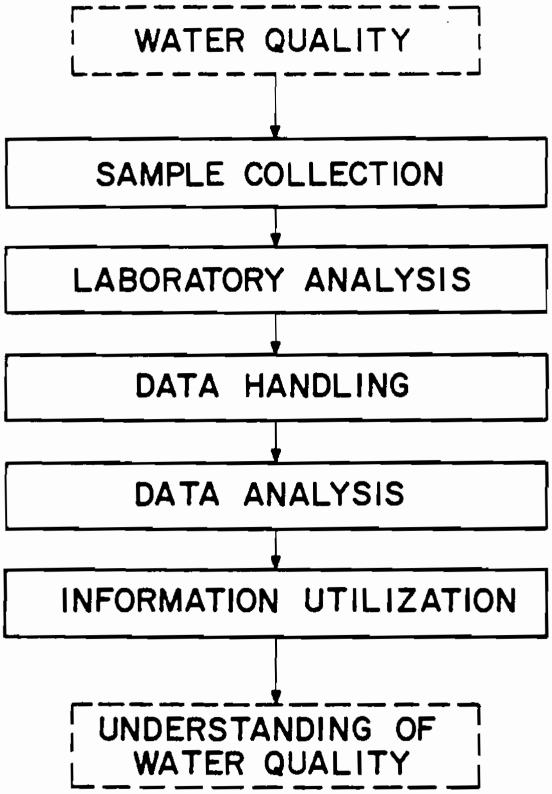

FIGURE 1. THE MONITORING SYSTEM BASED ON THE ACTIVITIES INVOLVED IN THE FLOW OF INFORMATION THROUGH A MONITORING PROGRAM.

routes to collect samples; (2) defining sampling procedures and equipment; (3) selecting laboratory operation and analysis procedures; (4) establishing a quality control program; (5) selecting a means to store and retrieve data; (6) developing data analysis software; etc. Operation design determines the number and type of personnel, facilities, and equipment needed to achieve the goals established in the network design. If the resources available for monitoring are exceeded in the operational design, it may be necessary to modify some aspect of the network design.

Network Design

Network design for water quality monitoring has received considerable attention in recent years as efforts have been made to improve the ability of monitoring programs to enhance understanding of the processes controlling the quality of water. Realizing that all phases of the hydrologic cycle are important, water quality monitoring is performed on surface waters, ground water, atmospheric moisture and, in a regulatory mode, on point and nonpoint effluents. The monitoring is either by fixed-station routine sampling programs or by special surveys of varying duration and intensity. In all cases (e.g., special ground water quality survey or routine fixed-station surface water quality monitoring program), there have been network design procedures proposed.

Montgomery and Hart (1974) and Sherwani and Moreau (1975) each discuss a number of aspects of designing a network to obtain information on trends in surface water quality, mainly flowing streams. Chamberlain et al. (1974) and more recently Heidtke and Armstrong (1979) have explored the role of water quality models in designing surface water quality monitoring programs to detect violations of stream standards. Sanders and Adrian (1978), Lettenmaier (1978), Wallin and Schaeffer (1979), Ward et al. (1979), Loftis and Ward (1979), and Dunnette (1980) are representative of a large number of studies on surface water quality monitoring dealing mainly with the sampling frequency aspects of network design. Sharp (1971) deals with location of sampling stations on rivers. Reckhow (1978) discusses water quality monitoring network design in lakes to determine nutrient budgets.

In terms of monitoring point source effluents, Grimsrud, et al. (1976), Bellanca (1976), and the U.S. Environmental Protection Agency (1979) represent efforts to establish design procedures for compliance monitoring. Eccles and Gruenberg (1978) discuss the design of a monitoring program to measure agricultural saline water discharges, a nonpoint water quality problem.

Ground water monitoring has received tremendous attention in the United States of America during the past year as its protection has been a top priority of the U.S. Environmental Protection Agency. Management of hazardous wastes has further increased this interest. Todd et al. (1976) and Everett (1980) present methodologies for designing ground water monitoring programs, but both suffer from lack of statistics. Nelson and Ward (1980) discuss ground water monitoring from a statistical perspective.

Many of the concepts of network design presented in the above studies apply to both fixed-station monitoring and special surveys.

Kittrell (1969) and UNESCO/WHO (1978) discuss additional considerations related specifically to special surveys.

To assist states in the United States in development of monitoring programs, the U.S. Environmental Protection Agency (1975, 1977) has published general guidelines. The reports, however, do not deal with all aspects of network design. The National Academy of Sciences (1977) criticized water quality monitoring in the United States saying that such monitoring lacks a strong statistical basis. The U.S. General Accounting Office is investigating the accountability of monitoring performed by the federal government.

All of the activity in network design has not resulted in any standard design procedures being developed. Sanders (1980) represents one attempt to identify and describe some fundamental concepts in network design. Network design, however, remains an art as much as a science. The specification of objectives, the conversion of objectives into criteria, and the selection of design methodologies and the means of translating data into information provides considerable room for subjective network design.

Operational Design

As the efforts to define and regulate water quality have intensified in recent years, so have the efforts to ensure that monitoring programs provide accurate and correct data and information. Training requirements for monitoring personnel have increased, more sophisticated equipment is available and quality control programs have expanded. As with network design, this has resulted in publications describing the new or newly organized, or agreed upon, procedures.

Huibregtse and Moser (1976), Shelley (1977), Diefendorf and Ausburn (1977) and Saxena and Nies (1979) are examples of publications dealing with the sampling of waters. Tarazi et al. (1970) and Schaeffer and Janardan (1978) describe differences between grab and composite sampling. The U.S. Environmental Protection Agency (1974), U.S. Geological Survey (1977) and American Public Health Association (1979) represent efforts to establish standard laboratory analysis procedures for water quality variables. Bicking et al. (1978) and Tracor Jitco, Inc. (1978) are reports which describe quality assurance programs and ways to evaluate overall laboratory performance.

In the data handling area, the Joint Committee on Water Quality Management Data (1967) is one of the earlier attempts to organize and categorize data for a management program. Kitch (1978) reports on the state of Pennsylvania's efforts to use these categories in establishing a statewide information system. Flemal et al. (1979) describe Illinois' water quality information system. Haseman et al. (1975) discusses information systems for water quality in a broad context. STORET is a national effort by the U.S. Environmental Protection Agency to handle data. NASQAN is a national effort by the USGS. The major difference is that STORET accepts data from any network while NASQAN is tied to over 500 sampling stations operated by USGS. STORET is simply a data handling system available to government agencies.

STORET also provides basic data analysis services to its users as well as handling data. Buda et al. (1976) describe an interactive system for analyzing water quality data. The U.S. Council on Environmental Quality operates an UPGRADE system for data analysis. In terms, however, of describing, in an overview manner all water quality data analysis procedures (i.e., statistics, modeling, indices, etc.), no references are available.

Likewise, there are no publications of which the author is aware, that describe, in an overview manner, the utilization of water quality information derived from monitoring programs.

Monitoring System Design Procedure

The systems perspective of monitoring provides a framework with which it is possible to develop a systematic approsch to monitoring program design. In this section, such an approach to design is proposed and described. Monitoring system design, of course, will never be quantified by a set of hard-and-fast rules or steps; therefore, the following is intended more as a general guideline than a set rules for design. In addition, the more subjective aspects of monitoring systems design (data utilization) are discussed along with the more quantifiable aspects (data collection) in order to give a broader perspective to system design.

Step 1. Determine Monitoring Objectives and Relative Importance of Each

The reasons for monitoring water quality should be determined as precisely as possible. This may require the system designer to formulate the objectives in his or her own words prior to approaching the ultimate users of the data and/or information. Such formulations provide a specific basis for discussion.

In many cases, the true monitoring objectives are extremely difficult to formulate. Moss (1980) suggests, in such cases, the use of surrogate objectives. Development of monitoring objectives, or surrogate objectives, requires the monitoring system designer to evaluate the entire management organization and, often, the philosophy underlying its operation. This projects the designer into an uncomfortable position and may result in him or her being told to leave management to the managers.

How can management be made to understand the importance of clearly defining monitoring objectives? The system's perspective illustrates clearly the interaction between data collection and data use and notes how ultimate accountability of the monitoring programs rests on the data's use. Such a view of monitoring will, hopefully, assist designers in getting access to monitoring objectives.

If the monitoring system is to meet several objectives, each should be determined and prioritized relative to the others. Conflicting objectives require the compromise of many design specifications later in the design process, and a relative weighing of objectives provides the designer with guidance in compromising. Once the specific objectives of a water quality monitoring system have been articulated, the designer must then translate these objectives into specific data requirements which can be fulfilled by the proposed design.

<u>Step 2</u>. <u>Express Objectives in Statistical Terms</u>

Translating objectives from words to statistics at the first of the design process permits the users of data and information to specify the accuracy they need in quantifiable terms while at the same time providing the designer with a more objective basis for future design calculations. If one of the monitoring activities is associated with enforcement or other statutory needs, it is very important that the design of the system and the resultant data obtained are compatible with existing and proposed legislation, both state and federal.

As noted earlier, water quality monitoring is a statistical sampling process and, thus, there will be uncertainty associated with the final results. The acknowledgment of this fact will greatly assist designers in developing meaningful and useful designs.

Step 3. Determine Budget Available for Monitoring and Amount to be Allocated for Each Objective

Realistic objectives cannot be formulated without acknowledging the limits within which one must work. Economic limits on the number of samples, number of stations, etc., will greatly influence the ability of a monitoring system to reach its statistically defined objectives. Recognizing the economic limits early will, again, permit the data users to participate in deciding where the compromises will be made. One compromise may be to trade off percent areal coverage (a large number of stations) for more intense sampling at a small number of stations.

For data users to see the statistical-versus-economic trade-offs, the designer should attempt early in the design process to quantify such trade-offs.

If the monitoring systems is multi-objective, an attempt should be made to determine the amount of the budget to be allocated to each objective--the ultimate measure of the importance a data or information user places on the objective.

Step 4. Define the Characteristics of the Area in Which the Monitoring is to Take Place

The geographical and hydrological characteristics of an area to be monitored need to be well defined prior to initiation of design calculations. Natural salt springs, precipitation, industrial concentrations, population centers, flow patterns, irrigation schedules, etc., will all influence the design process. The extent of influence will depend upon the objectives of the monitoring program.

Monitoring networks which must cover a number of watersheds, or portions thereof, require careful consideration of the geographical and hydrological differences and how these differences may preclude the use of one uniformly designed network for the entire jurisdiction. Instead the sub-basin objectives and budget allocations must be determined and each sub-basin then treated as its own separate network--a sub-network of the overall network.

Step 5. Determine Water Quality Variables to be Monitored

Variables measured by a water quality monitoring network are highly dependent upon the objectives, basin characteristics and economics of the network. A regulatory water quality monitoring program would perhaps be mainly interested in those variables stated in the stream standards. Likewise, an agency regulating a reservoir would, perhaps, be monitoring only those variables influenced by releases of the reservoir water. A monitoring program used to manage on-site home sewage disposal systems would be measuring nitrates and coliforms.

The design of a water quality monitoring system must center around water quality; however, water quality can be defined in terms of one variable or 1000 variables. Design of a water quality monitoring system, unless only one variable is considered, must account for the different statistical behavior of the different variables. This will require compromising of some form among the variables.

Both sampling location and sampling frequency can be developed independently of frequencies for analyzing samples in the laboratory. Both location and frequency are specified for the collection of the water sample--the analyses are made later. However, both criteria are affected by the behavior of the water quality variable being monitored. For example, sampling once a week at a single point in a river may be more than adequate for monitoring a relatively stable river temperature, but may be hardly adequate for monitoring rapidly varying coliform bacteria concentrations. Therefore, before a water quality monitoring program can be designed in a systematic fashion, the variables to be monitored should be specified so that their natural and/or man-made variation in time and space can be considered when designing the monitoring network. In addition to considering the water quality variables of interest, their respective units should be delineated as well. The network design varies tremendously if a daily mean (flow weighted) concentration is needed rather than an instantaneous grab sample concentration--the former being a result of several samples with flow measurements equally spaced during a 24-hour period, while the latter is only a single sample (generally in the daytime between 8:00 a.m.-4:30 p.m.).

In reality, the specification of the water quality variables to be monitored prior to initiating network design would be ideal. In practice, however, an already designed network is often given and then one must know or determine what water quality variables can be adequately monitored with the existing network.

Step 6. Determine Sampling Station Locations

The location of a sampling station in a water quality monitoring network is probably the most critical aspect of network design, but it is all too often never properly addressed. Expediency and cost compromises lead in many cases to sampling from bridges or near existing river gaging stations. Whether the single grab sample from the bridge or the gaging station is truly representative of the water mass being sampled is not known, but it generally is assumed to be representative by both the collectors and users of the water quality data. Using river stage for estimating discharge, measurement anywhere in the lateral transect would indicate the exact river discharge. However, this does not necessarily follow when measuring water quality variable concentrations. In fact research indicates the opposite, that a single sample will rarely be indicative of the average water quality in a river's cross section.

Sampling locations for a permanent water quality network can be classified into two levels of design: macrolocation and microlocation, the former is a function of the specific objectives of the network while the latter is independent of the objectives but a function of the representativeness of the water sample to be collected. The macrolocation specifies the river reach to be sampled while the microlocation specifies the point in the reach to be sampled.

The macrolocation within a river basin is usually determined by political boundaries (state lines), areas of major pollution loads, population centers, etc. Macrolocation can be specified, as well, according to percent areal coverage using basin centroids (Sharp 1971). This methodology locates sampling points in a systematic fashion, maximizing information for the entire basin with a few strategically located stations.

Once the macrolocations within a river basin are established, the microlocation is then determined. The microlocation point is the location of a zone in the river reach where complete mixing exists and only one sample is required from the lateral transect in order to obtain a representative (in space) sample. The zone of complete mixing can be estimated using various methodologies (Sanders et al. 1977).

If there is not a completely mixed zone in the river reach to be sampled, there are three alternatives: (1) sample anyway at a single point and assume it is representative (this is the general procedure being applied today); (2) do not sample the river reach at all, because the data which would be obtained does not represent the existing river quality, but only the quality of the sample volume collected; (3) sample at several points in the lateral transect collecting a composite mean, which would indeed be representative of the water quality in the river at that point in time.

If the sample was not representative of the water mass, the frequency of sampling as well as the mode of data analysis, interpretation and presentation and the realistic use of the data for objective decision-making would be inconsequential. In spite of this fact, criteria to establish station locations for representative sampling has received relatively little attention from agencies responsible for water quality monitoring.

Step 7. Determine Sampling Frequency

Once sampling stations have been located so that samples collected are representative in space, sampling frequency should be specified so that the samples are representative in time.

Sampling frequency at each sampling station within a monitoring program is a very important consideration in the design of a water quality monitoring network. A large portion of the costs of operating a monitoring program is directly related to the frequency of sampling. The reliability and utility of water quality data derived from a monitoring network is, likewise, related to the frequency of sampling.

Significant as sampling frequency is to developing an understanding of the processes controlling water quality, very little quantitative criteria designating appropriate sampling frequencies have been applied to the design of water quality monitoring networks. In many cases, professional judgment and cost constraints provide the basis for sampling frequencies. All too often, frequencies are the same at each station and based upon routine capabilities, once-a-month, once-a-week, etc. Given the lack of statistical training of many network designers, these may be the only practical means to implement a sampling program.

There do, however, exist many quantitative, statistically meaningful procedures to specify sampling frequencies at each station. The methods include specifying frequencies as functions of the cyclic variations of the water quality variable (Nyquist frequency), the drainage basin area and the ratio of maximum to minimum flow, the confidence interval of the annual mean (Ward et al. 1976, Sanders 1980), the number of data per year for testing hypotheses (Sanders and Ward 1978), and the power of a test measuring water quality intervention.

All of the aforementioned procedures can be applied to the design of a water quality monitoring network with each requiring a different level of statistical sophistication in terms of data requirements, necessary assumptions and limitations, and statistical expertise of the users. Sampling frequency calculations must deal with the fact that water quality variables may not be independent, but highly dependent and not identically distributed, but seasonally variable. Unfortunately, other than mean daily discharge, temperature and conductivity, data bases of water quality variables of sufficient number, reliability and length are not usually available for application of design procedures that account for these factors. Thus, more simple approaches are used with many assumptions being made.

Once a consistent sampling frequency criterion is selected, it can be utilized to objectively distribute sampling frequencies within a water quality monitoring network. For example, the expected half-width of the confidence interval about the annual mean approach can be applied basin-wide (for specifying sampling frequencies) in a consistent fashion by specifying equality of these expected half-widths at each sampling station. Thus, stations where water quality varies tremendously will be sampled more frequently than stations where the water quality varies little.

The expected half-widths of the confidence interval of the annual mean is not the only statistic that can be used to specify sampling frequencies; the expected half-width divided by the annual mean is a measure of relative error and may be more appropriate when assigning sampling frequencies in a network where water quality varies tremendously from river to river.

When developing sampling frequencies, one must keep in mind two very important cycles which can have immense impact on water quality concentrations--the diurnal cycle and the weekly cycle. The effect of the diurnal cycle (which is a function of the rotation of the earth) can be eliminated by sampling in equal time intervals for a 24-hour period and the effect of the weekly cycle (which is a function of man's activity) can be eliminated by specifying that sampling intervals for a network cannot be multiples of seven--occasional sampling on weekends would be necessary.

Perhaps, the major impact of network design (in terms of variables to be monitored, sampling location, sampling frequency, and the operational monitoring functions) is in the area of data analysis and, consequently, ultimate value of the monitoring information. Any sampling program that is to generate conclusive results from observing a stochastic time series (water quality concentrations) must be well planned and statistically designed. Statistically designed implies that the sampling is planned (in proper locations and numbers) so that the statistical analysis techniques chosen will be able to yield quantitative information. Thus, the data analysis techniques (level and type of statistics) to be used must be defined in order to know how to compute proper sampling frequencies, locations, etc.

Step 8. Compromise Previous Objective Design Results with Subjective Considerations

Station locations, sampling points, and sampling frequencies objectively computed in previous steps must now be evaluated in light of the access to the water, costs of acquiring the sample and rounding off frequencies to match practical sampling routes. Such compromising must be minimized, but to say that it should not be done is not realistic. Again trade-offs must be considered--having a crew sample on weekends must be balanced against the problem of nonrepresentative data, sampling from a bridge balanced against cost of obtaining access elsewhere. Such compromises should be recorded and attached to any reporting of data from the network.

Step 9. Develop Operating Plans and Procedures to Implement the Network Design

The actual day-to-day operation of a network includes many functions of which network design is a predecessor. These operational and informational functions, to be performed smoothly over time and changing personnel, must be well defined and documented. This requires the network designer to develop sample collection routes and schedules, sample collection procedures and forms, laboratory analysis recording procedures and forms, data handling procedures and forms, etc. The network design should be documented well enough so that the monitoring operation can easily function, in a consistent manner, regardless of the personnel involved since they will surely change over time.

Step 10. Develop Data and Information Reporting Formats and Procedures

The report formats and distribution procedures should be closely identified with the system's objectives, and, as a result, should be defined in consultation with the users of the results from the monitoring program.

Lack of reporting formats and procedures is a common problem with many monitoring systems today and is a sure sign that data collection may have become an end unto itself. Regular communication between gatherer and user ensures that a monitoring system's results are properly placed and meet the information needs for which the system was established.

Reporting formats will vary greatly, depending upon the monitoring system's purpose, the primary users and the budget available. For the same system there may be monthly reports of a one- or two-page nature, annual reports encompassing considerable detail and, therefore, length, and special reports needed to meet special information requests (e.g., 305 b reports required by the U.S. EPA of state regulatory monitoring networks).

Proper planning (monitoring system design) of reporting formats and distribution procedures can greatly reduce the staff time needed to generate reports. Computer plots of data, computer printouts in a reporting format, standard reporting forms, etc., are examples of ways to expedite reporting of data and information. Where timing of information distribution is critical, such procedures are almost always required if reporting is not to disrupt normal monitoring system operations.

Step 11. Develop Feedback Mechanisms to Fine Tune the Design

As part of the data and information reporting, specifically, and as part of the entire monitoring system design, in general, a means of receiving and utilizing feedback, both solicited and unsolicited, must be provided in order to "fine tune" the design. No water quality monitoring system design can be assumed fixed. There are always reasons for altering a design--changing objectives, changing technology, new data users, etc. There are some monitoring systems which may have components that change little over time (such as a network measuring long-term trends), but, in general, procedures to accommodate change must be incorporated into the design.

Feedback can be incorporated into the reporting of information by spelling out in standard form the means by which a reader can report any comments he or she may have regarding the data and/or information (and, thus, the design). Such unsolicited comment or feedback must be recorded, reported, evaluated and answered, if not to the person making the comment, to those responsible for the monitoring system's existence.

Solicited feedback can be obtained via planned questionnaires regarding the design. Soliciting feedback should be a regularly scheduled activity associated with the design. All aspects of the monitoring system should be included in such an evaluation.

Step 12. Prepare a Monitoring System Design Report

Monitoring system designers are generally system operators and/or managers, consequently, their design evaluations, procedures, etc., are performed and implemented without a report being prepared. Lack of design documentation creates many problems associated with water quality monitoring today (National Academy of Science 1977).

Whenever a designer is designing a new monitoring system or evaluating and modifying an existing system, he should carefully describe the design process and findings in a Monitoring System Design Report. Such a report would contain the results of the previously described 11 steps in the design process. If the design were contracted out, the monitoring personnel would expect such a report since the design has to be communicated between the designers and operators. Since water quality monitoring personnel are generally quite mobile, design and implementation done in-house, without documentation, often leaves those who follow with very little guidance as to the system's design. Documentation of designs is very much needed and should be done any time a new system is established or an existing system is evaluated.

IMPROVING MONITORING SYSTEMS

As has been alluded to several times earlier, there are a number of aspects of water quality monitoring which could be improved. The system's view of, and design framework for, water quality monitoring tend to focus attention on these problems in a manner not achievable when only a part of the system is examined. The following discussions describe areas where improvements could be made and the entire system would function more effectively and efficiently.

Monitoring programs are normally a part of an attempt to manage water quality. Such efforts to manage are instigated by government's attempts to regulate the externalities associated with water quality. As society, and government, identifies a new problem with water quality, a special program or agency is created to deal with the new problem. The monitoring programs designed to support this new management effort are narrowly focused around the problem. The larger picture of water quality is not the mission of the agency so it is not studied. This is especially true in the United States.

As a result of the above situation, the current understanding of water quality is problem oriented--rarely is the total picture of water quality processes even sought. The data is collected for a purpose, analyzed for that purpose, and stored. No one is responsible for tying all bits and pieces together into an overall picture of water quality. The science of water quality hydrology is basically nonexistent.

Managers of specific water quality control functions (planning, wastewater treatment plants, enforcement, research, etc.) optimize their own activities. They have few, if any, goals related to the overall quality of water. The "brush fire" aspect of water quality management often prevents regulatory agencies from assuming this role. In the United States, there are no other agencies charged with developing a basic understanding of water quality.

Regardless of the organizations doing water quality monitoring, the monitoring staff has perhaps the best opportunity to view water quality from a broader perspective. This view, however, often gets focused too closely on collecting data.

Collection/Utilization Balance

Within a monitoring program, the imbalance between data collection and data utilization is a serious problem. Data collection, over the years, has received the most attention in the literature. As noted previously, there are many articles describing how to collect a sample, how to analyze the sample, and how to process the data in a computer. There are precious few reports describing the range of procedures available to analyze data to develop a better understanding of the basic processes. Even fewer are available to assist the monitoring manager in preparing reports describing and presenting the information in a useful and readily communicated manner.

Water quality monitoring is expensive. Collecting data is expensive. Analyzing and using data are expensive. The utilization of data cannot occur unless it is first collected. Data can be collected and not used. If budgets are limited, data collection seems to get the available resources with utilization being cut--its productivity is not easily measured. Accountability in collection is much easier--so many miles were driven, so many samples processed, etc. Historically, little of the accountability of a monitoring program has been associated with the measurement of water quality. As has been said many times, data collection becomes an end in itself.

To be effective at describing water quality, a monitoring program must be carefully balanced between collection and utilization. As carefully as collection procedures are designed and operated, so should data utilization procedures. Resources for the total monitoring program should, likewise, reflect the balance. As people's functions are identified as sample collectors, laboratory technicians, so should there be data analyzers and technical journalists to get the information from the data and publicized.

Deriving information from water quality data is difficult at best. The science of water quality hydrology is not well defined and this makes for much of the difficulty in data utilization.

Scientific Understanding

Hydrology as a science has made tremendous strides over the past 15 to 20 years. The statistics and modeling applied to water quantity measurement, prediction and information utilization in all aspects of society (highway bridge design, flood plain planning, water supply forecasting, etc.) has evolved as the science has developed. There has been no comparable development in water quality hydrology. The statistics generally applicable to water quality are not known, nor are they hypothesized. Water quality models are not well developed for more than a few water quality variables.

The extra dimensions, numerous variables defining water quality and the interactions of the variables, make the extensions of many basic hydrological concepts to water quality very difficult. This means that development of a science of water quality hydrology will be more complicated and require considerably more effort. The fact that the need for such a science is just being comprehended, much less developed, is one of the main reasons water quality data utilization is so poorly defined.

A framework (e.g., standard, accepted approaches) for utilizing data in water quality needs to be proposed, debated, studied and ultimately agreed upon. For example, the Water Resources Council (1977) is the latest set of guidelines for determining flood flow frequency. Standard methods are suggested. They may not be agreeable to everyone in the field but they do meet a definite need in the application of hydrology to a practical problem. As the science develops further, the guidelines will undoubtedly change.

Work similar to that above has begun in the air quality area. Pollack (1975), Ott and Mage (1976) and Ott et al. (1979) are examples of studies of the applicability of statistical and physical models to air quality data. Kornreich (1974) is a proceedings of a conference dealing with statistical aspects of air quality data. Such meetings are desperately needed in the water quality field.

The analysis of water quality data has not been completely ignored. Hem (1970) is becoming a classic in the field. Lane (1975) and Steele et al. (1974) are examples of the type of examination of water quality data of which much more is needed.

There is a need to propose standard guidelines for analyzing water quality for various purposes, discuss the guidelines, and ultimately develop a consensus on which procedures represent the state of the art. Such an effort would not require the collection of more data at present--simply a thorough analysis of that currently available. There is a large amount of data that has not been analyzed for more than the very narrow purpose for which it was collected. This data, when examined to explain water quality in a larger context, could reveal a large amount of knowledge about the basic processes controlling water quality. An excellent example of this is the data collected by state regulatory water quality management agencies in the United States. This data is normally checked against a standard and stored. Very little, if any, effort is made to thoroughly examine and analyze the data. As usual, the problem stems from a lack of analysis resources--and unbalanced monitoring program. The regulatory agencies do not see a value to them in such an analysis. Who is to take the lead and examine the data? Who will develop the science of water quality hydrology?

Data Utilization Strategies

With its multiple dimensions, it is difficult to reduce water quality data to simple, easily understood information on water quality. As the data is reduced to information, many specific details about the water quality get masked and, consequently, many assumptions have to be made and explained. A classic example of this is the attempt to develop a water quality index in the United States. Many, many water quality indices have been proposed and debated; however, no consensus has been reached. Perhaps the development has not proceeded far enough to permit a consensus. In any case, the communication of water quality information to the public is greatly restricted by this lack of a consensus. Air quality indices have greatly enhanced to communication of air quality monitoring results. Ott (1978) is an excellent summary of water quality index development.

Within water quality management organizations, a well defined flow of data and information through the system would enchance the use of water quality data. Such a flow chart of data and information would specify who was to analyze the data, who was to report the information to whom, what decisions the information was to influence, and who was to make them. Ward et al. (1976) describe water quality data utilization strategies within a management organization.

Sanders and Ward (1978) and Schaeffer et al. (1980) discuss several problems created within a regulatory agency when water quality data do not have a well defined use. McAniff and Willis (1976) and Adrian et al. (1980) discuss the value of water quality data. Since value is related to use, these two studies have implications to the use of water quality data. Shnider and Shapiro (1976) present procedures for evaluating water quality monitoring programs; however, the study reflects the available literature, and, thus, is quite weak in the data utilization area.

For all monitoring systems, the strategies for use of data should be thoroughly designed before the monitoring is initiated.

Design Documentation

As noted in the 12th step of the step-by-step guidelines, and repeated here for emphasis, there is a need to document the design of all monitoring systems. Such documentation explains the underlying rationale for the data collection and use and, consequently, greatly enhances the value of the data. If a user knows the design of a monitoring system (network and operations), the data's accuracy and precision can more readily be evaluated.

Whenever a water quality monitoring system is established, there should be a report prepared detailing the system's design!

SUMMARY AND CONCLUSION

A system's view of water quality monitoring has been formulated, the design of such systems has been discussed from a network and operational perspective, step-by-step system design guidelines have been proposed, and several factors which, if changed, could improve monitoring system design have been discussed. Water quality monitoring was discussed in a general context; however, the bias of the author's experience in regulatory monitoring has probably skewed some of the presentations.

The purpose of the paper was to broaden the view of people involved in water quality monitoring. With the system's perspective, it is hoped that the important role water quality monitoring should play in enhancing management of water quality can be more readily visualized and that monitoring, from such a perspective, will ultimately provide the basis for a much more scientific understanding of water quality hydrology than that currently available.

The paper should not end without noting that hydrological monitoring systems, in general, have been undergoing analysis in recent years. Rodda (1974) discusses the role of data collection systems in the future development of hydrology. Dawdy et al. (1972) discusses the application of system analysis to network design. Moss et al. (1978), Moss (1980) and Steele et al. (1980) present overviews of hydrological network design and discuss future needs. Langbein (1979) analyzes a conference on the design of hydrological data networks.

With its extra dimensions, it is imperative that professionals in water quality monitoring expend considerably more effort on monitoring system design and operation than has been the case in the past. Only in this way, will water quality be understood.

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NITRATE POLLUTION OF GROUNDWATER RESOURCES--MECHANISMS AND MODELLING

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In the United Kingdom field studies of the unsaturated zones of 30 sites on the Chalk and 12 on the Triassic Sandstone aquifers have shown a correlation between arable farming regimes and high nitrate concentrations in interstitial pore water. A slow downward movement of nitrate has been proved at three sites. A mathematical model of solute transport through the unsaturated zone has successfully simulated the observed nitrate profiles at a number of the investigated sites. This vertical transport model has been incorporated in a catchment model which has been used to predict changes in groundwater nitrates, in some areas, to levels above 11.3 mg N/ℓ.

1. INTRODUCTION

About a third of the public water supplies in the United Kingdom are obtained from groundwater, the greater part of which are derived from two principal aquifers, the Chalk and the Triassic sandstones (Fig. 1). Groundwater, where available, is a cheap source of supply and generally of good quality. However, an increase in the nitrate concentration of some groundwaters in the United Kingdom has been reported in recent years (Foster and Crease, 1974; Severn-Trent Water Authority, 1976 a,b).

Over 100 public supply boreholes now produce groundwater with nitrate concentrations intermittently or continuously above 11.3 mg N/l. High nitrate concentrations in water supplies are of concern because of potential health risks (Comley 1945).

In 1974 the Water Research Centre initiated a programme of field investigations and laboratory studies with the objectives of:

i) determining the extent of nitrate contamination of the unsaturated zones of the principal aquifers in the United Kingdom.

ii) evaluating the mechanisms and rates of movement of potential pollutants, derived from the land surface, through the unsaturated zone to the water table, and

iii) estimating future trends in groundwater nitrate concentrations on both the local and regional scales.

Work on the Chalk and Triassic sandstone aquifers is essentially complete and attention is now being directed to the unconsolidated Greensand aquifer.

This paper presents the principal results of the field investigations and describes the models which have been developed to help interpret those results and provide predictions of future trends.

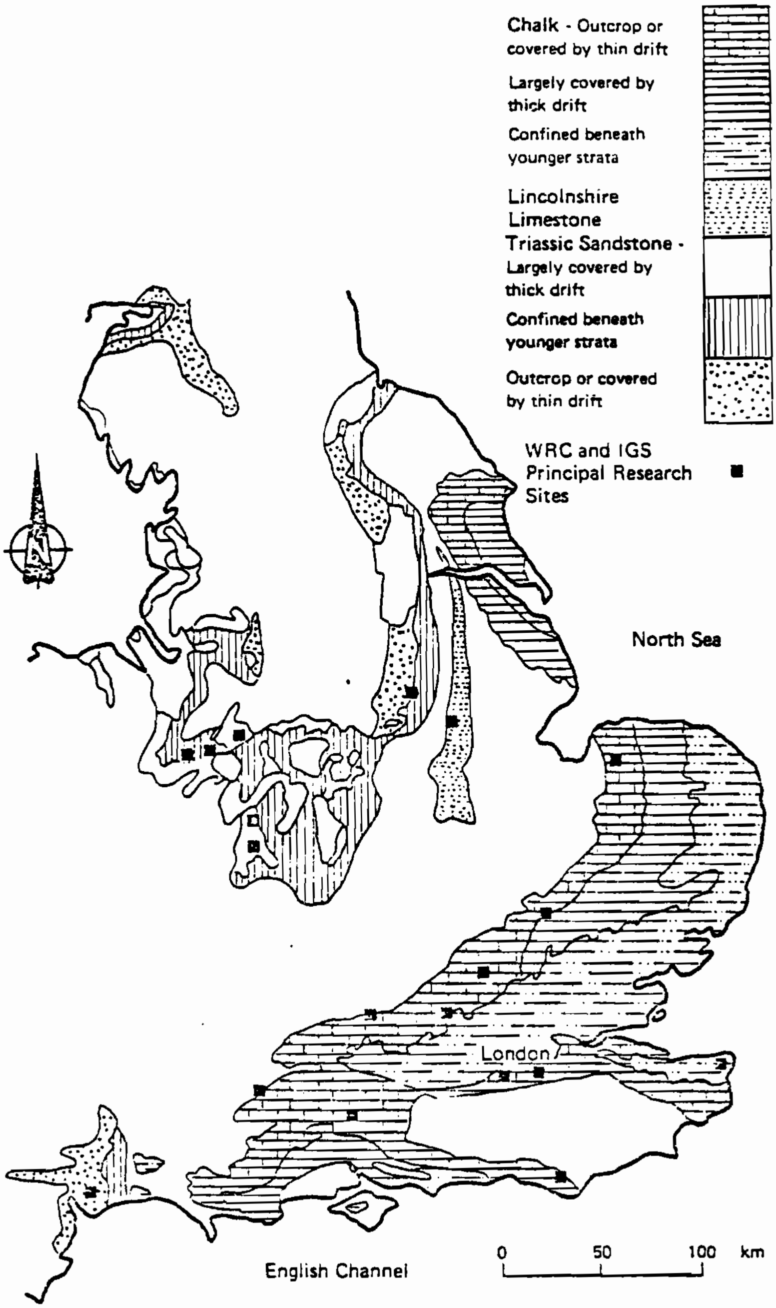


Fig. 1. Location map showing distribution of Chalk, Triassic Sandstone and Lincolnshire Limestone Aquifers and Sites of Research Work

2. SOURCES OF NITROGEN

The inputs of nitrogen to aquifers in the UK may be by direct discharges from wastes or sewage effluent or from leaching through agricultural soils. Locally the direct discharge of waste materials may give high concentrations of nitrate and ammonium in groundwater but regionally the effect is very small.

Nitrogen losses from the soil/plant system may be divided into gaseous losses, removal by the crop and leaching. The soils developed on the outcrop Chalk are characteristically shallow, well drained rendizas and associated brown calcareous earths, while those on the Triassic sandstones are well drained brown earths and podsols. Under arable regimes such soils are well aerated and losses by denitrification appear to be small whilst mineralisation of organic nitrogen is encouraged.

The rate of removal by crops is variable (Johnson, 1976) but a mean value has been estimated from published data. Fig. 2 shows for a variety of crops the relationships between N applied and N not accounted for in the crop, from which it is apparent that about 50% of the applied fertiliser nitrogen is lost to the crop. Under normal climatic conditions in the UK it is probable that a high proportion of the unaccounted for fertiliser nitrogen is assimilated by weeds and microflora during the growing season. The mineralisation of soil organic nitrogen following the ploughing of established grassland (Reinhorn and Avnimelech, 1974; Meints, Kurtz, Melsted and Pack, 1977) has been proposed as an important source of nitrates for leaching in the United Kingdom. It is noteworthy that the marked increase in arable acreages in Southern England from 1939 to 1946 was concentrated principally on the thin upland soils of the Chalk recharge areas, which had previously supported grass for sheep and cattle grazing. The quantity of nitrogen available for mineralisation may be several thousand kg per hectare (Reinhorn and Avnimelech, 1974). Measurements made at a Water Research Centre experimental plot on a 60 cm deep Chalk profile in Sussex, at which ploughing of virgin grassland first occurred in April 1978, have indicated that the soluble nitrate content of a fallow soil increased by about 200 kg N/ha during the period April to November when compared with a control plot under grass.

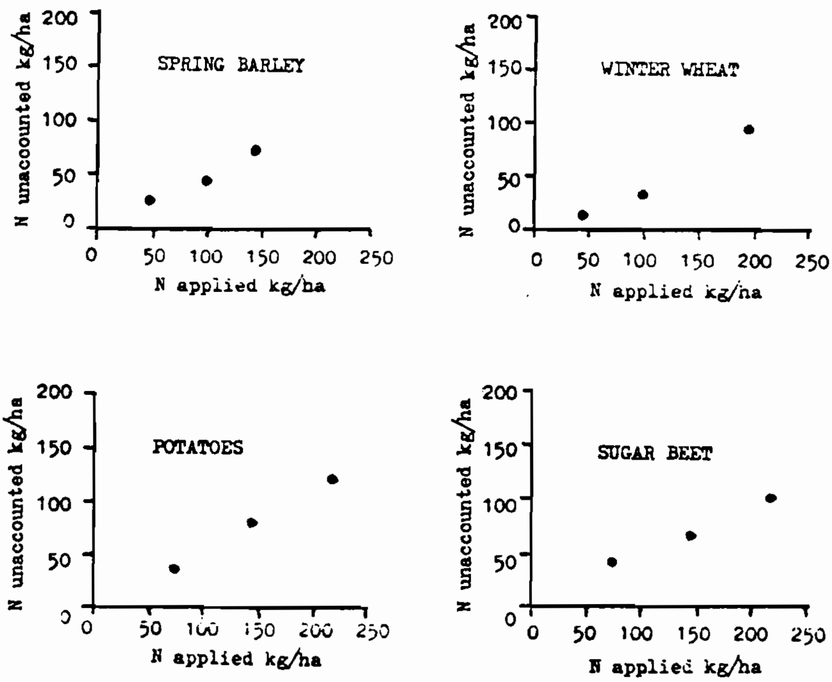


Fig. 2. Loss of fertiliser N to crop (Data from Johnson, 1976)

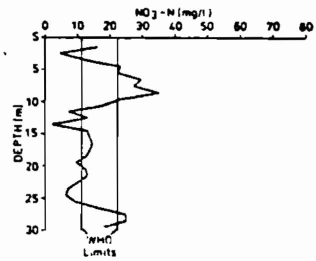


Fig. 3. Nitrate profile - Chalk arable

3. DIRECT MEASUREMENT OF NITROGEN IN GROUNDWATER SYSTEMS

3.1. DRILLING PROGRAMME

The Water Research Centre, the Institute of Geological Sciences and, latterly, the Water Authorities, have over the past few years been involved in an extensive borehole drilling programme. This has involved construction of more than 100 boreholes some to depths of up to 200 m at about 50 selected sites. Fig. 1 shows that most of the sites are on the Chalk aquifer, 12 are on the Triassic sandstones and one on the Lincolnshire Limestone. The sites were selected so as to cover a range of differing land use situations, meteorological conditions and hydrogeological environments.

Most of the boreholes were continuously cored and special drilling and sampling methods were employed to avoid contamination. Water for chemical analysis was extracted by high speed centrifugation and were analysed using standard AutoAnalyser techniques. Measurement of the tritium content of the waters have been carried out by the Harwell Laboratory using the method described by Otlet (1968).

3.2. RESULTS OF FIELD INVESTIGATIONS

Measurements in the Chalk and Triassic sandstone aquifers have revealed a correlation between high nitrate concentrations (often >20 mg N/l) in the interstitial waters of their unsaturated zones and arable farming regimes, whilst low concentrations (often <5 mg N/l) are characteristic of interstitial water beneath unfertilised permanent grassland and woodland. The vertical profiles of nitrate concentrations in the unsaturated zone beneath arable Chalk land are often sinuous and it is postulated that these variations with depth reflect changes in the rate of leaching from soils under different agricultural regimes. In addition to nitrate measurements the distribution of ammonium and nitrite were made at many sites. Determinations of the total elemental carbon, hydrogen and nitrogen in insoluble Chalk residues has also been undertaken, but only at a few sites.

3.3. UNSATURATED ZONE

3.3.1. Fertilised Arable

Uniform or relatively smoothly varying nitrate profiles were found to be characteristic of sites on the Chalk under continuous arable regimes with consistent fertiliser application rates (Young, Oakes and Wilkinson, 1979).

Sinusoidal variations of nitrate concentration with depth have been found to be well developed beneath Chalk sites at which arable cropping is periodically interrupted by grass leys (Fig. 3), this being most apparent at sites with long term (4 to 7 years) leys (Young, Oakes and Wilkinson, 1976). The nitrate profile beneath arable and arable/ley regimes in the Triassic sandstones (Fig. 4) have been found to follow a similar pattern, but to show more rapid and irregular variations with depth. This may be attributed to the modifying effects of the greater vertical and horizontal inhomogeneity of the Triassic sandstones compared with the Chalk.

At one site in South East England where the Chalk was covered by 11 m of clay high nitrate concentrations were found in the pore waters down to 20 m (Fig. 5), indicating that in this instance the surface deposit did not provide protection to the underlying aquifer.

3.3.2. Fertilised long term grasslands.

Nitrate profiles, in the Chalk and Triassic sandstones, beneath permanent grassland receiving fertiliser applications up to about 250 kg N/ha/yr have been found to be generally uniform at between 5 and 10 mg N/l (Young and Gray, 1978) but concentrations in the range 10 to 100 mg N/l have been measured in profiles beneath grassland with fertilisation rates greater than about 400 kg N/ha.

3.3.3. Unfertilised long term grassland and woodland.

Profiles measured beneath unfertilised grassland have consistently shown interstitial nitrate values of less than 6 mg N/l and commonly less than 1 mg N/l. A similar distribution appears to be present beneath established woodland.

3.3.4. Tritium

Tritium profiles have been determined at a large number of the Chalk sites. In all cases the peaked form of the profile was comparable with that determined by Smith, Wearn, Richards and Rowe (1970) in the Upper Chalk of Berkshire in 1968, who suggested that the peak concentrations recorded the position of infiltration during the winter of 1963/64, when thermonuclear tritium in rainfall reached maximum values (Otlet, 1978). The examples shown in Fig. 6a were measured in the Upper Chalk in Surrey between 1975 and 1977, and include both permanent unfertilised grassland and arable sites.

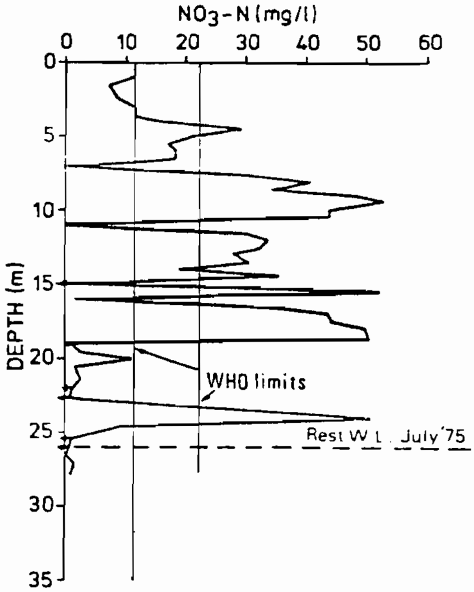


Fig. 4. Nitrate profile - Triassic sandstone arable

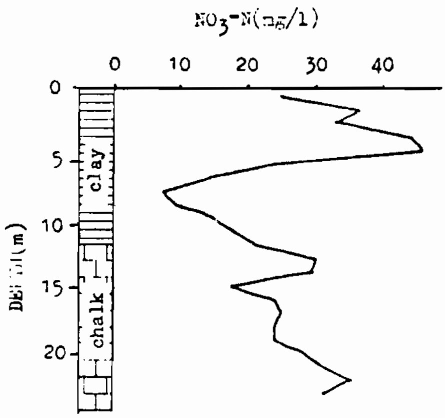


Fig. 5. Nitrate profile - Clay/Chalk arable

Profiles measures in the Triassic sandstone aquifers (Fig. 6b) are less well defined than those from the Chalk, but provide indications of peak concentrations at depths of about 20 metres.

3.3.5. Repeated Profiling

The question must be posed as to whether the nitrate and tritium profiles which have been measured result from a downward migration of solutes, or whether the positions of the peaks are controlled by hydrogeological factors such as the positions of bedding planes and zones of high and low permeability. Direct evidence of movement has come from repeat drillings at two sites on the Chalk and one on the Triassic sandstone. At a site in Kent, holes were drilled in a field, which has been in arable cultivation since the early 1900's, in October 1975 and again in October 1978. Nitrate and tritium profiles are shown in Fig. 7 and indicate a downward movement of about 2 m which is consistent with the low infiltration of this area.

3.4. SATURATED ZONE

In the Chalk and Triassic sandstones smooth concentration gradients have been recorded within the zone of groundwater level fluctuations, indicative of washing out of interstitial waters by saturated zone flows. As would be anticipated concentrations above and below the water table have frequently been found to be different at the interface between the vertical movements of the unsaturated zone and the horizontal flows in the saturated zone. Concentrations measured in pore water from the saturated zone have generally been less than 10 mg N/l, but values over 20 mg N/l have been recorded.

4. SIMULATION OF NITRATE MOVEMENT AND PREDICTION OF FUTURE TRENDS

Models of the movement of nitrate and other solutes through the unsaturated and saturated zones have been developed to help interpret the field measurements and to provide a means of predicting future trends.

4.1. VERTICAL FLOW MODEL

A vertical flow model has been developed in which solutes originating at the soil surface are leached downwards at a rate depending on the infiltration

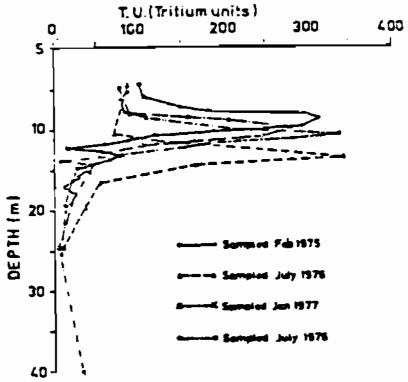


Fig. 6 (a) Tritium profiles in unsaturated Chalk in Surrey.

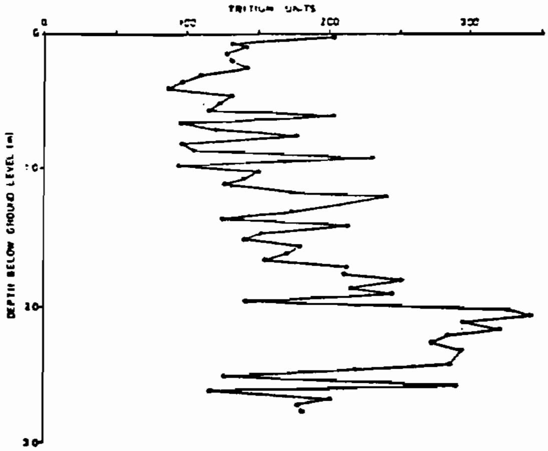


Fig 6 (b) Tritium profiles in unsaturated Triassic Sandstone.

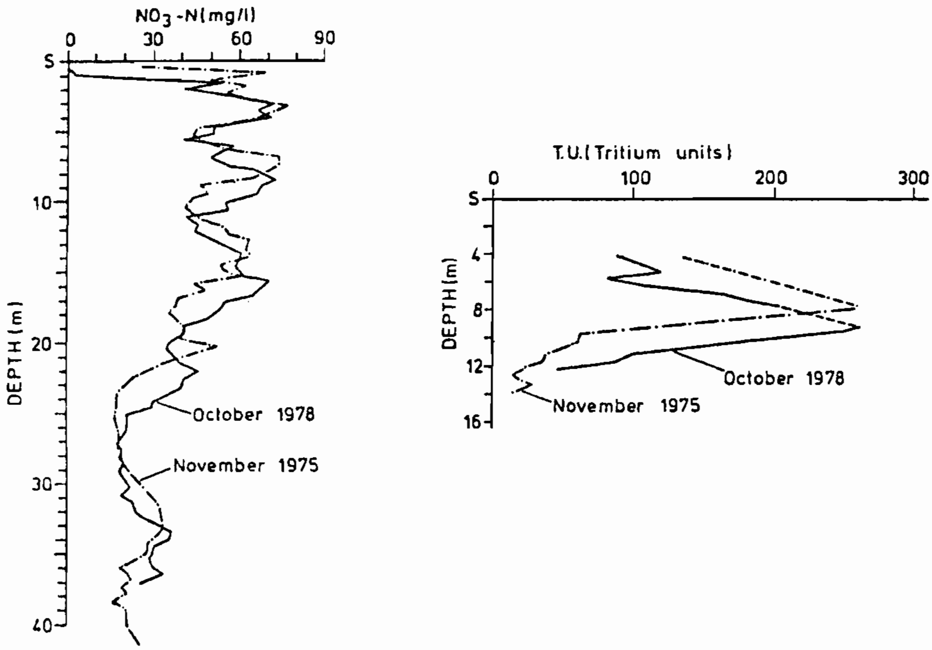


Fig.7. Nitrate and tritium profiles from repeat drillings at an arable site on the Chalk in Kent.

and the pore water content of the rock. It was assumed that a small fraction, typically 5-15% of the infiltrating water and solutes moves quickly down to the water table via the larger fissures. The remaining water and solutes fill up the pore space at the top of the unsaturated zone displacing downwards water and solutes already in the profile. Some attenuation of peak concentrations was assumed to occur and is modelled by a dispersion mechanism. The model works with a yearly time increment and requires for input:

i) annual infiltration rates for the period of simulation,

ii) land use history and fertiliser application rates for the period of simulation,

iii) pore water content which is assumed to be constant in time and with depth; this is a valid assumption in Chalk as shown by field measurements (Young et al, 1976).

The mass of nitrogen released each year in the soil layers for uptake by infiltrating water was assumed to depend on present and antecedent field use and fertiliser application. As shown in Fig. 2 about 50% of the fertiliser applied to root crops and cereals is not taken up by the crop, and it was assumed in the model that this quantity is leached downwards. Not all of this material is available in the year of application. Using Kolenbrander's (1975) work as a basis, it was assumed that mineralisation takes place over a three-year period. Table 1 gives an estimate of the nitrate available to infiltration as a fraction of the application rate.

TABLE 1. Model control rules for roots and cereals

Crop	N (kg/ha) available as a fraction of the application rate N (kg/ha)		
	Year of application	Following year	Next following year
Roots	0.35	0.10	0.05
Cereals	0.25	0.15	0.10

The effect on the model results of varying the distribution in time of the mineralisation rate was small as the major contribution to nitrate leaching comes from the ploughing of grassland. By matching the model-results to the observed N profiles in a number of boreholes it was possible to estimate the amount of N released by ploughing grasslands of various ages. Table 2 gives the corresponding model control rules.

TABLE 2. Model control rules for the ploughing of grass

Years in grass Prior to ploughing	N(kg/ha) released by ploughing Total	Year of ploughing	Following year	Next following year
1	100	60	30	10
2	190	114	57	19
3	240	144	72	24
4 or more	280	168	84	28

For each year of simulation the following sequence of computations is undertaken:

1. Evaluate the mass of nitrate per unit surface area available for leaching from the soil zone using the land use history and model control rules.

2. Divide the mass of nitrate leached by the infiltration to obtain the mean annual concentration.

3. Route a fixed fraction of the leachate directly to the water table, conceptually through the larger fissures.

4. With the remaining leachate fill the available pore space at the top of the unsaturated zone, displacing downwards water and solutes already in the profile.

5. Aggregate water and solutes so displaced from the base of the unsaturated zone with those moving rapidly downwards via the larger fissures to obtain the net fluxes at the water table.

6. Apply dispersion equation to the solute profile.

The model has successfully simulated nitrate profiles measured at a number of sites in the UK. Fig. 8 shows the simulation of a profile obtained from the Chalk in Hampshire and Fig. 9 shows the model simulation of the tritium profile, from the same drilling. Tritium inputs with infiltration were estimated with the aid of simple soil moisture model to take account of the seasonal fluctuation of tritium in rainfall.

Fig. 10 shows the results from the same model applied to the Chalk site in Kent to simulate the nitrate profile on October 1978. Land use data were available only from 1956, which means that the top 11 m only of the profile may be simulated with real data. Data for the preceding 50 years were synthesised by forcing the model to fit the lower part of the profile obtained for the earlier 1975 drilling. These data were used with real data for the years 1956-78 inclusive to generate the profile shown in Fig. 10. The comparison with measured concentrations is seen to be very good over the total depth.

4.2. CATCHMENT MODEL

A model of nitrate movement in the saturated zone has been built which has as inputs the nitrate fluxes across the water table generated by the vertical flow model. The catchment area is divided into 500 m x 500 m squares and nitrate and water routed through the system according to the prevailing hydraulic gradients. Nitrate reaching the water table during a yearly increment is assumed to mix fully with nitrate already in the saturated zone. This type of model is generally referred to as a fully mixed cell model, and is well suited to problems with diffuse source inputs.

The generation of nitrate fluxes across the water table using the vertical flow model is not a necessary prerequisite to catchment modelling. The principal effect of the unsaturated zone is to delay solutes in their movement to the water table. In the case of the Chalk aquifer in Eastern England the delay is about two years per metre of unsaturated thickness. For catchment modelling the nitrate fluxes across the water table may be generated directly from the time series of nitrate leached from the soil zone by applying lags dependent on unsaturated zone thickness and infiltration. The advantage of using the vertical flow model to simulate measured unsaturated zone profiles is the check on the hydrological data and the model control rules which such a simulation provides.

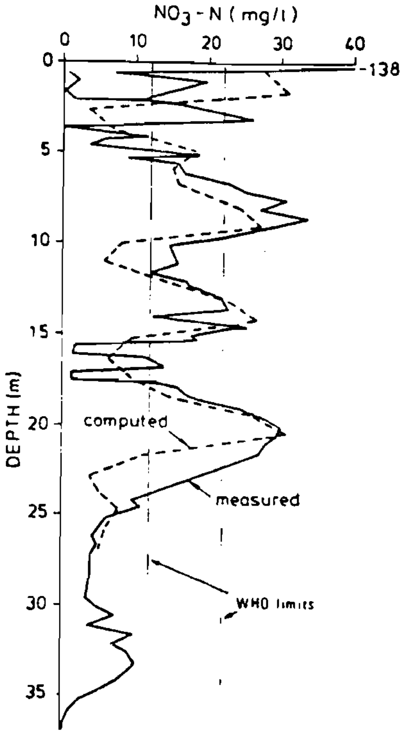


Fig. 8. Measured and simulated nitrate profiles - Chalk arable in Hampshire.

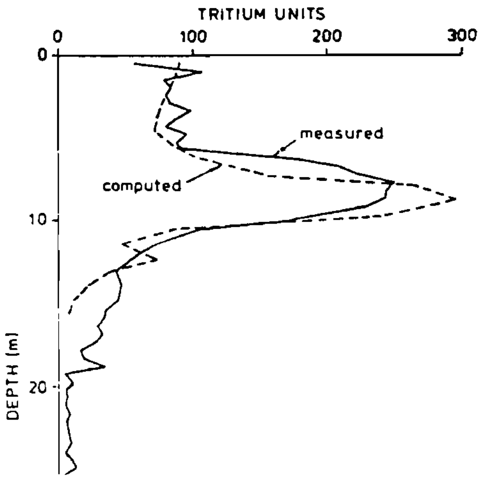


Fig. 9. Measured and simulated tritium profiles - Chalk arable in Hampshire.

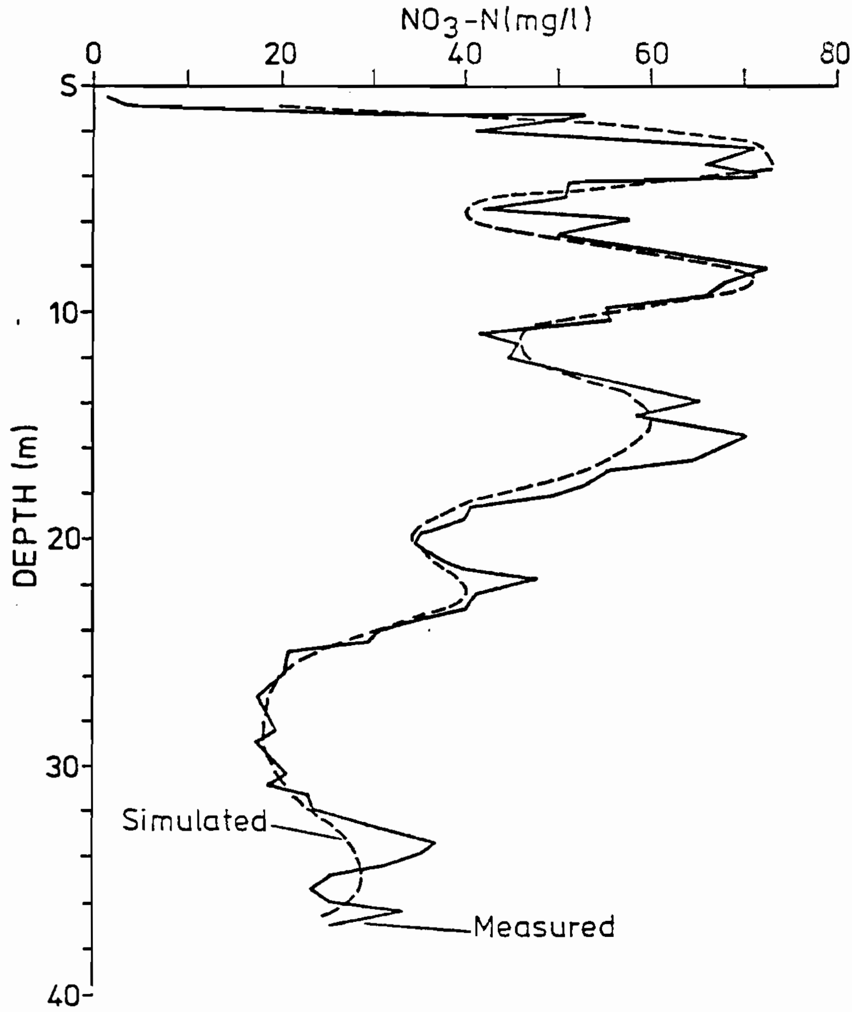


Fig. 10. Measured and simulated nitrate profiles - Chalk arable in Kent.

The catchment model works with a yearly time increment and requires for input:

i) annual infiltration rates for the period of simulation

ii) land use history and fertiliser application rates for the period of simulation for distinct land units within the catchment

iii) distribution of groundwater levels and pumped abstractions

iv) depths to water table

v) pore water contents of the unsaturated and saturated zones

vi) effective depth of flow in the saturated zone.

For each year of simulation the following sequence of computations is undertaken.

1. Evaluate groundwater flows from the water level distribution, infiltration rates and groundwater abstractions starting at the highest water level in the catchment and working through the nodes in order of decreasing water level.

2. For each node evaluate the nitrate flux at the water table either using the vertical flow model, or directly from the historical land use data by applying a lag dependent on the unsaturated zone water content and thickness, and the infiltration.

3. Starting at the node with the highest water level and working through the nodes in order of decreasing level evaluate the nitrate concentrations assuming nitrate entering each node, either from above or with subsurface flow, mixes fully with nitrate already stored in the saturated zone.

The model has been applied to Chalk, Limestone and Sandstone catchments and has been able to accurately simulate nitrate concentrations in pumped abstractions.

4.3. CHALK CATCHMENT

For the catchment in Kent from which the profiles of Fig. 7 were obtained the model has been run from the year 1800 up to the year 2100. Early data on land use and fertiliser application rates were obtained from parish and Ministry of Agriculture, Fisheries and Food records and by matching the vertical flow model to measured distributions of nitrate in the unsaturated zone. Present levels of fertiliser use were assumed to be maintained in the future. Fig. 11 shows a typical model output, giving nitrate concentrations in pumped groundwater discharge. There are a few measured values with which to compare the simulations and these are shown in Fig. 11. The predictions suggest that in this catchment nitrate concentrations will exceed 20 mg N/l within the next two decades and will stabilise at about 33 mg N/l.

4.4. SANDSTONE CATCHMENT

In a sandstone catchment in the Midlands a farmed area of nearly 2 km^2 has received a sewage discharge of 5000 m^3/d since 1880. The sewage is spread by a pipe system and allowed to seep into the soil through ploughed channels. Groundwater nitrate concentrations now exceed 14 mg N/l in parts of the catchment, although remote from the sewage spreading area the concentrations are below 5 mg N/l. A catchment model has been built, and run from 1880 up to 2100. Nitrate in the sewage effluent was treated in the same manner as inorganic fertiliser additions. Fig. 12 shows the model predictions at two major pumping stations in the catchment compared with measured values. Fig. 13 shows the areal distribution of nitrate in groundwater at the present time, from which the effect of the sewage spreading activities is clearly seen.

In Chalk and Sandstone catchments the model predictions of future trends are relatively insensitive to future changes in land use and fertiliser application rates because of the long transit time through the unsaturated zone. The transit times are between 0.5 y and 2 y per metre of unsaturated thickness for the two examples considered. It will be apparent that short term control of groundwater nitrate concentrations by reduction of fertiliser

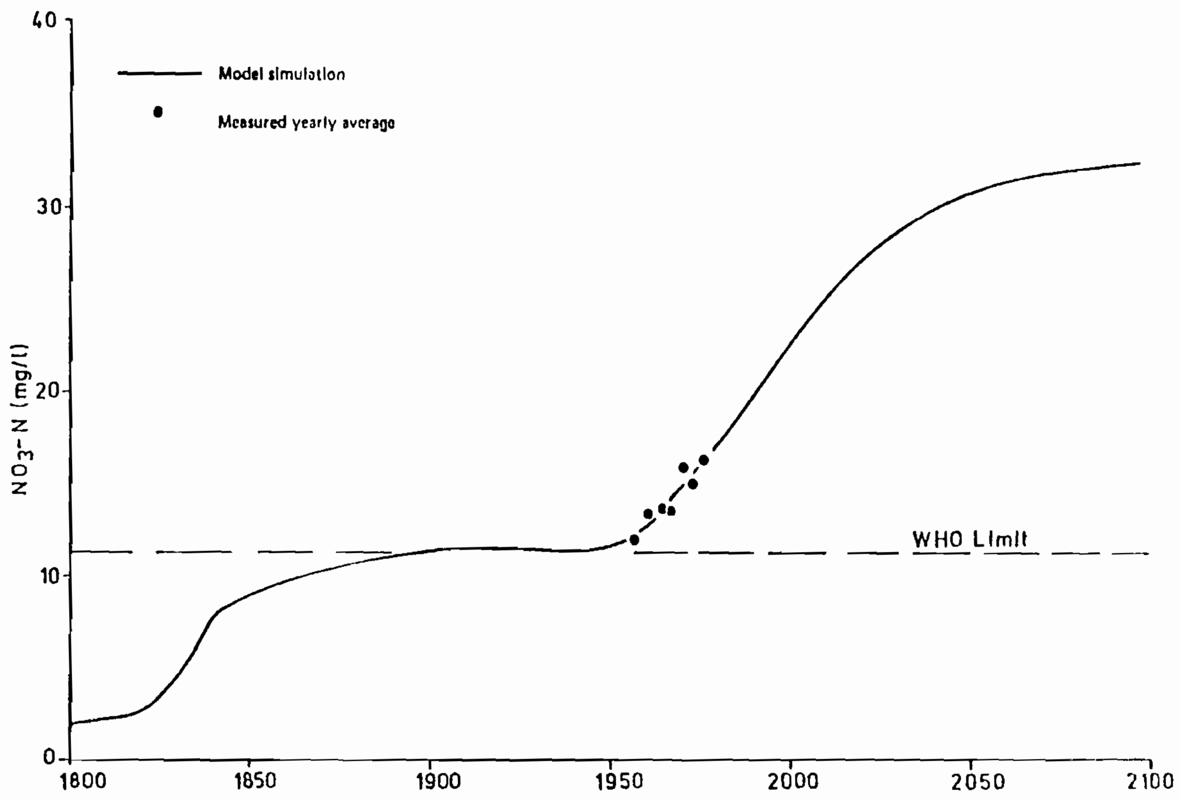
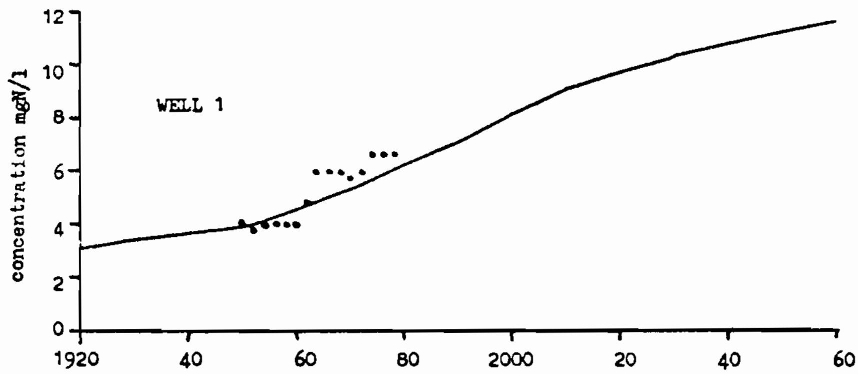


Fig. 11. Measured and simulated nitrate concentrations in pumped discharge - Chalk catchment in Kent.



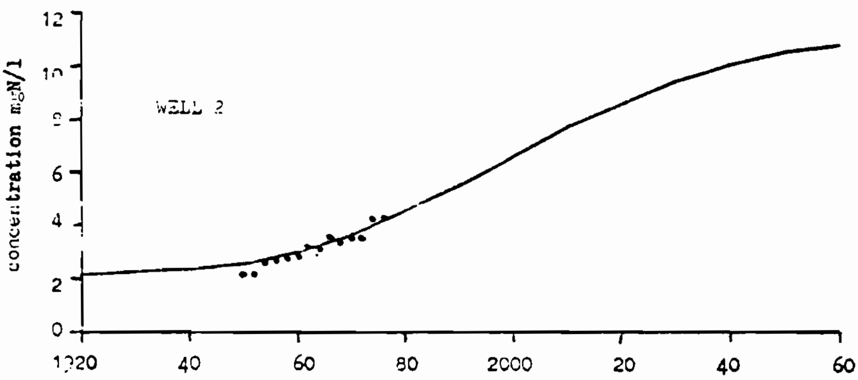


Fig. 12. Measured and simulated nitrate concentrations in pumped discharges - Triassic sandstone catchment.

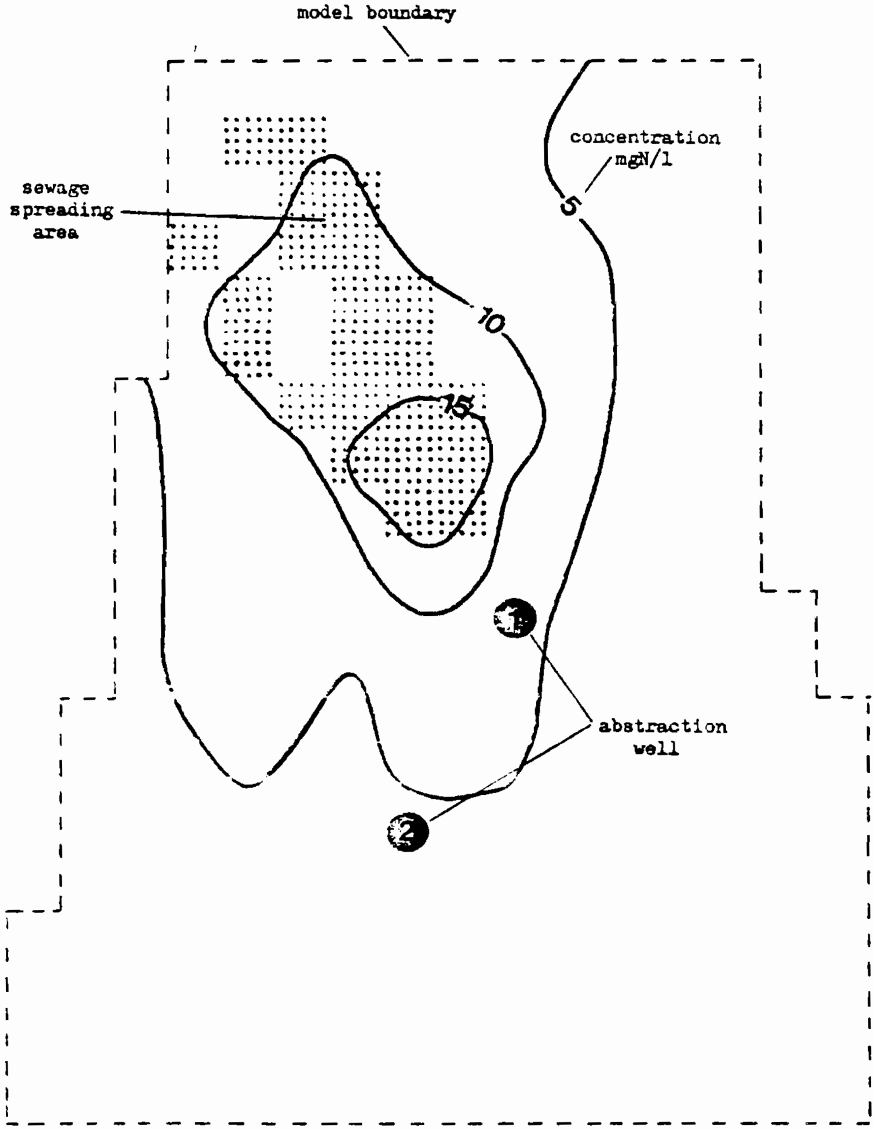


Fig. 13. Simulated groundwater nitrate concentrations for 1980 - Triassic Sandstone catchment.

applications in predominantly arable catchments is not practicable in general.

5. CONCLUSIONS

Field investigations have established that large quantities of nitrate are present in the unsaturated zones of the principal aquifers in the United Kingdom. A strong correlation has been found between high interstitial water nitrate concentrations in the unsaturated zone and arable farming regimes. Low nitrate leaching losses are associated with moderately fertilised, and unfertilised, grassland.

Experimental and field observations indicate that very large nitrate leaching losses are associated with the ploughing up of established grassland or long-term grass leys, as a result of the mineralisation of soil organic material.

Correlation of nitrate and tritium profiles through the unsaturated zone of the Chalk with historical variations in nitrate leaching rates due to changing husbandry practices and with the thermonuclear tritium content of rainfall suggests that a high proportion of both solutes moves slowly downwards. This has been confirmed by sequential sampling at three sites. A bypass mechanism exists to allow the rapid infiltration of intense rainfall to the water table; 5-15% of residual rainfall may reach the groundwater by this route.

A mathematical model has been developed which satisfactorily simulates measured tritium and nitrate profiles. The rate of downward movement is dependent on the local hydraulic properties of the aquifer and infiltration rates. In the Chalk the rate of movement is generally between 0.5 and 1.0 m/year and in the Sandstone up to 2 m/year. The quantity of nitrate leached from the soil zone is assumed to be one half of that applied as fertiliser, averaged over the year. Ploughed grassland leaches up to 280 kg N/ha and is a major contributor to increasing groundwater nitrate concentrations.

The mathematical model of vertical movement has been incorporated in a catchment model for groundwater flow in the saturated zone and allows predictions to be made of future nitrate concentrations in groundwater abstractions or spring discharge. The catchment area is divided into a number of cells in each of which the land use history and fertiliser application rates can be specified, and the model assumes complete mixing of nitrate over the full depth of flow in the saturated zone.

The model has been applied to Chalk, Limestone and Sandstone catchments and the results indicate that nitrate concentrations are generally not yet in equilibrium with land use and may continue to rise to levels in excess of 11.3 mg N/l (WHO lower recommended limit) before stabilising.

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MODELLING NITRATE POLLUTION IN WATER RESOURCE SYSTEMS: THE THAMES NITRATES PROJECT

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In view of the EEC Directive on the Quality of Water Intended for Human Consumption, nitrate concentrations threaten to impose a severe constraint on water resources in the Thames region. Thames Water adopted a three-phase approach when faced with rising nitrate concentrations in aquifers and rivers across the region. Before a detailed nitrates management policy could be selected (stage three), two stages of the project had to be completed. First, the broad region-wide magnitude of the problem had to be predicted in order to assess its urgency. Second, if there was sufficient cause for concern, a methodology had to be developed for evaluating the effectiveness of different regional nitrate management strategies. This paper describes these two phases of the Thames Nitrates Project.

INTRODUCTION

The Thames Water Authority is one of ten regional water authorities in England and Wales, responsible for water supply, sewage disposal and river management. It serves about twelve million people, sixty per cent of whom live in the London area. Over half of the water supplied is from the River Thames and its tributaries, the rest being drawn from groundwater sources. The London area is served largely by a group of reservoirs filled from the lower Thames and from one of its tributaries, the River Lea (Figure 1).

Although river quality in the Thames catchment has been generally stable, or, locally, improving in recent years, concentrations of nitrate have been rising. The mean annual concentration at Walton, an intake on the lower Thames feeding the London reservoirs (Figure 1), increased from 4.2 mg N/l in 1968 to 7.7 mg N/l in 1979 (Figure 2). In the winter of 1976/7 the concentration was above the WHO recommended limit of 11.3 mg N/l for sixty-two days. In recent years the potential impact of nitrates on water resources in the region has been an increasing cause of concern, heightened by the expected EEC Directive on the Quality of Water Intended for Human Consumption, published in July 1980. This legislation makes 11.3 mg N/l the maximum permitted concentration for water in supply, and requires compliance within five years.

FIGURE 1

THE THAMES CATCHMENT

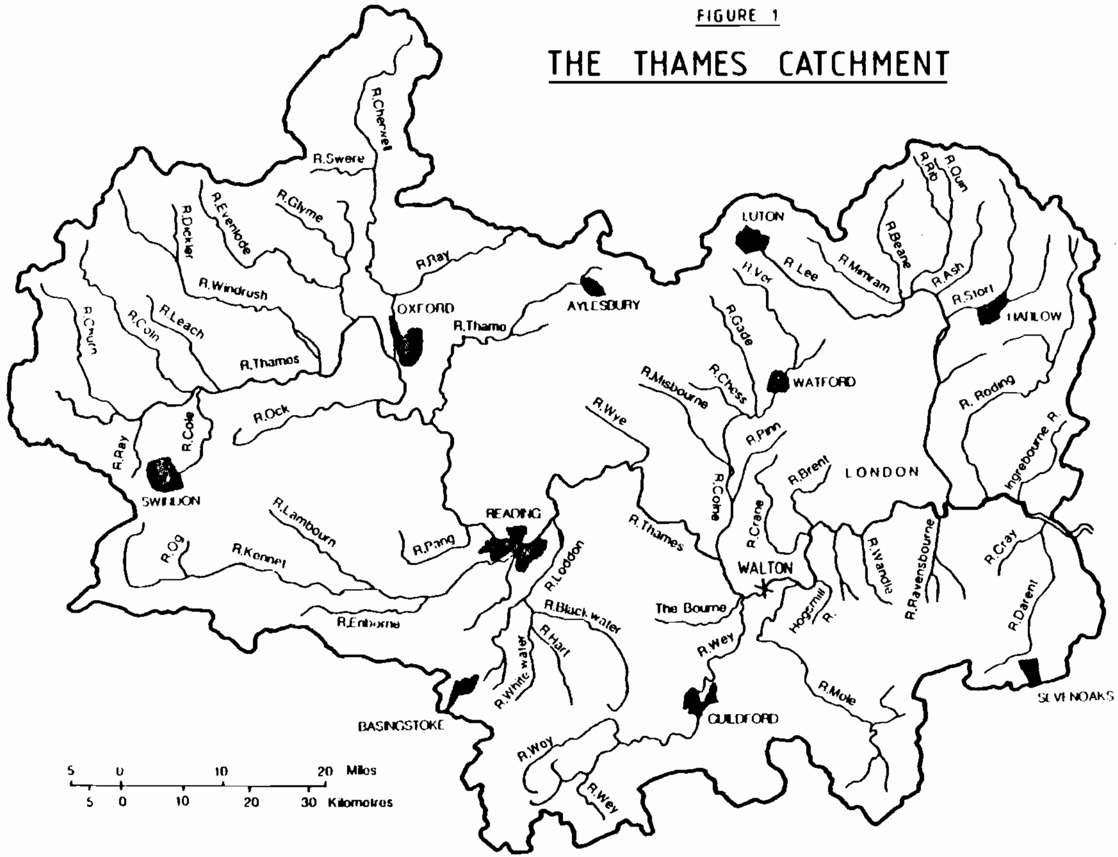
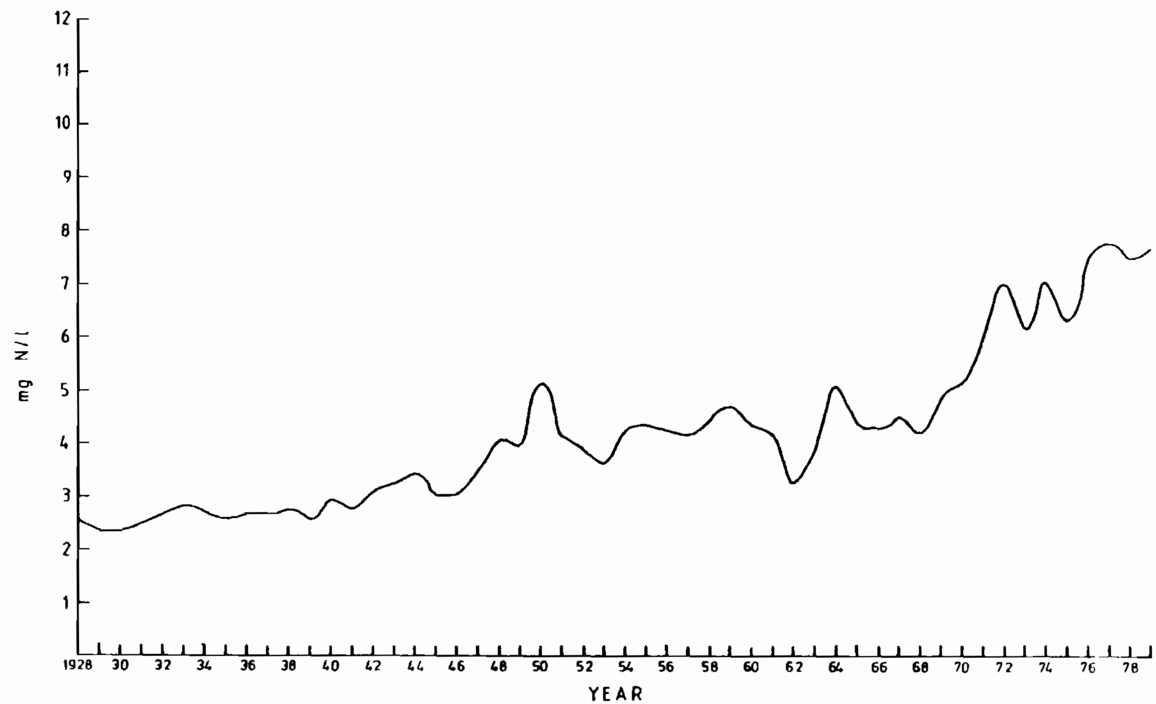


FIGURE 2

MEAN ANNUAL NITRATE CONCENTRATIONS AT WALTON 1928-1979



In order to devise a regional strategy for continuing to supply water within the nitrate limit, a three-phase approach was adopted whose aims were: a) to assess the potential severity of the problem ("problem identification"), b) to devise a technique for evaluating options, and c) to employ this technique in the selection of a preferred strategy. This paper describes first the preliminary exercise of problem identification, and then the methodological developments to which it led.

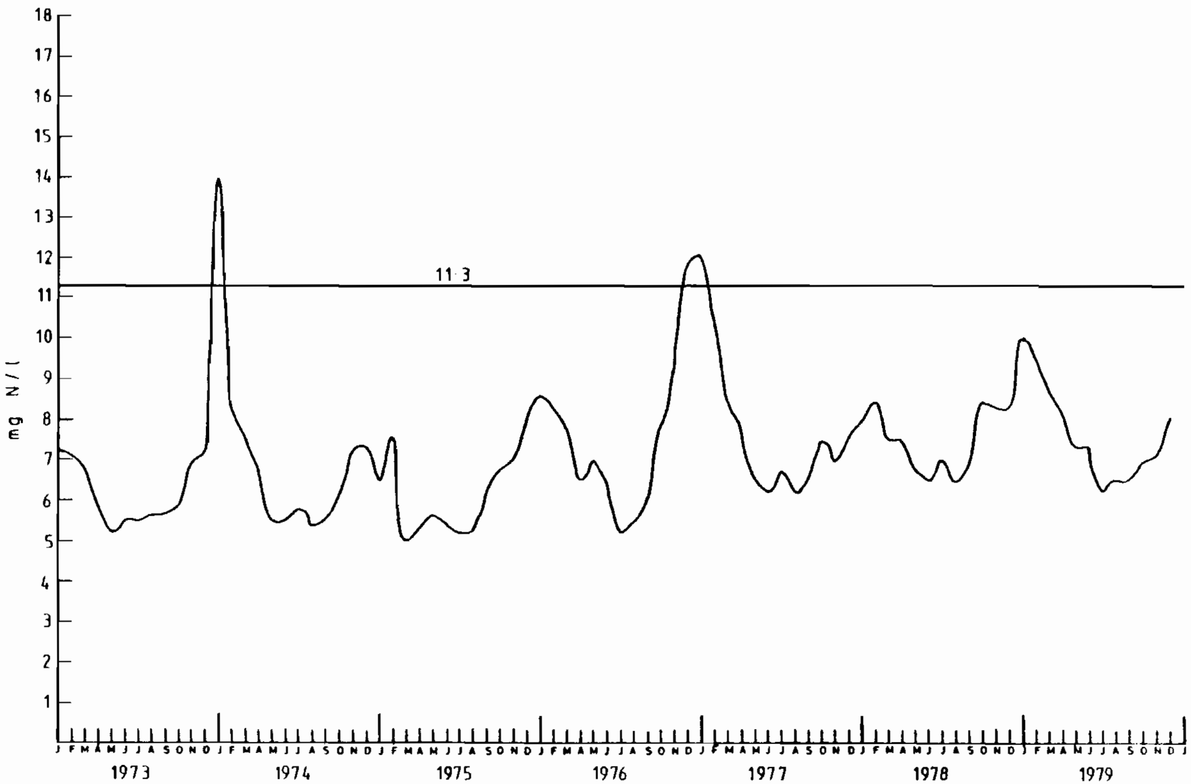
PROBLEM IDENTIFICATION

It was apparent that, for the Thames Water region, river nitrate concentrations were more likely to exceed the 11.3 mg N/l limit in the near future than were groundwater concentrations. Although surface water mean annual nitrate concentrations are not significantly higher than those of many groundwater sources, their seasonal variability leads to periods when the limit may be exceeded, even if the mean is well below the limit (Figure 3). Groundwater sources have generally stable concentrations through the year and therefore higher mean concentrations can be reached before water supplies are threatened. However, rising groundwater concentrations were considered to be a significant threat to resources through groundwater discharges to surface water which must contribute to increasing nitrate concentrations in rivers.

Problem identification required the prediction of nitrate concentrations at a site on the river system which was representative of major abstraction sites. The location chosen was Walton whose significance is described in the Introduction.

FIGURE 3

MEAN MONTHLY NITRATE CONCENTRATIONS AT WALTON 1973 - 79



Previous attempts at forecasting nitrate concentrations by extrapolation of existing records were considered inadequate since they give no consideration to the causes of the trend. The approach taken in this case was to develop a causal model, relating the observed series at Walton from 1929 to 1975 to estimated nitrogen inputs to the resource system over the same period (Onstad and Blake, 1980).

Nitrogen inputs to the hydrological system can be divided into nitrogen released in the soil zone and nitrogen entering rivers in the effluent from sewage treatment works. It was shown that historical population increase within the catchment and the resulting nitrogen load of the Thames from sewage effluent could explain very little of the variation in the observed series at Walton. Changes in soil nitrogen availability therefore appeared to be the main causal factor. Published statistics on land use, animal numbers, fertiliser use and crop yields were used to estimate the net inorganic nitrogen available each year in the soil zone (Figure 4).

Using time series analysis based on Box-Jenkins transfer function models, a mathematical relationship was derived between the "total available nitrogen" series, and the resulting mean annual river concentration at Walton:

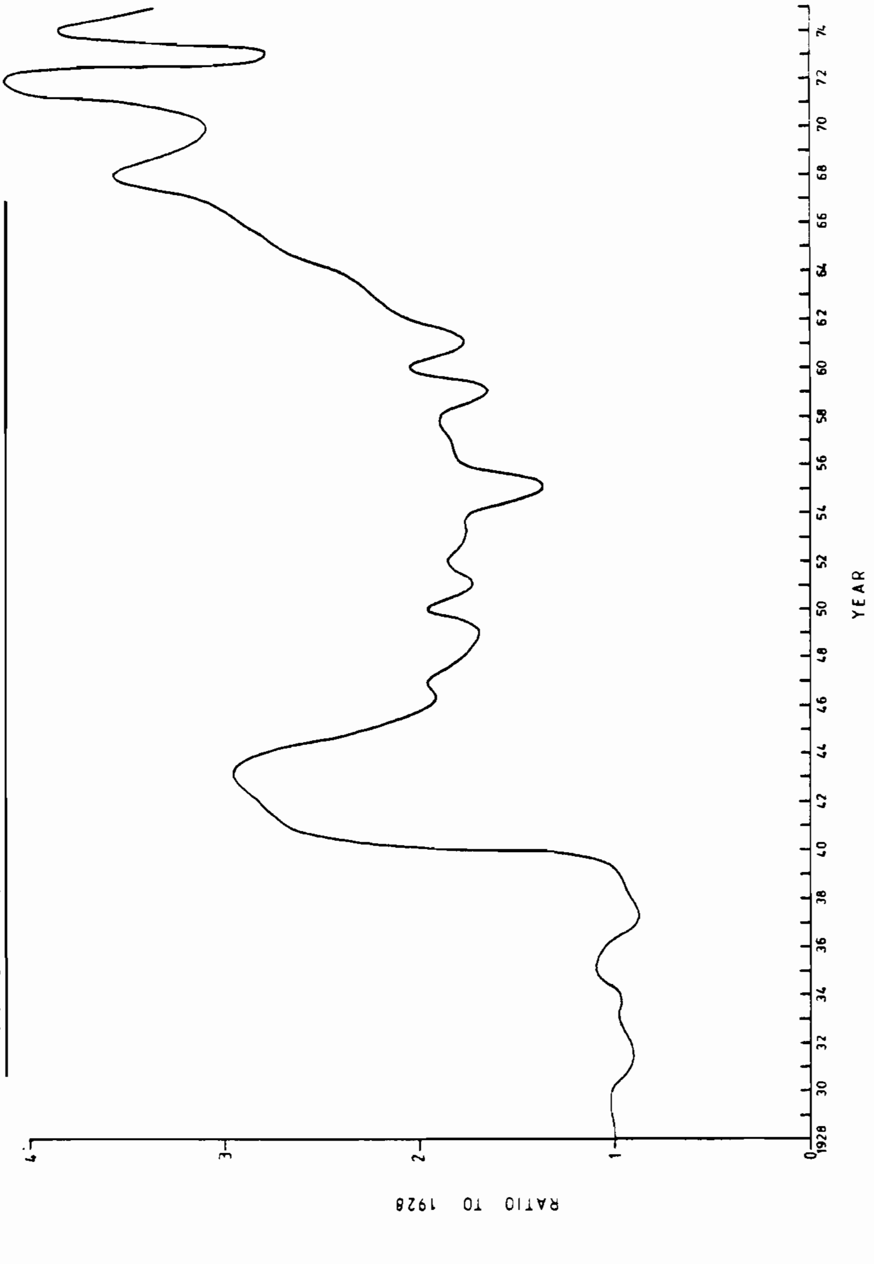
$$Y_t = 0.124\ Y_{t-1} + 0.727\ U_t + 1.060\ U_{t-7} - 0.240$$

where Y_t is mean annual nitrate concentration in mg N/l and U_t is inorganic nitrogen available in the soil expressed as a ratio to that available in 1928.

This relationship explains 78% of the variation in nitrate concentrations at Walton for the period concerned. It also reflects the two paths by which soil nitrate reaches watercourses, namely a rapid path through the soil layer only, and a slower path through the underlying aquifers. On an annual time scale these were represented in the mathematical model by an instantaneous and a lagged response in the river, the lag identified being one of seven years. The main features of the input series which helped to identify this relationship were the increase in soil inorganic nitrogen from 1940 to 1945 following wartime

FIGURE 4

MASS OF NITROGEN AVAILABLE IN SOIL ZONE 1928 - 1975



ploughing of grassland, and from 1963 onwards, due to increased inorganic fertliser applications. Considerably longer travel times of nitrate through aquifers have been suggested by those working at the Water Research Centre, but these were not identified in the forty-seven year series available. The model derived predicted concentrations over this period well enough for it to be used with some confidence for forecasting the future.

Using this relationship, and a range of assumptions on the future of agriculture in the Thames region, mean annual nitrate concentrations of the lower Thames were forecast. The range of predicted mean annual nitrate concentrations for the end of the century is 9.8 mg N/l to 15.3 mg N/l, depending on the assumptions made on the course of agricultural development.

The implication of a mean concentration above 10 mg N/l is that daily concentrations would be above 11.3 mg N/l for several weeks at certain times of the year. For the Thames, these periods are characteristically those of high flows associated with runoff from the soil. Until now it has been possible to stop abstracting water when concentrations rise above 11.3 mg N/l at any site but this practice clearly cannot be extended for long periods, especially since there is little spare capacity in the region's water resources.

A REGIONAL NITRATES STRATEGY : OPTIONS

The first phase of the project had confirmed that a strategy was needed for nitrates management within the Thames water resource system. Having recognised also the necessary emphasis on surface-derived water, several options were identified for reducing nitrate concentrations in the water supplied. River concentrations could be reduced at some abstraction sites by denitrifying effluent at certain sewage treatment works. Increased storage could be provided for high nitrate river water, to allow natural denitrification. At some surface water treatment works, facilities could be introduced for blending river water with lower nitrate groundwater or stored water. Groups of reservoirs serving one treatment works could be managed in such a way as to provide more low nitrate water at critical times. Finally, water denitrification plant could be installed at any treatment works.

These options had to be considered in association with those for a new resource in the Thames region, required to meet an increasing demand for water. Hydrological evaluations were proceeding for a number of sources, including a new reservoir for direct supply or river regulation, increased artificial recharge of groundwater and increased river augmentation by pumping from groundwater. A technique was needed for both assessing potential new sources from a nitrates viewpoint, and evaluating options for nitrates management throughout the water resource system.

METHODOLOGY : CONSIDERATIONS

Three main factors influenced the choice of a methodology. First, the Thames water resource system is complicated, requiring simulation to investigate the impact of changes in the system, its inputs and its management. Since simulation is demanding of computing time, the model had to be computationally as economic as possible. Second, Thames Water had already developed a water resource model (Sexton et.al. 1979), designed for hydrological evaluation of the system. It was desirable that a quality model be compatible with this. Finally, there was a tight time constraint imposed not only by the EEC Directive but also by the need to contribute to the impending decision on a new source.

These considerations established the main characteristics of the model. It was to be developed from a number of general components, based largely on existing nitrates models and calibrated using data for the Thames system. The components needed were: soil, aquifer, sewage treatment works, channel, reservoir and water denitrification plant. Process models for the sewage works, reservoir and water denitrification plant components were already being developed within Thames Water. For the remaining components, the Water Research Centre and the Institute of Hydrology were invited to adapt their existing nitrate models for particular parts of the Thames water resource system.

The space scale for soil, aquifer and river reach components was decided partly on the basis of the expected sensitivity of the system at points of interest to the spatial variability of inputs. However, since this sensitivity was not known with certainty at the outset, it was desirable to include as much detail as was feasible. The resulting spatial structure of the model is shown in Figure 5.

A way of integrating the components to reduce overall computing time was devised, and is shown in Figure 6. It was clear that in investigating options for nitrates management, attention would focus on the two main water supply rivers, the Thames and Lea, and their associated reservoirs. This part of the system was therefore identified as a free-standing sub-model ('B'), which could simulate the impact of any of the above strategies on the quality of water supplied. A range of inputs to this sub-system would be provided by sub-model 'A', which would simulate the impact of different agricultural conditions and of different sewage denitrification strategies on nitrate concentrations of tributaries at their confluence with the main rivers. Figure 6 indicates also the

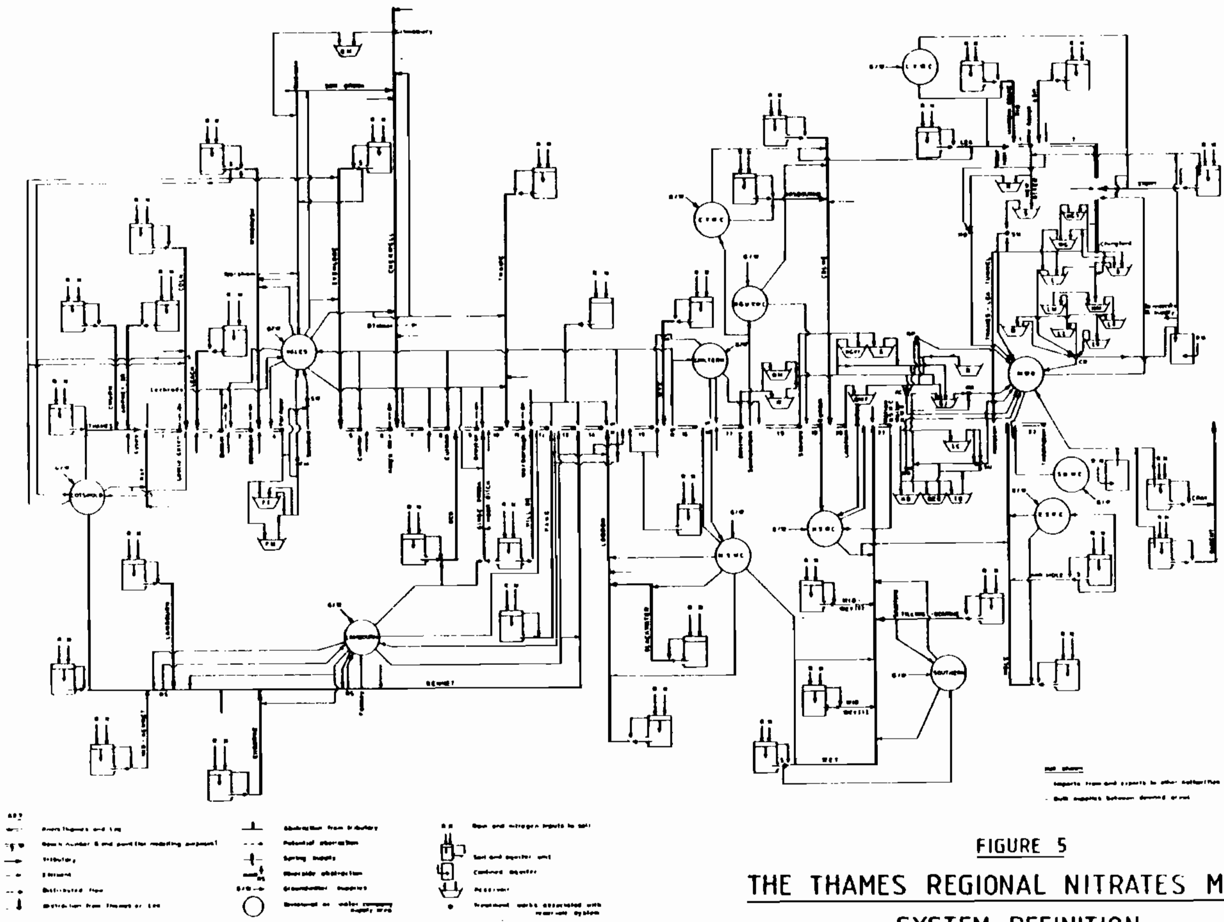


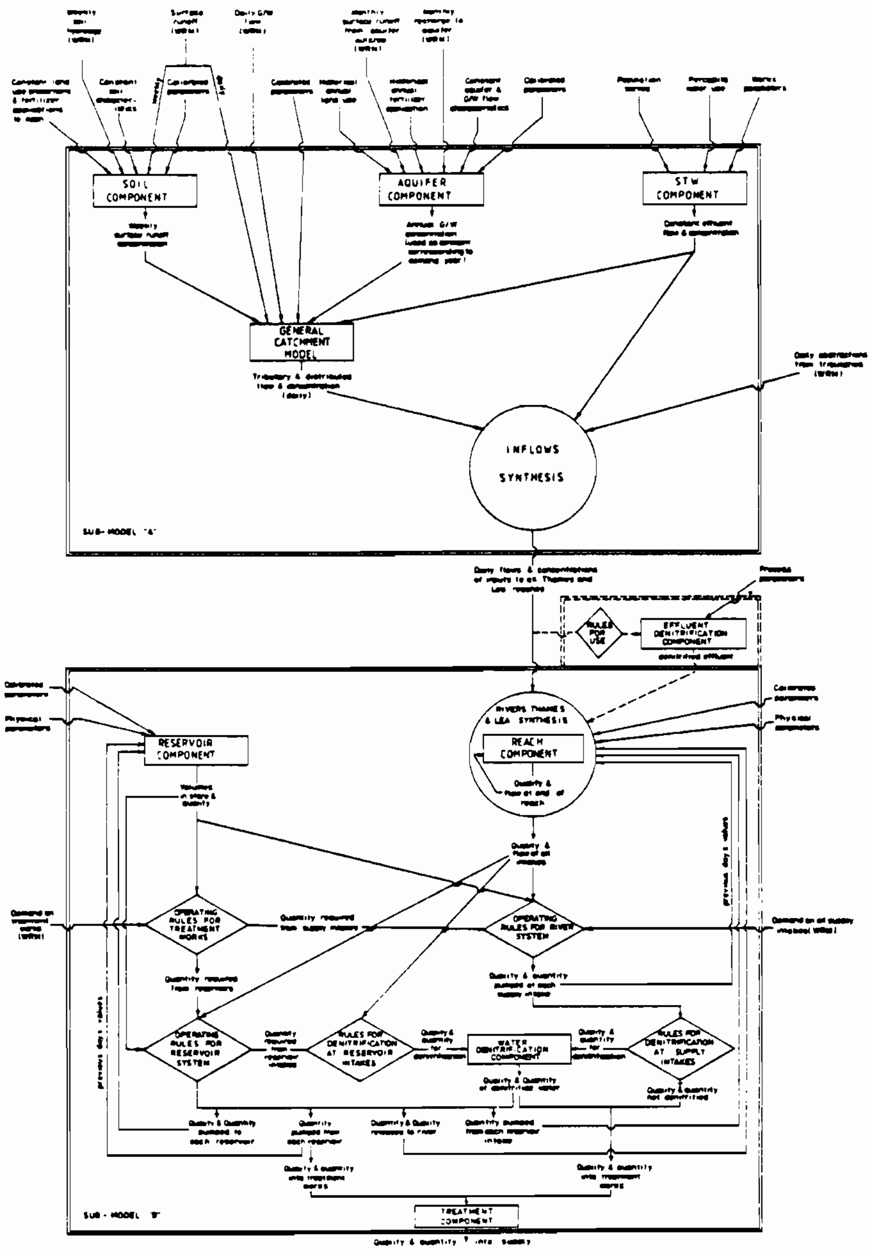
FIGURE 5

THE THAMES REGIONAL NITRATES MODEL

SYSTEM DEFINITION

FIGURE 6

THE THAMES REGIONAL NITRATES MODEL - STRUCTURE



particular time increments to be used for each component model, reflecting the observed general variability of processes within that component.

RUNNING THE MODEL

Thames Water is collaborating with the Ministry of Agriculture to produce forecasts of the extent and use of agricultural land in each tributary catchment. The most significant factors for total nitrogen release in the soil are likely to be the balance between cereals and grassland, the application rate of fertiliser to cereals and the frequency of ploughing of temporary grass. These forecasts along with those for water demands and the population served by sewage works will be input to the model, together with sixty years of hydrology generated by the Water Resource Model from many inputs including historical rainfall. The performance of the system over this sample period of hydrology under different strategies will be defined by the quality and quantity of water supplied as output by the model.

In parallel with model runs, cost estimates for each system configuration and mode of operation will be made. A strategy will be selected on the basis of a performance-cost ranking.

OBSERVATIONS

The project is clearly an ambitious one in its scope and complexity. It must be recognised however that investment decisions will have to be made by Thames Water within the next few years in order to comply with the nitrate limit of the EEC Directive. It is desirable that these decisions be based on a systematic assessment of our nitrate management options, comparing their impact on the future water resource system as a whole, in order to maximise the effectiveness of the investment.

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AN INTEGRATED PHYSICAL-ECONOMIC SYSTEMS ANALYSIS OF IRRIGATED AGRICULTURE

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This paper presents an integrated physical-economic methodology for the evaluation of nonpoint source pollution control technologies in an irrigated agricultural setting. The economic model determines the optimal allocation of resources and provides the capacity to assess the socio-economic impacts of nonpoint source control. Its physical counterpart estimates effects on the quality and quantity of groundwater as a result of the normative cropping pattern derived in the economic model. Four alternative control policies are then evaluated: continuation of current practices, mandatory tailwater recycling, mandatory irrigation scheduling, and recycling in concert with scheduling. Although the results presented emphasize the nitrate pollution of groundwater, the production of drainage effluent, salinity, surface runoff and changes in subsurface drainage are also examined. Results are described in aggregate for the region by parameter and time period and by spatially differentiated production units. These results show the need for integrating the physical and socio-economic aspects of the agricultural production system within a single analytical methodology in order to generate policy relevant information.

INTRODUCTION

This paper presents the philosophical, methodological and empirical basis for the systematic analysis of environmental policies. While the specific focus of this research has been the problem of nonpoint source pollution control, it is imperative that any policy analysis be conducted within a systems framework. Otherwise, there is no assurance that the proposed policy will be either effective in attaining its objectives or without unintended impacts upon other policy targets. Resource and environmental planning cannot be separated. Planning for resource use without recognition of the environmental goals or objectives of society may result in resource allocations which are socially suboptimal. Similarly, planning for environmental quality without assessing the suitability, availability, and productivity of the resource base may impair the efficiency and distribution of output.

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Current events certainly imply more serious restraints on public fiscal resources in the coming decades with no compensating reduction in the demand for new and expanded public programs and policies. Clearly, this dictates increasing the efficiency of operation of institutions. Specifically, in natural resource management, the explicit dynamic and often simultaneous nature of these systems must be recognized and incorporated into any policy analysis. Failure to account for these system attributes may result in inefficient or ineffective policy actions (Dudek, 1979). Increased emphasis needs to be placed upon the design of policies that are not only effective in attaining social objectives, but do so efficiently without disruption to the rest of the system. The current partial analysis of resource problems leads to policy recommendations whose net effects upon the system are ambiguous. This research demonstrates the critical role of system dynamics and institutional design in public policy effectiveness.

This paper is organized into four broad sections. The first of these establishes the critical role of the institutional environment in defining the feasible policy set and in conditioning the planning approach. The second section presents the integrated physical-economic analytical methodology. Each of the modeling components are introduced and described in the context of their interaction within the system. The operation of the analytical system is demonstrated for four policy management alternatives in the third section. The final section of the paper is devoted to a discussion of the conclusions derived from the analysis.

THE INSTITUTIONAL ENVIRONMENT AND PLANNING PHILOSOPHY

For over 20 years, economists have been commenting on the methods, procedures, and criteria employed in water and related land resource planning and development. This input has been substantial and has had significant impact on the conduct of such resource analyses. An example of this impact in the United States is the formulation of Principles and Standards by the Water Resources Council which require Federal and Federally-assisted planning activities to consider the co-equal objectives of national economic development and environmental quality (U.S Water Resources Council, 1973). Increasingly, an awareness of the complex interdependencies between bio-physical and socio-economic systems has stimulated the initiation of comprehensive resource planning methods and the abandonment of single-purpose approaches. However, irrespective of the recognition given to these concepts, examples of their practical application in resource planning efforts are scarce. Currently, opportunities exist in the United States for the integration and simultaneous completion of several water and land resource planning activities. These individual legislatively mandated planning requirements are not currently integrated or coordinated.

Water Quality Legislation

While many planning and assistance programs exist in the United States, the one with the most potential for widespread effect on agricultural resources is the Areawide Waste Treatment Planning and Management program of Public Law 92-500, the Federal Water Pollution Control Act Amendments of 1972. Public Law 92-500 was enacted with an overall objective of restoring and maintaining the chemical, physical and biological integrity of the Nation's waters. It sets forth goals, requirements and deadlines for achieving this objective and calls for eliminating the discharge of all pollutants into navigable waters by 1985. An interim goal was established for attaining, wherever possible, water quality suitable for the protection and propagation of fish, shellfish and wildlife and for recreation in and on the Nation's waters. This interim goal is to be achieved by July 1, 1983.

Public Law 92-500 requires that water resource development, land use planning and environmental policies be coordinated, integrated and updated in a continuing planning process. This requirement will be accomplished within the structure of Section 208 which requires each state to formulate an areawide waste treatment management plan. Each plan will include; (1) the identification of agriculturally related nonpoint sources of pollution and (2) the specification of procedures and methods to control, to the extent feasible, such sources. These control procedures are termed "Best Management Practices". Under Section 305 of this same Act, each state is required to develop estimates of the environmental impact and the economic and social costs and benefits associated with attainment of the law's objectives. In addition, each state must maintain a continuing planning process consistent with the provisions of the Act.

Planning Conflicts

Owing to the complexity of the socio-economic issues, to differences in the objectives of the planning agencies and to the existence of contradictory legislation, the integrated objectives of comprehensive legislation like PL 92-500 may not be met. Specifically, water resource development and land use plans within a region could be substantially affected by the nature of nonpoint source controls. Alternatively, the development of marginal lands for agricultural production, or the provision of water at subsidized prices increases the difficulty and cost of achieving reasonable environmental goals. The environmental impact from changes in the quantity of unused cropland will be significant. In particular, such land is important in wildlife production and in satisfying recreation and open space demands. Further environmental damage may be caused by utilizing unused cropland with generally high erosion potential.

Preserving agricultural land is often cited as a requirement to meet future food demands. However, pursuing such a goal may have profound effects on water quality. For example, the creation of agricultural preserves could concentrate production and thus compound nonpoint source pollution. Nonpoint controls for the production of sediment which require changes in cultural practices may call for different forms of pesticide and fertilizer applications. Such efforts could complicate attaining desired reductions in chemical residues and plant nutrient levels in receiving waters. Further, as the demand for water resources increases, the issues of water supply and water quality become increasingly interrelated. The loss of water from inefficient irrigation systems not only wastes water, but the water that is returned may be polluted by sediments, salts and agricultural chemicals.

The interaction between water quality and quantity is clear. As water use continues to increase, return flows and sewage effluents will increase. For example, groundwater contamination is increasingly regarded as a serious problem. Since aquifers typically recover very slowly from such contamination, groundwater degradation may be considered semipermanent. Such contamination, whether real or potential, poses a health threat to the populations deriving municipal supplies from groundwater. Continued degradation of groundwater can also affect municipal, industrial or agricultural uses of such supplies. This could have significant implications in the economies of the affected communities and result in substantial changes in land use.

If agricultural income stability and enhancement is a policy objective, nonpoint control measures may have a counteracting effect.

These policies may impact on those least able to pay or in some cases reduce income to a negative level (Dudek and Horner, 1980). There could be large regional disparities in impacts upon agricultural income depending on how controls are costed and administered. Nonpoint pollution controls are likely to take the form of a set of performance expectations. Actions may be taken by public agencies to achieve these performance levels through uniformly applied "best management practices," treatment methods, or physical structures that have no assurance of being least-cost options. Tax reductions have been employed as one mechanism for the preservation of agricultural lands in the rural-urban fringe. These benefits could be offset by the costs associated with nonpoint source pollution control. Thus, these potential conflicts have implications for the viability of agricultural communities.

These comprehensive planning problems require the specification of an analytical system and planning model which considers the simultaneous interaction and trade-off between efficiency and equity. The system must simulate the impacts of varying economic incentives, establishing resource controls, initiating public resource investment, varying resource quality, market structure and alternative institutions. It is to the precise specification of such a methodology that we now turn.

THE INTEGRATED METHODOLOGY

The economic information required for resource planning and evaluation can be discussed under four interrelated headings; commodity demand, commodity supply, resource demand and resource supply. Given the planning setting, the necessary set of information and the analytical models to be employed, it is useful to see how the basic economic concepts of demand and supply are empiricized. These concepts can readily be identified as flows of information between different components of the analytical system as indicated in Figure 1. These same flows of information between models are shown in more precise detail in Figure 2.

The demand for regional commodity production is projected from estimates of changes in population, income and U.S. trade policies. The production model determines the amount of land and productive inputs to achieve the projected level of commodity demand which in turn is affected by land use policy decisions. The specific location of commodity production and resource use is determined by the water quality model and projections of the amount and quality of irrigation return flows result. Alternative BMP's can be simulated in the appropriate model to determine their impacts both in physical and economic terms. Each model will be explained below in more detail.

Commodity Demand

The process begins with the specification of commodity demand from the projections model. In order to accurately portray changes in the irrigated crop economy of the valley over time, projections of changes in technological coefficients, yields and the demands for crop commodities facing the region are needed (see Figure 2). Particularly crucial to the determination of resource problem effects is the projection of commodity demands. While there exist a wide variety of projection approaches varying with the quantitative techniques employed, the area of interest and their ultimate application, the basic approach employed is that reported by King, Carter and Dudek (1977).

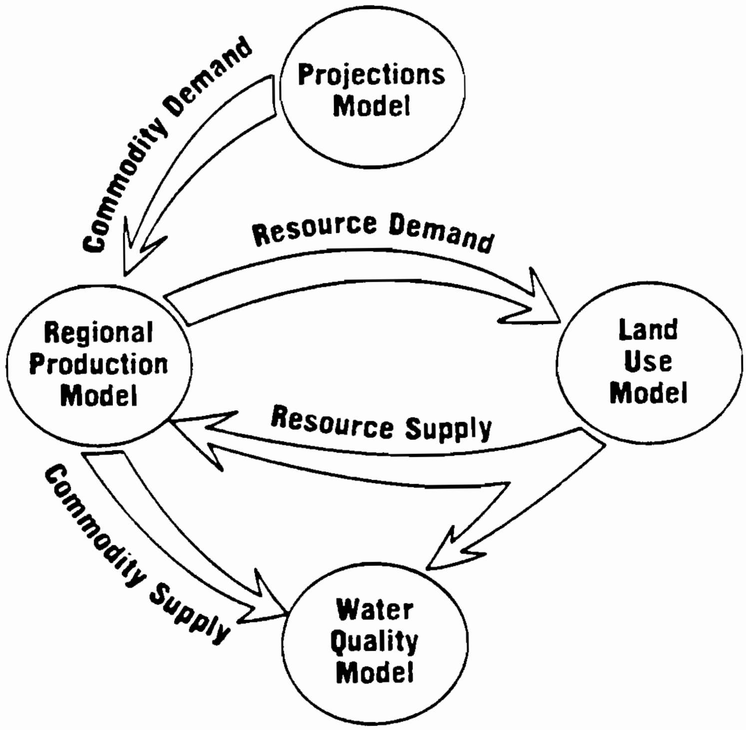


FIGURE 1. INTERACTIONS AMONG COMPONENTS OF THE IRRIGATED AGRICULTURAL ANALYTICAL SYSTEM

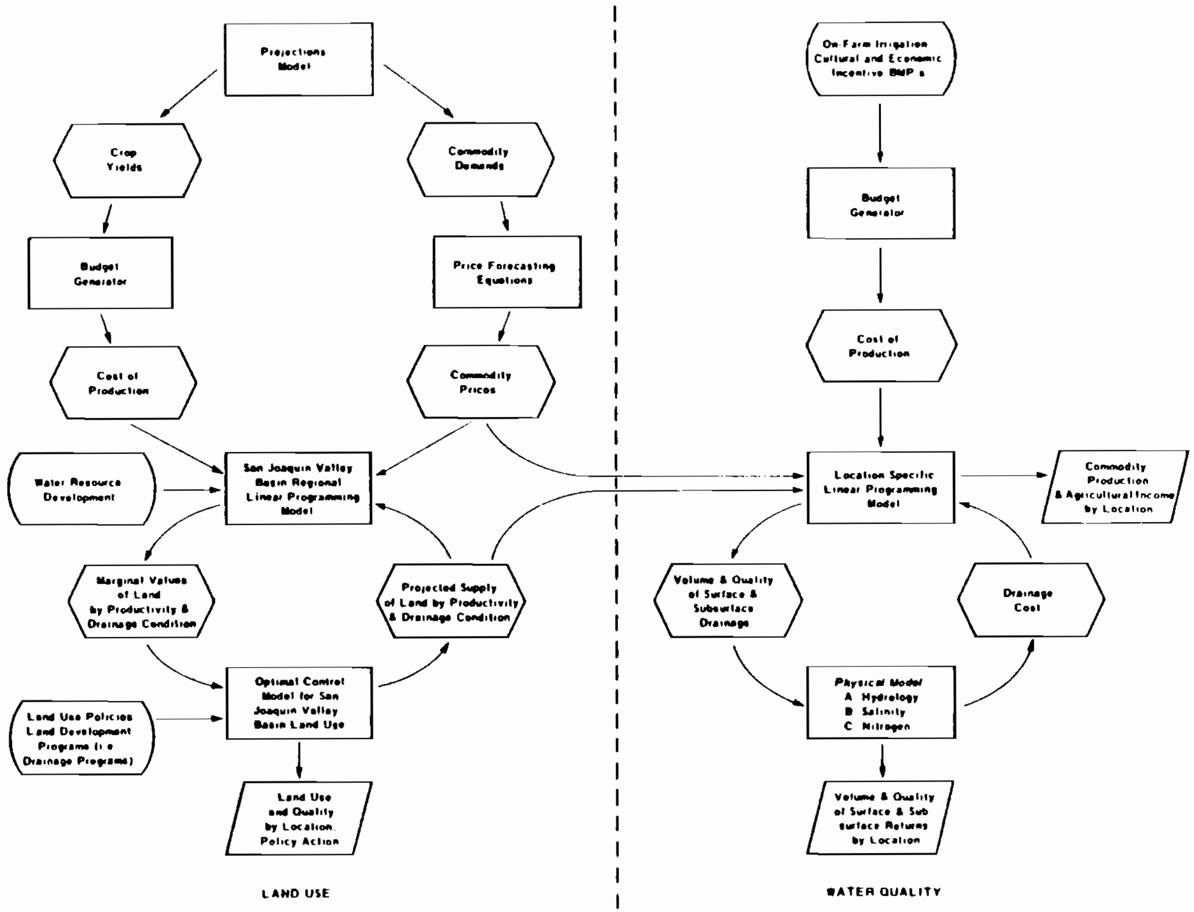


FIGURE 2. INTEGRATED LAND USE, WATER RESOURCE AND WATER QUALITY ANALYTICAL SYSTEM

Essentially, projections of national demand are disaggregated to the state or regional level through the projection of shares of national production as in Dean, et al. (1970). The principal shifters of the domestic component of national demand are population and per capita income (Dean and King, 1970), but trends in per capita consumption, presumed to capture changes in tastes and preferences, may also be included. The addition of projected export-import balances produces estimates of national demand requirements, i.e., given favorable economic conditions, no radical changes in price-cost relationships and the assumption of perfectly elastic supply, these are the commodity quantities which would have to be produced in order to satisfy domestic and export demands. Thus, these projections are conditioned upon the levels specified for key parameters — population, per capita consumption, per capita income, exports and imports. Exploiting the dual relationship between quantities and prices, the quantity demand projections are translated into their price dimension through price forecasting equations (see Figure 2). These prices form one component in the regional linear programming model's objective function.

Commodity Supply

Commodity supply response is estimated by the Regional Production Model using the San Joaquin Valley Basin regional linear programming model. This model has been developed by the United States Department of Agriculture's California River Basin Planning Staff as part of an ongoing analysis of the water and related land resource problems of the area (McKusick, 1974). The linear programming model was developed to analyze the impact of deteriorating drainage conditions on production patterns in the study area and to evaluate program, policy or project measures designed to remedy the problem.

The model is static, i.e., designed to produce solutions for a single specified period. The objective function maximizes net agricultural crop returns in the basin subject to the availability of land resources by soil group, drainage condition and location, the availability of surface and groundwater for irrigation, labor, crop production restrictions and the underlying production technology.

The San Joaquin Valley is divided into two component subbasins, each separately modeled. Each subbasin is subdivided into an east and west side on the basis of differing salinity and drainage characteristics. For 30 principal crops, production activities differ by soil group (of which there are 18), drainage condition (three discrete categroies) and location. Technological coefficients on an annual per acre basis are specified for yields, applied water requirements (which include estimates of the leaching requirement and embody an implied irrigation efficiency), harvest and nonharvest labor, fertilizer (in nitrogen equivalent terms) and gas and diesel fuels.

Resource Demand

The regional linear program is used to generate the demands for land and water resources. These resource demands are the basis of land use valuations within the control model. The optimal control model is used to simulate the dynamics of changes in land use over time for the Valley in reaction to changes in agricultural commodity demands, population, resource productivity and availability and environmental and land use policies. The control model generates optimal land uses over time, the social values resulting from these allocations and the opportunity costs of changing those resource uses.

Resource Supply

The control model, in turn, predicts the availability of land resources for agricultural production which are required in the regional linear program. Projections of commodity demands from the system previously described are introduced into the linear program and a new cycle of model interactions is initiated. In this manner, the process is repeated over the planning horizon with the result being a close approximation of the optimal patterns of land use over time in the Valley.

Resource and commodity supplies are also utilized by the water quality model. This analytical subsystem consists of two specific models sequentially linked to simulate agricultural production and environmental adjustments that occur as a result of an environmental policy (Horner and English, 1976). The first is a linear programming model that derives an optimal cropping pattern, water application technologies, water use and fertilizer in 300 subregions of the valley. The production patterns derived in the linear programming model serve as data inputs to the physical model. The physical model estimates the subsurface vertical and horizontal water movements and surface runoff created by the irrigation activity. The costs for collection and disposal of return flows, the costs for installing tile drainage to relieve high water tables and yield reductions from high water tables are also calculated by the physical model. The change in production costs for subsequent periods are then adjusted in the linear programming model. Solutions from the models are derived annually and are iterated a sufficient number of times to simulate the environmental adjustments from a change in water and land use as a result of alternative control policies. This presents a necessarily brief overview of the integrated methodology. For a more thorough discussion of its underlying theoretical basis see Horner and Dudek (1980).

The Water Quality Model

Since the primary focus of this conference is the nitrate pollution of groundwater and the analysis of management alternatives, it is important to elucidate the precise specification of the water quality model. As previously indicated, the model consists of two location specific components. One is a linear programming model that derives optimal cropping patterns and use of water and fertilizer, subject to physical, institutional, resource and environmental constraints. These results serve as inputs to the second, or physical, model. The physical model is partitioned into three specific interdependent submodels that analyze (1) hydrology, (2) salinity and (3) nitrogen (see Figure 2). These submodels estimate the effects on the quantity and quality of groundwater as a result of the normative cropping patterns derived in the LP portion of the model. This recursive procedure represents one year and the procedure is repeated as often as necessary to simulate the agricultural production and environmental conditions that occur during the implementation of a policy. Alternative policies to achieve water quality goals are analyzed and compared on the basis of the cost of their attainment. Cost is defined as a reduction in farm income, reduction in commodity production and/or reduction in the income of agriculturally related sectors in the region.

The area selected for study is the San Joaquin Valley Basin of California. The basin is topographically bounded by the Stanislaus River Watershed in the north, the Tehachapi mountains in the south, the Sierra Nevadas on the east and the Coast Range to the west. Its total area comprises 28,372 square miles. Hydrologically, the basin is divided into the San Joaquin and Tulare subbasins on the basis of essentially different drainage characteristics, i.e., the Tulare is a closed basin. Since average rainfall is only 11 inches at Modesto and declines to 6 inches at Bakersfield, the climate is arid. Dominant aspects of the geology of the area include the existence of extensive groundwater aquifers with an estimated capacity of 80 million acre-feet (Johnson, et al., 1975) and an impermeable Corcoran clay layer which confines the lower water-bearing zone in parts of the basin.

Prominent among the resource problems in the study area are salinity and drainage. Of the 4.68 million acres irrigated in the San Joaquin Valley Basin in 1972, it is estimated that 1.5 million acres suffered from drainage problems. This figure is projected to rise to 1.75 million by 1985. It is estimated that 1.33 million acres in the basin applied irrigation water with greater than 1,000 ppm of total dissolved solids (USDA California River Basin Planning Staff, 1976). These physical phenomena of drainage and salinity are inextricably linked in arid region irrigated agriculture. Without management of these problems, the region can expect high water tables and contamination of groundwater by salts and nitrogen.

In the model, the area is divided into approximately 300 cells that are homogeneous with respect to soil type, productivity, drainage condition, depth to the impermeable layer, elevation and water supply. Cells vary in size from about 1,500 to 70,000 acres. These homogenous cells provide a common basis for the physical and economic model.

The Economic Model

The economic model is used to determine the optimal allocation of resources and to compare results from alternative policies designed to achieve nonpoint source control. Linear programming is used for this portion of the analysis. Production costs for 1974 are used to derive objective function values which with corresponding commodity price levels are used to determine net revenues. Groundwater and water supplied by irrigation districts are priced externally from the production activities. Water prices reflect each district's projected rates including groundwater pumping costs.

The Hydrologic Model

The hydrologic model is used to estimate, on an annual basis, groundwater depth, tile drainage flow rates and associated salinity and NO_3-N content on the basis of projected rainfall, streamflow, irrigation water use, cropping patterns and drainage systems. The assumptions of the model are: (1) irrigation water and rainfall infiltration are the primary sources of recharge to the unconfined aquifer, (2) impermeable layers are assumed to exist when the restricting layer has a permeability of one-tenth or less of the surrounding soil and (3) the soil and water conditions of each cell are homogeneous. A typical cell grid and the corresponding spatial relationships are illustrated in Figure 3.

Cell i has water flowing within the cell and between adjoining cells as dipicted in Figure 4. The change in the amount of water in cell i is estimated by utilizing the mass conservation principle:

$$\Delta V_i = (\sum_j Q_{ij} + Q_{fi} + Q_{di} - Q_{ei})\Delta t \qquad (1)$$

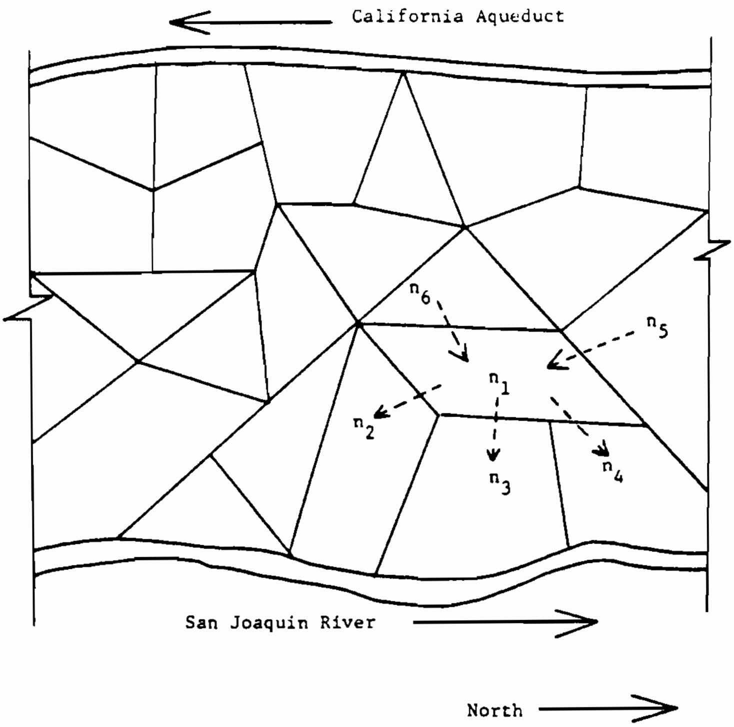
where $\Delta V_i = \Delta V_{\text{unsaturated zone}} + \Delta V_{\text{saturated zone}}$

Δt = Time period

and the other variables are as defined in Figure 4. Equation (1) can be rewritten as:

$$\Delta V_i = A_i Z_i \Delta\theta_i + A_i(\Delta h_i) \qquad (2)$$

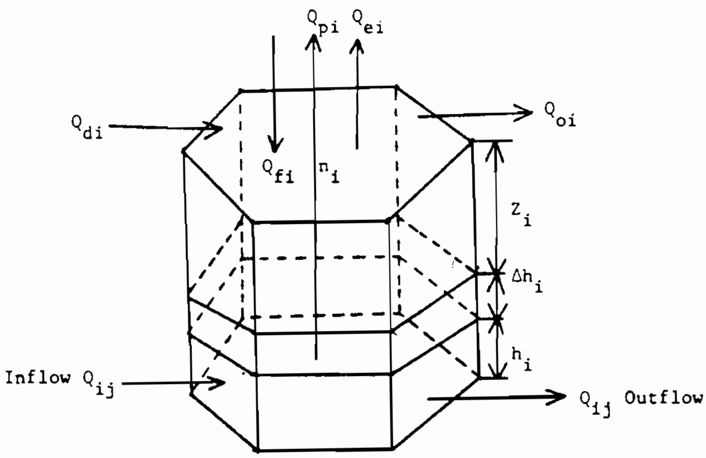
where $\Delta\theta_i$ = the change in the soil moisture content of the unsaturated zone



n_i = Cell i, i = 1,2,...,N.

N = Total cells in Subbasin

FIGURE 3. CELL SPATIAL RELATIONSHIPS IN THE SAN JOAQUIN LOCATION-SPECIFIC PHYSICAL MODEL



Where: Q_{fi} = infiltration rate at n_i.

Q_{ei} = evapotranspiration rate at n_i.

Q_{pi} = pumping rate of subsurface drainage.

Q_{di} = rate of irrigation water applied.

Q_{oi} = surface outflow (positive if rainfall rate is greater than infiltration rate).

Z_i = depth of unsaturated zone.

h_i = initial water table depth.

Δh_i = change in water table depth in time period t.

Q_{ij} = subsurface flow between n_i and n_j.

FIGURE 4. THREE DIMENSIONAL VIEW OF CELL n_i

A_i = the area of cell i

The change in the volume of water in the unsaturated zone is determined by:

$$\Delta V_{unsaturated} = A_i Z_i \Delta\theta_i = (Q_{fi} - Q_{ei} - Q_{ui})\Delta t \tag{3}$$

where Q_{ui} = rate of excess water percolating into the groundwater reservoir.

The change in the volume of water in the saturated zone is determined by:

$$\Delta V_{saturated} = A_i \Delta h_i = (\sum_j Q_{ij} + Q_{ui} - Q_{pi})\Delta t \tag{4}$$

Then by setting:

$$a_{ij} = \frac{A_i\ (\alpha_i - \theta_i^*)}{\Delta t} + \sum_j X_{ij} \tag{5}$$

where α_i = soil porosity of cell i

θ_i^* = the unknown soil moisture content of cell i

$$a_{ij} = X_{ij}, \quad \text{for } j = 1, \ldots, n \tag{6}$$

Now, C_i, the unknown equilibrium water level, may be expressed as:

$$C_i = \frac{A_i(\alpha_i - \theta_i^*)}{\Delta t} h_i - \sum_j X_{ij} h_i + \sum_j X_{ij} h_j + Q_{ni} - Q_{pi} \tag{7}$$

where $X_{ij} = \frac{K_{ij}\ D_{ij}\ W_{ij}}{2X_{ij}}$ is the rate of water movement from n_i to n_j or n_j to n_i

Then the linear equation for cell i is:

$$a_{ii}\ h_i^* + \sum_j a_{ii} h_j^* = C_i, \quad \text{for } i = 1, \ldots, n \tag{8}$$

By writing one equation of the form of (8) for each cell, a set of simultaneous linear equations can be formed. This set of equations can be represented in matrix form as:

$$[A]\ [H^*] = [C] \tag{9}$$

The Guassian elimination technique is then used in solving this problem.

Tile drainage effluent volume for each cell can then be obtained by:

$$Q_{pi} = A_i q_i \Delta t \tag{10}$$

where q_i = a drainage rate. Also, the water table in a cell will change according to:

$$h_i^{t+\Delta t} = h_i^* - q_i \Delta t \tag{11}$$

where $h_i^{t+\Delta t}$ will be used as the value for h_i in the next time period.

Salinity Model

The hydrologic model must be solved before the salinity model, as it is necessary to know the groundwater flow rates between cells and the drainage effluent volume from each cell. The change in salinity content is then estimated by using the salt mass conservation principle:

$$\frac{d(u_i S_i)}{dt} = \sum_j S_{ij}\, Q_{ij} + S_{fi} - S_{pi}\, Q_{pi} + S_{di}\, Q_{di} \tag{12}$$

where S_i = the salt concentration in ppm.

The salt balance for each cell is described by:

$$b_{ij}\ S_i^* = R_i \tag{13}$$

where b_{ij} = the salt concentrations of cell i and all cells adjacent to it.

S_i^* = unknown

$$R_i = \frac{V_{it}}{V_{it+1}} S_{it} + \frac{\Delta t}{V_{it+1}} (S_{fit+1} Q_{fit+1} + S_{dit+1} Q_{dit+1})$$

The set of n equations can be developed for the Basin as:

$$[B] [S^*] = [R] \tag{14}$$

which can be solved by the technique used to solve the hydrologic model. Once the salt concentration S_i^* is known, the salt concentration of the drainage is:

$$TDS = S_i^* Q_{pi} \Delta t, \qquad i=1, \ldots, n \tag{15}$$

Nitrogen Model

The predicted movement of NO_3-N from the root zone to the unsaturated zone to the groundwater reservoir and then into the tile line system will follow the approach developed by Pratt and Andriano (1973). Nitrogen in the soil primarily originates from inorganic fertilizers, organic matter and irrigation water (N_a). It is mainly removed by crops (N_c), drainage water (N_d) and denitrification (N_u). A positive N_u value means that nitrogen has moved into the atmosphere by denitrification or volatilization, whereas a negative value indicates other nitrogen inputs were not considered during the nitrogen mass balance process. The following conditions are assumed in this model:

1. The nitrogen consumed by the harvested crop and removed from the soil system is approximately equivalent to the nitrogen content in the primary and secondary harvested crop.
2. The crop's nitrogen consumption is linearly proportional to the yields produced by varying the amount of nitrogen fertilizer applied.
3. Excess nitrogen (N_e) occurs as NO_3-N in the water percolating from the unsaturated zone to the saturated zone.

4. The values of N_e and percolating water (Q_u) are determined independently of each other, but are interdependent in determining the amount of nitrogen leached through the root zone (N_p), i.e., $N_p = N_e$ as $Q_u > 0$ and $N_p = 0$ as $Q_u = 0$.

The nitrate-nitrogen model is highly dependent on the output of the hydrologic model. In order to meet these basic assumptions, the model must treat each cell in two parts, i.e., the upper unsaturated zone and the lower saturated groundwater reservoir. The conditions in the upper unsaturated zone determine the amount of downward percolating water (Q_{ui}) to the groundwater reservoir and its associated nitrogen (N_p) content. Once Q_{ui} and N_p are determined, the flow rate of nitrogen between neighboring cells can be found by solving the system of simultaneous equations representing the basin. Each equation describes the nitrogen mass balance of each cell.

$$d_{ij}n^{*}_{ai} = W_i \tag{16}$$

where d_{ij} = the nitrogen concentrations for n_i and surrounding cells

n^{*}_{ai} = unknown

$W_i = N_{it+1}(1 + H_iQ_{pit+1}) - H_i \sum_j Q_{ijt+1} N_{ijt+1}$

For the basin, this relation can be expressed in matrix forms as:

$$[D]\ [N^{*}_{a}] = [W] \tag{17}$$

Equation (17) is then solved and the nitrogen concentration of the drainage water is given by:

$$G = n^{*}_{ai}Q_{pi} \quad i=1,\ldots,n$$

These three sets of equations (9), (14) and (17) constitute the physical system component of the water quality model (see Figure 2). The economic component of this model is a set of location specific linear programming models of irrigated agricultural production. These components, recursively iterating, provide the specific mechanism by which alternative water quality management policies may be evaluated.

A SYSTEMS ANALYSIS OF ALTERNATIVE NONPOINT SOURCE CONTROLS

Four alternative irrigation water quality policies were analyzed for the San Joaquin Valley. These policies are not the complete set of feasible alternatives for the reduction of environmental impacts associated with irrigation return flows. However, these policies have been suggested as possible programs to reduce water use and minimize the impact on receiving water bodies. Budget constraints on the project made it impossible to evaluate other policies that might focus on economic incentives affecting water and fertilizer use and/or changes in the institutional structure of water resource allocation. Examples of those policies will be explained and discussed in the final section of this paper.

Policy Definitions and Assumptions

The first alternative is to do nothing, i.e., what are the future effects on water quality of continuing existing resource use practices. This policy option was simulated in the analytical system by assuming that cropping patterns, agricultural costs and prices and technologies will remain constant in the future. This future state of the economy is not likely to occur, but these restrictions are imposed on the analytical system in order to isolate the effects of changing (or not changing) the method or timing of irrigation water applications.

The second policy analyzed was the imposition of a mandatory tailwater recycle program. This water conserving practice involves the capture of surface runoff from irrigations and their return to the head of the field for use as a source of irrigation water. Tailwater could be recycled in most areas of the San Joaquin Valley for approximately $11.00 per acre-foot in 1974 and was being implemented in those areas that were experiencing irrigation water shortages or high water prices (Kinney et al., 1977). Typical tailwater recycle systems can deliver approximately 15 to 25 percent of a total water application to the head of the field depending on the soil type and the degree of monitoring of the irrigation application.

The imposition of an irrigation scheduling service was analyzed as a third water quality improvement option. Irrigation scheduling is a water management technique for systematically determining the timing and amount of each irrigation in individual fields. This technique is presently being used by U.S. government agencies, corporations and individual farmers in the Western U.S. to assist in reducing water use and irrigation return flows (English, et al., 1980). It was assumed that scheduling will improve application efficiency by about 5 percent at a cost of approximately $5.00 per acre.

The fourth policy was to simultaneously require that all tailwater be recycled and that all irrigations be scheduled within the region. This policy would represent a concerted effort to control on-farm irrigation practices. All of the assumptions that were in effect for the first three policies apply to the combined tailwater recycle and scheduling option.

Temporal Effects of Water Quality Policies

The analytical system is designed to simulate public policies affecting the use of land and water resources and to display the projected variable changes on a temporal and spatial basis. To illustrate the temporal dimension of the system, changes in the shallow water table, quantities of subsurface drainage, quantities of surface runoff, salt load in return flows and nitrogen in return flows are reported for the two hydrologic subbasins contained within the study area.

Shallow Water Table

The area that will require subsurface drainage in the San Joaquin Subbasin was projected to increase from about 450,000 acres in 1974 to 750,000 acres by 1990 if no policy action is initiated (Figure 5). If a scheduling program is implemented, the area requiring drainage is projected to be approximately equivalent to that that would result under the continuation of existing practices. However, substantial increases in the area with shallow groundwater tables is projected under programs requiring tailwater recycling or the combination of recycling tailwater and irrigation scheduling. This is in contrast to results in the Tulare Subbasin which indicate that the shallow groundwater table will not differ substantially under the four water quality policies (Figure 6).

These apparently divergent results for the two subbasins can be attributed to the source of irrigation water supplies and how the four policies change water use and pumping patterns. The practice of recycling tailwater substantially reduces the pumping of groundwater, which in the San Joaquin Subbasin is the marginal water supply due to higher cost and lower quality (Table 1). Reduced pumping from the unconfined aquifers results in higher groundwater tables without a reduction in deep percolation beyond the root zone. Alternatively, an irrigation scheduling program does reduce deep percolation with a small reduction (6.6%) in water use. In the Tulare Subbasin, a higher proportion of groundwater is pumped from confined aquifers.

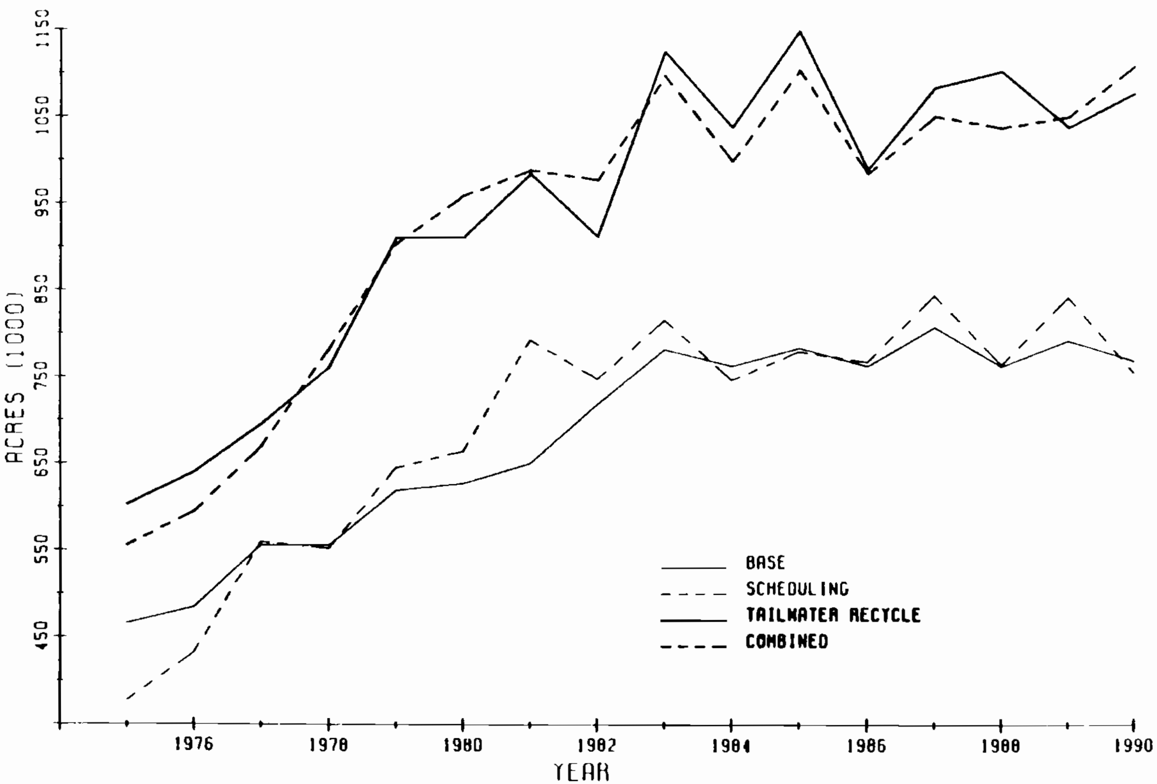


FIGURE 5.
AREA WITH A WATER TABLE OF 5 FEET OR LESS, SAN JOAQUIN SUBBASIN, 1975-90.

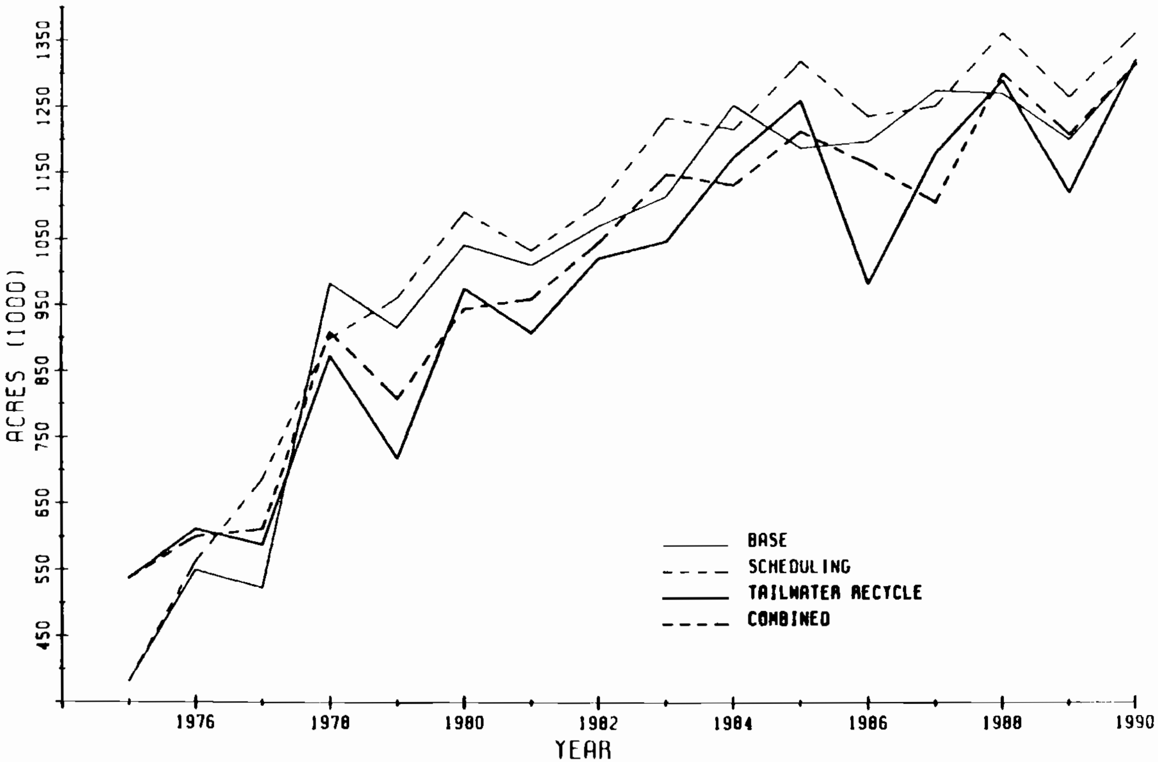


FIGURE 6.

AREA WITH A WATER TABLE OF 5 FEET OR LESS, TULARE SUBBASIN, 1975-90

Table 1. Projected Water Use by Source for Alternative Water Quality Policies, San Joaquin Subbasin, 1990

	Groundwater Use			Surface Water Use	Total Water Use
	Confined	Unconfined	Total		
	(1,000 Acre-Feet)				
Policy					
Base	1,291	1,729	3,020	3,419	6,439
Tailwater Recycle	951	1,210	2,161	3,350	5,511
Scheduling	1,090	1,526	2,616	3,396	6,012
Combination	900	1,113	2,013	3,321	5,334

Subsurface Drainage

The quantity of subsurface drainage water resulting from the increasing area with shallow water tables was projected to increase approximately 100 percent from 1974 to 1990 in both subbasins. These results were obtained when existing practices were continued into the future and when the irrigation scheduling program was employed (Figures 7 and 8). This contrasts sharply with the results obtained if the tailwater recycle policy is implemented. An approximate eightfold increase in drainage effluent volume is projected in the San Joaquin Subbasin and a fourfold increase in the Tulare Subbasin with recycling. These results are expected since tailwater recycling reduces pumping from the unconfined aquifer which consequently requires that more groundwater be removed by subsurface drainage systems.

Surface Runoff

The quantity of surface runoff projected for the two subbasins did not change substantially over time except in 1978 and 1982 when greater than normal rainfall was simulated to occur (Figures 9 and 10). By reducing surface runoff from irrigated agricultural operations, a tailwater recycle program can be effective in reducing sediment, pesticide and herbicide loadings to surface water bodies. Most of the surface runoff projected under the recycling program is composed of natural flows from precipitation that occurs during the winter months.

Salt Load

Surface return flow salt load is defined as the salt contained in natural runoff, surface irrigation return flows (tailwater) and subsurface drainage. The salt load projected for the tailwater recycle program was

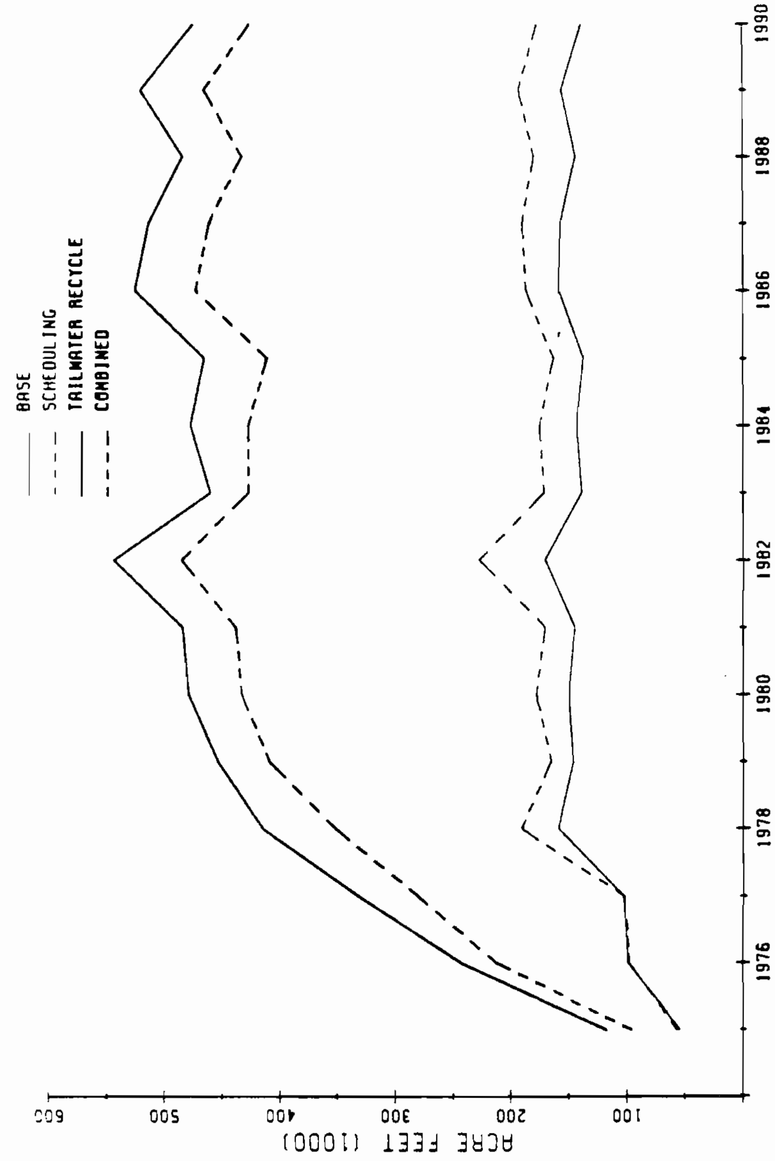


FIGURE 7.

QUANTITY OF SUBSURFACE DRAINAGE, SAN JOAQUIN SUBBASIN, 1975-90

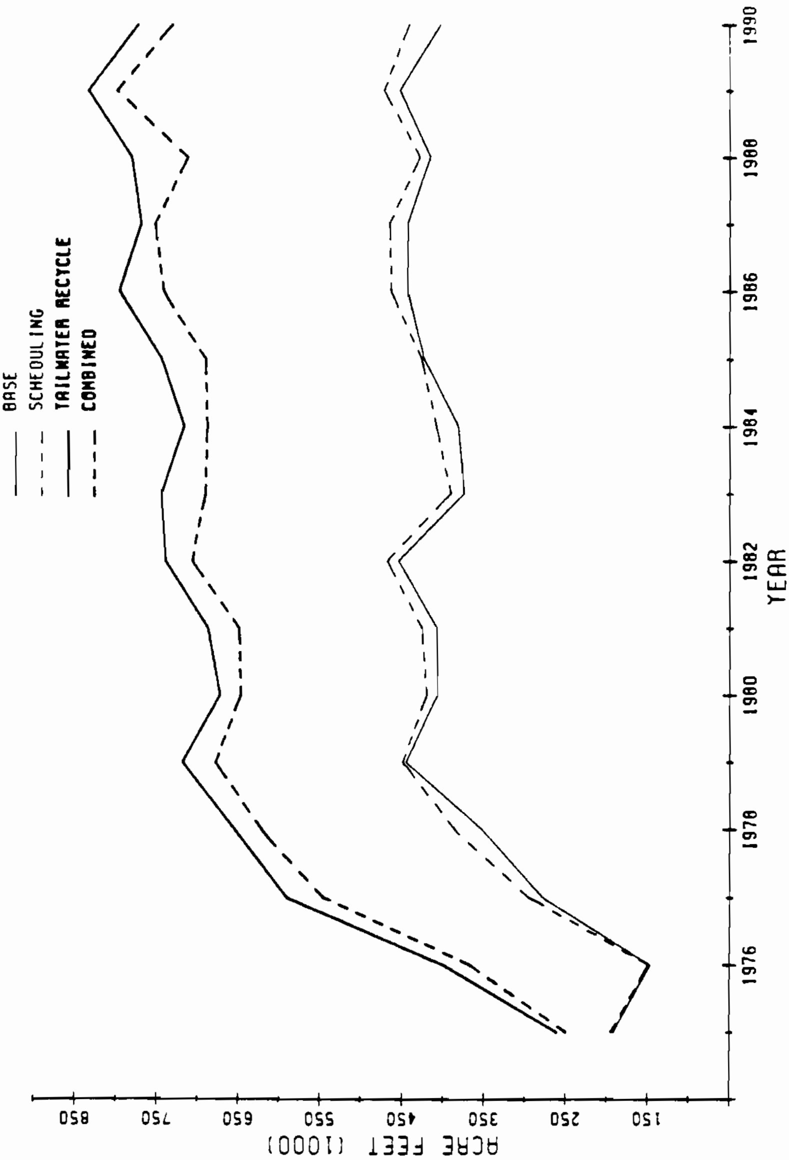


FIGURE 8.

QUANTITY OF SUBSURFACE DRAINAGE, TULARE SUBBASIN, 1975-90

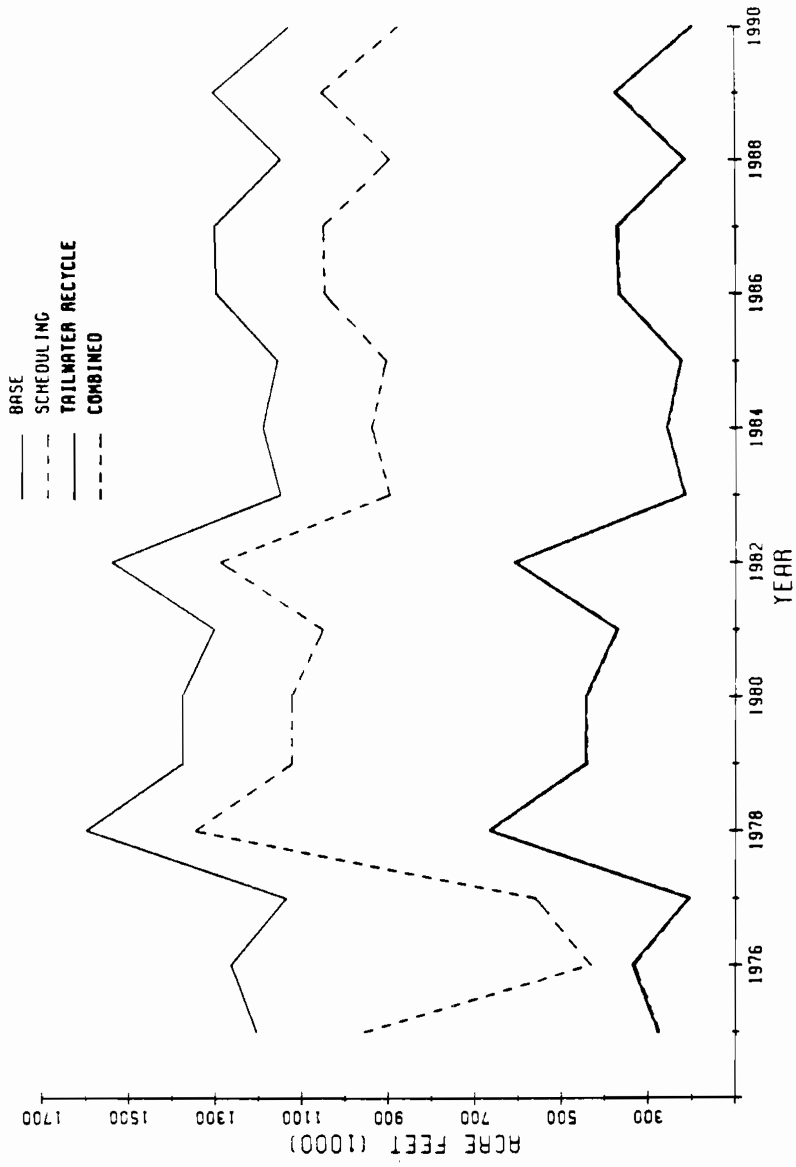


FIGURE 9.

QUANTITY OF SURFACE RUNOFF, SAN JOAQUIN, SUBBASIN, 1975-90

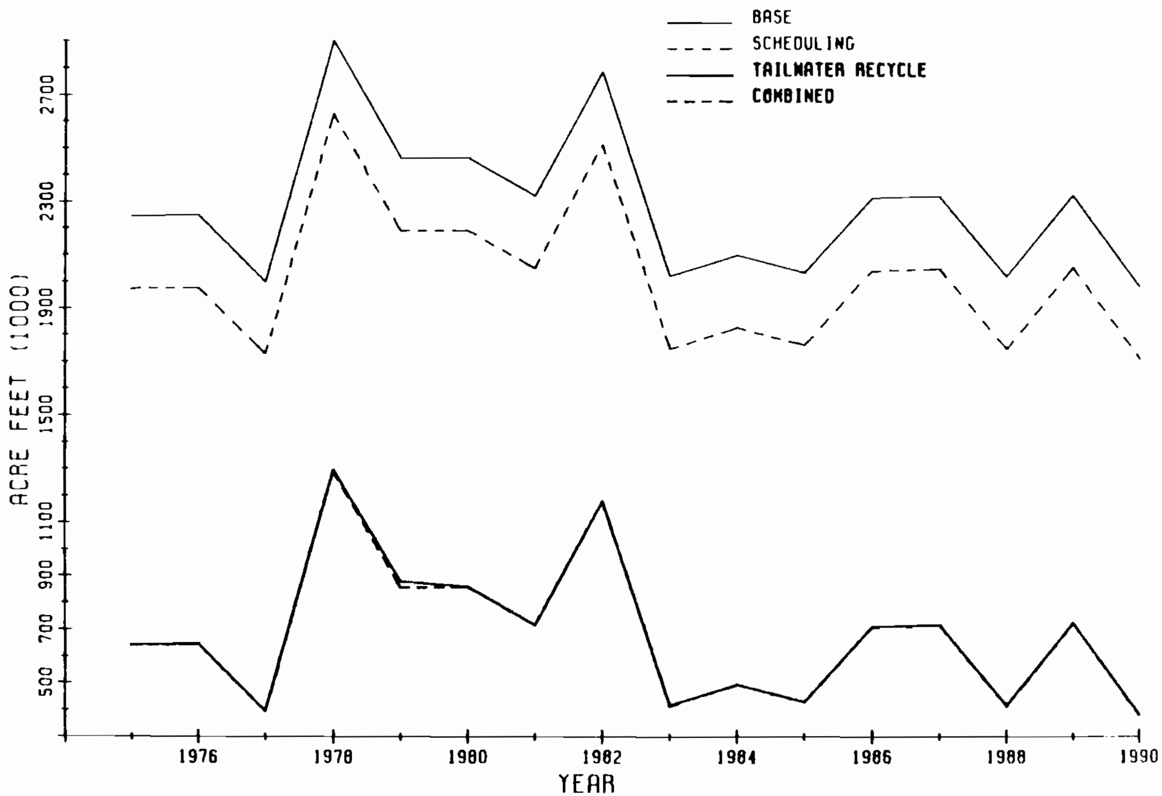


FIGURE 10.

QUANTITY OF SURFACE RUNOFF, TULARE SUBBASIN, 1975-90 marginally less than that under the base condition (existing practices) or the scheduling policy (Figures 11 and 12). This result may seem counterintuitive given that the tailwater recycling program achieved the least reduction in subsurface drainage among the policy alternatives analyzed. However, this seeming anomaly can be explained by examining the relative magnitudes of surface and subsurface drainage displayed in Table 2. Comparing existing practices (base) and the tailwater recycling policy, one finds surface return flows reduced from a projected 3.2 million acre-feet in 1990 to a 0.4 million acre-feet residual which is essentially rainwater. This dramatic reduction more than offsets the relatively modest increases in projected subsurface drainage (0.5 million acre-feet to 1.225 million acre-feet). The net result is an overall reduction in salt load.

Nitrogen Load

The bulk of the nitrogen load in return flows originates from excess nitrogen fertilizer in the root zone and natural deposits that are leached by deep percolation. Nitrogen load, therefore, is closely correlated with the pattern of subsurface drainage results (Figures 7 and 8). The quantities of return flow nitrogen load over time is depicted in Figures 13 and 14. Variations in the nitrogen load and the amount of subsurface drainage are caused by differences in cropping patterns and the amount of water leaching through the root zone. For example, irrigation efficiencies in the Tulare Subbasin are higher than those in the San Joaquin Subbasin due to reduced water supplies and higher water costs. These differences result in increases in the concentration of nitrogen in deep percolation in the Tulare region, but a smaller average per acre nitrogen load. Consequently, the impact

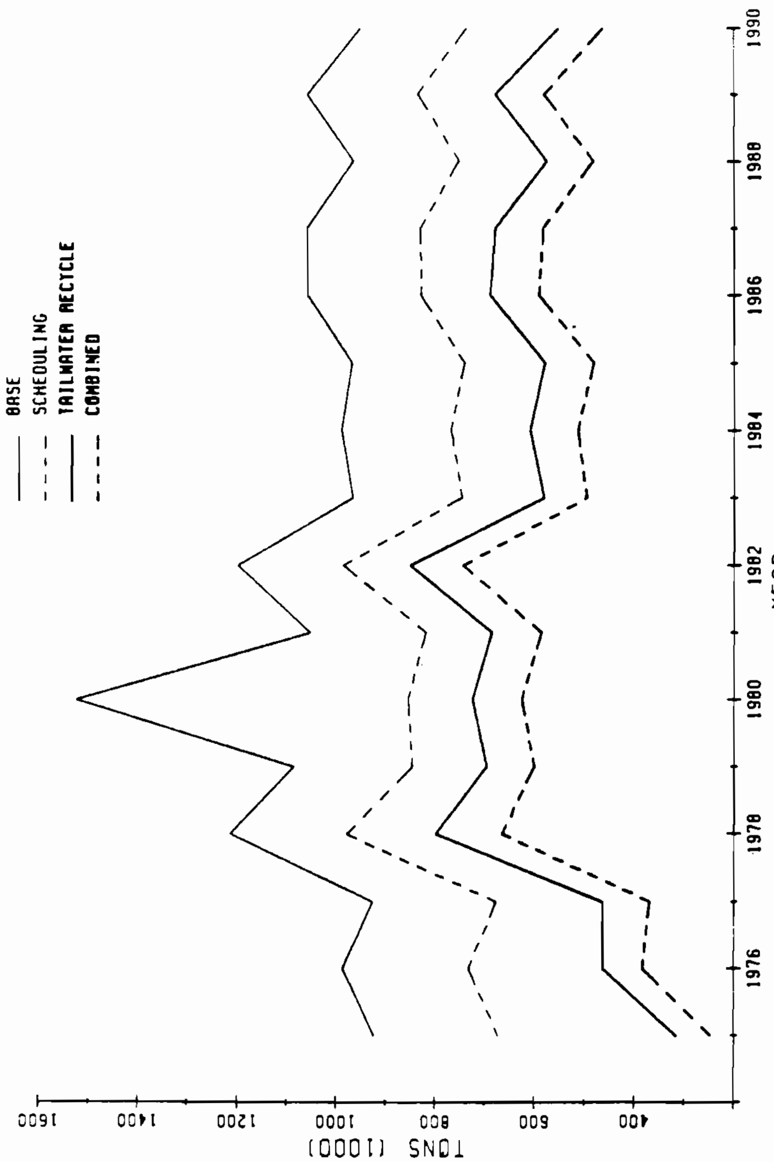


FIGURE 11.

SURFACE RETURN FLOW SALT LOAD, SAN JOAQUIN SUBBASIN, 1975-90

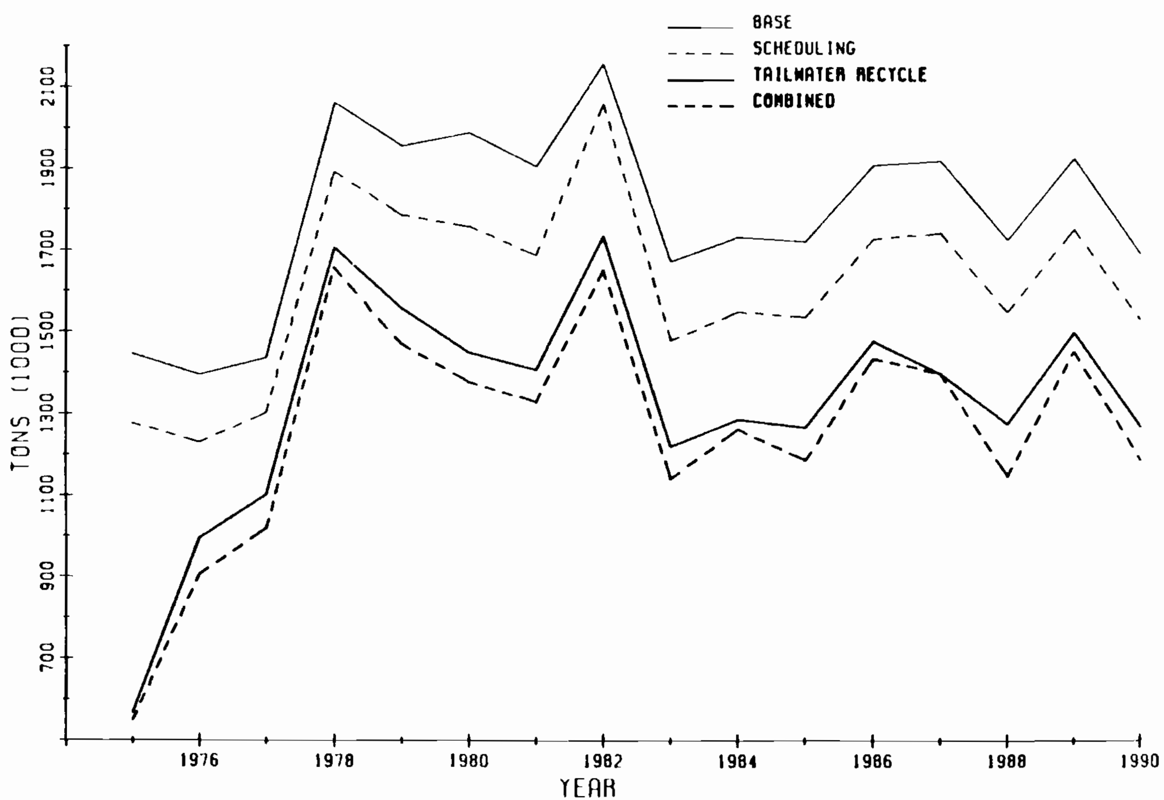


FIGURE 12.

SURFACE RETURN FLOW SALT LOAD, TULARE SUBBASIN, 1975-90

Table 2. Projected Irrigation Return Flows for Alternative Water Quality Policies, 1990

	Subsurface			Surface			Total
	San Joaquin	Tulare	Total	San Joaquin	Tulare	Total	
	(1,000 Acre-Feet)						
Policy							
Base	150	350	500	1,200	2,000	3,200	3,700
Tailwater Recycle	475	750	1,225	150	250	400	1,625

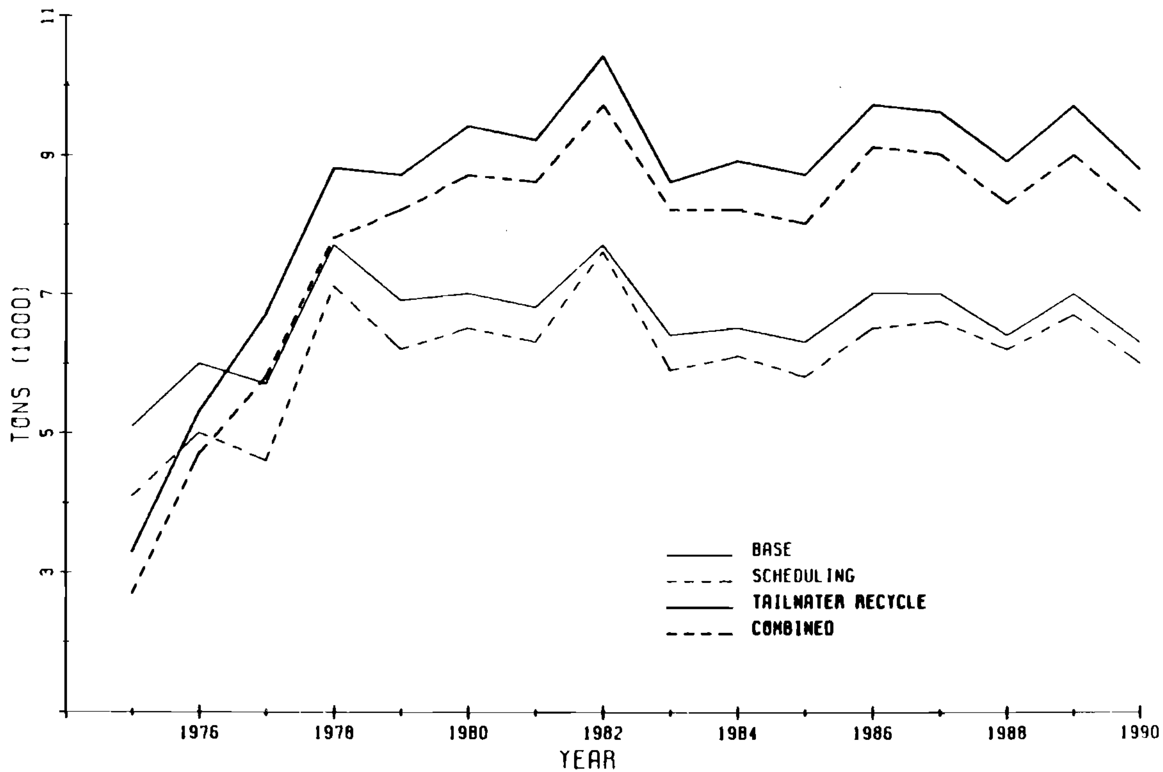


FIGURE 13.
SURFACE RETURN FLOW NITROGEN LOAD, SAN JOAQUIN SUBBASIN, 1975-90

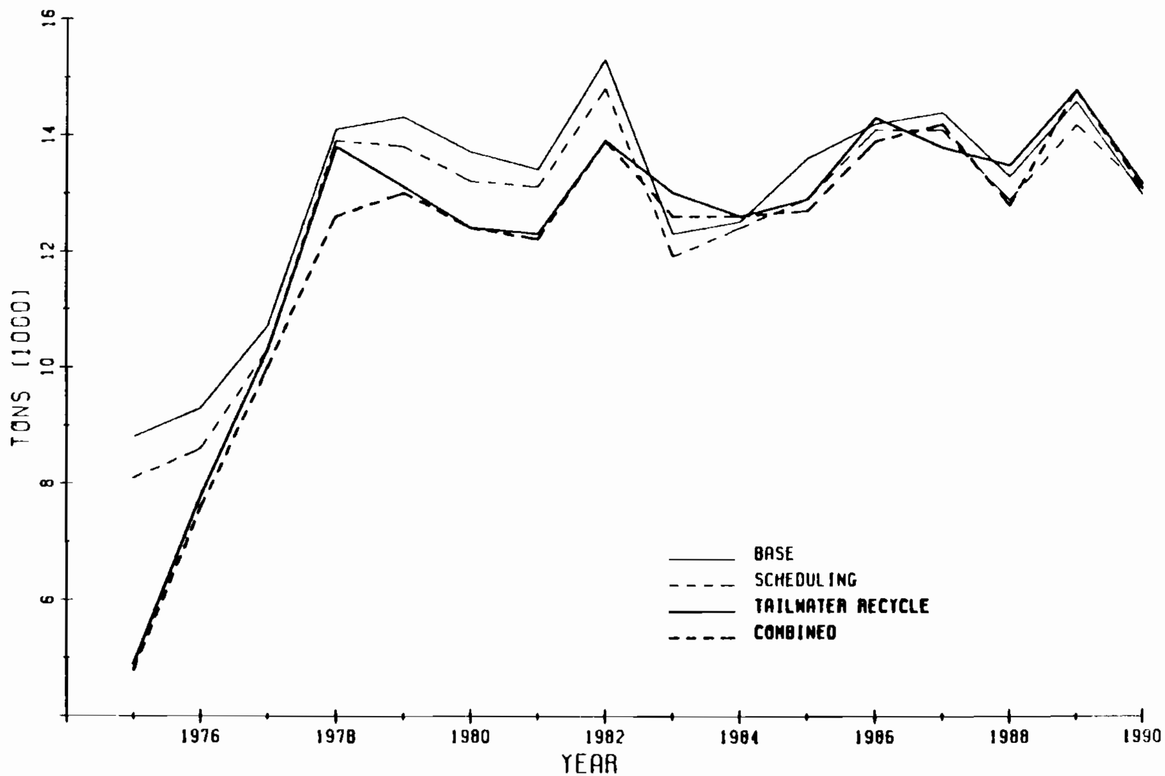


FIGURE 14.

SURFACE RETURN FLOW NITROGEN LOAD, TULARE SUBBASIN, 1975-90 of tailwater recycling on the return flow nitrogen load in the Tulare Subbasin is not as great as in the San Joaquin Subbasin. However, these results are aggregate, being reported at the regional level in order to illustrate the changes that occur over time. Nitrogen concentrations by location are of greater importance when municipal water supplies are being taken from these same groundwater aquifers. Therefore, a spatial display of nitrogen concentrations exceeding the drinking water standard in California has policy significance.

Spatial Effects of Water Quality Policies

In California, water supplies used for human consumption may not contain more than 10 ppm of nitrate nitrogen. This is twice the recommended United States Public Health Service standard for NO_3-N concentrations in drinking water. In the San Joaquin Subbasin, a substantial area has groundwater supplies exceeding the standard. It should be noted that these concentrations were sampled from irrigation wells located close to the center of each cell. These wells were pumping from the unconfined aquifer and are assumed to be representative of groundwater conditions within the homogeneous planning unit. Wells of this type are normally used for irrigation, not domestic purposes. However, the 10 ppm standard for NO_3-N serves to illustrate the spatial distribution of nitrogen concentrations and how they are affected by alternative water quality management policies.

High Groundwater Nitrogen Concentrations

In 1974, most of the irrigated area west of the San Joaquin River had groundwater nitrogen concentrations that exceeded 10 ppm (Figure 15). Most of the soils in this area originated from alluvial fans from the Coastal Mountain Range located on the Western edge of the study area. The parent material for these soils is extremely high in natural nitrogen deposits.

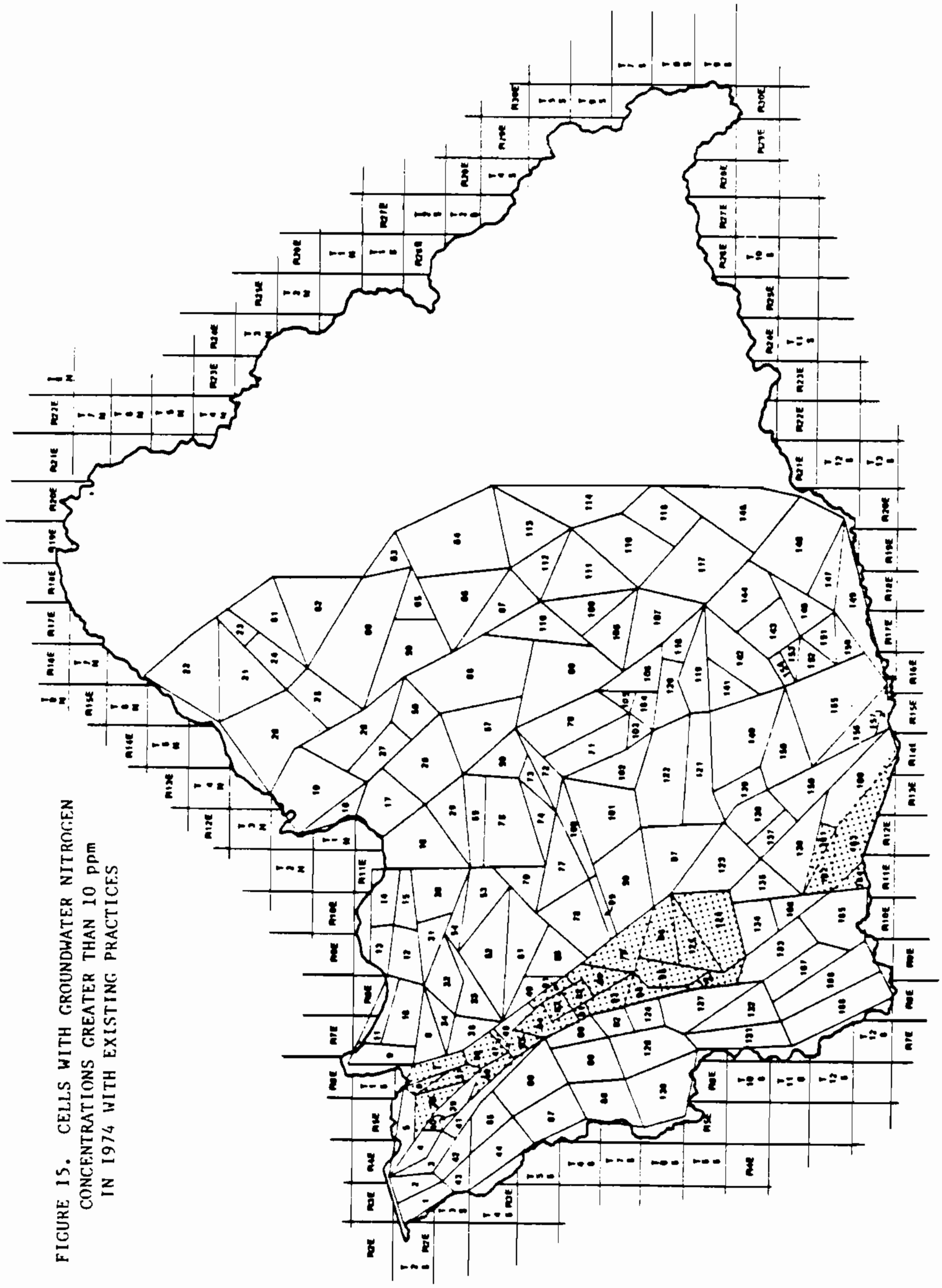


FIGURE 15. CELLS WITH GROUNDWATER NITROGEN CONCENTRATIONS GREATER THAN 10 ppm IN 1974 WITH EXISTING PRACTICES

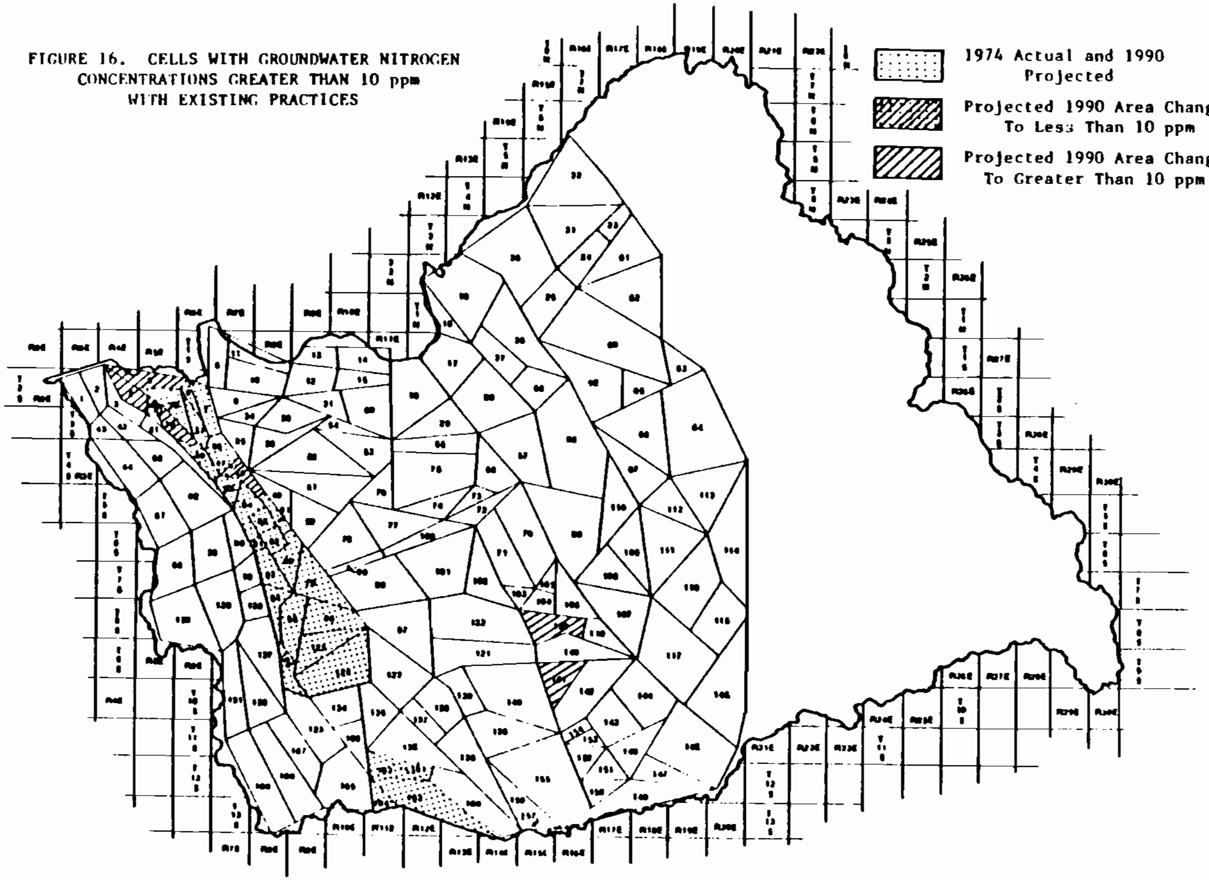
Five additional cells have groundwater nitrogen concentrations that exceed 10 ppm if current conditions are projected to continue through 1990 (Figure 16). Most of this increase can be attributed to the leaching of excess nitrogen into groundwater and the concentrating effect of using these same waters for irrigation.

Imposing a tailwater recycle program in the San Joaquin Subbasin resulted in just one additional cell exceeding the 10 ppm standard in 1990 (Figure 17). Under this nonpoint management alternative, recycled tailwater is substituted for groundwater with generally higher nitrogen concentration. Since deep percolation is roughly equivalent under tailwater recycling and continued current practices, one would expect nitrogen concentrations to be lower.

Irrigation scheduling was not as effective in keeping nitrogen concentrations from increasing as the tailwater recycle option. Scheduling techniques improve water application efficiency and therefore reduce deep percolation. This reduced quantity of water leaching the soil profile results in higher nitrogen concentrations in return flows to the unconfined aquifer. By 1990, the scheduling option resulted in four additional cells exceeding the nitrogen standard. This was just one less than was projected under base conditions (Figure 18).

Only two additional cells were projected to exceed the 10 ppm standard by 1990 under the combined policies of tailwater recycling and irrigation scheduling (Figure 19). This was almost as effective as the tailwater recyle option alone. It is critical to note that each policy should not be evaluated solely on the basis of performance with respect to a single parameter even though it may represent a dominant policy objective. For example, it may be important to locate areas where groundwater nitrogen

FIGURE 16. CELLS WITH GROUNDWATER NITROGEN CONCENTRATIONS GREATER THAN 10 ppm WITH EXISTING PRACTICES



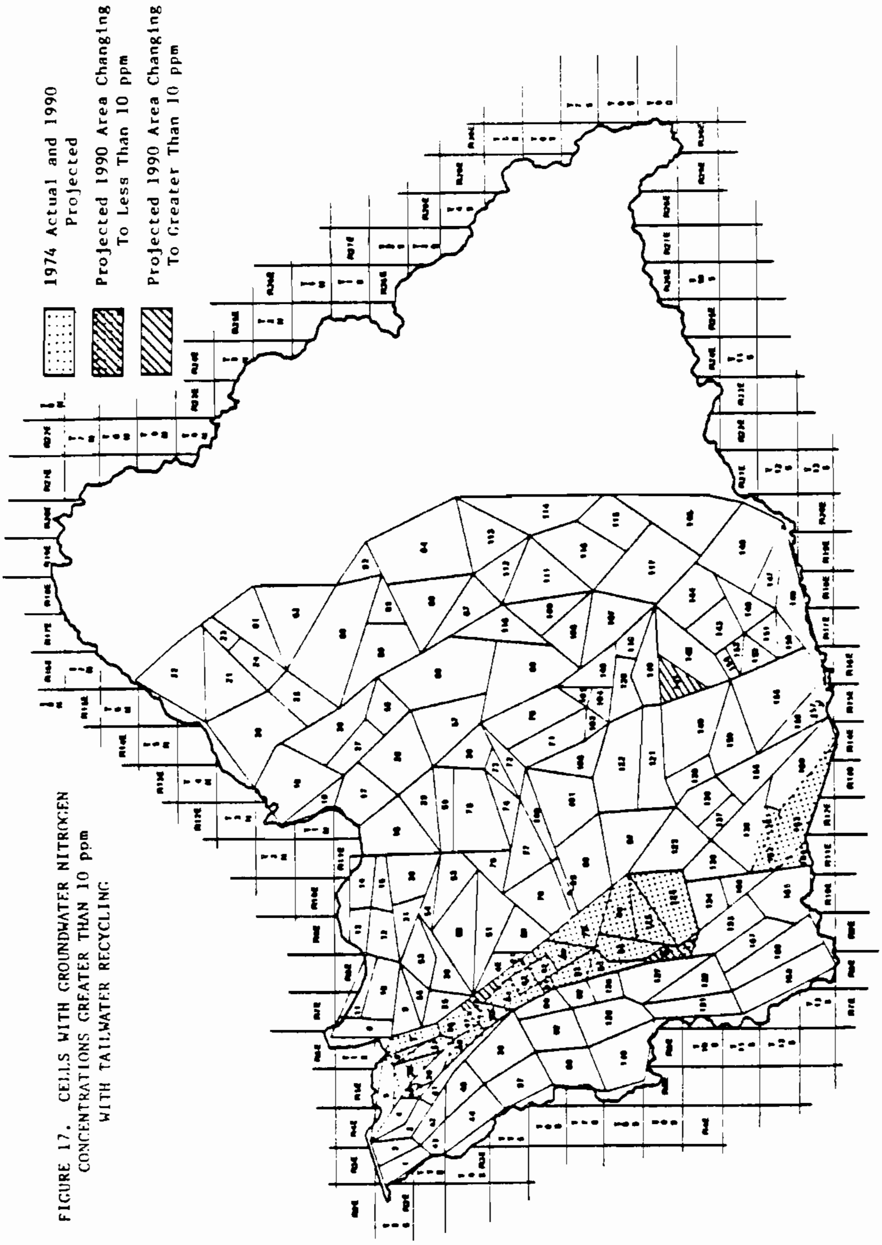


FIGURE 17. CELLS WITH GROUNDWATER NITROGEN CONCENTRATIONS GREATER THAN 10 ppm WITH TAILWATER RECYCLING

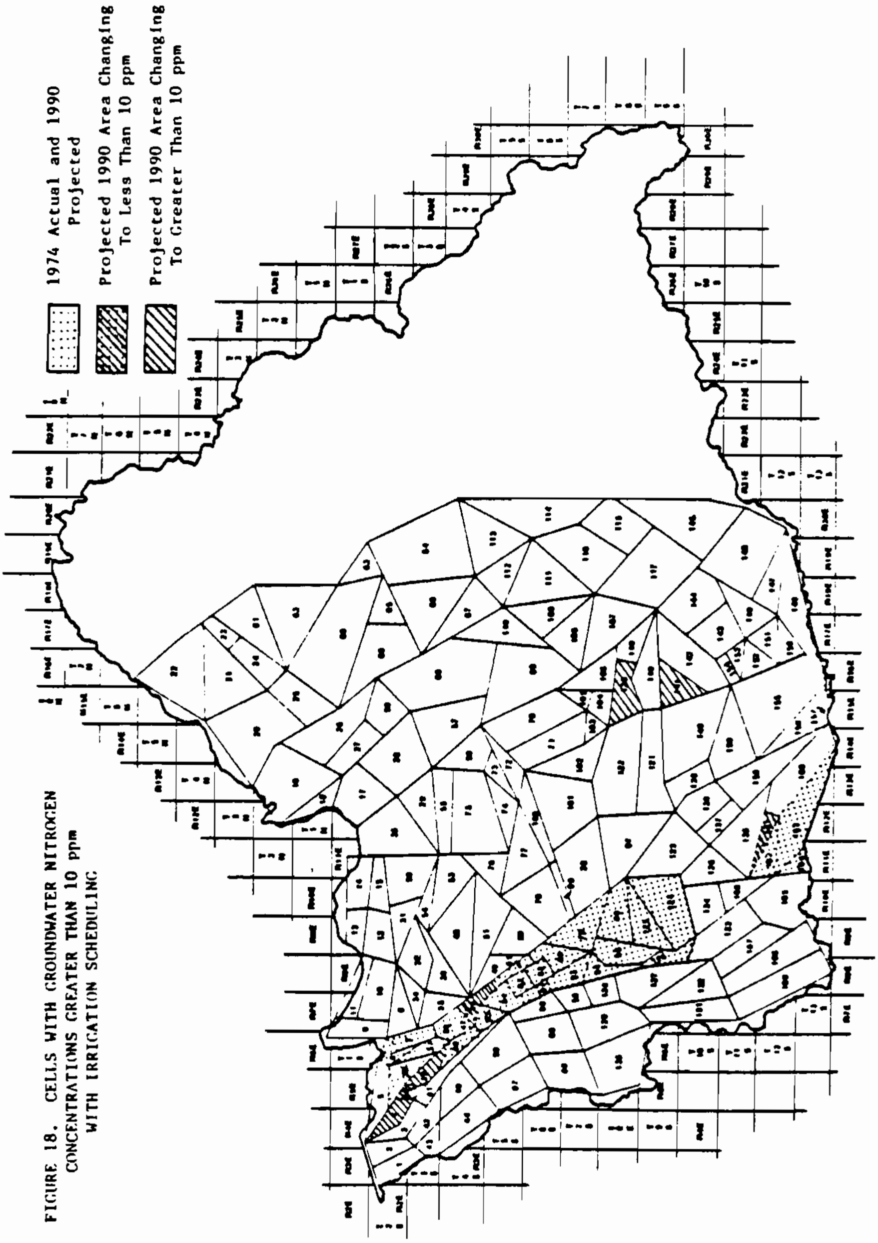


FIGURE 18. CELLS WITH GROUNDWATER NITROGEN CONCENTRATIONS GREATER THAN 10 ppm WITH IRRIGATION SCHEDULING

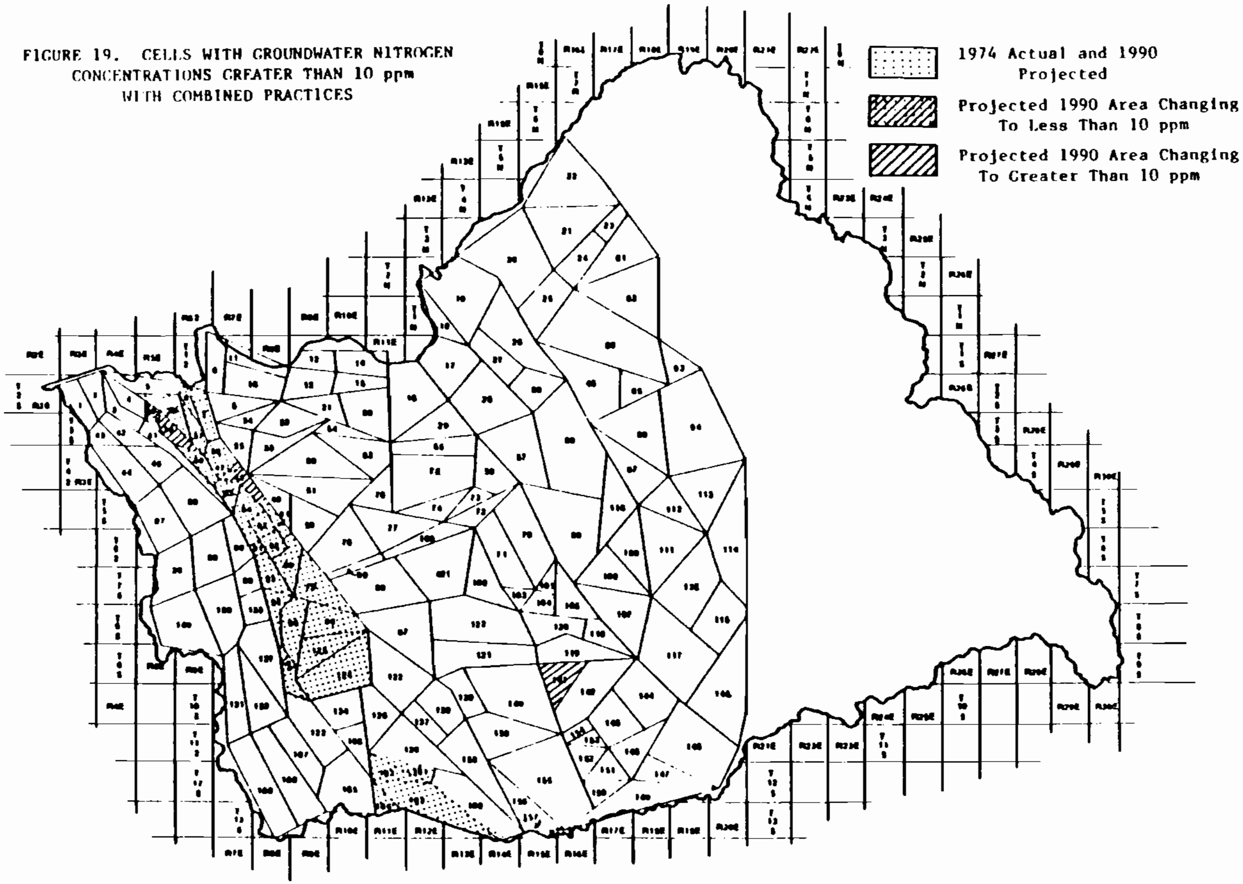


FIGURE 19. CELLS WITH GROUNDWATER NITROGEN CONCENTRATIONS GREATER THAN 10 ppm WITH COMBINED PRACTICES

concentrations are rapidly increasing and to devise corrective action to change those trends despite the fact that these areas may not currently exceed standards. In Figure 20, cells that experienced a doubling in groundwater nitrogen concentrations are depicted. It is interesting to note that only one of the cells (#120) previously appeared as exceeding the 10 ppm standard. A detailed examination of nitrogen concentrations and land and water use by cell provides some insights for policy action. Of the 15 cells that experienced a doubling of nitrogen concentrations, six of the cells had at least 75 percent of the land cropped and pumped at least 70 percent of the total irrigation water used (Table 3). Twelve of the cells had at least 75 percent of the land irrigated. This intensive cropping and irrigation activity and the relatively low initial concentration of groundwater nitrogen are responsible for the rapid increase in concentrations. Clearly, policies other than those analyzed in this study may be needed to to alter these projected outcomes.

CONCLUSIONS

The nitrogen concentration of groundwater has been used as an example of a resource problem that may or may not be mitigated through public action. The primary point of this analysis is that partial analyses will not give public decision makers sufficiently precise information to make optimal policy choices. It is imperative that the analytical system be comprehensive and flexible in order to provide the opportunity to test alternative water quality policies and evaluate their effects upon related resource uses. Nitrogen pollution of groundwater is an excellent case in point.

FIGURE 20. CELLS IN WHICH GROUNDWATER NITROGEN CONCENTRATIONS DOUBLED BETWEEN 1974 AND 1990

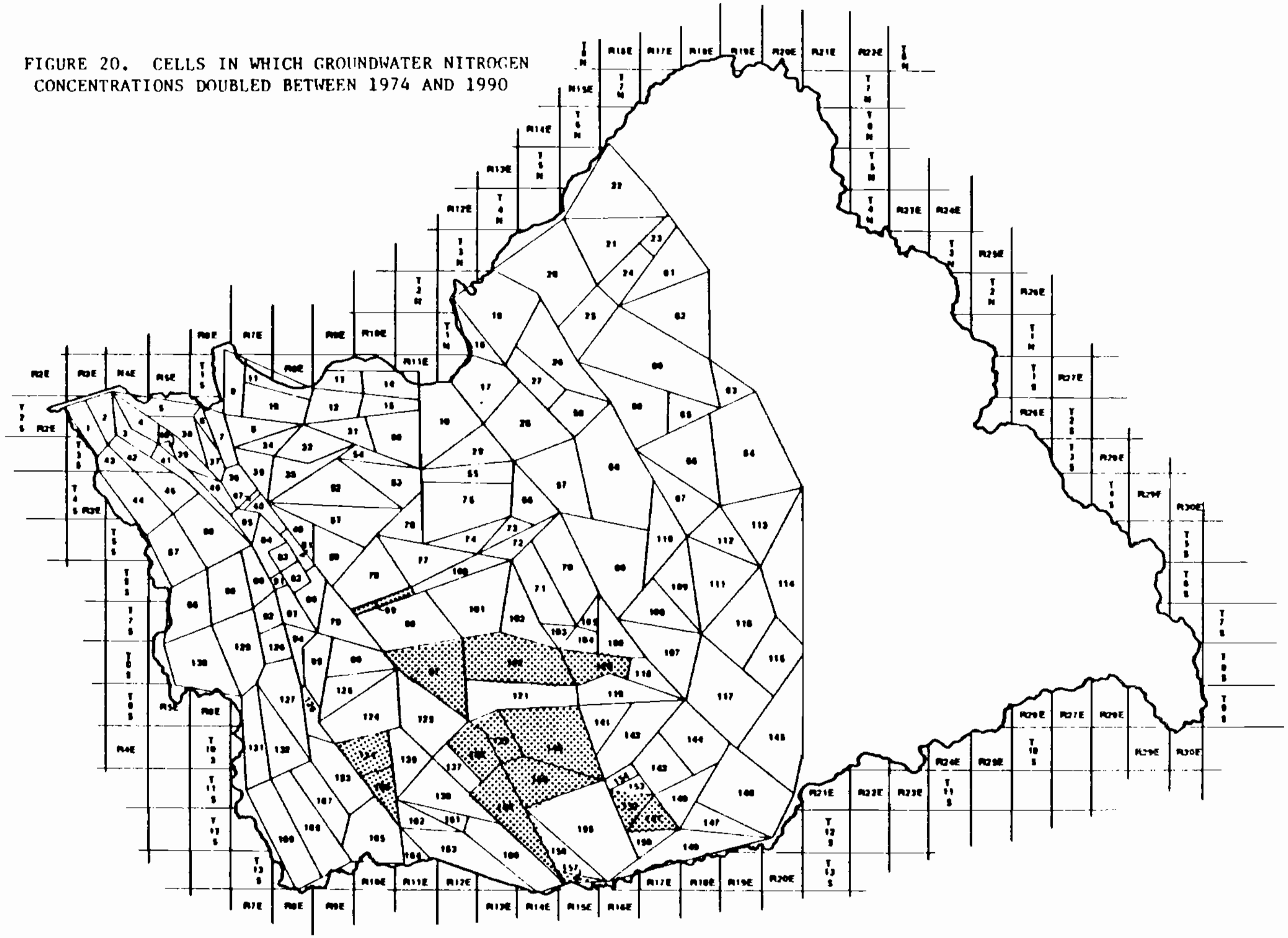


Table 3. Projected Increases in Groundwater Nitrogen Concentrations by Water Quality Management Policy from 1974 to 1990

Cell	Water Management Policy: Base	Tailwater Recycle	Sched-uling	Combi-nation	Percent of Cell Irrigated	Percent of Total Irrigation Water Pumped
	----% Change From 1974 to 1990------					
97	123	93	121	96	38	73
99	144	108	141	112	99	74
120	531	359	509	362	83	34
122	241	197	246	209	80	63
123	109	85	110	90	77	87
134	174	136	173	141	83	20
138	103	75	99	78	98	70
139	205	152	197	152	68	37
140	332	243	312	247	94	23
151	441	391	439	401	98	68
152	130	110	129	113	74	99
157	170	159	170	162	93	100
158	100	56	82	52	96	38
159	103	93	102	95	62	62
166	105	81	106	89	93	100

Two conclusions can be inferred from the results of this analysis. First, high groundwater tables and the nitrogen load in return flows may be adversely affected by imposing irrigation water management practices that reduce groundwater pumping. Second, the reduction of groundwater nitrogen concentrations may be an extremely costly water quality objective to achieve by changing irrigation practices. The analysis and comparison of a much wider range of resource management policies is clearly needed. Previous studies have shown that nitrogen in irrigation return flows can be reduced at least cost by implementing a system of effluent charges (Horner, 1975). The impacts of such externality pricing could be simulated and tested within this analytical system. Land use changes may also be required in order to reduce the rapid increases in groundwater nitrogen concentrations. Potential nitrogen losses from the root zones of shallow rooted crops that require high fertilization can be substantial. Accounting for this pollution potential in crop selection may be possible and desirable.

Clearly, there exists a wide array of technologic and institutional possibilities for the management of nonpoint sources of pollution. The problem of assessing the effectiveness of these alternatives in mitigating water pollution, however, is a very complex and demanding problem. One must not only establish the physical capacity of the management alternative to control the problem, but also the socio-economic impacts of implementing the control strategy. Each of these assessments mandates an integrated systems view of the problem.

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